Watershed Health in Wilderness, Roadless, and Roaded Areas of the National Forest System

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Photo by John McCarthy

Introduction

This white paper presents the results of a national-scale overlay of watershed condition data on three general types of land management categories in the 193 million-acre National Forest System – designated Wilderness, Inventoried Roadless Areas, and all other lands. The findings presented here are made possible by new information about watershed conditions generated through the U.S. Forest Service's Watershed Condition Framework (USDA Forest Service 2011a).

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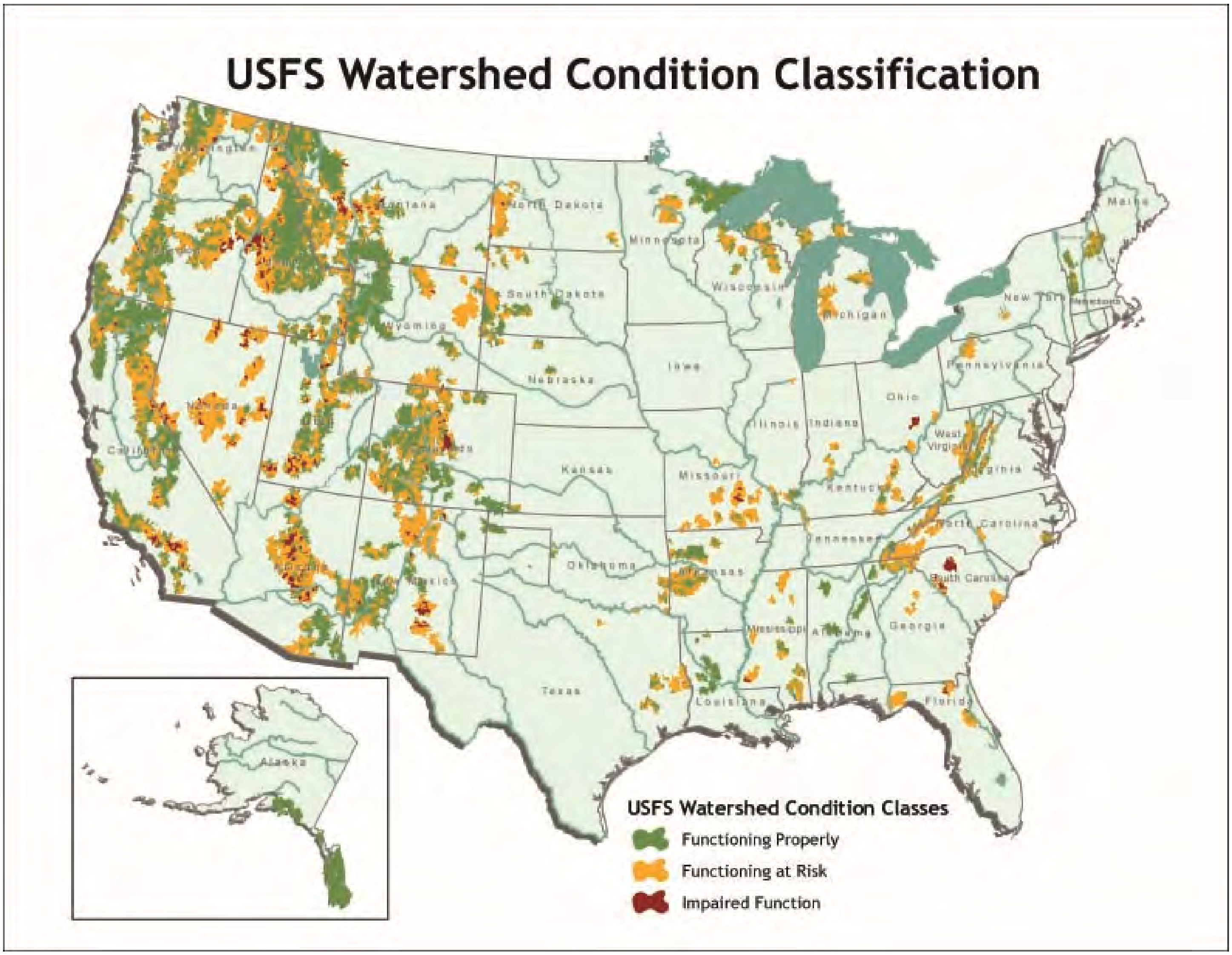
Watershed Condition Framework

On June 3, 2011, Secretary of Agriculture Tom Vilsack announced the release of a national map that characterizes the health and condition of National Forest System lands in more than 15,000 watersheds across the country (USDA Press Office 2011). The U.S. Forest Service's Watershed Condition Classification Map is the first step in the agency's six-step Watershed Condition Framework process, and is the agency's first national assessment of watershed health across all 193 million acres of National Forest System lands. It is also the first time that the Forest Service has created a process to allow data from local watershed assessments to be collected and evaluated at the national level.

The Forest Service's watershed framework identifies three watershed condition classifications: Class 1 for "properly functioning", Class 2 for "functioning at risk," and Class 3 for "impaired." These represent watersheds that display high, medium, or low "geomorphic, hydrologic,and biotic integrity relative to their natural potential condition" (USDA Forest Service 2011a). The national Watershed Condition Classification Map is displayed in Map 1.



Map 1. This map of the USFS Watershed Condition Framework illustrates a new assessment of watershed health across all lands of the National Forest System.

The condition class mapping was undertaken by local Forest Service interdisciplinary teams using a national set of 12 watershed condition indicators, which are listed in Table 1. Each of the dozen indicators was assessed through a simple score card approach using a defined set of numeric, descriptive, or map-derived attributes. For example, the Aquatic Habitat condition indicator was evaluated using three attributes: habitat fragmentation, large woody debris, and channel slope and function.

WATERSHED CONDITION INDICATORS	
Watershed Quality	Riparian/Wetland Vegetation
Water Quantity	Fire Regime or Wildfire
Aquatic Habitat	Forest Cover
Aquatic Biota	Rangeland Vegetation
Roads and Trails	Terrestrial Invasive Species
Soils	Forest Health

Table 1. A list of the twelve watershed condition indicators that contributed to the USFS Watershed Condition Framework.

Detailed instructions for applying the indicators and associated attributes and for computing the watershed condition scores are contained in a technical guidebook (USDA Forest Service 2011b). Recognizing the wide variety of ecological settings across the National Forest System, the assessment process relied on local professional expertise and judgment to interpret the indicators and assess watershed condition. The Forest Service's sampling of 15,000 watersheds provides a detailed data source and opportunities for robust national-level analysis.

Land Management Categories

The National Forest System can generally be divided into three broad land management categories: designated Wilderness areas, Inventoried Roadless Areas, and all other lands (commonly referred to as the "managed landscape" or "roaded areas"). The proportion of National Forest System land within each of the three categories is displayed in Figure 1 and the location of these land management areas is shown in Map 2.

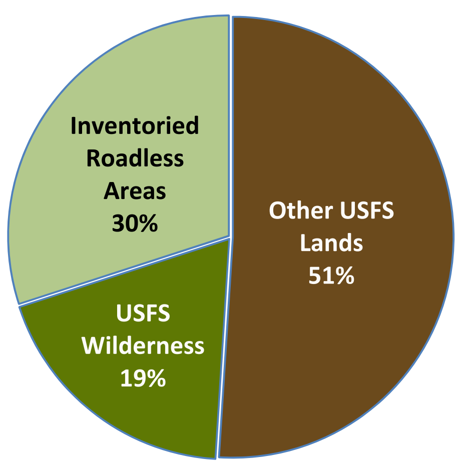
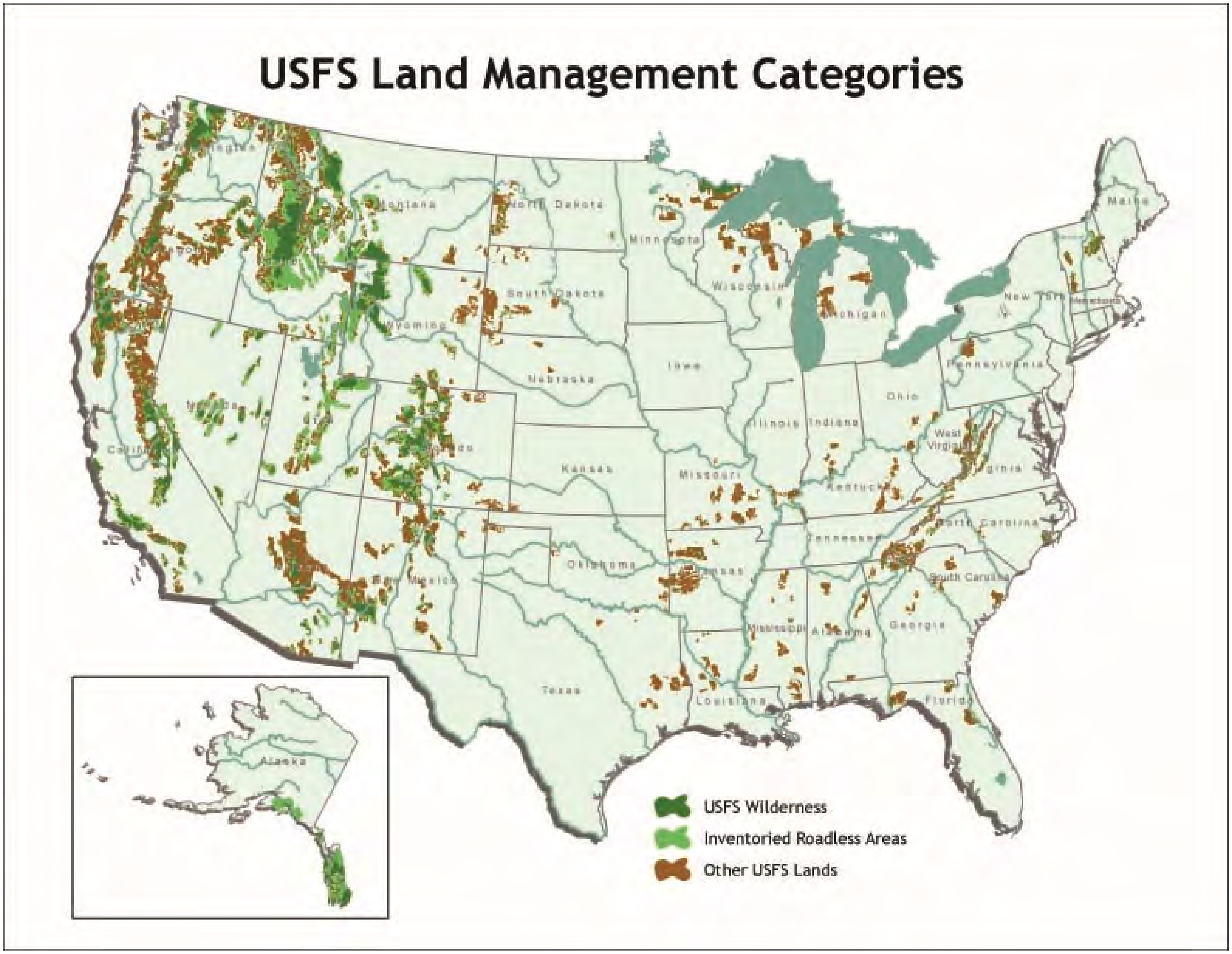


Figure 1. Land management categories as a percentage of the total National Forest System.



Map 2. This map illustrates three levels of protection for National Forest System lands: congressionally designated Wilderness, lands identified as roadless under the 2001 Roadless Area Conservation Rule, and all remaining National Forest System lands.

Congress has designated a total of 439 national forest Wilderness areas, covering 36.2 million acres or 19 percent of the entire National Forest System. The Wilderness Act of 1964 does not specifically mention watershed health or water quality as selection criteria or management objectives. Instead, the Act defines Wilderness as "undeveloped Federal land retaining its primeval character and influence, without permanent improvements or human habitation, which is protected and managed so as to preserve its natural conditions."[5] The Act generally prohibits road building, logging, mining, and motor vehicles. Wilderness designation provides the strongest level of legal protection among the three land management categories used in this analysis. However, livestock grazing is generally allowed in all three land categories.

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[5] 16 U.S.C. 1131(c).

Inventoried Roadless Areas include 58.5 million acres that the Forest Service identified as warranting long-term administrative protection under the 2001 Roadless Area Conservation Rule (USDA Forest Service 2000). Representing 30 percent of all National Forest System lands, Inventoried Roadless Areas are unevenly distributed across the country, but are located in most national forests and grasslands. The 2001 Roadless Rule's definition of roadless area characteristics lists high quality or undisturbed soil and water and sources of public drinking water as among the resources that are often present in or characterize Inventoried Roadless Areas.[6] The Roadless Rule generally prohibits road building and commercial logging, but – unlike the Wilderness Act – does not regulate mining or motorized recreation. Thus, the Inventoried Roadless Areas represent an intermediate level of protection between Wilderness and roaded portions of the National Forest System.

The remaining 99 million acres or 51 percent of the National Forest System encompass a wide variety of lands with different land management histories and objectives. Representing the "managed landscape," these lands contain the vast majority of the 370,000 miles of roads in the National Forest transportation system. Much of the land has been logged, reforested, mined, and otherwise managed for commodity extraction or other commercial uses, but some areas are lightly roaded and contain relatively intact old-growth forests and riparian vegetation. Management direction and environmental safeguards are primarily contained in the local land and resource management plans developed by the Forest Service pursuant to the National Forest Management Act of 1976 and the Multiple-Use Sustained-Yield Act of 1960. During the past decade, Congressional laws and agency policies have increasingly emphasized restoration of national forest lands and resources.[7]

Purpose and Limitations

The purpose of this analysis is to evaluate, quantify, and display at a national scale the spatial relationships and correlations between the three watershed condition classes and the three land management categories discussed above. It is not intended to identify causal relationships; therefore, we have not attempted to identify and isolate any potentially confounding variables, of which there are undoubtedly many. Nor have we attempted to conduct any geographic analysis smaller than the national scale (such as by Forest Service regions, individual states, or specific national forests). While we recognize that there is potential for significant regional and local variability in these relationships, such smaller-scale evaluation is beyond the scope of this analysis. Suggestions for additional analysis along these lines are presented in the section on Further Research.

Methods

This analysis was completed using Geographic Information Systems (GIS) data from a variety of sources. The U.S. Forest Service (USFS) supplied both the Watershed Condition Framework (WCF) and Inventoried Roadless Areas (IRA) datasets.[8] The agency did not supply the USFS boundary information, as the detailed data layer used in the WCF analyses was not available for public distribution. Several USFS boundary data layers were considered in its place, and the U.S. Geological

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[6] 36 C.F.R. 294.11.

[7] Examples include the Healthy Forest Restoration Act of 2003, the Collaborative Forest Landscape Restoration Act of 2009, and congressional appropriations since 2007 for the Legacy Roads and Trails Remediation Program.

[8] Watershed Condition Framework: http://www.fs.fed.us/publications/watershed/. IRA, lower 48 states: http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsm8_037469.html . IRA, Chucagh National Forest: http://fsgeodata.fs.fed.us/rastergateway/alaska/chugach/roadless.html. IRA, Tongass National Forest: http://seakgis.alaska.edu:8080/geoportal/catalog/main/home.page.

Survey Protected Areas Database had the most detailed boundaries in a national-scale dataset.[9] Lastly, Wilderness.net, a multiple government agency partnership,[10] supplied the Wilderness dataset.[11]

Each source dataset was processed initially to a common map projection (Albers Equal Area Conic), as well as error-checked for obvious spatial incongruities, and queried if needed to extract the records needed for overlay analysis.

Additional processing was required to finalize the IRA dataset. When this dataset was overlaid with the Wilderness dataset, there were significant areas of overlap. Since the Wilderness and IRA designations are mutually exclusive, these datasets should not have these overlapping areas. It was assumed that the overlapping areas could be attributed to the greater accuracy and up-to-date status of the Wilderness dataset, and that many of the IRA designations in the IRA dataset had in fact been re-designated as Wilderness. For this reason, all the overlapping areas in the IRA dataset were removed, but retained in the Wilderness dataset. In addition, the polygons in the resulting IRA dataset were analyzed and those losing greater than or equal to 92 percent of their area (representing about 23 polygons and 16,874 acres nationwide) were also removed.

The Wilderness, IRA, and USFS boundary datasets were then merged and further processed to facilitate analysis. The analysis consisted of a spatial overlay between the merged dataset and the WCF dataset, followed by a frequency analysis to determine national acreage totals by land designation classes and WCF condition classes.

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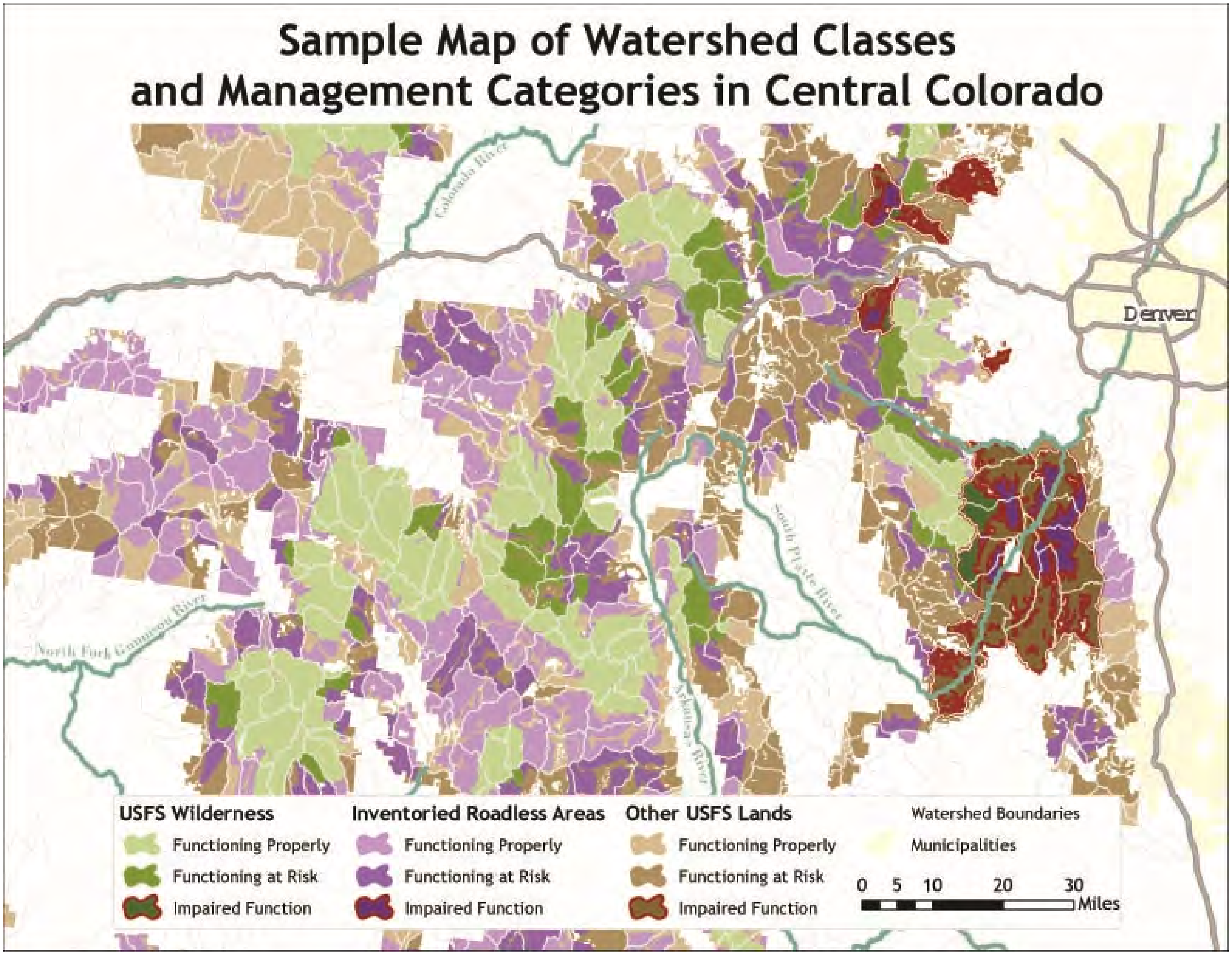
9 USGS PAD: http://gapanalysis.usgs.gov/data/padus-data/padus-data-download/

10 Wilderness.net is a collaborative partnership between the College of Forestry and Conservation's Wilderness Institute at The University of Montana, the Arthur Carhart National Wilderness Training Center, the Aldo Leopold Wilderness Research Institute, the US Bureau of Land Management, the US Fish & Wildlife Service, the US Forest Service, and the US National Park Service.

11 Wilderness: http://www.wilderness.net/index.cfm?fuse=NWPS&sec=geography



Map 3. This map illustrates the nine possible combinations of three watershed condition classes and the three land management categories for a sample of National Forest lands in Colorado.

Map 3 provides an illustrative example from Colorado of the GIS overlay analysis of watershed conditions and land designations. The analysis produced nine combinations of the three different watershed condition classes (Properly Functioning, Functioning At Risk, and Impaired) and the three land designations (Wilderness Areas, Roadless Areas, and Other National Forest Lands). The Colorado example map displays several of the nine combinations of watershed conditions and land designations on national forest lands in the vicinity of Denver.

Results

Our GIS overlay analysis found a strong spatial association at a national scale between watershed health and protective land designations in the National Forest System. The overall results are displayed in Table 2 (by acreage) and Figure 2 (by percentage).

	Properly Functioning	Functioning At Risk	Impaired Function	Total Land in USFS Management Category
USFS Wilderness	29.0	6.7	0.5	36.2
Inventoried Roadless Areas	36.7	19.2	1.2	57.1
Other USFS Lands	37.4	56.9	4.6	98.9
Total USFS Land in Condition Class	103.1 (54%)	82.8 (43%)	6.3 (3%)	192.2 (100%)

Table 2. Acreage of land in the nine possible combinations of watershed condition classes and land management categories in the National Forest System (in millions of acres).[12]

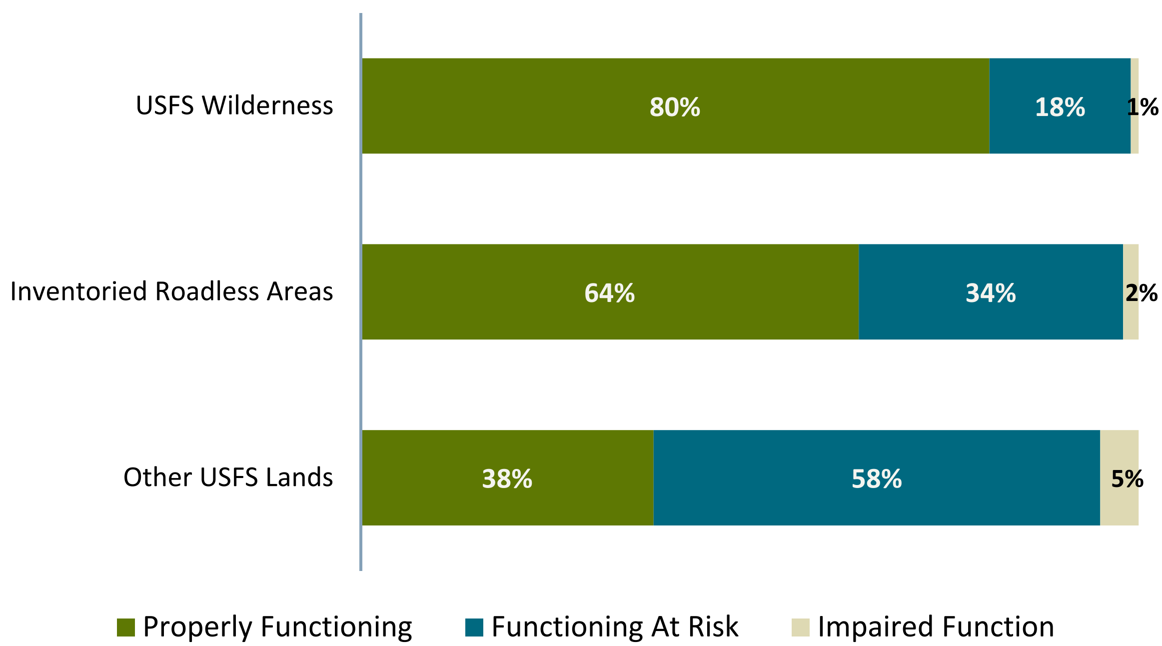


Figure 2. For each land management category, this chart shows the percent of its area in each of the three watershed condition classes.

The Watershed Condition Framework data identifies 54 percent of all NFS land in properly functioning watersheds, 43 percent in watersheds functioning at risk, and just 3 percent in impaired watersheds. However, these proportions are not evenly distributed across the three land designation categories.

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[12] Acreage figures are calculated from the best available GIS data at the national scale, but do not always exactly match USFS published acreage figures. However, variances in acreages are small relative to the acreages of overlapping watershed classes and lands protection categories that are the subject of this paper.

Designated Wilderness areas are most frequently spatially coincident with healthy watershed conditions. Eighty percent of the land within designated Wilderness is located in properly functioning watersheds, while 18 percent is in at-risk watersheds and just 1 percent is in impaired watersheds.

Watershed conditions in Inventoried Roadless Areas are not as healthy as in designated Wilderness, but almost two-thirds of their area is still in properly functioning condition. Sixty-four percent of the IRA acreage is in properly functioning watersheds, 34 percent is in at-risk watersheds, and 2 percent is in impaired watersheds.

Finally, other Forest Service lands – which make up slightly more than half of the National Forest System – tend to have the least healthy watershed conditions. While 38 percent of the managed landscape is in properly functioning watersheds, most of the roaded lands are in watersheds that are either functioning-at-risk (58 percent) or impaired (5 percent).

Discussion

The results of this GIS overlay analysis suggest that watershed conditions tend to be best in areas protected from road construction and development. National forest lands that are protected under the Wilderness Act, which provides the strongest safeguards, tend to have the healthiest watersheds. Watersheds in Inventoried Roadless Areas – which are protected from road building and logging by the Roadless Area Conservation Rule – tend to be less healthy than watersheds in designated Wilderness, but they are considerably healthier than watersheds in the managed landscape. Of course, an area's physical characteristics and management history may well have a greater impact on watershed condition than its current legal status. Wilderness areas are typically large tracts of wild land designated by Congress on the basis of their pristine natural features, including the absence of roads and clearcuts. Roadless areas, by definition, contain at least 5,000 acres that are generally free of roads and associated development.

As noted in the Introduction, this analysis does not attempt to identify causal relationships, since there are many other variables besides land designations that could be at play. Factors such as elevation, temperature, and precipitation might explain differences in watershed conditions better than land designations do. Some of the associations may simply be a function of the way in which the Watershed Condition Framework assessment index was constructed. For example, one of the twelve assessment indicators was road and trail condition, which included attributes of road/trail density, maintenance, proximity to water, and risk of mass wasting. Since Wilderness and roadless areas typically have no roads, this part of the assessment process may tend to bias the results toward better condition ratings in those areas.

However, the relationship between forested wild lands and watershed health is well documented in the scientific literature. For example, the Forest Service's Interior Columbia River Basin Ecosystem Assessment (1996) found a positive relationship between unroaded areas and "strongholds" of high-quality habitat for salmon, steelhead, bull trout, and other key salmonid species. An evaluation of the role of Wilderness areas in conserving aquatic biological integrity in western Montana concluded that "the importance of wilderness in aquatic conservation is extraordinary" (Hitt and Frissell 2000). In contrast to Wilderness and roadless areas, "the roaded, intensively managed landscapes of the other national forest lands have been closely correlated with heavily sediment-laden streams and dramatic changes in flow regimes" (DellaSala et al. 2011). The Forest Service's environmental impact statement for the Roadless Rule explains that the presence of roads has a major influence on stream and watershed conditions: "Without the disturbances caused by roads and the activities that they enable, stream channel characteristics are less likely to be adversely altered compared with stream channel conditions in roaded areas" (USDA Forest Service 2000).

Further Research

The Forest Service's Watershed Condition Framework Condition Classification Map opens up new research opportunities to improve our understanding of how to maintain and improve healthy watersheds. Our analysis is an example of how the Watershed Condition Framework data can be applied at a national scale to correlate watershed condition and land management categories. Following are a few examples of further research needs and opportunities:

Similar, finer-scale analyses could be done at a more local level – such as for a single state or national forest – using the same management categories or other map-based categories that are relevant to a local jurisdiction – such as forest plan management areas.

GIS overlay analysis could explore the relationship between watershed health and various physical characteristics such as elevation, precipitation, slope, soils, and fire history.

Statistical multi-variate analysis could help explain differences in watershed condition by isolating certain attributes, such as determining the extent to which road density affects the condition classification of Wilderness and roadless areas.

Further research could examine vulnerability of watershed condition to climate change, species invasion, uncharacteristic fire, and other anticipated changes.

Conclusion

Covering more than 15,000 individual watersheds across the National Forest System, the Forest Service's Watershed Condition Classification Map provides a useful means of comparing watershed conditions with a variety of geophysical features, management histories, and other variables. Our nationwide GIS overlay of the three watershed condition classes with three broad land management designations – Wilderness, Inventoried Roadless Areas, and roaded areas – found a strong spatial association between watershed health and protective designations. This finding is consistent with previous scientific studies of aquatic resources in roaded and unroaded landscapes. Regional and finer-scale analyses of the watershed condition and land designation data would improve our understanding of the factors that determine watershed health.

Acknowledgements

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THE WILDERNESS SOCIETY

Science & Policy Report

Managing the Landscape for Fire: A Three-Zone, Landscape-Scale Fire Management Strategy

Background

Fire has shaped America's public lands for millennia. From ponderosa pine forests that burn every few decades to spruce-fir stands that erupt into flame every few centuries, most forests have evolved with fire and depend on periodic blazes for their health and regeneration. Fire is such an important force in U. S. ecosystems that vegetation and fire cannot be described independently.

Just as vegetation and fire are intimately connected, land management and fire management must also be inextricably linked. In the last decade, policymakers and forestry experts have come to recognize that a century of fire suppression policies have created a "crisis" in forest health, starving fire-dependent ecosystems of regular fire cycles and creating unhealthy fuel loads that can lead to unnaturally large wildfires in some places. All too often, however, land and resource management plans (LRMPs), the documents that guide all major decisions affecting federal lands, are devised with only cursory consideration of the important ongoing role of fire in the landscape. Even though broad scientific consensus now exists regarding the crucial role fire plays in ecosystem sustainability, few LRMPs specifically address fire management needs.

Because of the intimate connection between land and fire, LRMPs must

In recent years, federal maintenance funding has fallen far short of the amount necessary to maintain the more than 700 miles of Forest Service roads throughout the Hells Canyon National Recreation Area.

themselves be fire plans, and land managers must work to accommodate fire in the development of all LRMPs. If LRMPs fail to account for the role of fire on a landscape scale, other management failures are sure to follow. For example, timber production schedules must take into account the certainty of fire, else inevitable fires will foil expectations by consuming growing stock and reducing future harvests. Similarly, landscape-scale objectives, like the maintenance of sufficient wildlife habitat to sustain viable populations, can only be achieved by relying on the landscape-scale process of fire. LRMPs must be developed to account for natural fire and use it wherever possible to achieve plan objectives. Public lands, with their large tracts of undeveloped areas, provide federal agencies with a vital opportunity to use natural fire to achieve social and ecological goals.

Key Points

- Land and Resource Management Plans (LRMPs) too often fail to incorporate fire management as an essential part of the planning process, giving only cursory consideration to the important role of fire in the landscape.
- Federal agencies should standardize the inclusion of fire-management goals into LRMPs by using a three-zone strategy that helps managers determine the appropriate level of mitigation against, preparation for, and response to, the inevitable wildland fire.
- The "Community Fire Planning Zone" (CFPZ) is the area within a half-mile of communities in which fire should generally be excluded. Land managers should seek opportunities to improve public safety through infrastructure improvement and fuel treatment to protect homes.
- The Restoration Planning Zone (RPZ) extends a few miles beyond the CFPZ to a distance where it is safe to consider additional management approaches as a supplement to aggressive initial attack. Within the RPZ, prescribed fire and mechanical thinning may be used to protect critical resource values and restore conditions that are resilient to inevitable fires.
- The "Fire Use Emphasis Zone" (FUEZ) is the area beyond those zones where the full range of management responses to fire (from suppression to allowing natural fire) is possible. In these wilderness, roadless, and remote roaded areas, priority should be placed on Wildland Fire Use for Resource Benefit (WFU) when conditions allow.
- The development of such landscape-scale fire management classifications requires creation of a map that clearly demarcates the three zones using a combination of readily available national and statewide GIS spatial data and local expertise.

Landscape Fire Planning Zones

Land and resource management plans are, at their core, documents that define relationships between landscapes and people. In any landscape, there are three situations with regard to communities and fire.

- First, there are those situations where fire has the potential to cause great damage to people and homes, and should always be excluded. Areas where wildlands come into contact with communities — the wildland-urban interface — are an example.
- Second, there are places where fire can be used as a tool to reduce fuels and restore ecosystems, but only under tightly prescribed conditions.
- Third, there are places where fire poses little risk to people and resources, and natural fires can actually help achieve management objectives, such as fuel reduction and provision of wildlife habitat.

We recommend that federal agencies develop a landscape-scale, three-zone fire management strategy across each administrative unit that reflects these three situations, and that they incorporate these zones into all LRMPs.

The "Community Fire Planning Zone" (CFPZ) exists immediately adjacent to communities and is managed for their protection.

The "Restoration Planning Zone" (RPZ) occurs beyond the CFPZ for some distance (a few miles) and is managed to minimize unplanned fire (through suppression or containment) but also to restore conditions that are resilient to inevitable fires.

Beyond those zones, the full range of management responses to fire (from suppression to allowing natural fire) is possible, but a priority is placed on Wildland Fire Use for Resource Benefit (WFU). This area is called the "Fire Use Emphasis Zone" (FUEZ) to reflect the preference for WFU when conditions allow.

By developing LRMPs with fire in mind, LRMPs can serve as practical templates for subsequently developed Fire Management Plans (FMPs). FMPs are planning documents required by policy for all lands with burnable vegetation (USDA Forest Service et al. 2001). They provide the strategic foundation for all fire-related management activities on a given land management unit before, during, and after a wildland fire. FMPs are developed to aid implementation of the LRMP and must be consistent with all land designations made in the LRMP.

These three planning zones can improve management of public lands by focusing resources where they are most valuable and helping to restore natural processes to those lands that can benefit from the restoration of natural fire regimes.

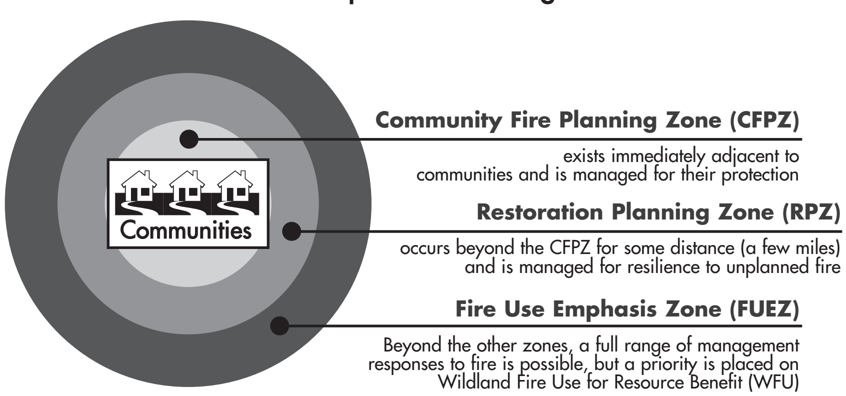
The Community Fire Planning Zone (CFPZ)

The highest priority of fire management must be the protection of people and their homes, and LRMPs must be structured to support this goal. Thus, the first step in designing a plan that addresses fire is to identify the "Community Fire Planning Zone," the area around communities that should be managed to protect homes and structures from wildland fire. This zone is sometimes called the "wildland-urban interface," but Community Fire Planning Zone (CFPZ) better conveys the overriding objective of community protection. The CFPZ is that area in and around communities that should be examined for opportunities to improve public safety through infrastructure improvement and fuel treatment to protect homes. It will not be necessary to treat fuels everywhere within that zone, but quantifying the extent of the area where communities are at risk from wildland fire can help focus community protection efforts.

It has been demonstrated that the most effective way to protect homes is to build them out of fire-resistant materials and aggressively reduce nearby fuels. The simple principle behind this notion is that homes will not burn if they do not ignite, regardless of what happens to the sur-

FIGURE 1.

Landscape Fire Planning Zones



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this is from the fire report photos rick sent to me

rounding forest, and research by the U.S. Forest Service has shown that a very narrow "home ignitability zone" of approximately 60 meters determines whether a home will burn. By clearing highly flammable fuels near homes, thinning small-diameter trees within 60 meters of homes, and building with non-flammable materials, especially roofs, fire risk to homes can be dramatically reduced (Cohen and Butler 1998, Cohen 2000).

Beyond the 60-meter home ignitability zone, communities may wish to thin trees to create "defensible space" within which firefighters may work safely, to reduce the probability of crown fire and to protect scenic views or watershed quality. Nowicki (2002) applied rules of thumb developed by fire physicists and fire safety personnel to conclude that community protection zones of 400 meters could provide an area that would allow firefighters to work safely to protect structures.

In 2003, The Wilderness Society released The Wildland Fire Challenge report (Aplet and Wilmer 2003), which suggested that a buffer distance of a half-mile may be necessary to provide the latitude needed to adjust community fire planning zones to terrain, taking advantage of natural fuel breaks such as cliffs and rock outcrops. While there may occasionally be situations that require extension of the CFPZ to greater distances, we encourage the federal agencies generally to employ a CFPZ up to one-half mile beyond communities (Wilmer and Aplet 2005).

If there are situations where extending the width of the CFPZ helps improve community safety, it may fairly be asked, "Why limit the width of the CFPZ at all?" The answer is that management for community protection may compromise other resource objectives. Treating fuels to protect homes may result in unnatural forest conditions that compromise wildlife habitat, water quality, and aesthetics. It is therefore important to limit the CFPZ to the area where it will do the most good to protect homes. Narrowing the width of the CFPZ also helps to focus limited resources (money, personnel) where they will have the greatest impact.

It is important to emphasize here that this logic does not argue for clearing a half-mile buffer around every community. Rather, the CFPZ is the area within which to look for opportunities to treat fuels to protect homes. Not every type of vegetation will need to be treated, and there are some vegetation types, such as chaparral and subalpine forest, within which thinning will be only marginally effective at lowering the probability of crown fire. However, treatment near homes (and the use of fire-resistant building materials) can be very effective at increasing the chance that a home will survive the inevitable crown fire.

Efforts to map the wildland-urban interface or CFPZ have shown that community protection is predominantly a private land challenge, but where the CFPZ overlaps with federal land, there is an important role for the federal agencies (Wilmer and Aplet 2005). Management within the CFPZ consists of actions that minimize the threat of fire to homes. Obviously, paramount among those actions is aggressive suppression when fires start. The CFPZ is a place where, ideally, fire is excluded. This task is enhanced by sufficient suppression infrastructure, such as hydrants and access roads, as well as suppression forces ready to attack at a moment's notice. It is also enhanced by fuel treatments, such as mowing and pruning, to minimize fine fuels that contribute to rapid fire spread.

But absolute fire exclusion is, unfortunately, wishful thinking. We will never be able to keep fire out of the CFPZ completely. Accordingly, precautions must be taken so that, when fire does eventually burn, that fire poses a minimal risk to homes. Such precautions include reducing tree density (thinning) near homes to reduce heat output during fires. Reducing heat output may keep homes from igniting and give firefighters the space they need to protect structures. Fortunately, many of these precautions have been formalized for public education through programs such as FIREWISE (see www.firewise.org).

Historian Stephen Pyne (2003) has called structure loss in the CFPZ "a dumb problem to have" because it is preventable. Within the CFPZ, we know what must be done to minimize fire risk; we simply need the will to do it. Pyne imagines a future in which people are "active agents in shaping the fire regime of their surroundings, not simply passive victims and whining litigants."

Becoming an "active agent" can be achieved in two ways. First, homeowners must manage their property to minimize risks to their homes and their neighbors. Second, community members, including the federal agencies, must work together across ownerships to develop plans that meet community fire protection needs.

The Community Wildfire Protection Plan (CWPP) process, established in the Healthy Forests Restoration Act of 2003, provides an excellent opportunity for citizens and agency managers to work together to achieve common goals for the CFPZ. CWPPs are to be developed by multiple stakeholders to identify and prioritize areas for hazardous fuel reduction and to recommend measures to


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reduce structural ignitions. Because CWPPs must be considered in the evaluation of federal fuel reduction projects, federal agencies should be part of every CWPP process involving communities whose CFPZ overlaps with federal land. Where these processes have not already begun, we encourage federal agencies to pull stakeholders together to develop these plans.

Various resources exist to help facilitate this engagement, including "A Handbook for Wildland-Urban Interface Communities: Preparing a Community Wildfire Protection Plan," developed by the Society of American Foresters, the National Association of State Foresters, the National Association of Counties, and the Communities Committee of the Seventh American Forest Congress.[1] The "Leaders' Guide for Developing a CWPP" by the International Association of Fire Chiefs, the National Association of State Foresters, and The Wilderness Society is also an excellent resource.[2]

The Restoration Planning Zone

The Restoration Planning Zone (RPZ) extends beyond the CFPZ to a distance where it is safe to consider additional management responses to fire as alternatives to aggressive initial attack. Within the RPZ, suppression will be the response to unplanned ignitions, but fire may also be introduced intentionally to achieve management objectives. There, the primary management objectives are the protection of critical resource values, such as recreation sites, experimental forests, and research natural areas, and the maintenance of forest composition and structure that is resilient when the inevitable fire occurs. Generally, this means modifying fuels to protect specific resources and restoring ecosystems, based on an understanding of the historical range of variability (Landres et al. 1999). These objectives can be accomplished under a variety of management prescriptions for different land uses, from commodity production to roaded recreation to roadless areas to passive or active restoration.

While some may argue that the RPZ should be as broad as possible to facilitate restoration across the maximum extent of the landscape, there are many practical reasons to constrain the RPZ. First, the larger the RPZ, the more land must be managed under an obligatory suppression response, which has proven to be more difficult and expensive over time. Constraining the RPZ allows suppression forces to focus on a smaller portion of the landscape where they can be most effective. Second, restoration work is expensive and simply cannot be done everywhere. So far, restoration work has not paid its own way, and for the foreseeable future, it will need to be supported through taxpayer investments. Sound fiscal management requires that those investments be limited.

Finally, to be effective, restoration must be focused on the places where it is needed most. Throughout the West, the landscapes that are most in need of restoration are those immediately adjacent to communities, often at the base of adjacent mountain ranges. These dry, low-elevation forests of ponderosa pine, Douglas-fir, and various oaks have been the most altered by fire exclusion, and are the most in need of thinning to restore a fire-tolerant forest structure. Constraining the RPZ to the area within a few miles of communities will focus restoration efforts where they will yield the greatest benefit.

Management within the RPZ may be aimed at a number of objectives, including commodity production, viewshed conservation, recreation, and scientific study, but except in specific locations,

[1] http://www.safnet.org/policyandpress/cwpphandbook.pdf

[2] http://www.iafc.org/Grants/documents/CWPP_rev062005.pdf


CREDIT??

Fuel treatment on the Deschutes National Forest. Small-diameter trees were thinned to restore a fire-resilient forest structure. Fine fuels created by the thinning operation will be subsequently burned to reduce fire hazard.

such as campgrounds and experimental forests, management should adhere to principles of ecological restoration. One such set of principles can be found in the article "A Citizen's Call for Ecological Restoration: Forest Restoration Principles and Criteria" by DellaSala et al., published in Ecological Restoration in 2003. This article contains a number of sound ideas that should be applied to restoration planning. At the center of the document are three "core principles" upon which a good restoration plan should be based:

1. Ecological Forest Restoration Core Principle: Enhance ecological integrity by restoring natural processes and resiliency.
2. Ecological Economics Core Principle: Develop and employ the use of economic incentives that protect or restore ecological integrity.
3. Communities and Work Force Core Principle: Make use of or train a highly skilled, well-compensated work force to conduct restoration.

A LRMP is a solid restoration plan if it restores processes, such as vegetation development or characteristic hydrology and fire, not just forest structure, if it is based on an economics that recognizes ecological costs and benefits, not just market values, and if it contributes to the long-term viability of communities with a culture of environmental sustainability.

The Citizens' Forest Restoration Principles (DellaSala et al. 2003) offers a useful framework for forest restoration that, if incorporated into a broadly inclusive, collaborative planning process, can yield a comprehensive restoration plan. A simpler but also helpful set of guidelines is offered by Brown and Aplet (2000) in their paper "Restoring Forests and Reducing Fire Danger in the Intermountain West with Thinning and Fire." They offer several goals for restoration planning that can be summarized as follows:

1. Focus on water and watersheds
2. Account for rare ecosystem elements
3. Protect riparian areas
4. Focus on low elevations
5. Thin the smallest trees
6. Treat fine fuels with prescribed fire
7. Avoid disturbing soils
8. Avoid creating new roads and protect roadless areas

These simple principles can form the basis of a sound program for the Restoration Planning Zone and should be employed in the development of a LRMP.

The Fire Use Emphasis Zone

In the Fire Use Emphasis Zone (FUEZ), the full suite of management responses (including suppression and containment) may be appropriate under any given condition, but the intent is to maximize opportunities for Wildland Fire Use for Resource Benefit (WFU) where possible. WFU — managing naturally-burning fires in designated, remote sections of the landscape — is widely accepted by scientists and policymakers as an important tool for helping to restore forest health and mitigating the escalating costs of fire suppression. However, in practice, WFU is rarely implemented because it is viewed by fire managers as too risky (Parsons 2000). The only way that the benefits of WFU can be realized over substantial areas is to allow natural fires to burn wherever safe. Designating a FUEZ — the area determined through rigorous analysis to be far enough away from communities that fire will not threaten structures or other highly valued resources — should increase managers' confidence to opt for WFU in the event of a natural ignition.

In order to implement WFU, federal policy requires having a Fire Management Plan (FMP) in place; without an FMP, all unplanned ignitions must be suppressed. Even with a plan in place that authorizes the use of fire in a given area, however, weather conditions, personnel availability, and other variables must be considered before a manager can make a definitive decision to use wildland fire to improve ecosystem condition. Once the initial decision is made, fire managers must constantly monitor and re-assess conditions to see if the fire begins to move out of prescription, at which point suppression will be ordered.

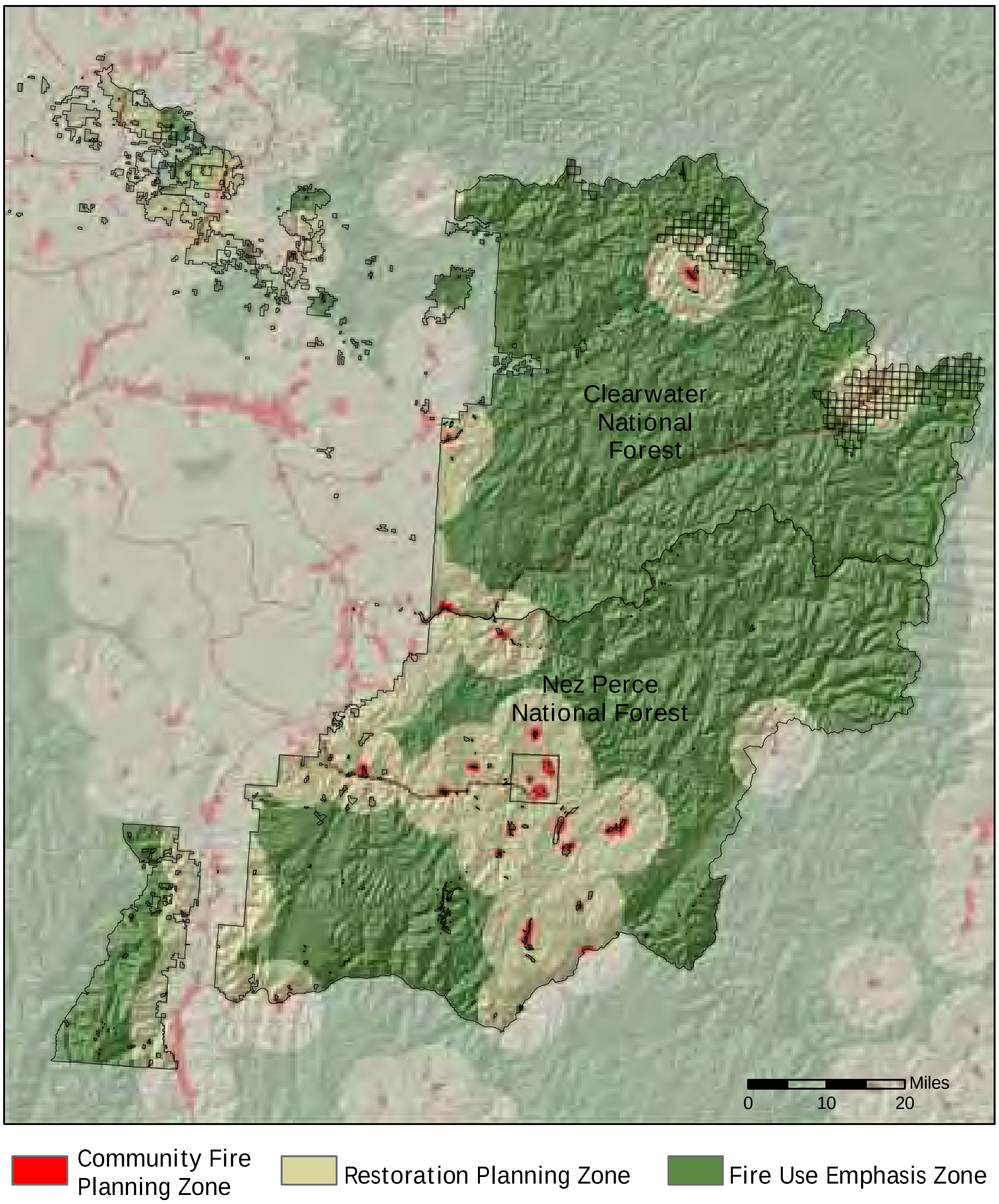
Identifying the specific conditions under which WFU might be appropriate requires detailed scientific and spatial analyses. Even in remote areas, such as the FUEZ, forest conditions, weather and wind factors may preclude the safe use of fire. WFU is only appropriate where the results of fire are likely to produce resource benefits. Generally, this requires a determination that fire behavior will be natural or historically typical for the location. To provide a sufficient basis for fire management, a LRMP would not need to include these detailed analyses, but the plan must provide sufficient latitude to allow fire planners to identify the appropriate places for WFU in the subsequent FMP. Such latitude can be provided by making the FUEZ as big as possible.

Management prescriptions appropriate for the FUEZ range from wilderness and protection of roadless character in the roadless landscape to active restoration and protection of recreation sites in the roaded portion. Throughout this landscape, prescribed fire may be used to achieve a composition and structure that can accommodate natural fire. This is especially true for the roaded portion of the landscape, where existing roads can be used (possibly after thinning of adjacent fuels) to systematically reintroduce

Example: Clearwater and Nez Perce National Forests

The following map displays the Community Fire Planning Zone in the vicinity of the Clearwater and Nez Perce National Forests in Idaho, its overlap with the Forests, the Restoration Planning Zone within five miles of the CFPZ, and the Fire Use Emphasis Zone beyond the RPZ. On these Forests, the CFPZ amounts to only 3 percent of the landscape; the RPZ makes up about 29 percent, about a third of which is wilderness and roadless land, and the remaining 69 percent of the Forest is FUEZ. Seventy-nine percent of the FUEZ consists of wilderness and roadless areas, providing ample opportunity to apply Wildland Fire Use.

Landscape-Scale Fire Management on the Clearwater and Nez Perce National Forests



fire to the landscape.

In the roadless landscape, including wilderness, a higher burden of proof must be met prior to manipulation, including the use of prescribed fire. The Wilderness Act specifically requires meeting that burden through a Minimum Requirements Analysis, but the special values of roadless areas also demand that a high standard be met. As with suppression action, the Wilderness Act does not specifically prevent fuel management in wilderness, but actions proposed for any part of the roadless landscape must be carefully planned using excellent science and an inclusive public process. Because remote areas tend to be in higher-elevation, cooler vegetation types, little of the FUEZ is likely to be in low-severity-fire forest types that may require thinning or prescribed fire before natural fire will yield resource benefits. The vast majority will be in less-frequent fire regimes that will likely benefit from natural fire.

Fire management in the FUEZ should seek to maintain the natural character of the area, even in the roaded portion, and minimize impacts to aquatic, terrestrial, or watershed resources. Accordingly, Minimum Impact Suppression Tactics should be used throughout the FUEZ when suppression is the appropriate management response.

Mapping the Fire Landscape

Developing a LRMP that supports landscape-scale fire management requires the creation of a three-zone map representing the Community Fire Planning Zone, the Restoration Planning Zone, and the Fire Use Emphasis Zone. Creating such a map is a relatively simple matter that relies on a very few readily available spatial data sets:

1. U.S. Census 2000 data at the block level, representing the number of houses in each block.
2. High-resolution land ownership data.
3. Federal land administrative data showing the locations of wilderness, roadless areas, research natural areas, campgrounds, etc.
4. High-resolution vegetation cover data, representing non-wildland cover types and wildland vegetation types.

Mapping Methods

To develop a map of the CFPZ, we recommend identifying communities denser than one house per forty acres (the minimum density of a wildland-urban interface community, according to the January 4, 2001 Federal Register notice[3]) based on housing density calculated from modified Census 2000 blocks. Census blocks can be modified by subtracting public land and recalculating housing density based on the area of non-public land. Next, communities can be buffered by a half-mile to approximate the CFPZ (see discussion above). The buffered communities can be further refined by removing non-wildland cover types (water, barren, rock, agriculture, and urban land) from the buffered communities based on cover classes from the National Land Cover Dataset (http://landcover.usgs.gov/natllandcover.asp) or the best available locally derived cover data. Removal of these non-flammable cover types from the CFPZ helps keep fire protection planning focused on the portion of the landscape where treatment opportunities are greatest. The final map of the CFPZ represents natural vegetation within one-half-mile of communities. The portion occurring on federal land should be identified in the plan for treatment according to plans developed collaboratively between communities and the federal agencies.

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[3] "Urban wildland interface communities within the vicinity of Federal lands that are at high risk from wildfire" (*Federal Register* 66(3): 751-777, January 4, 2001).

PHOTO BY BRYAN DAY

Prescribed burn in 2000, Upper Snake River District,

Because this mapping method utilizes large national and statewide datasets, errors are bound to occur at local scales of application. One such error is the identification of unoccupied private parcels as communities when those private parcels are within a census block that meets the density threshold for selection as a "community." Because of the potential for errors, we highly recommend that the CFPZ be generated through a combination of GIS techniques and local expertise.

In general, the RPZ need not extend beyond about five miles from the CFPZ. While there will be cases where restoration is desirable beyond this distance, the majority of restoration opportunities will be found at the lowest elevations, in dry forests near communities. By establishing a five-mile-wide RPZ, restoration planning can be focused on the "frontcountry," where the need is clear and where there is less controversy over the use of thinning. With time, restoration efforts may be extended beyond the RPZ but these cases are a lower priority for the foreseeable future (i.e., the life of the plan).

Within a five-mile RPZ, a fair amount of the area is expected to be wilderness and inventoried roadless area. While restoration treatment in wilderness is not prohibited by the Wilderness Act, the need for any proposed manipulation of wilderness carries a high burden of proof, which must be detailed in a Minimum Requirements Analysis. Such a burden of proof should, with rare exception, make wilderness a low-priority candidate for treatment. Similarly, the Roadless Area Conservation Rule[4] and the "Bosworth letter"[5] place a high standard on entry of roadless areas. Both the Scientific Findings of the Interior Columbia Basin Ecosystem Management Project[6] and the EIS for the Roadless Rule[7] note that roadless areas are among the least eco-

4 http://roadless.fs.fed.us/documents/rule/rule_fedreg.html

5 http://roadless.fs.fed.us/documents/1230_Roadless_Ltr.htm

6 http://www.fs.fed.us/pnw/publications/icbemp.shtml

7 http://roadless.fs.fed.us/documents/feis/

logically altered parts of the landscape. Thus, roadless areas should also be lower-priority candidates for restoration.

While wilderness and roadless areas should be mapped as low priority, some vegetation types seem to be good candidates for restoration. Forest types that historically experienced frequent fire have been identified in the scientific and management literature as the highest priority for fuel treatment. The Cohesive Strategy (Laverty and Williams 2000) sets a national programmatic goal to "[c]oncentrate projects in the shorter interval fire-adapted ecosystems" such as ponderosa pine forests that historically experienced frequent fire. Within these forests, stands of old-growth ponderosa pine with an understory of dense saplings have especially high restoration potential.

We expect that, in mapping priority areas for restoration, agency managers will feel under considerable pressure to utilize existing methods for discriminating Fire Regime Condition Class (FRCC) (Schmidt et al. 2002). We highly recommend against this course of action. Initial criticisms of FRCC methods are discussed by Aplet and Wilmer (2003), and we believe FRCC methods will not stand up to future scientific scrutiny. Rather than relying on these flawed methods, we suggest that agencies map short-interval, fire-adapted ecosystems, such as low-elevation ponderosa pine forests, as the highest priority places to assess project-specific restoration potential on a case-by-case basis.

While WFU is often confined to wilderness, there is no reason why fire cannot be used outside wilderness as well, wherever safe. Thus, the FUEZ may be mapped as everywhere beyond the RPZ, i.e., everywhere that is more than, for instance, five miles from the Community Fire Planning Zone. Within this area, wilderness, roadless areas, and remote roaded land provide excellent opportunities to plan for fire use. The extent of the FUEZ will vary regionally, depending on the degree of regional development. In some places, it may be virtually non-existent, while in others, it may dominate.

In some cases, fire plans may be in place at scales broader than the LRMP. For example, the Bureau of Land Management has been developing statewide FMPs to provide the context for land management planning. In such cases, we believe that the three-zone approach still provides a workable way to implement fire management goals identified at the broader scale.

Scoping Questions

The preceding sections of this brief have presented a framework for considering wildland fire management during development of a Land and Resource Management Plan (LRMP), identified sources of data, and provided methods for allocating land to three fire management priorities. In making land use decisions, federal agencies have an obligation under the National Environmental Policy Act (NEPA) to take a "hard look" at the environmental consequences of a proposed action, and the requisite analysis "must be appropriate to the action in question." The implications of wildland fire for the implementation of a LRMP demand that fire management be given the "hard look" required by NEPA. In the process of developing a Draft LRMP and evaluating the environmental consequences of alternatives, agencies should address the following issues:

- Obtain all data necessary for the development of a map-based fire management plan, and include in the draft plan an inventory of all data possessed by the agency relevant to the preparation of a map-based Fire Management Plan.
- Describe in detail the methods

used to identify the area managed for community protection, including providing all data used in that assessment.

- Describe in detail the methods used to assess restoration potential, including providing all data used in that assessment.
- Describe in detail the methods used to identify the area where Wildland Fire Use will be considered, including providing all the data used in that assessment.

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Fire Management *today*

Volume 70 • No. 1 • 2010

Managing Wildland Fire

United States Department of Agriculture
Forest Service

Share Your Story...

This issue of Fire Management Today explores the new Interagency Guidance for Implementation of Federal Wildland Fire Management Policy. Striving to achieve sound natural resource management, apply the best available science, and collaborate among agencies, wildland fire management agencies changed their strategy in 2009 to allow fires to be managed concurrently for multiple objectives and to allow boundaries of fire management objectives to shift as fires move across the landscape. Along with this new implementation guidance come stories of success—acres burned, fires contained, and resource goals attained—and stories of frustration—communication errors, funding complications, and challenging management scenarios.

Share your stories from the 2009 and 2010 fire seasons and your lessons learned about managing wildfire for resource benefits with the FMT community. Send photos, excerpts, and articles to Monique LaPerriere, managing editor, at FireManagementToday@fs.fed.us.

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Fire Management today

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On the Cover:

On The Cover:

A lightning storm over the Beaverjack Fire, a wildland fire use fire (now termed a wildfire) in the Selway-Bitterroot Wilderness, viewed from Hells Half Acre Lookout in the Bitterroot National Forest, ID. Photograph by Mark S. Moak, professor at Rocky Mountain College in Billings, MT, and lookout at Hells Half Acre, 2005.

The USDA Forest Service's Fire and Aviation Management Staff has adopted a logo reflecting three central principles of wildland fire management:

- *Innovation:* We will respect and value thinking minds, voices, and thoughts of those that challenge the status quo while focusing on the greater good.
- *Execution:* We will do what we say we will do. Achieving program objectives, improving diversity, and accomplishing targets are essential to our credibility.
- *Discipline:* What we do, we will do well. Fiscal, managerial, and operational discipline are at the core of our ability to fulfill our mission.

Firefighter and public safety is our first priority.

Contents

Anchor Point

by Tom Harbour
Director, Fire and Aviation Management
Forest Service, Washington, DC

Managing Wildfire for Resource Benefits

Change has come to wildland fire use (and its precursor, prescribed natural fire). The Federal Interagency Wildland Fire Community now has only two kinds of fire: wildfire (unplanned fire) and prescribed fire (planned fire). According to the 2009 "Guidance for Implementation of Federal Wildland Fire Management Policy," the Forest Service and U.S. Department of the Interior agencies can now manage wildland fires for multiple objectives concurrently, and the objectives can change as the fire moves across the landscape. This means that where fire is a major component of the ecosystem, naturally ignited fires can be managed to achieve resource benefits where the impacts to landscape are tolerable. What does that mean to us? Currently, in simple terms, wildland fire management is comprised of two types of fire. First, there are those fires we plan and ignite; we refer to them as prescribed fires. Then, there are unplanned fires, the ones we call wildfires, which can be started either naturally (by lightning strikes) or unnaturally (by humans). Although wildfires are, by definition, unplanned, we conduct a planning and analysis process, closely linked to land management plans, in which we decide ahead of time if we want to allow some naturally occurring fires to burn in order to either reap a positive resource benefit or to allow fire to burn within tolerable limits set by the agency administrator.

Naturally caused wildfires can enhance many natural resource values when we allow fire to play its natural role while we protect private property and social values. For centuries, these lightning-caused fires have resulted not only in the enhancement of land conditions, but in better places for wildlife to live and roam. Simply stated, in some cases, fire on the landscape is beneficial, and resource managers need to become more active in allowing it to be part of the natural landscape.

> All fires have risks, but we have developed sophisticated tools that will assist us in predicting what a fire will do.

That's not to say that managing wildfires for resource benefits comes without risks. All fires have risks, but we have developed sophisticated tools—and are developing more—that will assist us in predicting what a fire will do—where it will go and how it will act.

Managing wildfires as an ecosystem process is a relatively new fire management strategy for most of us throughout the Forest Service. However, there are some forests with long-standing histories of this practice, referred to in the past as wildland fire use, or prescribed natural fire. On national forests such as the Gila in the Southwest Region and the Bitterroot in the Northern Region, wildfires have been managed for resource benefits since 1972. Managers and the public are beginning to see the advantages of allowing fire to play a natural role in some defined areas, the same role it played more than 100 years ago.

Climate change continues to challenge the Nation and our national forests. Fire season comes earlier and stays longer each year. Fires burn with more intensity. They are more damaging and dangerous to our firefighters, the public, and people's properties. When appropriate, management of wildfires for resource benefits is one component of fire management that can help us improve the condition of the land where, ultimately, we will be better able to control those unwanted fires when they happen.

We have individuals who specialize in managing naturally ignited wildfires within the Forest Service, but all of us need to be aware of and support the new interagency strategy, in which fires can be managed for multiple objectives. We will continue to suppress human-caused fires at the lowest cost and with the fewest negative consequences possible. Naturally caused wildfires will not be used to benefit natural resources everywhere—not every location is appropriate. But, under the right conditions, wildfires can be a tremendous asset to effectively move us toward our motto, "*caring for the land and serving people.*" ■

Implementing Federal Wildland Fire Policy—Responding to Change

Richard Lasko

Federal wildland fire policy has significantly changed since the 1935 introduction of the "10 a.m. policy," whereby all wildland fires were to be contained by 10 a.m. on the day following ignition. Although revisions to policy and implementation guidance have often been the result of tragic loss-of-life events or notably destructive fire seasons, other factors have provided an impetus to examine relationships between wildland fire policy and Federal land managers' mandate to protect life and property while managing ecosystems. The exponential growth of the wildland-urban interface—a result of rapid development in and near wildland areas—coupled with the dramatic increase in wildland fire frequency (fig. 1), intensity, and size (fig. 2), and an increasing need to use fire to meet natural resource objectives provided the latest incentives to take a fresh look at the guidance for implementation of Federal wildland fire policy.

Continuing the quest to provide land managers with relevant Federal wildland fire policy, the interagency fire community field-tested potential modifications to the 2003 "Interagency Strategy for the Implementation of Federal Wildland Fire Management Policy." Based on information from the field test and discussions with the fire community, fire management agencies modified the Implementation Strategy and removed the categorical distinction between wildland fire use and wildfire. Field deployment of this change began in 2009.

Richard Lasko is the assistant director, Fire and Aviation Management, Fuels and Fire Ecology, Forest Service.

A revision to the 2003 Interagency Strategy removes the distinction between wildland fire use and wildfire. This will enhance a fire manager's ability to implement Federal Wildland Fire Management Policy by allowing consideration of the full range of positive and negative attributes of a fire.

Implementing Federal Wildland Fire Policy—Changes Since 1988

The Yellowstone National Park fires of 1988 reinvigorated the debate over management of wildland fire and raised public awareness that fire is a necessary disturbance for the overall health and diversity of many ecosystems. The fires of the 2000 fire season stimulated further debate and fostered acceptance for the idea that fire exclusion had increased fire hazards in vegetation types historically characterized by frequent, low- to mixed-severity fire regimes. The 2000 fire season also nurtured the concept that fire exclusion is not operationally or

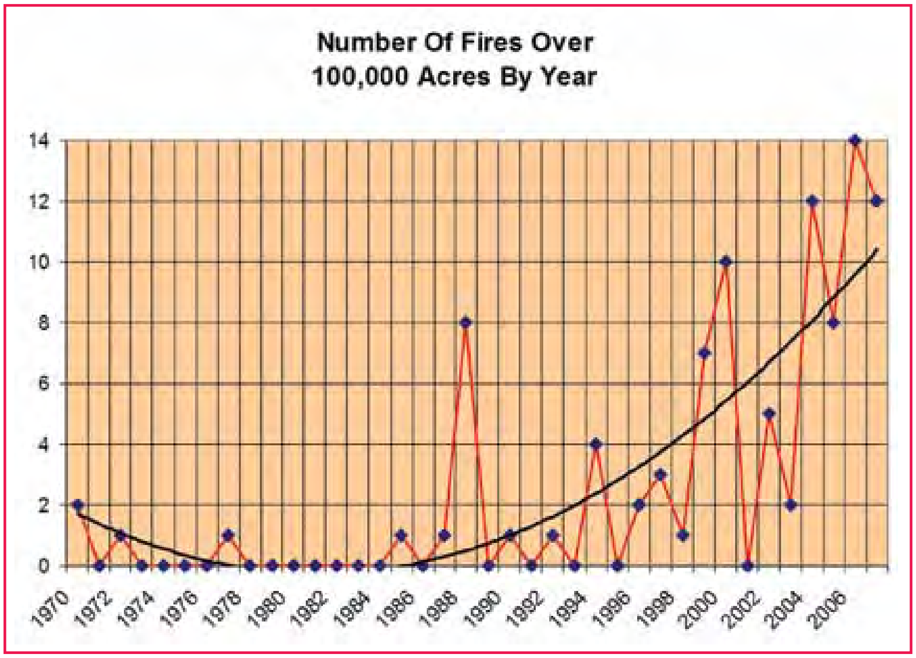

Figure 1—*The number of fires greater than 100,000 acres (40,500 ha) in size has increased dramatically over the years.*

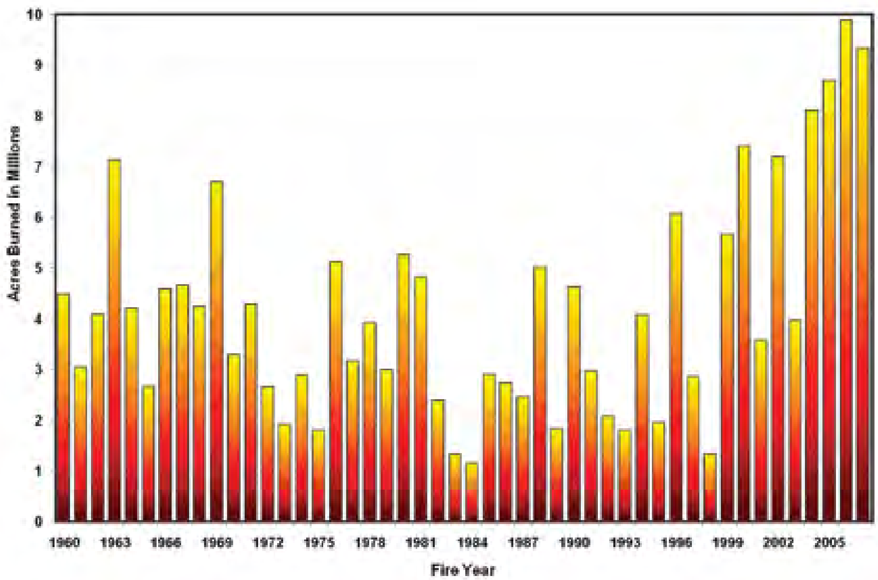

Figure 2—*Acres burned, in millions, 1960–2007.*

ecologically desirable in infrequent, stand-replacing fire regimes. This discussion led to the development of the "National Fire Plan," part of a national program linking fire research with land management practices to address the changing forest conditions.

In 1995, the "Federal Wildland Fire Management Policy" addressed the role of fire as a natural disturbance and moved fire planning toward integration with resource management. Natural ignitions could be managed to achieve natural resource benefits and maintain fire-dependent ecosystems. The 1995 policy also introduced the appropriate management response concept, which was further refined in the 2001 "Review of Federal Wildland Fire Management Policy."

The 2003 "Interagency Strategy for the Implementation of Federal Wildland Fire Management Policy" broadened the scope of fire management to balance fire suppression with management for ecosystem sustainability. It defined the alternative strategies available to manage unplanned natural ignitions: manage a fire to achieve resource benefits **or** (author's emphasis) manage a fire to reduce losses and minimize suppression costs. While all person-caused fires were to be managed as wildfires and treated as such, land and resource management plans or fire management plans could identify the appropriateness of using natural ignitions to achieve resource benefits through wildland fire use. Regardless of the chosen strategy, the 2003 Interagency Strategy required that Federal land managers respond to all wildland fire events with an appropriate management response, which allowed the use of any tactic (or combination of tactics), from monitoring to intensive management actions, to achieve a defined strategic objective.

Impetus for Change

The 2003 "Interagency Strategy for the Implementation of Federal Wildland Fire Management Policy," divides unplanned fire events into two categories: wildland fire use and wildfire. The distinction between the two categories is often obscured, especially when tactical actions implemented on a wildfire to minimize loss may be essentially the same as those implemented for a wildland fire use event to achieve resource management objectives.

The distinction imposed by the two categories presented difficulties in addressing the biophysical, temporal, and spatial complexities of wildland fire events. The fact is that the effectiveness and efficacy of a fire management strategy in protecting public values and achieving natural resource goals is highly situational. As fire moves across the landscape, scenery, structures, and valued resources are threatened at the same time that land management benefits are realized.

Success of a fire management strategy is dependent upon an intricate web of conditions. Fire managers encounter changing levels of risk as fires occur throughout the season. Actions that may be successful and sensible under one set of conditions may be unachievable or unrealistic under more extreme conditions of weather and terrain or with regard to the national and regional priorities that dictate availability of fire management resources. Costs of a management action may be inordinately high in relation to the resources protected or improved.

Engaging the Future

The 2008 field test of modifications to the 2003 "Interagency Strategy for the Implementation of Federal Wildland Fire Management Policy," and the subsequent dialogue and collaborative engagement with many of our partners and the public provided the opportunity to carefully reconsider the 2003 Implementation Strategy. The 2009 revision to the 2003 "Interagency Strategy for the Implementation of

Federal Wildland Fire Management Policy" removes the categorical distinction between wildland fire use and wildfire. The revision provides fire managers with the flexibility to respond successfully to changing conditions and address the complexities of the wildland fire environment encountered on a fire event. This will enhance a fire manager's ability to implement Federal Wildland Fire Management Policy by allowing consideration of the full range of positive and negative attributes of a fire while developing and implementing realistic, cost-effective actions to accommodate changing conditions as a fire moves across the landscape and through time. ■

Web Sites On Fire

Ecosystem Restoration Through Fire

A diverse group of volunteers is promoting the use of controlled fire to restore and maintain ecosystem health on the Mendocino National Forest and surrounding lands. This campaign, called "Restore the Mendo," has generated support from local governments, landowner associations, and individual citizens as well as State and national environmental groups.

The Web site at <http://www.restorethemendo.org> explains the benefits of low-intensity fires to homeowners, landowners, and others. The site provides information about fire management objectives, recent management actions, and positive results and responses. The Web site features video testimonials and a 30-second commercial used for local television spots in an ongoing effort to make prescribed fire an accepted part of maintaining the local landscape and its resources. Links to participating organizations, other fire information sites, and publications are provided.

Watching the Red. Mandi Unick keeps an eye on burnout operations on the Cub Creek Complex, Lassen National Forest, CA. The lightning-caused fire burned more than 19,000 acres in northern California. Photo: Aaron Black-Schmidt, Squad Leader, Columbia River Division Initial Attack Crew, Wenatchee-Okanogan National Forest, June 2008.

Organizational Learning Contributes to Guidance for Managing Wildland Fires for Multiple Objectives

Thomas Zimmerman and Tim Sexton

Since the inception of organized fire suppression in the early 1900s, wildland fire management has dramatically evolved in operational complexity; ecological significance; social, economic, and political magnitude; areas and timing of application; and recognition of potentially serious consequences. Throughout the past 100 years, fire management has matured from a single-dimensional program focused solely on control and immediate extinguishment to a multidimensional program. Throughout this period, fire managers have adapted their responses to changing conditions, emerging knowledge, and increasing experience. Now, they can utilize the full spectrum of responses to wildland fire to achieve both protection and ecological benefits based on objectives described in the applicable land and resource management plans and fire management plans.

Social pressures and organizational biases have created barriers to program development for wildland fire management.

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As organizational learning has affected the entire wildland fire management program, its influence on the management of wildland fires for resource benefits has accounted for significant advances, directly contributing to the program's evolution and growth, including:

- Expanded knowledge and understanding of fire ecology and the natural role of fire;
- Continual adjustments to the Federal wildland fire management policy;
- Focused planning, procedures, and precision;
- Advanced risk assessment of management knowledge and capabilities;
- Expanded and improved directions and magnitude of operational procedures;
- Increased management of fires as an ecological process, with implementation scales expanded beyond wilderness areas and into all fire regimes and vegetation types;
- Improved capability to manage fires for multiple objectives, and to redefine those objectives throughout the life of a fire;
- Improved capability to manage fires across a wider fire behavior range; and
- Implemented after-action reviews to observe, evaluate, and document accomplishments, successes, and failures.

The expanded knowledge of fire's natural role has markedly facilitated the increased use of wildland fire to accomplish beneficial ecological effects. Management of naturally caused wildland fire to protect, maintain, and enhance resources and, as nearly as possible, to function in its natural ecological role, is one of many management responses supported by the new "Guidance for Implementation of Federal Wildland Fire Management Policy" (USDA and USDI 2009).

What we know today about management of wildland fires to meet resource objectives evolved from decisions made nearly 40 years ago about the use of fire in wilderness areas, national parks, and other lands. This progressive thinking and the associated adaptive responses have extended fire man-

Table 1—*Critical tasks important to organizational learning.*

Task	Specific Activity	Outcome
Acquire new information.	• Collect information; • Consolidate program history and —current status; and • Develop shared vision.	• Information and existing information from personal sources documented; • Information accuracy validated; • Current policies, procedures, and processes reviewed; and • Program goals and purposes better defined.
Analyze the best procedures.	• Analyze program development; • Examine past performance; • Establish standards and baselines; and • Analyze interdependency of all program elements.	• Programmatic needs identified; • Past practices, both good and bad, both limiting and facilitating, evaluated; • Past experiences that need to be replicated or eliminated identified; and • Best practices that lead to superior performance and accomplishment identified.
Apply knowledge, processes, technology, and proven practices.	• Experiment with new knowledge applications; • Experiment with new technological applications; • Incorporate best knowledge and technology into business; • Address problem solving; and • Transfer knowledge.	• Continual flow of new ideas, knowledge, and technology into application established; • Distinction between factual information, perceptions, and personal viewpoints recognized; • Knowledge, principles, guidelines, procedures, practices, etc., transferred through all available methods to practitioners; and • Application through the use of a dynamic learning environment improved.
Archive overall processes and results.	• Document program development, practices, and organizational growth; and • Ensure the retention of critical information.	• Information transfer processes improved; • New practices, experiences, and knowledge, both positive and negative, documented; and • All information for future reference and application retained.

agers' knowledge and experience. We now think of management of naturally caused ignitions as an essential tool for achieving beneficial ecological effects.

Organizational Learning

Organizational learning has contributed to continuous and programmatic development of the guidance for management of wildland fires and has increased the ability of personnel to manage fires for multiple objectives by:

- Recognizing the importance of consolidating program examination;

- Acquiring new information;
- Analyzing the best procedures;
- Applying knowledge, processes, technology, and proven practices; and
- Archiving the overall processes and results and using the information to improve program effectiveness.

Fire managers recognize the importance of examining the results of management responses to wildland fire and applying the information to improve program effectiveness. However, organizations are sometimes controlled by social influences that hinder innovation and administrative mandates that limit response.

Barriers to Managing Wildland Fire as an Ecological Process

Social pressures and organizational biases have created barriers to program development for the management of wildland fires as a natural process. Such internal and external forces have led to divisiveness and a lack of clear and concise messages, direction, and goals. This situation has stifled overall organizational growth, restricted productivity, and has most certainly fueled negative public attention.

Public and governmental responses to specific fire situations have promoted agency reluctance to advance wildland fire management and resulted in procedural statements, operational guidance, and other circumstances intended to limit the magnitude and slow implementation of change in fire management. The conviction that

The conviction that all wildland fires can and should be suppressed is long standing, but mixed success in achieving this provides widespread support for defining multiple fire management objectives.

all wildland fires can and should be suppressed is long standing, but mixed success in achieving this provides widespread support for defining multiple fire management objectives. This belief has limited fire managers from full utilization of "emerging knowledge" of fire's natural role, fire effects, and the ramifications of fire exclusion in the development of management responses.

Administrative barriers have existed throughout the history of wildland fire management. Use of wildland fires to support ecological processes has been viewed as an action that is distinctly separate from wildland fire management and with different operating standards. Internal policymaker resistance to changes that advocate expanded use of wildland fire have surfaced in every review and revision of wildland fire management policy.

Managing wildland fire to achieve land and resource management goals continues to be riddled with misperceptions and misinformation, which have limited both programmatic growth and overall effectiveness. As more credibility has been placed on identifying best practices for wildland fire management, efficiency and accomplishment have improved; yet despite this development, resistance still affects resource agencies to some degree today.

Changing Perspectives

Today, organizational learning promotes a broader understanding and awareness that is beginning to change outdated thinking and reduce barriers. Organizational learning is spurring policy revisions, directing funding, and relaxing fiscal constraints for managing wildland fires for multiple objectives. The 2009 "Guidance for Implementation of Federal Wildland Fire Management Policy" allows wildland fires to be managed concurrently for many objectives and allows personnel to redefine those objectives as conditions change. Additionally, public perceptions and support have improved, workforce limitations have been reduced, and safety concerns have been addressed.

Finally, fire's role in a healthy ecosystem is receiving positive recognition. Management of wildland fire for ecological benefits has grown from a wilderness-only application to one that spans all land-use situations with marked increases of land types considered suitable for application and expanded operational capabilities.

References

USDA and USDI. 2009. Guidance for Implementation of Federal Wildland Fire Management Policy. Washington, DC: U.S. Department of Agriculture and U.S. Department of the Interior: 20 p. ■

Table 2—*Specific examples of organizational learning benefits that support the management of wildland fire for resource benefits.*

Changes and Advancements	Learned Outcome	Fire Management
Expanded knowledge of fire and its natural role	• Better understanding of wildland fire as a natural process and of its role in restoring and maintaining healthy ecosystems; and • Understanding that many ecosystems contain plants that depend upon periodic fire presence for their continued existence and that many of the effects of fire are positive.	• Significant knowledge base of literature and reference materials established; The Fire Effects Information System Web site <http://www.fs.fed.us/database/feis> provides fire managers with an array of reference and support for land management and project planning; and the Wildland Fire Decision Support System <http://wfdss.usgs.gov/wfdss/WFDSS_Home.shtml> assists fire managers and analysts in making strategic and tactical decisions for fire incidents.
Continual adjustments of policy	• Understanding that wildland fire policy must provide flexible and responsive direction for wildland fire management—without unnecessary constraints, and readily adapting to emerging knowledge, technology, and science.	• Accountability for long-term unplanned fire events managed for resource benefits that consider preparedness levels and fire management plan completion; • Prescribed natural fire eliminated as a strategy; • Wildland fire use eliminated as a defined and separate entity from other wildfires; • Approval of naturally caused ignitions to be managed as an ecological process, and to be managed for multiple objectives. • Fiscal procedures established that are conducive to greater use of wildland fire for resource benefits; • Standardized qualification of all fire management activities; and • Specific policy elements in the areas of science, planning, fire management, and ecosystem sustainability.

Changes and Advancements	Learned Outcome	Fire Management
Improved planning processes	• Successful application of fire to ecosystems depends upon detailed planning at all levels from the land management plan to the fire management plan and into specific fire implementation action planning.	• Guidance to incorporate fire effects and the natural role of fire information into land management plans; • Land management processes that guide fire management planning and implementation; • Fire management plans that translate and support land management plans and on-the-ground action; • The Wildland Fire Decision Support System, providing the most detailed and comprehensive fire management planning and implementation information for fire use decision and tactical action to accomplish the strategic objectives of an unplanned igntion managed for resource benefits; and • A process developed with a focus on efficient long-term risk assessment, strategic planning, and tactical implementation instead of short-term, tactical operational implementation.
Risk assessment and decision support tools	• Acceptance of the importance of assessing risks associated with wildland fire management in terms of values, hazards, and probability in order to more adequately determine if the level of risk can be accepted and successfully mitigated or eliminated; and • Recognition of the importance of obtaining better information, reducing uncertainty, assessing potential fire outcomes, evaluating consequences of failure, determining probabilities of success, evaluating potential costs, and identifying values to be protected to better support decisionmaking.	• Significant advances in predicting fire behavior spread and intensity, analyzing climatological and meteorological data, and assessing rare weather occurrences; • Advances in predicting fire effects, smoke production, and smoke dispersal; estimating fire-spread areas; identifying values at risk; and evaluating probabilities of the fire spatial extent; • Enhanced experience and knowledge in utilizing this kind of information in support of fire management decision-making, planning, and implementation; and • Improved decisionmaking processes.

Changes and Advancements	Learned Outcome	Fire Management
Increased management of wildland fires for ecological benefits	• Balanced fire management program with multiple management objectives; • Recognition of the value and importance of managing wildland fire for resource benefits; and • Recognition of the role wildfire can play in long-term restoration programs.	• Improved understanding of wildfire and its primary and secondary benefits; and • Expanded fire management accomplishments, strengthened ecosystem maintenance and restoration, increased vegetation mosaics, decreased long-term wildfire potential, increased community protection, and advanced land management practices.
Development of operational procedures	• Better understanding that operational mitigation actions must include the full range of firefighting responses and tactics as appropriate to the specific situation; and • Understanding that successful wildland fire management requires detailed planning that defines threats, operational mitigation actions, constraints, number, and types of resources needed, and contingency actions.	• Increased capability to respond to wildland fire under a wider range of jurisdictional situations and individual management areas; • Ability to acquire and utilize all firefighting resources as needed to respond to wildland fires, regardless of objectives; and • Established dedicated resources for use in managing wildland fire for resource benefits.
Expansion beyond wilderness	• Acceptance of the use of wildland fire to protect, maintain, and enhance resources and, as nearly as possible, to function in its natural ecological role as an effective management practice in wilderness and nonwilderness; and • Realization that successful management across all landscapes is dependent upon continued and proactive collaboration among Federal and State agencies, private organizations, and private landowners.	• Increased vegetation mosaics, decreased long-term wildfire potential, and increased community protection capabilities resulting from the expansion of the use of wildland fire as an ecological process outside wilderness; and • Expanded fire management accomplishments, strengthened ecosystem maintenance and restoration, community protection strategies, and advanced land management practices achieved by managing naturally caused ignitions to accomplish resource benefits beyond wilderness to across all land-use situations, where applicable.

Changes and Advancements	Learned Outcome	Fire Management
Management across wider fire behavior ranges	• Understanding of the need to include wildland fire management across all fire regime classes and diverse situations, depending on land management direction and constraints; and • Understanding that the success of managing wildland fire for resource benefits is measured by fire effects and not solely by fire type and behavior.	• Growing experience with managing fire in all fire regime classes and all fire behavior scenarios; and • Successful examples of management of high-intensity stand replacement wildland fires.
Use of After Action Reviews	• Immediate illumination of both successes and failures; • Awareness of the importance of timely and frank assessments of actions and presentation of outcomes regardless of success or failure; and • Understanding the importance of documenting both successes and failures in fire management planning and implementation.	• Immediate feedback to program efficiency; • Facilitated progression toward a high-reliability organization; and • Established dynamic feedback mechanism supporting improved and advanced processes, procedures, and policy.
Documentation	• Understanding the importance of archiving both successes and failures in fire management planning and implementation; and • Understanding the value of saving examples and practical knowledge.	• Markedly improved and advanced training; and • A substantial record of accomplishments, examples, case studies, etc., accessible to fire management practitioners.

Wildland Fire Behavior Case Studies and the 1938 Honey Fire Controversy

Martin E. Alexander and Stephen W. Taylor

Over the past 90 years, fire research has contributed to our understanding of wildland fire behavior through laboratory and field experiments, physical and empirical modeling, numerical simulations, analyses of individual fire reports, and wildfire case studies. Although basic research on combustion is essential to a full understanding of fire behavior, such research would not be very useful without actual field experience gained and case study documentation (Brown 1959).

In general terms, what is a case study? Contributors on *Wikipedia* (<http://www.wikipedia.org/>) propose that case studies "provide a systematic way of looking at events, collecting data, analyzing information, and reporting the results." With the renewed interest in carrying out research on active wildfires (e.g., Lentile and others 2007a), it's worth reexamining the features of a good case study.

To this end, this article summarizes the findings from the case study of the controversial Honey Fire of 1938, originally published in *Fire Control Notes* by Olsen (1941)—one of the first comprehensive case studies of a wildland fire undertaken by fire behavior researchers. This account was reprinted in the Fall 2003 issue of *Fire Management Today*, the first of three special issues devoted to the subject of wildland fire behavior (Thomas and Alexander 2006).

> The story of the Honey Fire and the ensuing controversy is as much about human behavior as it is about fire behavior.

The Story of the Honey Fire

The story of the Honey Fire and the ensuing controversy is as much about human behavior as it is about fire behavior. In broad outlines, the situation was as follows. A fire behavior research crew happened upon a newly started wildfire, but rather than engaging in any suppression action, the crew began documenting its behavior. This course was taken partly because the crew had advance clearance to do so. The fire became one of the largest fires in the region that year and was finally contained by local fire suppression forces. The research crew's decision to not fight the Honey Fire raised some eyebrows. Later, a member of the research crew published a case study that not only analyzed the fire's behavior but also critiqued the actions of the suppression forces. That article, in turn, provoked a harsh outcry.

Synopsis of the Honey Fire Case Study

Chronology and Behavior

The major run of the Honey Fire took place on January 25, 1938, on the Catahoula Ranger District of the Kisatchie National Forest in north-central Louisiana (fig. 1). A total of 494 fires were to burn more than 12,800 acres (5,180 ha) on the Kisatchie National Forest in 1938 (Burns 1982), and the Honey Fire was one of the many human-caused fire occurrences that year. Interestingly enough, Burns (1982, 1994) did not mention the Honey Fire in her historical accounts of the Kisatchie National Forest.

The Honey Fire was the result of careless actions on the part of freight train employees disposing of burning waste along the east side of the Louisiana & Arkansas Railroad, approximately 1.5 miles (2.4 km) north of Bentley, LA, at around 9:50 a.m. The lookout at the Catahoula Tower, located 2 miles (3.2 km) to the east, detected the fire within 2 minutes, a very acceptable discovery time (Bickford and Bruce 1939b).

Carl Olsen, a forester with the Southern Forest Experiment

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Timeline and Tactics

Initial Fire Behavior and Attack

The fire started at 9:50 a.m. on the east side of the Louisiana & Arkansas (L & A) Railroad (point A). Crew 1 (a pumper truck and 2 men) and Crew 2 (a fire boss and 12 men) were dispatched to the fire's presumed point of origin. When they arrived, the fire had a perimeter of 2,640 feet (805 m) and was spreading at about 360 feet per minute (110 meters per minute). Crew 2 began to work the north flank of the fire. The pumper truck could not be used because of wet ground and was redeployed to join Crews 3 and 4 (a total of 31 men), who had started backfiring along the west side of Tower Road. The fire boss then split Crew 2, taking five men (Crew 2A) overland to the west firebreak, and leaving seven men (Crew 2B) at the north flank. By 10:30 a.m., the fire reached the Civilian Conservation Corps (CCC) camp and Tower Road, where it was stopped at the line created by the backfires and the pumper truck. Crews 3 and 4 then joined Crew 2A on the west firebreak and began backfiring and attacking the north flank of the fire near the head. At 10:44 a.m., the wind shifted to the southwest, creating a new head (point B), which by 10:53 a.m. had spread to the west firebreak, where it was held by the backfiring operation; however, all of the constructed line on the north flank was lost.

Later Fire Behavior and Tactics

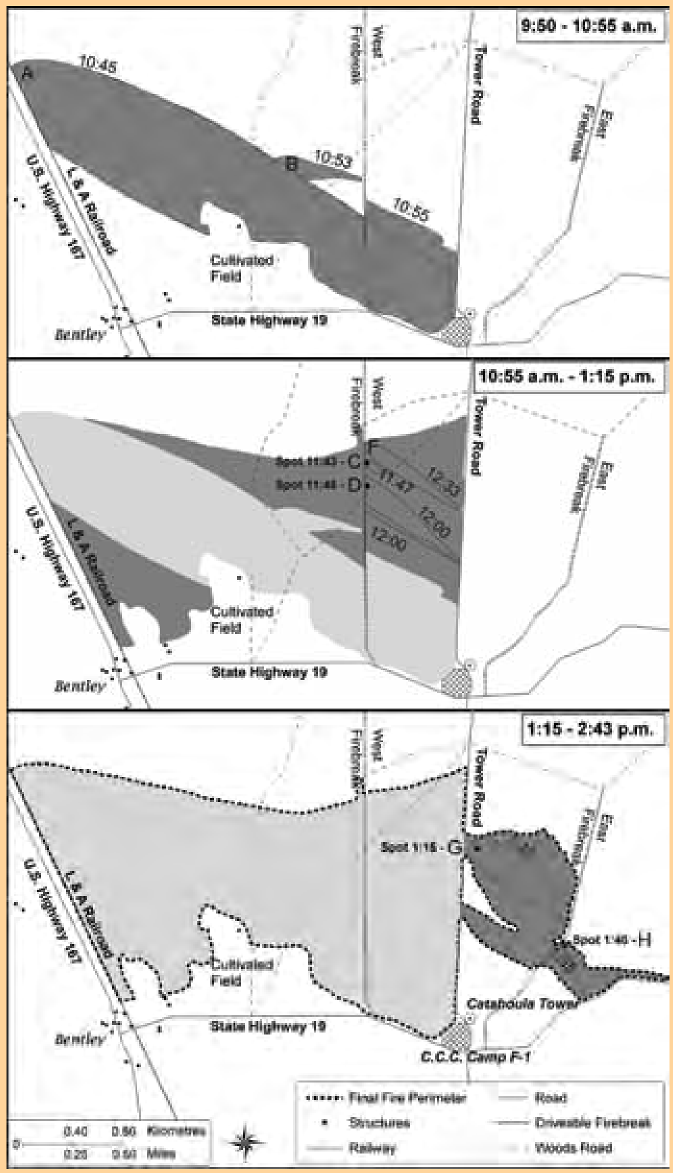
After the wind shift, the north flank, from the tail to the west firebreak (now effectively the head), was left to burn freely, which resulted in fire spread to and spotting across the west firebreak with new heads developing between the west firebreak and Tower Road (points C, D, and F). Crews continued patrolling and backfiring along the east and west firebreaks, Tower Road, and Highway 19. The south flank of the fire was stopped by patrols (22 men), a cultivated field, backfiring against Highway 19, and a wind shift to the southwest.

Final Attack

During the final attack on the fire, crews reinforced the backfires on the Tower Road and east firebreak (although spot fires at points G and H occurred across the Tower Road and east firebreak) and worked the north flank from the rear or tail of the fire to the head, mopping up as they went, aided by the pumper truck and additional crews. The fire was contained at 2:43 p.m. by a force of 19 supervisors and 129 men. The fire was mopped-up and declared out some 4 hours later.

Suggested Strategy and Tactics

Olsen made many positive comments on preparedness, dispatch time, equipment, and crew morale under trying conditions. However, he felt that, given the extreme fire behavior during the fire's initial run, indirect attack by backfiring was the only feasible control measure and valuable time had been lost in direct attack at the point of origin. He suggested that if the pumper truck and crews 2, 3, and 4 had begun aggressive backfiring earlier along the west firebreak, the fire might have been held there. He also suggested that the fire boss and crew leaders should not have worked directly on the line alongside their crews, but should have been more engaged in directing and managing the firefighting operation.



Maps of fire progress and summary of fire suppression activities and general fire behavior associated with the major run of the 1938 Honey Fire (adapted from Olsen 1941).

Station of the Forest Service, and three others (A.H. Antonie, R. Brooks, and C.A. Bickford) were members of a research crew assigned to study the behavior of free-burning wildfires in the region (Harper 1937, Olsen 1938). Normally, the crew was dispatched with initial attack forces. However, in the case of the Honey Fire, the crew happened to arrive on scene (at 9:53 a.m.) within 3 minutes of the fire's origin; they had been traveling about a mile (1.6 km) behind the train south along U.S. Highway 167, which ran parallel to and west of the railroad tracks (see description on previous page).

Within 2 minutes of happening upon the initiating fire, the four-person crew began mapping the fire perimeter (fig. 1) in order to determine rates of fire spread and fire size, collecting fuel and soil samples for analysis of moisture content, recording fire weather data, and making notes on various fire behavior characteristics (e.g., flame size and spotting distances). Unfortunately, to our knowledge, the crew took no photographs during or immediately after the fire. The technology of the time would not likely have permitted the research crew to have radio communication with the local fire suppression organization (Gray 1982).

At one point, the Honey Fire advanced almost 2 miles (3.2 km) during a 30-minute interval following ignition, and the fire eventually burned a total area of 1,092 acres (442 ha) before containment at 2:43 p.m. on the day of origin. The Honey Fire's documented rate of advance ranged from 330 to 463 feet per minute (101 to 141 meters per minute). Spot fires over 200 feet (61 m) in advance of the main head were observed. Computed fireline intensities, determined after the fact and based on these observed spread rates and estimated fuel consumption, ranged from 6,660 to 9,295 British thermal units per second per foot (23,050 to 32,170 kw/m) with corresponding flame lengths averaging 26 to 30 feet (8 to 9 m) (Byram 1959). However, flames at the head of the fire "frequently reached out in long tongues extending 100 feet [30 meters] or more" (Olsen 1941), no doubt in response to momentary gusts of wind (table 1).

When should the observer drop the camera and notebook and pick up a shovel or pulaski?

Environmental Conditions

The fire started in an area that was "typical of open cut-over longleaf pine land in the Upper Coastal Plain" (Olsen 1941), the predominant fuel being a heavy stand of cured broomsedge grass (*Andropogon* sp.) resulting from more than 3 years' accumulation. Available fuel loads would have been in the order of 3.4 tons per acre (7.6 tonnes per hectare), based on the sampling carried out by Bruce (1951).

Although air temperatures were considered "crisp" at 45 to 50 degrees Fahrenheit (7.2 to 10 degrees Celsius), moderately low relative humidities prevailed (26 to 33 percent). The moisture content of the fine, dead, fire-carrying fuels was determined to be about 12 percent. Winds were moderately strong and gusty (table 1), and shifted about 90 degrees, from northwest to southwest, during the initial major run.

Fire Suppression

The Civilian Conservation Corps and Work Projects Administration provided 129 firefighters and 19 supervisory personnel for suppression duty on the Honey Fire. They used a single 350-gallon (1,325-L) pumper truck along with the standard fire tools of the day—swatters or flaps (Sykes 1940), backpack pumps, fire rakes, fusees, and axes. Some photographs illustrat-

Table 1—*Onsite wind speeds measured during the major run of the 1938 Honey Fire (adapted from Olsen 1941)*

Duration and exposure	mph	km/h
Average at 3.5 feet (1.1 m) above ground	9.7	15.6
Average at 20-foot (6.1-m) open standard	15	24
Average at 33-foot (10-m) open standard	17	27
Maximum 1-minute average at 3.5 feet (1.1 m) above ground	16.6	26.7
Maximum 1-minute at 20-foot (6.1-m) open standard	25	40
Maximum 1-minute at 33-foot (10-m) open standard	29	47

Note: The 20-foot (6.1-m) and 33-foot (10-m) open wind speeds used for fire danger rating and fire behavior prediction in the United States and Canada, respectively, were estimated from the observation at 3.5 feet (1.1 m), as per Lawson and Armitage (2008).

A Suggestion To Help Improve Fire Suppression Tactics*

The morale and determination of all men were excellent, and in many cases remarkable. Virtually all of them used their flaps and backpack pumps effectively, showing that the training they had received was very much worthwhile. During the hot flank attacks, however, the flapmen [i.e., firefighters using swatters that are commonly used in containment of grass fires] relied heavily upon the pumpermen spraying water to knock down the flames. The men should be trained to rely less upon water in fighting the flanks by having the crew leaders temporarily stop suppression and rest the crews when the wind shifts on a flank, resulting in a very hot fire to fight. More line on the flanks will be extinguished and held by resting a crew while the fire is burning intensely and then efficiently directing them when the heat and flames have diminished.

Two firefighters attack a spot fire in 4-year-old rough using swatters or flaps, South Carolina. Photo: George K. Stephenson, Forest Service, 1944.

Firefighters use backpack pumps and a swatter or flap on a small grass fire, Georgia. Photo: Clint Davis, Forest Service, 1942.

Civilian Conservation Corps crew undertaking suppression action on a wildfire with backpack pumps and handtools, Ozark National Forest, Arkansas. Photo: Bluford W. Muir, Forest Service, 1938.

*Excerpt from Olsen (1941).

ing firefighting scenes of the era and general geographical location associated with the Honey Fire are presented here.

Communication on the fireline would have been difficult under the circumstances. There would have been no radio communication capability between the local district office and the fire boss or among the fire suppression crews (Gray 1982).

In addition to observing and recording the fire's development and chronology, Olsen's crew documented the fire suppression activities and the fire's resistance to control (e.g., arrival time, suppression tactics, amount of constructed and held line, and general difficulties experienced by the firefighters). No firefighters were killed or injured during the Honey Fire, but Olsen (1941) acknowledged that, after the wind shifted, "the danger of a crew getting trapped by the high, oncoming flames was great" along the left flank of the fire.

The Controversy That Followed

Roy Headley, who served as head of fire control for the Forest Service from 1919 to 1942, was interested in analyzing the accounts of large fires for the lessons that they might provide. For the year 1938, the Honey Fire was the third largest of the 13 Class E fires (fires greater than 300 acres [121 ha] in size) in the Southern Region of the Forest Service and 1 of 5 large fires on the Kisatchie National Forest. A little more than a third of the area burned by the Honey Fire had been planted with slash pine seedlings about a year earlier. Wildfires had been and continued to be a chronic problem for the reforestation pro-

Lessons Learned in Large Fire Management*

Such an infinite variety of problems are involved in the management of large fire jobs that thoughtful men seldom fail to learn from each one something which should be guarded against in the future, something which should be done differently, some cherished belief which must be modified or abandoned. For 35 years I have been working on or observing suppression jobs, but I still learn something from every fire I reach.

Roy Headley, circa 1942. In "Re-thinking Forest Fire Control," Headley (1943) summarized the lessons he had learned from a long and distinguished career in fire control administration with the Forest Service. Photo: courtesy of Stephen J. Pyne, Arizona State University.

Sometimes, alas, we "learn the same lesson over and over"—or do we? For example, I have learned throughout many years that there is some flaw in our management of larger fires which keeps us from getting a reasonable output of held line from a crew of a given size. Plenty of other people have learned the same thing. But, untrained as we are in the science and art of management, we have not found ways to act satisfactorily on what we have learned. Our learning has too often failed to lead to productive action.

The first essential in such matters is to grasp the need for change, the nature and importance of a problem, the chance to introduce something better. With that fact in mind, the outline for 1938 reports on larger fires requested a record of lessons learned by the man or men who had most to do with each fire. Some of the most suggestive answers received are quoted in this article. … All fire-control men may benefit by the lessons learned on these fires. Perhaps these notes will help reduce the number of times lessons have to be "relearned" by different men—or by the same men.

*Excerpt from Headley (1939a), which was published when Roy Headley headed the Division of Fire Control, Forest Service, Washington, DC.

gram that began in 1930 when the Kisatchie National Forest was first established (Burns 1982, 1994).

In his analysis of the Honey Fire, Headley (1939b) felt that the fire boss had failed to recognize the severity of the burning conditions that prevailed at the time and thus failed to select an appropriate strategy and tactics for containing the fire, namely backfiring from existing roads and firebreaks (Cooper 1969; Riebold 1956). Yet as Cheney and Sullivan (2008) have rightly pointed out, there are inherent dangers with backfiring that limit the chances of success. At the time, the fire boss was required to rely solely on his general knowledge and experience; no guide to judging fire potential relevant to the fuel type was available at the time. Less than 2 years later, Bickford and Bruce (1939a) produced what evolved into the Coastal Plain Forest Fire Danger Meter for the Southern and Southeastern United States (Jemison and others 1949).

Olsen and his fellow crew members were criticized for not immediately attempting to suppress the fire. However, the forest supervisor had previously agreed that this research crew was free of any obligation to undertake any fire suppression action so that the best possible fire behavior data could be obtained. It's unlikely that they could have done much anyway: "With two fences and a railroad between them and the fire, there is no doubt that their truck was unusable on this fire" (Olsen 1941). Furthermore, when the research crew arrived on the scene, the fire had already advanced more than 100 feet (30 meters) from its point of origin and "was very definitely too big for them to hold with hand tools alone" (Olsen 1941).

Olsen's (1941) account of the Honey Fire included considerable commentary on the actions taken by fire suppression personnel in addition to his description of fire behavior and the associated fire environment. This commentary was presumably in part the result, according to the editor of *Fire Control Notes* at the time, of a board review held by the regional forester that provided additional information to the Southern Forest Experiment Station for use in its study of the Honey Fire (Olsen 1941).

Olsen (1941) indicated that one of his objectives in publishing his case study was "to offer constructive criticism and suggestions as a guide in planning suppression action for future fires burning under similar conditions." He also offered many positive observations.

Despite his good intentions, Olsen was criticized in an article published in 1942 in *Fire Control Notes*. Barry (1942) chastised the fire behavior research crew for not attempting to control the fire; he also deemed it inappropriate for fire research personnel to analyze or critique the efforts of the fire suppression personnel involved after the fact. Further, Barry asserted that such actions could have serious repercussions on the image and morale of the organization and that only those fires that had escaped initial attack should be the subject of fire behavior studies.

Reflections

Wildfire case studies are invaluable in providing fire behavior data for developing and evaluating fire behavior models (e.g., Pearce 2002, Townsend and Anderson 2006) and as a source of training material (Alexander 2002). The recent report on the 2006 Billo Road Fire in New South Wales, Australia, by Cruz and Plucinski (2007) is a good example of this traditional role of wildfire case studies. Documentation of the effects of fuel treatments on fire behavior in relation to fire suppression effectiveness (e.g., Murphy and others 2007), highlighting firefighter safety incidents (e.g., Pearce 2007), and fostering institutional memory of local, historically significant fires (e.g., Ward 2005) represent other valuable contributions. Case studies of prescribed fires (e.g., Alexander 2006) are just as valuable as their wildfire counterparts. A combination of case study knowledge, experienced judgment, and simulation modeling of fire behavior is seen as the most effective approach to appraising fire potential and predicting wildland fire behavior (Alexander 2007, Alexander and Thomas 2004).

Lessons-Learned Analyses of the Honey Fire*

In this case the fault lies with the fire boss in his failure to recognize extreme fire conditions that existed on January 25, and to modify his attack to fit these extreme conditions. If he had recognized the danger, or had means other than his general knowledge and experience to guide him in selecting the correct method of attack, the fire would have been controlled much easier, and with a somewhat smaller acreage. Instead of attempting a direct attack, had he backfired all existing roads and firebreaks facing the oncoming fire, the fire would have been controlled at about 700 acres [280 ha] and the slash-pine plantation inside of the fence would have been saved. The amount of held line per man-hour would have been at least tripled. One answer is a well-constructed, fire-danger meter which will leave as little as possible to the judgment of the fire boss on the fire line.

The only method of controlling this fire at a smaller acreage after it had started would have been an immediate attack by the indirect method by backfiring. Under such conditions, tank trucks and specialized equipment are of very little value. A strip of burned ground at least 400 feet [120 m] wide is necessary to stop the heads of such a fire.

The fire was started by the L. & A. Railroad train which was temporarily stalled at the point of origin. The Louisiana State law requires that the railroad free their right-of-way from combustible material. The forest [Forest Service] has never been able to force the L. & A. to do this. The railroad officials have been warned, both in person and by letter, many times. Also, they have paid suppression cost and damages for other fires caused by their railroad. Railroad business is rather poor, and the officials took the attitude that they could not afford to keep rights-of-way clear as required by law. Reimbursement of damages and suppression costs amounting to $2,160.62 has been asked for.

Since this fire occurred, however, the railroad officials have decided it is cheaper to clear the right-of-way than to pay damage and suppression costs. Both the L. & A. Railroad and Missouri-Pacific Railroad Cos. have cleared their rights-of-way of combustible material within the forest boundary. For the first time in the history of the Kisatchie Forest, we will enter the 1938-39 fire season without the constant hazard of railroad fires.

Fusees used for backfiring in some of the tool boxes had absorbed enough moisture from the air to be worthless. The wet or damp fusees could not be detected by casual examination. Some delay in backfiring was caused by these dud fusees. Fusees cost only about 9 cents a piece, and this failure could have been eliminated by simply replacing old fusees with new ones every 30 days.

*Excerpt from Headley (1939b), which was published when Roy Headley headed the Division of Fire Control, Forest Service, Washington, DC.

Criticism of the Actions of the Wildfire Behavior Documentation Crew on the Honey Fire*

A reading of the article by C.F. Olsen, entitled "An Analysis of the Honey Fire," in the October 1941 issue of *Fire Control Notes*, brings to attention a situation hard to imagine. Of course, it is practically impossible for us at this remote location to visualize all the factors; nevertheless, after making generous allowances, I still experience an unpleasant jolt when I think of what happened.

There were two branches of the same department involved in the suppression of a fire, one interested in determining how the fire would behave on a bad burning day, the other charged specifically with the responsibility for stopping its spread.

The branch interested in behavior arrived at the Honey Fire first, 3 minutes after its origin according to the article. A four-man fire-behavior crew had been traveling on a paralleling highway about a mile [1.6 km] behind a train that stopped to service a hot box. The train crew carelessly threw some burning waste into dry grass and the behavior crew happened along 3 minutes later. They found it "definitely too big for them to hold." The decision of the fire-behavior crew—equipped with a car having various fire-fighting tools—to refrain from an attempt to check or retard the spread of this fire when it was approximately 100 feet long is hard to understand. We would expect more from four untrained men off the street as a quality of citizenship. Forest Service guard-training instructions have emphasized for years that there is always something that even a single guard can do to retard the spread of a fire, although it may be obvious that a frontal attack is impossible. The failure to make some attempt in that direction on the part of this fire-behavior crew indicates that they did not believe in such a theory. Won't the morale and fighting spirit of our temporary guards be lessened by such an example? The public, too, may find such action, or lack thereof, confusing.

If the fire-behavior crew admitted that they were unskilled in fire fighting and limited their report to factors of weather and rate of spread, their disregard for attempting control action could be overlooked to some extent.

The fact that suppression foremen, who apparently did their best to stop this fire, were subjected to criticism by such men indicates an oversight in personnel management that cannot help but decrease spirit and morale in a marked degree. Moreover, the fire-behavior crew has been permitted to make capital of their questionable action by printing the results of their study.

There is no quarrel with the policy of conducting fire-behavior studies, and the men assigned to that duty should not be expected to take part in the suppression work on fires that have escaped first control efforts. However, there should be no tolerance of a policy permitting an organized crew of men to travel about the country looking for fires to study unless they are willing to lend a hand in an effort to check the spread of small fires pending the arrival of regular suppression crews.

It is hoped that in the future this fact will be made clear to all, so that even though a fire cannot be entirely stopped, it may be retarded, thereby permitting arriving suppression crews to handle it more easily. That kind of action will make far better reading than the one referred to above, and the results after the fire is out will go far toward strengthening the spirit and morale of the whole organization.

*Excerpt from Barry (1942), which was published when E.F. Barry was a staff assistant on the Flathead National Forest, Northern Region (Region 1), Forest Service.

The value of the fire behavior documentation of the Honey Fire that Olsen (1941) provided is unquestionable. As Van Wagner (1971) has pointed out, "some valuable reference data can be collected by being at the right place at the right time" through wildfire monitoring and documentation. This is especially true during periods of extreme burning conditions, which are often impractical or impossible to simulate with outdoor experimental fires, in the laboratory, or by computer simulation. At the time, Olsen's article was the most comprehensive published wildfire case study of its kind. Over time, many others have used his data and information in their own fire research studies and for other purposes,

On Wildfire Case Studies and Firefighter Safety

I confess that I like case studies. They are the kind of thing historians are used to dealing with. We don't expect to find general laws: we accept the particularity of experience. Moreover, the case study is a story. That's why I think it's especially useful for safety. Nobody remembers guidelines the way they remember a story, which is the next best thing to actually experiencing the events.

Dr. Stephen J. Pyne (2008)
Global Wildland Fire Historian

including the present article. For example, the Honey Fire was one of five wildfires that Anderson (1983) used to evaluate his two elliptical fire shape models.

Olsen's (1941) documentation of the fire suppression decisions and actions on the Honey Fire are also valuable, though controversial. His case study analysis of the Honey Fire provides lessons for fire managers and researchers alike and raises issues that are still pertinent today, including some of the following ethical questions:

- Should case studies document fire control activities as well as fire behavior and compare model predictions and accepted knowledge against observations?
- When should the observer drop the camera and notebook and pick up a shovel or pulaski?
- When is it appropriate for a researcher to critique the decisions and actions of firefighters and fire managers or analyze how a fire should have been suppressed?
- Is it incumbent upon researchers to raise questions and point out deviations from standard operating procedures and discuss potential reasons for doing so?

A clear understanding of what happened during a fire is often "hard to acquire because it is obstructed by the natural human desire to save face, fear of disciplinary action, fear of being made a goat, and lack of confidence in the competence and impartiality of men who may judge the record," as pointed out by Headley (1943). However, a case study is not intended for "taking people to task for errors in judgment, but solely to ensure that the lessons that have been learned contribute to the success of future fire suppression operations" (Luke and McArthur 1978).

Implications

The general value of wildland fire behavior case studies has been discussed at length (Alexander and Thomas 2003a, 2003b, 2006). However, case studies are commonly seen as the "poor cousins" of fire science, occasionally tolerated but seldom encouraged in the scientific and technical peer-reviewed literature, although exceptions do exist (e.g., McRae 1986, Noble 1991). This situation contrasts with that of other professions, such as engineering, medicine, business, and law, where case studies are well accepted (Henderson and others 1983). For example, the *New England Journal of Medicine* has published an ongoing series of case studies since 1923 (Falagas and others 2005) and the Harvard Business School is renowned for the use of the case study method in the classroom (McNair 1954).

On Criticism and Wildland Fire Suppression

The one contemporary issue that interests me most in this article is sensitivity to the concept of criticism—constructive or otherwise.

We still have not, I'm afraid, learned to use criticism to its full benefit. Many fire managers and leaders in today's firefighting ranks are especially fearful of criticism from official sources—especially as it relates to firefighter safety. After-action reviews, risk refusal, lessons learned, accident prevention analysis and other tools are being successfully used to counteract resistance to constructive criticism, but much more work is needed. It will always be so as long as firefighters remain a proud, self-assured bunch, and they want to control fires in risky environments.

The source and purpose of criticism is key here. The threat of "witch-hunts," real or imagined, will keep criticism a sensitive subject. Direct criticism from research is no exception, even with good intentions.

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> We can only speculate whether the gain was worth the adversity that Olsen and his crew faced afterward.

Case studies can bring to light unusual or perplexing problems that might otherwise be neglected and, by telling a story, can ground what would otherwise be dry theory into a meaningful context (Hallenbeck 2005). However, case studies can be among the worst of the literature, offering few conclusions. Additionally, extrapolating conclusions from a single case is usually unwise, and attempting to solve a difficult case after the fact can become an exercise in self-aggrandizement (Hallenbeck 2005).

The role of the fire researcher as an independent observer established by Olsen (1941) and others more than 70 years ago continues to be used today. For example, current work by rapid-response researchers focuses on gathering data related to fire behavior and fire effects (Lentile and others 2007a, 2007b).

Similar activities have been undertaken in the past, especially in documenting free-burning fire behavior (e.g., Hardy 1983, USDA Forest Service 1993, Wilson and Davis 1988). In fact, Forest Service pioneer fire researcher Harry T. Gisborne is believed to have published the very first attempt at a comprehensive wildfire case study in his description of the Quartz Creek Fire (Gisborne 1927), which occurred on the Kaniksu National Forest adjacent to the Priest River Experimental Forest in northern Idaho during the summer of 1926; Kay (1927) published a less detailed documentation of several fires that occurred the following summer in Western Canada. This was followed by several other pioneering case studies in North America in the early 1930s (e.g. Jemison 1932, Dauge 1934, Shaw 1936).

Documenting or analyzing fire suppression strategies and tactics has not been undertaken as part of rapid response research to date, despite the fact that fire behavior may be influenced by fire suppression and that fire suppression actions are arguably an important part of the record. Although further analysis of human factors and activities on a fire opens the door to controversy, it may nonetheless provide valuable information and learning tools for fire managers. Taking a page from the *New England Journal of Medicine* and developing a mechanism to analyze and publish a regular series of peer-reviewed case studies of fire behavior and fire suppression activities would be a valuable addition to both the fire management and fire research professions. This would serve to complement the suggestion of creating operational wildland fire behavior research units (Alexander 2002).

Perhaps the idea of fire researchers critiquing human decisionmaking and actions would be viewed by fire managers as taboo, although there doesn't seem to have been any past reluctance to publish positive assessments (e.g., Countryman 1969, Kurth 1968, Scowcroft and others 1967). Nevertheless, we suspect a certain sensitivity still exists in having fire researchers second-guess fire operations personnel. This might be overcome in part by involving practitioners in the analysis.

Parting Thoughts

As fire behavior research professionals, we admire the determination that Olsen and others showed in their approach to systematically documenting the Honey Fire. It must have been extremely difficult for Olsen to complete his case study article in the face of the criticism that followed the control of the Honey Fire.

We can only speculate whether the gain was worth the adversity that Olsen and his crew faced afterward. Despite their express freedom to study fire behavior, the question of whether or not to engage in initial attack must have constituted a major moral dilemma. Obviously, the crew sincerely believed in the value of their research, and such dedication to the task is commendable. Would you have done the same?

Acknowledgments

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Vehicle and equipment used in fire behavior studies by fire research staff of the Southern Forest Experiment Station during the mid to late 1930s on the Harrison Experimental Forest, De Soto National Forest, MI. From left to right, the instruments are Foxboro pyrometer, thermocouple wire, thermocouple switch dial, storage battery, compass and Jacob staff, 8-pen thermograph recorder, portable recording hygro-thermograph, hand aspirated psychrometer, anemometer, and wood carrying case. In the truck compartments there are glass jars for fuel samples, cans for soil samples, a chain, and cloth of varying colors for plot markings. Photo: T.T. Kohara, Forest Service, 1937.

Remembering (or Discovering) the 1988 Yellowstone Fires

Any member of the wildland fire community younger than 21 years old was not even born when the Yellowstone fires of 1988 took place. And many of those who were involved have since gone on to retire from active service or are about to. Thus, a report recently published by the Wildland Fire Lessons Learned Center (WFLLC) will no doubt be of value to both generations in remembering, or in fact discovering, the past. The WFLLC report is entitled "The 1988 Fires of Yellowstone and Beyond as a Wildland Fire Behavior Case Study" and was written by Dr. Marty Alexander. This report is based in part on the opening remarks made by the author at the fire behavior fuels and weather session of The '88 Fires: Yellowstone and Beyond conference held 22–27 September 2008 in Jackson Hole, WY. Dr. Alexander served as the co-organizer and co-moderator of the session. A copy of the WFLLC report is available for download at: <http://www.wildfirelessons.net/documents/alexander_Yellowstone88_FB.pdf>.

A crowning forest fire begins to descend upon the Old Faithful complex in Yellowstone National Park on September 7, 1988. Photo: Jeff Henry, National Park Service, courtesy of the Yellowstone Digital Slide File.

The Effects of Climatic Change and Wildland Fires on Air Quality in National Parks and Wilderness Areas

Don McKenzie

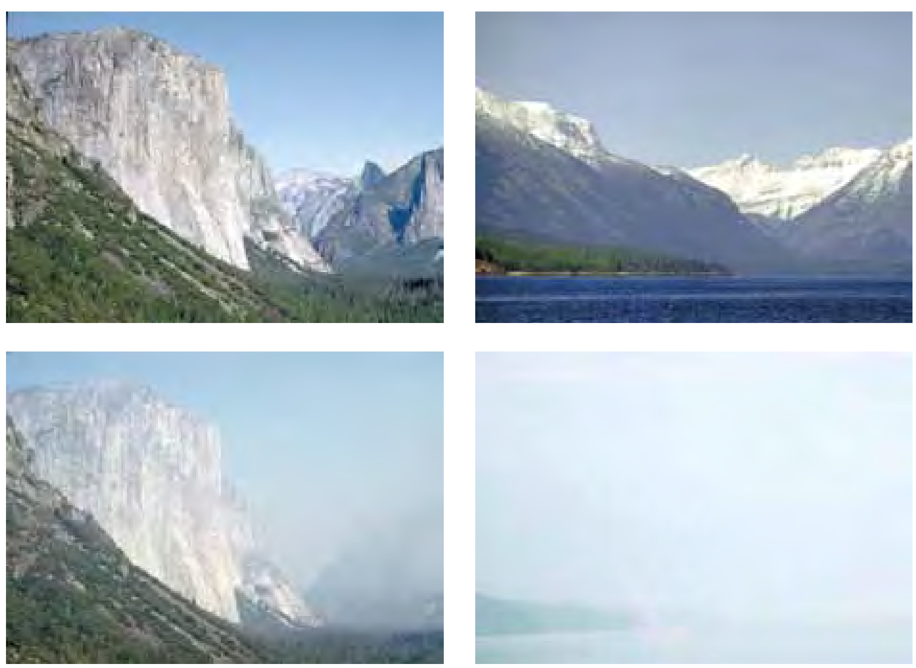
How will climatic change and wildfire management policies affect public land management decisions concerning air quality through the 21st century? As global temperatures and populations increase and demands on natural resources intensify, managers must evaluate the trade-offs between air quality and ongoing ecosystem restoration. In protected areas, where wilderness values are paramount, public land agencies have adopted the policy of using wildfires to benefit natural resources, allowing naturally ignited fires to burn unless they present additional threats, such as fire risk to structures or degraded air quality.

Effects on Air Quality

Fire effects on air quality can be both local and regional. Smoke exposure at fires and immediately downwind from fires can cause respiratory problems even in healthy people, but exposure is especially problematic for those with asthma or other chronic respiratory problems. Particularly hazardous are the particulate emissions smaller than 2.5 microns (2.5 x 10-6 m) in diameter ($PM_{2.5}$), which can be breathed more deeply and cross protective membranes in the lungs. These same particulates and other elements of the smoke plume can impair visibility hundreds of miles downwind from emissions sources (Malm 1999). In the Western United States, regional haze from fires and other sources reduces visibility in most of the protected areas at some time during a typical year. The worst days, in terms of visibility, are usually associated with smoke from wildfires.

To maintain air quality, we need to understand not only present-day emissions from fires but also how conditions may change over time in response to future climatic changes, land use, and management strategies. Fire regimes will likely evolve in response to temperature increases and associated vegetation changes (McKenzie and others 2004). The annual area burned by wildland fire is expected to increase across the Western United States

In the Western United States, regional haze from fires and other sources reduces visibility in most of the protected areas at some time during a typical year.



Yosemite (left) and Glacier (right) National Parks experiencing near-pristine (top) and severely degraded (bottom) visibility. Photos courtesy of the IMPROVE Web site. [Web site <http://vista.cira.colostate.edu/IMPROVE/>.]

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and Canada (Flannigan and others 1998, McKenzie and others 2004, Gedalof and others 2005).

Fires in many ecosystems are already becoming larger and more severe than under historical conditions because of increasingly severe fire weather, unnatural fuel buildup from fire suppression, or both (Agee 1997, Allen and others 2002). Increases in area burned and fire severity increase biomass consumption, smoke emissions, and atmospheric dispersion of particulates and aerosols that produce regional haze.

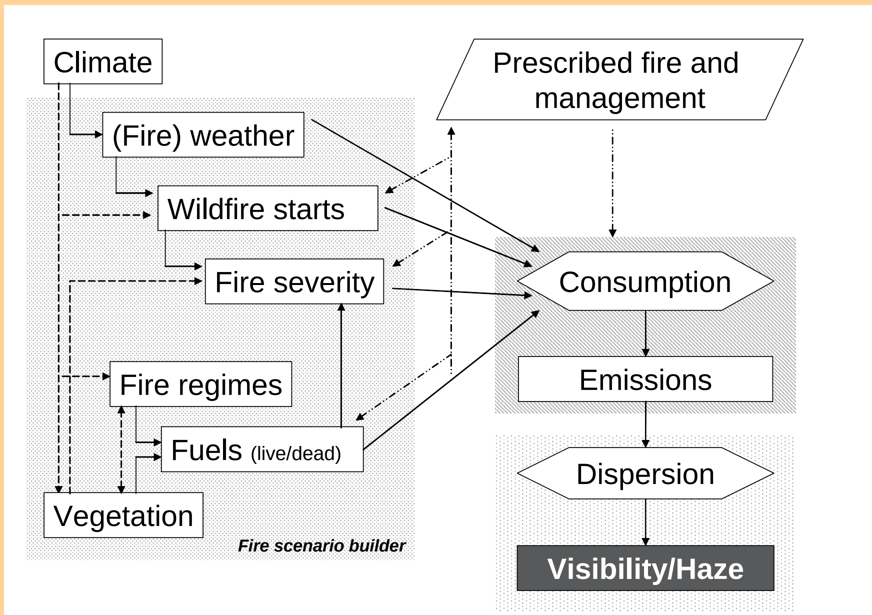
Air Quality Trade-Offs

There are many obstacles to returning the Nation's wildlands to their natural fire regimes, as noted by other authors in this issue. In many regions, such as the Pacific Northwest, air quality restrictions are one of the major impediments even to well controlled prescribed fires. These restrictions are based on the hazard of smoke exposure to local communities. Local effects, and the prospect of generating unacceptable visibility impairment in protected areas many miles away, make the management of wildfires for resource benefits less available as a fire management tool.

In one study, colleagues and I simulated smoke dispersion and regional haze from the wildland fires of 2003 in the Pacific Northwest with an integrated model of fire starts, combustion, emissions, and dispersion. We found that wildland fires in Oregon and Washington produced significant regional haze downwind at Glacier National Park in Montana and the Bob Marshall and Selway-Bitterroot Wilderness Areas in Montana and Idaho (fig. 1).

Fire Scenario Builder: A Tool for Predicting Regional Haze From Wildland Fire

Haze-producing emissions are sensitive to weather patterns and the nature of fire occurrence, which can be offset by management efforts. The fire-scenario builder uses real-time regional meteorology to simulate regional haze under current conditions and allows for the projection of wildfire events. A fuel-mapping module links vegetation data to a fuel classification system. A framework of emission, consumption, dispersion, and trajectory models reads the fire event data and the fuel mapping and calculates smoke emissions, plume rise, and regional-scale dispersion. Associated research is reported in McKenzie and others (2006).



Thinking Locally, Reacting Globally

Fire managers in national parks and wilderness areas are faced with background levels of reduced air quality, which exacerbate the conflict between air quality and other wilderness management goals. The contribution of wildfires to haze, in particular those wildfires allowed to burn as a natural ecological process, may be overestimated in some areas, leading to management choices hostile to the expansion of the use of wildfires for resource benefits. In some cases, wildfires may be the sole source of smoke, whereas in others it may be a minor contributor alongside agricultural and industrial pollution and haze from distant wildland fires.

Climate Change and the Use of Wildfires as an Ecological Process

How will wildland fire affect visibility in the future? With a warming climate, statistical models and simulation models suggest that wildland fire areas will increase in

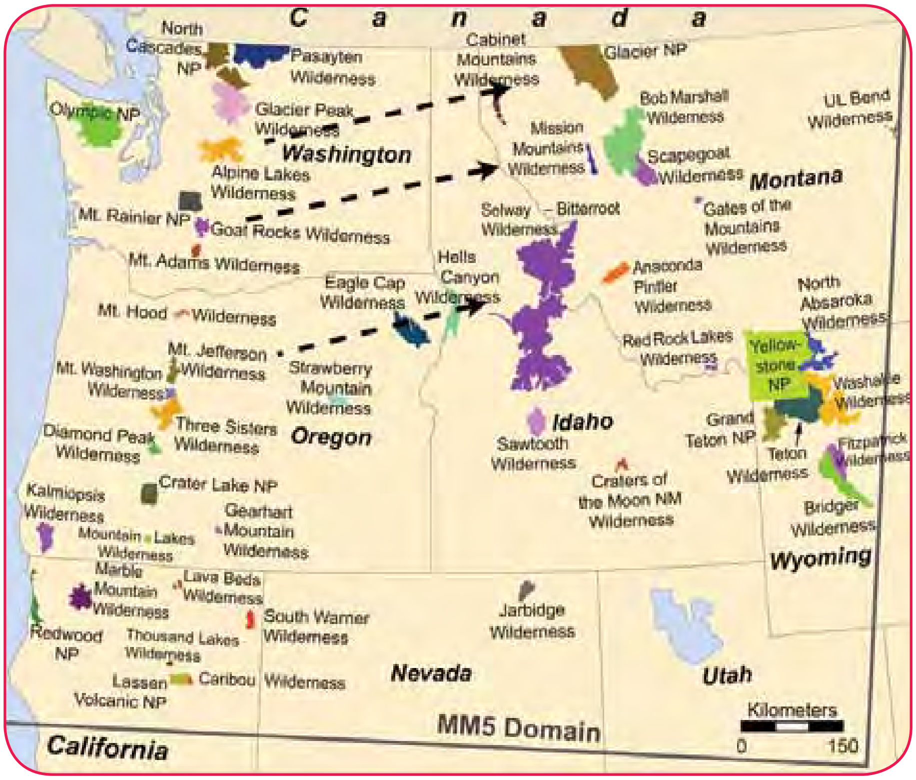


Figure 1—*Class I wilderness areas in the Pacific Northwest. Arrows indicate approximate flow patterns of smoke emissions from wildland fires in Washington and Oregon. From McKenzie and others 2006.*

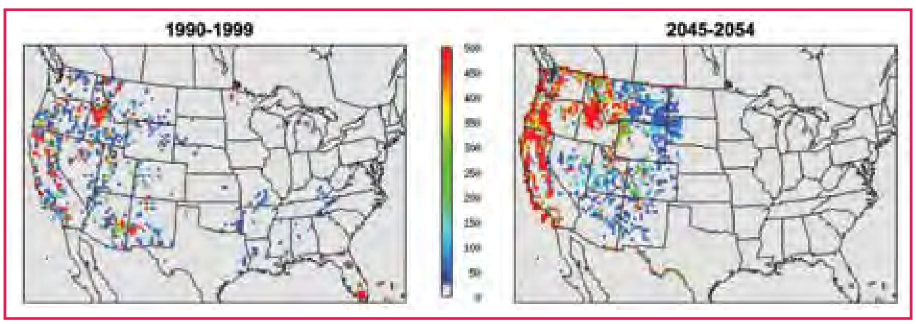


Figure 2—*Total emissions of $PM_{2.5}$ (tons) from wildland fires simulated over a future decade (2045–2054) compared to estimates from fire records (1990–1999). Simulations were restricted to the West; the observational data covered the conterminous United States.*

the Western United States (fig. 2). We can, therefore, also expect the contribution of fire to regional haze and reduced visibility to increase.

Emissions are projected to increase, especially in the westernmost States. Given current patterns of smoke dispersion, in which haze from fires in Washington, Oregon, and California significantly degrades visibility in national parks and wilderness areas to the east, Idaho and Montana will continue to be affected by regional haze, thereby compromising the role of naturally ignited wildfires as an ecological process.

Given the expected complexity of future management and policy decisions, multidisciplinary approaches are needed to guide management alternatives in the face of dynamic ecosystems and a warming climate. Examining prescribed fire scenarios or other means of fuel reduction allows us to estimate the potential value of fuel treatments on multiple-use lands for enabling ongoing application or expansion of managing wildfires for resource benefits in protected areas. Understanding trade-offs between air quality and ongoing ecosystem restoration, and precise quantitative estimates of the effects of fuel treatments, will help land managers across the West make informed choices.

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The 10 Standard Firefighting Orders and 18 Watch Out Situations: We Don't Bend Them, We Don't Break Them...We Don't Know Them

Bryan Scholz

Most of us don't know the 10 standard firefighting orders and 18 watch out situations, the "10 & 18," by heart. Judging by our fatality reports and close calls, it shows.

In 1956, Forest Service Chief Richard McArdle convened a task force to study 16 fires that occurred from 1937 to 1956. These fires had 79 fatalities due to burnover. The resulting 1957 report to the Chief (Moore and others 1957) identified 10 factors that were common to many of these fires:

1. Unexpected fire behavior—basic elements not understood; indicators of change in usual fire behavior not recognized; local fire weather forecasts not obtained, inaccurate, or not understood.
2. Instructions—not followed, not clear, or not given.
3. Foremanship—lost control of personnel at critical time.
4. Line supervision—overhead busy on minor jobs, not available when major decisions had to be made.
5. Communication—not available, not used, or broken down.
6. Firefighting strategy and tactics—control effort made in wrong location or without adequate margin for safety; detailed line location incorrect.
7. Scouting—not done, not thorough, too dependent on air scouting.
8. Escape plan—not formulated, not explained, not executed.
9. Lookout posting—routine practice not followed.
10. Organization—humans and machines committed to action without adequate supervision, or without adequate tie to the rest of the organization.

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> Knowing the "10 & 18" is the best tool we have to protect ourselves from bad decisions. It is the best tool we can give to our rookies to protect them from our bad decisions.

To address these critical factors, the report presented a list of 10 "standard firefighting orders" and recommended:

> "These orders are to be committed to memory by all personnel with fire control responsibilities.
>
> "Military organizations have had long experience in training men to remember certain fundamental instructions and to react even in emergencies in accordance with those instructions. One device by which such discipline is achieved is that of 'general orders,' which all men of the unit are required to memorize. On some of the fires we reviewed, men who knew better just did not pay adequate attention to good firefighting practices that seem like small details, but could become the critical item in an emergency. The use of a form of standard orders starting immediately would be a long step in the direction of assuring attention to the fundamentals" (Moore and others 1957).

Shortly after the standard firefighting orders were incorporated into firefighter training, the 18 watch out situations were developed to complement them (USDA Forest Service 2008a).

Fifty years later, fire has found no new way to hurt us. We continue to make the same mistakes. From Mann Gulch to South Canyon to Cramer, we put ourselves into places where there is unburned fuel between us and the fire, or where we can't see the main fire and we're not in contact with someone who can. We make decisions that are not based on current and expected fire behavior.

In "A Trend Analysis of Fireline 'Watch Out' Situations in Seven Fire Suppression Fatality Accidents" (Morse 2004), 84 separate hazardous conditions or events were identified in the fatality reports. Morse states, "In each of seven fatality events, a single overlooked 'watch out' appeared to be the major contributing factor."

In a September 2004 report to the Chief, the Office of Inspector General (OIG) analyzed the fatality reports for the Cramer, Thirtymile, and South Canyon Fires. The OIG found that "fire suppression personnel violated all of the [standard firefighting] orders and failed to mitigate most of the watch out situations. Each fire had rapid growth unexpected by management; fire suppression personnel employed questionable or improper tactics and did not adjust their tactics as necessary" (USDA Office of Inspector General 2004).

This is not just a problem during wildfire suppression. In 2006, 10 people assigned to the Little Venus Fire on the Shoshone National Forest in Wyoming as part of a fire use module were entrapped by the fire and deployed fire shelters. Members of this fire use module did a great service to their profession by contributing openly and honestly to the after-action review, especially by reminding us that a fire managed in part for ecosystem benefits (those previously called wildland fire use events) is still a wildfire, and the same rules apply. From the review:

> "This incident...differs from past deployments in that the involved personnel were not actively engaged in the performance of an operational fireline assignment when the deployment occurred. They were enroute to a camp location to debrief with a crew they were replacing and would not have been given a fireline assignment until the next operational period."

"The 10 standard firefighting orders must be firm rules of engagement. They cannot be simple guides, nor can they be 'bargained.' They are the result of hard-learned lessons. Compromise among one or more of them is always the common denominator of tragedy. On Dude, South Canyon, and Thirtymile, these orders were ignored, overlooked, or somehow compromised. The orders mean little once we are in trouble, and because of that we must routinely observe them and rely on them before trouble confronts us."

—Jerry Williams, former director, Fire and Aviation Management (2002)

> "Many individuals did not have a thorough understanding of the purpose and objectives of their fireline assignments; many did not have a good awareness of the weather, its influence on fire behavior, and resource disposition; an understanding of planned contingencies; working knowledge of personnel assigned to the fire and the chain of command; and assumptions were made that led to failure to realize deficiencies in the organization and implementation. As a result, this lack of situational awareness created instances of confusion, incomplete information sharing, and contributed to complacency."

> "There were numerous instances where personnel indicated their perceptions that wildland fire use and wildfire suppression were two separate events, even on a single wildland fire such as the Little Venus Fire."

The reasons for not recognizing the 18 watch out situations and not following the 10 standard firefighting orders are complex, and have much to do with human factors. But whatever the reasons, judging by our fatality reports and close calls, we continue to act like we don't know the "10 & 18," and the reason is, a lot of us don't. This doesn't make sense. We should be required to prove, every year, that we know the "10 & 18" by heart in order to get an incident qualifications card ("red card"). Knowing the "10 & 18" is the best tool we have to protect ourselves from bad decisions. It is the best tool we can give to our rookies to protect them from our bad decisions.

Some people think that the new foundational doctrine for fire suppression (USDA Forest Service 2005) replaces the "10 & 18." While this is not its intent, there is language in the doctrine that confuses the issue. The doctrine describes the "10 & 18" as "universal principles of suppression operations… principles [that] guide our fundamental fire suppression practices, behaviors and customs, and are understood at every level of command." However, the doctrine then states that they "…are not absolute rules. They provide guidance in the form of concepts and values." This is an unfortunate contradiction. Either the "10 & 18" are universal and fundamental, or they are not. Either we base all of our actions on current and expected fire behavior or we don't. And if we're not going to base all our actions on current and expected fire behavior, then what are we going to base them on?

Some people think that "lookouts, communications, escape routes, and safety zones" (LCES) replace the "10 & 18." I had the privilege of hearing one of the first lectures that Paul Gleason gave about his concept of LCES, and it was not his intent that LCES replace the "10 & 18." The establishment of LCES on the fireline is dependent on recognizing the watch out situations and following the standard firefighting orders. The use of LCES is a dynamic system; it exists and moves in space and in time, as the fire moves and as the firefighter moves. LCES "must be continuously evaluated as fire conditions change" (USDA Forest Service 2008b). But the system will not work unless it is based on current and expected fire behavior, and a firefighter who doesn't know that standard order can't follow it.

There is a perception among some firefighters that following the "10 & 18" reduces our tactical options, but there is no fire suppression tactic that is prohibited by "10 & 18." For example, downhill line, 1 of the 18 watch out situations, is a potentially hazardous situation whose risk is mitigated by following the standard firefighting orders. Downhill line is not prohibited; in some situations, it is safer.

"Safety first" is a simple, clear expression of the fundamental value of our profession.

There is concern that the orders are not measurable and quantifiable. So what? They are clear and concise: "keep calm," "give clear instructions," and "know what your fire is doing." While most mission statements, vision statements, and value statements are ambiguous or grammatically challenged, "safety first" is a simple, clear expression of the fundamental value of our profession.

Fifty years ago, some smart, experienced firefighters identified the common hazards of the fireline and came up with a set of rules to mitigate those hazards that is elegant in its simplicity. It is one of the best things that the Forest Service has ever done. We should honor the memory of those firefighters by seeing that "the orders are committed to memory by all personnel with fire control responsibilities."

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From Another Perspective—The 10s, 18s, and Fire Doctrine

Larry Sutton

The following comments are offered as response to the article, "The 10 Standard Firefighting Orders and 18 Watch Out Situations: We Don't Bend Them, We Don't Break Them...We Don't Know Them;" they are meant to continue the discussion on this important topic. My impression of some of the points the article makes might be summarized as follows:

1. If all firefighters memorized the "10 & 18," we would have fewer fireline fatalities;
2. Historic investigation reports have reached the correct conclusion that firefighter mistakes cause firefighter fatalities, and the same reports accurately point out what those mistakes were;
3. The standard orders need not be measurable and quantifiable; and
4. Foundational doctrine for fire suppression somehow contradicts or confuses the intent or purpose of the "10 & 18."

We all want firefighters to come home safely after every shift, on every fire. Yet we recognize that the environment in which we operate contains many hazards, some of which can be difficult to detect or predict until it's too late. The problem with relying too much on memorization of rules to keep us safe is that we are presupposing that a firefighter's mind will retrieve the appropriate piece of memorized information for any situation, even under stress, and make it available just when needed. Unfortunately, human minds under duress just don't work that way. Even if they did, a firefighter would still have to consider multiple possible courses of action, decide, and then act under conditions involving time pressure, fatigue, and incomplete information. These "human factors" are extremely important to any complex human endeavor like wildland firefighting, which is why the approach of simple memorization of rules will ultimately be ineffective. It is easy to memorize words without understanding their implications.

The problem with relying too much on memorization of rules to keep us safe is that we are presupposing that a firefighter's mind will retrieve the appropriate piece of memorized information for any situation, even under stress, and make it available just when needed.

Furthermore, we have to look at what is being memorized. Standard order #3 is frequently mentioned: "Base all actions on current and expected behavior of the fire." The problem with this order is that you can follow it and still be killed! All that is required is for the fire to do something unexpected. In fact, that is the true common denominator of fire behavior on tragedy fires: what the fire actually did wasn't what firefighters thought it was going to do. An investigation report that says that specific fire behavior could have been or should have been predicted is itself an interpretation: investigators have the advantage of hindsight. What actually happened was that the fire moved faster, or went in a different direction, or burned with more intensity than firefighters thought it would. Is this a shortcoming on the part of the firefighters? Not necessarily. Unpredictability is not predictable: even the most sophisticated fire behavior prediction tools currently available cannot always replicate observed fire behavior.

Unfortunately, accident investigation reports have historically done a poor job of reconstructing the "whys" of an accident. Why did the firefighters' decisions make sense to them at the time? Simplistic causal factors have been cited, such as the "violation" of a standard order requiring firefighters to have an escape route. Often, firefighters did have one or more escape routes, but they were inadequate when needed. We need to know why firefighters thought an escape route would be adequate when in fact

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it proved not to be. Most reports haven't told us that, even when firefighters survived a burnover.

The standard firefighting orders and watch out situations focus on preventing burnovers, but they are no guarantee of safety from fire behavior-related hazards, and they do not address the other four-fifths of accidents that kill firefighters. Accident data show that burnovers account for approximately 21 percent of all wildland firefighter fatalities. The other 79 percent are from causes unrelated to fire behavior, including aviation (23 percent), driving (23 percent), heart attacks (22 percent), and hazard trees/rocks (4 percent) (see "Wildland Firefighter Fatalities in the United States, 1990–2006," available at <http://www.nwcg.gov/pms/pubs/pms841/pms841_all-72dpi.pdf>).

> The foundational doctrine for firefighting is based on the premise that the best tools we have are firefighters' brains using all our best practices for safe firefighting, not a set of hard and fast rules to cover all situations.

It's very important for firefighters to clearly understand what the standard firefighting orders represent. First, we need to be clear about whether or not they are, in fact, "orders": standards that must be followed at all times. Second, if we consider them to be mandatory orders and use them as a yardstick to judge firefighter behavior when things go wrong, then they must be "measurable and quantifiable." But, is it even possible for the standard orders to be measurable and quantifiable? It seems clear that while the standard orders and the 18 situations are extremely useful as principles or best practices, they are, in fact, subjective and circumstance-dependent enough that they cannot function as true standards by which firefighters should be judged in a post-accident investigation. In the past, occupational safety and health investigators have agreed to have standard order "violations" removed from the record. There is also now case law (*Backfire 2000 vs. United States of America*, 2006, available at <http://wildfirelessons.net/documents/CJ_Molloy_ruling_memo.pdf>) describing the standard orders as "vague principles" and calling the language used in them "…the language of discretion, not of specific mandatory actions or protocols."

For example: should you automatically disengage if you can't maintain prompt communications with your supervisor? How are "prompt communications" defined? Is it really possible to know what your fire is doing at all times, when you are on one division of an 80,000 acre (30,000 ha) fire? It's important to know what's happening on your division and adjoining divisions for the safety of your crew, but it's often a practical impossibility to know what's happening with the whole fire unless you're an operations section chief. Even then, you'd only have a general idea—you wouldn't know about every spot fire on every division. The standard orders cannot be absolute rules. We must recognize them as best practices for safe firefighting and teach them that way.

The foundational doctrine for firefighting is based on the premise that the best tools we have are firefighters' brains using all our best practices for safe firefighting, not a set of hard-and-fast rules to cover all situations. Simply put, the standard orders and watch outs alone *aren't enough* to keep firefighters from harm. There is no silver bullet in managing the risks confronting wildland firefighters; there is just a large toolbox of principles and best practices for safe and effective firefighting, coupled with firefighters' discretion.

Doctrine was never meant to replace the standard orders; lookouts, communications, escape routes, safety zones (LCES); or other published guidance. Doctrine is the leaders' intent: a common set of values that can guide our actions in a variety of situations. It's noteworthy that, while the idea for standard orders came from military organizations, so did the idea for operational and strategic doctrine, something that exists today in all branches of the U.S. military. Furthermore, the general orders in the military, upon which the standard orders were modeled, are just that: general orders, not specific ones. The general orders have to do mainly with soldiers' conduct while on guard duty—they are not a set of prescriptive rules to be followed in any given tactical situation. The military places a high value on individual soldiers' initiative and creativity in those situations, just as we do for our firefighters.

As for LCES, that too is dynamic guidance. Brad Mayhew, a former hotshot, developed a variation on LCES that he calls “F LCES Δ.” The “F” stands for fire behavior, which urges you to consider the potential “worst case scenario.” LCES is looked at to determine if it’s adequate for that worst case. And the “Δ” (delta) represents change—it is there to remind you to consider “what’s changing now” as well as “what might change later.” (For a more thorough discussion, see http://www.firerescuemagazine.com/pdfs/WUI_04.pdf.)

These topics will be discussed and debated by firefighters forever. It’s important for firefighters to learn and understand—not just memorize—the standard firefighting orders and watch out situations, LCES, and all the other tools of our trade. Well-educated firefighters and capable leaders who are able to maintain situation awareness and continuously make sense of their environment are safe firefighters. But we’re kidding ourselves if we think that any single rule set will serve to keep everyone safe on every fire. There is no such thing as a “safety guarantee” in the dynamic wildland fire environment. ■

Introducing the Virtual Incident Procurement (VIPR) System

Beginning with the 2009 fire season, the Forest Service is using the Virtual Incident Procurement (VIPR) system to acquire certain types of contracted equipment for incident management. The VIPR system is a Web-based Forest Service application that awards and administers preseason Incident Blanket Purchase Agreements or I–BPAs (formerly called Emergency Equipment Rental Agreements or EERAs; EERAs are used for at-incident sign ups and are not part of VIPR).

Solicitations for wildland fire equipment are posted on the FedBizOpps Web site: <https://www.fbo.gov/>. Vendors may easily sort and find solicitations issued through VIPR, e.g., “VIPR I–BPA for Mobile Laundry in the Intermountain Region.” Computer-based forms submitted to VIPR are used to respond to solicitations. Vendors who wish to participate will need appropriate computer access and an eAuthentication account.

For more information about VIPR, including how to set up an eAuthentication account and what equipment categories are being solicited, visit <http://www.fs.fed.us/business/incident/vipr.php>.

The Potential for Restoring Fire-Adapted Ecosystems: Exploring Opportunities To Expand the Use of Wildfire as a Natural Change Agent

Gregory H. Aplet and Bo Wilmer

Fire has shaped America's forest ecosystems for millennia. From ponderosa pine woodlands that burn every few years to subalpine forests that erupt into flame every few centuries, most forests have evolved with fire and depend on periodic blazes for health and regeneration. Fire is such an important force that vegetation ecology and fire cannot be described independently.

Just as vegetation ecology and fire are intimately connected, land management and fire management are inextricably linked. Policymakers and forestry experts recognize that, after a century of fire suppression, there is a crisis in forest health: fire-dependent ecosystems starved of regular fire cycles now have unhealthy fuel loads and experience unnaturally large wildfires (Laverty and Williams 2000, Aplet and Wilmer 2005).

In response, forest managers seek to restore fire to fire-dependent ecosystems using both management-ignited and natural fires. The management of natural fires as a natural change agent in designated, remote sections of the landscape is widely accepted by scientists, managers, and policymakers. It is a tool for restoring forest health and mitigating the escalating costs of fire suppression (USDA Forest Service and others 2001). But despite its broad acceptance, in practice, wildfires are rarely used to benefit natural resources. Many people consider allowing wildfires to burn for resource benefit to be appropriate only in national parks and wilderness; even some fire managers view this management option as too risky (Parsons 2000, Black and others 2008). If the benefits of wildfire are to be realized, use of wildfires as a natural change agent must be applied over large areas wherever safe. The fire management approach we suggest would greatly expand the use of wildfires for resource benefit across significantly larger areas of the Western landscape.

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> Just as vegetation ecology and fire are intimately connected, land management and fire management are inextricably linked.

A Three-Zone Approach

Three situations exist on any landscape with regard to communities and fire:

1. Where fire has the potential to cause great damage to people and homes, and fire should always be excluded;
2. Where people are uncomfortable with the close proximity of natural fire but fire could be used as a tool to reduce fuels and restore ecosystems under tightly prescribed conditions; and
3. Where fire is distant enough from communities that it poses little risk to people and resources and natural fires can be used to help achieve land management objectives.

These three situations are compatible with a three-zone, landscape approach to wildland fire management (DellaSala and others 2004, The Wilderness Society 2006). Under this approach, a *community fire planning zone* (zone 1) consists of the area immediately adjacent to communities and is managed for community protection. A *wildfire resilience zone* (zone 2) exists beyond zone 1 for a few miles and is managed not only to minimize unplanned fire through direct attack or containment but also to restore conditions that are ecologically resilient to fire. Beyond zone 2, the full range of management responses to fire (from direct attack to monitoring) is possible, but emphasis is placed on the use of fire for resource benefit. In this *fire use emphasis zone* (zone 3), management of fire as a natural process is a priority when conditions allow. Public land managers may use these three planning zones to focus resources where they are most needed and to restore natural processes to the landscape where practical.

Because the highest priority is the protection of people and their homes, the first step in designing a plan to promote the management of fire as an ecological process is identifying the *community fire planning zone* (Wilmer and Aplet 2005). Although sometimes called the wildland-urban interface, the term community fire planning zone better conveys the overriding objective of community protection for the area. Areas designated as zone 1 should be examined for opportunities to improve public safety through public education, infrastructure improvement, and fuels treatment (Cohen 2000, Nowicki 2002). Delineation of community areas at risk from wildland fire can help focus community protection efforts.

The *wildfire resilience zone* would extend from the *community fire planning zone* to a distance considered safe for possible fire use. Within zone 2, suppression would be the response to unplanned ignitions, but fire could be introduced intentionally to achieve management objectives. The primary management objectives in zone 2 would be (1) protection of critical resource values such as recreation sites, experimental forests, and research natural areas, and (2) maintenance of ecological resiliency through modification of forest composition and structure. Generally, this means fuels would be modified to protect specific resources and restore ecosystems (Landres and others 1999, Brown and Aplet 2000).

Opportunities for expanded management of wildland fires for resource benefit exist in the *fire use emphasis zone*. The full suite of management responses (including suppression and containment) is available under any given condition, but the preference would be to maximize opportunities for managing wildfire for resource benefit wherever possible. Delineation of zone 3 would require rigorous analysis to determine if an area is far enough away from communities such that fire would not be expected to threaten structures or other highly valued resources. Zone 3 delineation should increase managers' confidence to select this management option in the event of a natural ignition.

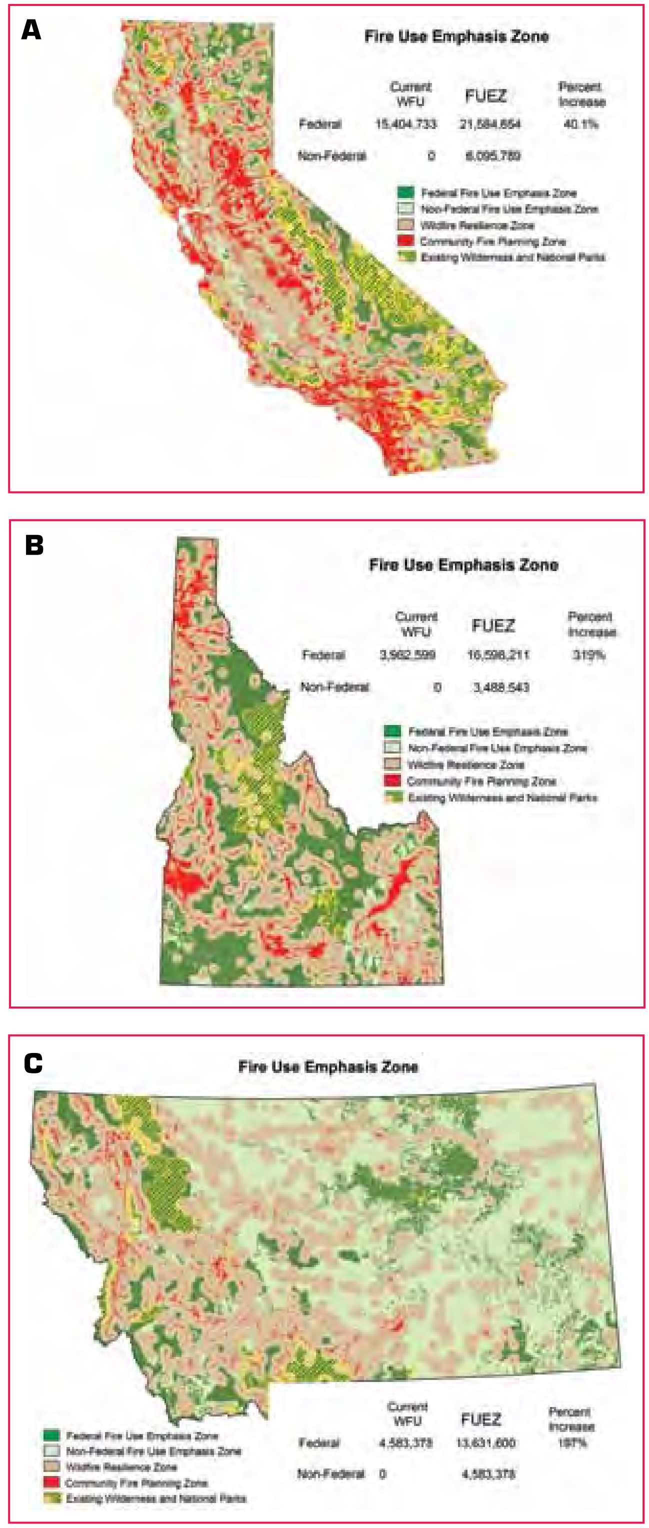

Figure 1—Comparison of current opportunities for using wildfires for resource benefits with an expanded fire use emphasis zone (FUEZ) in California (A), Idaho (B), and Montana (C). Current opportunities to use wildfire as a natural change agent based on existing national parks and wilderness are represented by yellow cross-hatching. Fire use emphasis zones (zone 3) are represented in dark green (Federal lands) and light green (non-Federal lands). The wildfire resilience zones (zone 2) are shown in pink. Community fire planning zones (zone 1) are shown in red.

Mapping the Zones

To represent the three-zone approach and identify opportunities for expanded use of wildland fire as a natural change agent, we mapped areas meeting the definition of a wildland-urban interface community.* Using housing data from Census 2000 and ownership data for California, Idaho, and Montana (three States representative of conditions in the Western United States), we identified locations meeting the housing density threshold for definition as a community. We removed public land (where houses generally do not occur) from census blocks and calculated where housing density within a census block exceeded one house per 40 acres (16 ha) on private land. We assigned those communities a ½-mile buffer to complete delineation of zone 1. A ½-mile buffer is codified in law (Healthy Forests Restoration Act of 2003) and provides a practical zone in which to look for opportunities to reduce home ignitability through fuels reduction, emphasis on use of fire-resistant building materials, and education efforts (Wilmer and Aplet 2005).

A buffer extending 5 miles around zone 1 represents the wildfire resilience zone (zone 2). In practice, the extent of zone 2 would have to be negotiated through participatory public planning; a 5-mile buffer was chosen as a starting point for this analysis because it seems a reasonable approximation of the discomfort zone within which it is unrealistic to expect people to accept natural fire. From ½ to 5 miles outside of communities also provides a reasonable area for fuels treatments that should be the focus of restoration work in the dry forests of the Western United States. In some cases, restoration would be desirable beyond this distance, but most opportunities to reduce fuels in dry forests at low elevations for restoration purposes exist within a few miles of communities. By limiting zone 2 to a 5-mile wide buffer, restoration planning can be focused on the "frontcountry," where the need is clear and there is less controversy over the use of thinning.

Managing the landscape under a three-zone, landscape-scale fire management strategy could dramatically increase the area on which natural fire could be managed for resource benefit, without fear of property loss.

We classified the remainder of the landscape beyond zone 2 as the *fire use emphasis zone*. We assessed opportunities for expanded management of wildfire by comparing the extent of zone 3 with an approximation of the current opportunities for managing wildfires for resource benefit, defined by the boundaries of existing national parks and wilderness areas in California, Idaho, and Montana.

Fire Use Emphasis Zone

Currently, 15,404,733 acres (6,234,074 ha) of national parks and wilderness areas in California are available for using wildfires as part of land management (fig. 1A). Under the three-zone approach suggested above, the estimated fire use emphasis zone would encompass 21,584,654 acres (8,935,000 ha) of Federal land (a 40-percent increase over the current situation) and 6,095,789 acres (2,466,878 ha) of private land, most of it in the mountains to the west of the Central Valley. Together, lands in this zone would amount to 27.5 percent, about one-quarter, of the area of California.

In Idaho, national parks and wilderness cover less than 4 million acres (1.6 million ha) (fig. 1B). Our estimated fire use emphasis zone would increase the amount of Federal land available for using wildfires as part of land management by 319 percent to 16,598,211 acres (6,717,057 ha), and identify 3,488,543 acres (1,411,763 ha) of non-Federal land, mostly in southeast Idaho, where natural fire could be considered as a management option. Zone 3 in Idaho would represent 37.6 percent, over one-third, of the State's area.

In Montana, the situation is even more dramatic. Montana currently has 4,583,378 acres (1,854,827 ha) of national parks and wilderness (fig. 1C). The delineated zone 3 would almost triple the amount of Federal land suitable for using wildfires as part of land management to 13,631,600 acres (5,516,512 ha), but an even larger change would be the inclusion of almost 29 million acres (11.7 million ha) of private land in the eastern two-thirds of the State. All told, zone 3 would represent 45.6 percent, almost one-half of the area of Montana.

*"Urban wildland interface communities within the vicinity of Federal lands that are at high risk from wildfire" (Federal Register 66(3): 751–777, January 4, 2001).

Land Management and the Management of Wildland Fire For Resource Benefit

Our calculation shows that managing the landscape under a three-zone, landscape-scale fire management strategy could dramatically increase the area on which natural fire could be managed for resource benefit without fear of property loss. The fire use emphasis zone would start at a distance of 5½ miles from delineated communities. In practice, this distance could be modified by individual community and scientific input, but these numbers do suggest ample opportunity for expanded use of wildfire in the West.

In order to implement the use of wildfire as a management strategy, Federal policy requires the existence of a management plan that recognizes a beneficial role for fire; currently, all human-caused ignitions must be suppressed. Even with an approved fire management plan that authorizes the use of naturally caused wildfire for resource benefit in a given area, weather conditions, personnel availability, and other variables would have to be considered before a manager could make a definitive decision to use wildland fire to improve ecosystem condition. Once the initial decision was made, fire managers would have to constantly monitor and re-assess conditions and order suppression where appropriate.

Identifying the specific conditions under which management of wildfire as a natural change agent might be appropriate requires detailed scientific and spatial analyses. Even in remote areas, forest conditions, weather, and wind factors may preclude the safe use of fire. The use of wildfires is appropriate only where the results of fire would benefit resources. For example, benefits are unlikely where invasive weeds now carry frequent, intense fire into plant communities in which fire was historically rare. Generally, ensuring resource benefits requires a determination that fire behavior will be natural or historically typical for the location. To provide a sufficient basis for fire management, a land management plan would not need to include these detailed analyses but must provide sufficient latitude to allow fire planners to identify the appropriate conditions for management of wildfires for natural resources in the subsequent fire management plan. Such latitude could be provided by delineating zone 3 as widely as possible.

> Wilderness, roadless areas, and remote roaded land provide excellent opportunities to plan for management of wildfire as a natural ecological process.

Management prescriptions appropriate for zone 3 range from addressing wilderness concerns and protection of roadless character in a roadless landscape to active restoration and protection of recreation sites in roaded areas. Prescribed fire could be used throughout zone 3 to achieve a composition and structure that can accommodate natural fire. This is especially true for roaded areas, where existing roads could be used (possibly after thinning of adjacent fuels) to systematically reintroduce fire to the landscape.

In the roadless landscape, including wilderness, managers must prove that proposed actions will not degrade roadless or wilderness character prior to manipulation, including the use of prescribed fire. The Wilderness Act requires a "minimum requirements analysis," a deliberate review to determine the least disruptive method necessary to accomplish the objective. The special values of roadless areas also demand that special care be taken. The Wilderness Act does not specifically prevent suppression action or fuel management in wilderness, but actions proposed for any part of the roadless landscape must be carefully planned using best available science and an inclusive public process. Because remote areas tend to be in higher elevation montane and subalpine forests, open deserts, and arid shrublands, little of zone 3 is likely to be in the low-severity fire forest types that may require thinning or prescribed fire before natural fire will yield resource benefits. The majority of zone 3 areas would include forests typified by less frequent fire regimes that would likely benefit from natural fire as long as fire regimes have not been altered by invasive species, human ignitions, or other causes.

Fire management in zone 3 should seek to maintain the natural character of the area, even in any roaded portion, and minimize impacts to aquatic, terrestrial, or watershed resources. Accordingly, minimum-impact suppression tactics should be used throughout zone 3 when suppression is the appropriate response.

Management of wildfires for resource benefit has historically been confined largely to wilderness areas and national parks, but there is no reason why fire cannot be used outside wilderness, wherever safe. Thus, the *fire use emphasis zone* may be mapped as everywhere beyond zone 2. Zone 3 in our examples includes any location further than 5 miles from the wildland-urban interface. The extent of zone 3 would vary regionally, depending on the degree of regional development. Opportunities for use of wildfires may be virtually nonexistent in some places, and in other areas, those opportunities may dominate. Wilderness, roadless areas, and remote roaded land provide excellent opportunities to plan for management of wildfire as a natural ecological process.

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Working Toward a Fire-Permeable Landscape—Managing Wildfire for Resource Benefits in Remote, Rural, and Urban Areas of Alaska

Mary Kwart and Morgan Warthin

Wildland fire is a recurring, significant, natural process in the boreal forest and tundra ecosystems of Alaska. These ecosystems surround Alaskan cities, towns, native villages, remote homes, and historic properties, rendering them susceptible to wildland fire. In 2004 and 2005, two of Alaska's three most severe wildland fire seasons on record, fires burned more than 11 million acres (4,444,000 ha), an area greater than that of Massachusetts and Connecticut combined. Now, fire managers must think of values at risk in terms of their permeability to wildland fire and begin to promote a fire-permeable landscape: one in which fire and values at risk coexist. Managing wildfires as an ecological process and natural change agent is the first of many steps toward achieving that landscape.

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> Fire managers must think of values at risk in terms of their permeability to wildland fire and begin to promote a fire-permeable landscape in which fire and values at risk coexist.

A Tool for Alaska's Fire Managers

The Alaska Interagency Wildland Fire Management Plan sets priorities for the assignment of firefighting resources statewide and provides a range of initial responses to wildland fire through the use of fire protection categories called "management options" (Alaska Wildland Fire Coordinating Group 1998). The four management options—*critical, full, limited*, and *modified*—are tied to the proximity of the fire to values at risk; they determine priorities for fire suppression needs and indicate where using naturally caused wildfires to benefit natural resources is appropriate.

Lands managed under the *critical* management option—where human lives, inhabited property, housing developments, or National Historic Landmarks are at risk—are the first priority for the assignment of suppression forces. Lands under the *full* management option—where uninhabited property or cultural, historical, or high-value natural resources are at risk—have second priority. Fires on *limited* management option lands are generally managed for resource benefits unless they threaten values on adjacent lands.

The *modified* management option is more flexible and provides a level of management between the *full* and *limited* options. A predetermined conversion date is used as part of the *modified* management option to determine whether initial attack on wildland fire is appropriate. Fires that start before the conversion date normally receive initial attack. On the conversion date, the Alaska Wildland Fire Coordinating Group assesses the current fire danger indices and fire activity to determine whether it is appropriate to convert to a noninitial attack response strategy. Fires starting after the conversion date might not be selected to receive initial attack and can be managed to accomplish resource management goals and reduce long-term suppression costs.

Most of Alaska's park units and wildlife refuges managed by the U.S. Department of the Interior, National Park Service (NPS) and U.S. Fish and Wildlife Service (FWS) have fire management plans that approve management of some wildfires for resource benefits on lands in the *limited* management option and on lands in the *modified* management option following the conversion date. If suppression actions have not been initiated and the criteria for an alternative response have been met, the agencies can also use naturally caused wildfires on lands in the *modified* management option before the conversion date, and those on lands in the *full* management option, for resource benefits.

Managing wildfires as an ecological process and natural change agent is the first of many steps toward achieving a fire-permeable landscape.

Fires used to protect, enhance, or maintain resources are managed with the expectation that they will be of long duration. Fire managers use long-term assessment methods and tools to help determine where the fire might burn, to identify long-term management actions, and to identify trigger points that will initiate actions for preventing the fire from burning into areas of higher protection priority or for protecting specific features. Fire managers face unique challenges: the incidence of wildland fire may be increasing on the landscape and Alaskan values at risk are varied, widely dispersed, and often difficult to access. Highlights of these challenges and their solutions follow.

Using Wildfire as an Ecological Process in Rural and Urban Alaska*

During the 2005 fire season, the Kenai National Wildlife Refuge in south-central Alaska managed two wilderness fires: the Irish Channel Fire and the Fox Creek Fire. Both were within sight of a major recreational road system and several Kenai Peninsula communities.

The Irish Channel Fire, ignited by lightning on July 6, burned on the south shore of 25,000-acre (10,100-ha) Skilak Lake within plain view of touring motorists. The fire burned in deep duff under white spruce and hemlock. Smoke was visible from the Sterling Highway, a main route into the Kenai Peninsula. The Irish Channel Fire was managed under a stage 1 wildland fire implementation plan (WFIP) analysis level for 12 days. When continuing dry weather indicated that active fire behavior and perimeter growth would continue, the WFIP analysis level progressed to a stage 2. Although not directly on a road network, the fire was directly west of a floatplane- and boat-accessible lodge on the shores of Skilak Lake. Final fire size was 925 acres (374 ha).

The Irish Channel Fire burned within view of a heavily used recreation road system. Photo: Paul Slenkamp, FWS, 2005.

The Fox Creek Fire, discovered the evening of July 11 by detection aircraft, was 392 acres (159 ha) at size-up and actively burning parallel to 73,000-acre Tustumena Lake. The weather on the Kenai Peninsula had been hot and dry, and the fire was burning by passive crowning in stands of black spruce and beetle-killed white spruce. Although the fire was within designated wilderness, the smoke column was in plain view of the town of Soldotna, which has a year-round population of about 4,000 and twice that during busy summer weekends, when recreationists arrive from Anchorage.

The Fox Creek Fire smoke column was consistently visible from central Soldotna. Photo: Jim Hall, FWS, 2005.

Smoke from the Fox Creek Fire was also visible within the communities of Kasilof, Clam Gulch, and Ninilchik. *Suppression action was taken only to protect specific values at risk, such as the Caribou Hills Recreation Area directly west of the fire, which contained over 200 structures with no road access*.

Because of the fire's potential to grow and threaten structures in the Caribou Hills, the Kenai National Wildlife Refuge and Kenai-Kodiak Area Forestry decided to order a "short" Alaska type 2 incident

*The wildland fires described in this article were managed under the 2003 "Interagency Strategy for the Implementation of Federal Wildland Fire Management Policy." The 2009 "Guidance for Implementation of Federal Wildland Fire Management Policy" replaces that strategy and no longer uses the terms "wildland fire use," "fire use incident," or "fire use manager" to describe naturally ignited fires managed for resource benefits. Terminology from 2003 policy was retained in this article to provide an accurate description of how these specific fires were managed.

Lessons Learned From The Fox Creek and Irish Channel Fires:

1. The fire use manager for the two fires worked as a liaison between the suppression service provider (Alaska Department of Natural Resources–Division of Forestry, Kenai-Kodiak area) and the land manager (Kenai National Wildlife Refuge) to revalidate the WFIP daily. This allowed both the suppression service provider and the refuge manager to be involved in the WFIP process, alleviating understandable anxiety about an unfamiliar process.

2. The incident commander for the Fox Creek Fire, the suppression fire management officer, and the refuge manager gathered around a fire area map showing vegetation, land management boundaries, and the latest fire perimeter. They collaboratively drew a maximum manageable area, which proved to be a good choice and remained intact for the duration of the fire.

3. The type 2 team provided successful management of the fire under a wildland fire use strategy, and, when they transitioned to a type 3 organization, the team ensured that the refuge manager and the type 3 incident commander agreed on a plan of action and organization.

4. Managing the impact of smoke on nearby communities was a constant challenge. Besides being visible to local Kenai Peninsula communities, a wind shift blew smoke into Anchorage (population of about 270,000). Managers and incident commanders on the Fox Creek and Irish Channel fires documented their work and followed the guidelines in the "Smoke Effects Mitigation and Public Health Protection Proposal" (see Alaska Wildland Fire Coordinating Group 2007), which the Alaska Wildland Fire Coordinating Group prepared in response to public concerns about smoke impacts from the record-breaking 2004 fire season.

5. It was important to have wildland fire use messages prepared and ready for use by incident information officers and staff who were not familiar with management of fires for resource benefits. A temporary staff answered a bank of phones so that information could be clearly and consistently communicated to the public.

6. Aerial resources were critical to success. The two Canadair CL-215 air tankers proved invaluable during the successful burnout operations. With the fire in such close proximity to a large lake, these "scooper" planes could make quick turnarounds, providing wetline and spot fire support as the burnout progressed. Maintaining scarce aerial resources while multiple suppression fires were active throughout the State was a constant challenge.

management team to help manage the wildfire. A fire use manager was already on site. The Fox Creek Fire spread extremely quickly through one of the largest contiguous fuel beds on the Kenai Peninsula—about 125,000 acres (50,600 ha) of beetle-killed white spruce and live, highly flammable black spruce.

While the Kenai National Wildlife Refuge and Kenai-Kodiak Area Forestry were transitioning with the type 2 team, the fire progressed quickly to a stage 3 WFIP analysis level. Within a few days, the fire grew to 25,189 acres (10,194 ha) with about 150 people performing suppression, support, and monitoring. The final fire size was 26,300 acres (10,640 ha), the largest wildfire on the Kenai Peninsula since 1969.

Using Wildfire To Manage Resources in Remote Alaska

Although many NPS fire management units in Alaska comprise extensive and remote tracts of fire-dependent ecosystems, values at risk dot the landscape. For instance, there are about 325 known cultural resources in Denali National Park and Preserve, but cultural resource inventories are incomplete, and this number represents only a small fraction of the total sites. In 2005, Denali National Park and Preserve sustained five naturally ignited wildfires that were used to benefit natural resources, totaling 118,034 acres (47,767 ha). To varying degrees, each of those wildfires threatened a value at risk.

Thunderstorms ignited three wildfire sites on June 16 in the remote northwestern portion of Denali National Park and Preserve.

The NPS Western Area Fire Management officer (a fire use manager type 2) managed the fires with support from a staff of six.

Denali National Park and Preserve wildland fire use and Denali Mountain. Photo: NPS Western Area Fire Management Staff, 2005.

Over several days, the McKinley River wildland fire use fire grew to 112 acres (45 ha). While completing "Wildland Fire Relative Risk Assessment, Step 1: Determining Values" from the McKinley River wildland fire use WFIP, the fire management officer determined that the McKinley River to the west and the Kantishna River to the north were sufficient natural barriers to prevent the fire from entering the full management option area (native allotments) around Lake Chilchukabena. However, the historic town site of Roosevelt, a cultural resource with several structures that needed protection, was located roughly 10 miles northeast of the fire. The park had proposed restorative stabilization plans for the structures and did not want to lose them.

To lessen the wildland fire threat to the historic site, Western Area Fire Management staff flew by helicopter to Roosevelt, brushed out thick alders, willows, and spruce, and created defensible space around the numerous structures. Sprinklers and hoses were used to wet down the area. The McKinley River wildfire was declared out on July 12 and never advanced towards Roosevelt.

Western Area Fire Management not only managed wildfires for natural resources in Denali National Park and Preserve but also in Noatak National Preserve. Four wildland fire use fires, totaling 17,945 acres (7,262 ha), occurred in the national preserve. The largest, the Goiter Fire, totaled about 8,000 acres (3,200 ha). Because of the remote nature of the fire and the fact that no values were threatened, the fire remained at a stage 1 WFIP analysis level and was monitored through aerial surveillance by the Bureau of Land Management Alaska Fire Service every few days.

Aerial view of Roosevelt following defensible space treatment. Photo: NPS Western Area Fire Management staff, 2005.

The Noatak National Preserve, located north of the Brooks Range, is characterized by immense sweeps of tundra strewn with ponds and marshes. The northernmost reaches of spruce forest that exist in the far west region of the preserve constitute less than 1 percent of the total vegetative cover of the preserve. Major portions of Noatak National Preserve are within the northernmost lightning belt of interior Alaska, where fire plays a critical role in ecosystem sustainability.

Periodic tundra and boreal forest fires act as a mechanism to select plants and animals that are adapted to fire-caused change. Without fire, organic matter accumulates, the permafrost table rises, and ecosystem productivity declines; vegetation communities become less diverse, and their value as wildlife habitat decreases. Fire rejuvenates these subarctic and arctic systems: it removes some of the insulating matter and elicits a warming of the soil; vegetative regrowth quickly occurs, and the cycle begins again. Wildland fire is a key environmental factor on the Noatak National Preserve, an appropriate area for using wildfires as a natural ecological process.

Conclusion

Managing naturally ignited wildfires specifically for natural resource benefits allows land managers to maintain the important role of fire across the Alaskan landscape even as they protect values at risk—whether homes at the wildland-urban interface adjacent to wilderness areas, a remote residence, or a historically significant cultural site within a national park and preserve. Using wildfires as an ecological process will promote fire permeability and will help maintain the character of the landscape while accommodating values and resource use.

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Fire Effects Information System: New Engine, Remodeled Interior, Added Options

Jane Kapler Smith

Some of today's firefighters weren't even born when the Fire Effects Information System (FEIS) (Web site <http://www.fs.fed.us/database/feis>) "hit the streets" in 1986. Managers might remember using a dial-up connection in the early 1990s to access information on biology, ecology, and fire offered by FEIS.

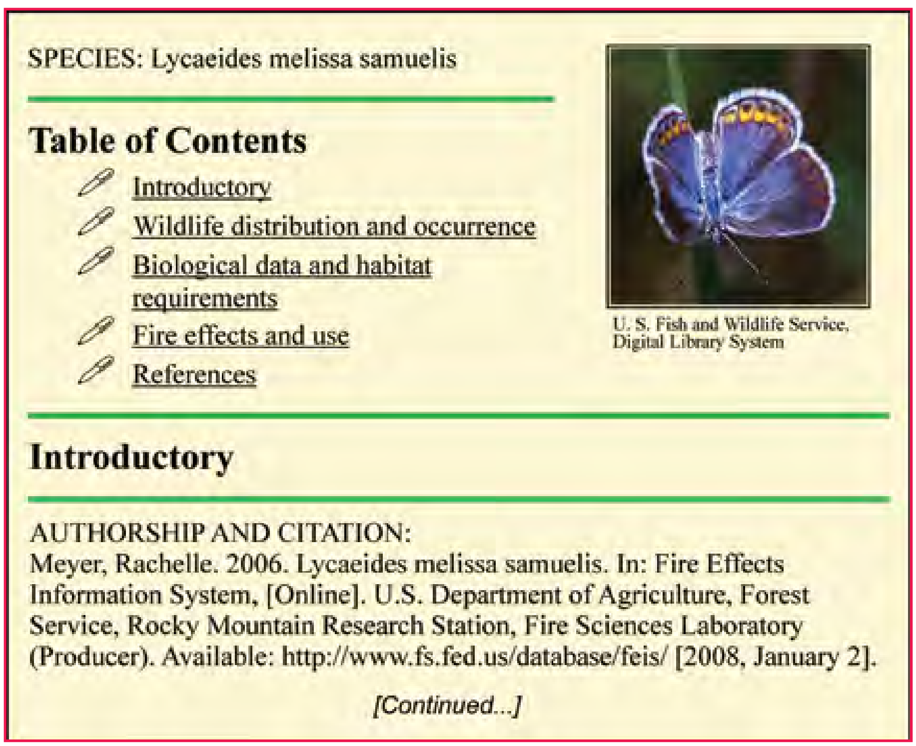
For more than 20 years, FEIS has synthesized scientific information on fire ecology and fire effects for managers. The resulting "species reviews" describe patterns in research results, point out conflicting results and possible reasons for disagreement, identify knowledge gaps, and provide thorough documentation and a complete bibliography. Species reviews cover the available knowledge on fire-related questions such as:

- Will changes in abundance after fire be short lived or long term?
- Will increased productivity provide food essential for wildlife?
- Will increases in one species interfere with regeneration of others?
- Is rejuvenation by fire the only way to ensure long-term species presence?

FEIS reviews also offer extensive information on biology and ecology that can help readers make inferences about responses to fire.

FEIS reviews also offer extensive biological and ecological information that can help readers make inferences about responses to fire. For example, the review of rush skeletonweed, an invasive forb, reports successful sprouting from deep rhizomes after injury, so the review infers that it may be able to recover after a fire, possibly even a severe one, by sprouting (Zouhar 2003).

The usefulness of FEIS is not limited to fire. Because reviews give thorough descriptions of species


SPECIES: Lycaeides melissa samuelis

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U. S. Fish and Wildlife Service, Digital Library System

Introductory

AUTHORSHIP AND CITATION:
Meyer, Rachelle. 2006. Lycaeides melissa samuelis. In: Fire Effects Information System, [Online]. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (Producer). Available: http://www.fs.fed.us/database/feis/ [2008, January 2].

[Continued...]

Figure 1—*Opening page of species review in Fire Effects Information System showing table of contents (top) and citation (bottom). This review (Meyer 2006) contains nearly 20 pages of information and 76 citations.*

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FEIS Tips

If you locate a species review through the FEIS search window, your first screen shows mainly the citation and taxonomic information. You'll want the complete review, so *click on any link in the table of contents* before downloading.

Don't limit your use of FEIS to the *Fire Ecology* and *Fire Effects* sections of a review. Many facts reported in *Botanical and Ecological Characteristics* pertain directly to management issues. Examples include vegetative regeneration, response to nonfire disturbance, seedbed and establishment requirements, and successional patterns.

Go online to get the best that FEIS has to offer. Recycle those ancient printouts in your file cabinet. Since 2000, more than 100 new reviews have been added to the system, more than 150 old ones have been rewritten, and small changes have been made in at least 250 reviews. This means nearly 50 percent of the database has been improved in the past 7 years—and more improvements are coming.

If you use FEIS for environmental planning documents, cite individual species reviews rather than the entire database. Each review has its own date and author; so, when you cite reviews individually, you tell readers exactly what information you used and how current it is.

biology and ecology, including regeneration and succession, they can be used for land use planning, restoration and rehabilitation planning, wildlife and range projects, and related environmental assessments. A person who is unfamiliar with a particular geographic region can use FEIS to get a quick orientation to the ecology of dominant species.

> Fires that are used to protect, enhance, or maintain resources are managed with the expectation that they will be of long duration.

While the fundamental purpose of FEIS is unchanged, the content and technology have advanced since its establishment. FEIS moved from the now-retired Data General* computer to the Internet in 1996. Additions, corrections, and revisions have been continuous, guided by input from a 20-member advisory committee and supported by the Forest Service Office of Fire and Aviation Management, the Joint Fire Science Program, the National Wildfire Coordinating Group, and the U.S. Department of the Interior. Other contributors include the National Forest System and individual agencies in the U.S. Department of the Interior, including the Bureau of Land Management, the National Park Service, and the U.S. Fish and Wildlife Service.

FEIS now contains reviews of more than 1,100 plant and animal species and subspecies, native and nonnative. The system is nationwide in scope, covering hundreds of species in every region of the United States. Nearly one-half of all fire-related environmental impact statements prepared by Federal wildland managers now cite FEIS. Recent changes that can help managers and fire specialists are discussed below.

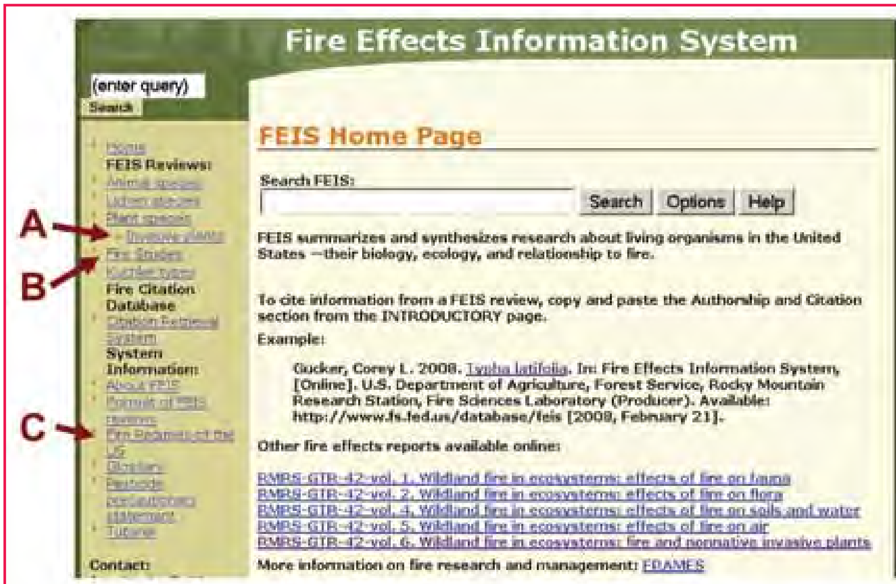

Figure 2—*Homepage of Fire Effects Information System shows (A) link to information on invasive species; (B) list of fire studies in FEIS, including research project summaries, fire case studies (located within species reviews), and downloadable research papers; and (C) link to list of fire regimes for the United States.*

Excerpt from Research Project Summary (Gucker 2005) describing effects of prescribed fire on graminoids in a rough fescue prairie. (The RPS includes a separate table describing fire effects on 19 forb and 3 shrub species.)*

Percent cover of graminoids species at the end of the second growing season after prescribed fire (Archibold and others 2003)				
Common Name	Unburned	Spring	Summer	Fall
Grasses				
thickspike wheatgrass	0.1	0	0.1	0.1
slender wheatgrass	1.3	0.5	0.1	0.3
rough fescue	11.3	13.2	7	8.8
spikeoat	0	0	0	0.1
porcupine grass	5.6	4.9	3	2.2
prairie Junegrass	0	0.2	0	0.1
green needlegrass	0.5	0.2	0.7	1.5
western wheatgrass	0	0	0.1	0.2
Kentucky bluegrass	6.8	0.2	1.3	5.4
Sedges				
needleleaf sedge	0.4	0.1	0.3	0.7
sun sedge	1.4	2.6	3.2	3.7
obtuse sedge	1	1	0	0

*Yellow identifies species that are cross-linked with FEIS reviews. Blue identifies species *not* reviewed in FEIS; a search on these species in FEIS retrieves the research project summary.

New Engine

FEIS users sometimes stalled out in the database's file structure before finding needed information on ecology and fire. Now, the system is rebuilt so that every review starts with a table of contents and links to all sections in order (fig. 1). This organization allows readers to quickly access topics of interest.

Remodeled Interior

Reviews covering 60 nonnative invasive plant species and subspecies were revised or added to FEIS between 2001 and 2006. A list of all invasives covered in FEIS (more than 100 species) is available through the homepage (fig. 2A).

The FEIS team recently completed a project that began in 2004 to update 100 FEIS species reviews and add reviews covering 100 additional species. Updates include:

- Rewritten reviews on the spotted owl, Table Mountain and pitch pines, several western oaks, and Jeffrey pine, all originally written in the late 1980s and early 1990s;
- New reviews on bear huckleberry, bog birch, and several cacti, lichens, and mosses;
- New reviews on the great gray owl, Indiana bat, eastern box turtle, red-headed woodpecker, fisher, and black-tailed prairie dog; and
- A review of the first insect species in FEIS, the Karner blue butterfly (fig. 1) and its obligatory forage species, the wild lupine.

FEIS reviews describe the fire regimes thought to have influenced the species in past centuries. When FEIS was established, reviews addressed fire regimes only for dominant species. At the request of managers, FEIS began in 2000 to report historic fire intervals for the habitat of each species reviewed. These reports were initially organized by plant community but not linked to a comprehensive national classification. Reviews completed since mid-2007 include new, more complete fire regime descriptions for a comprehensive list of vegetation types (fig. 2C). These descriptions were developed from data collected for the LANDFIRE Rapid Assessment (2007) and will be updated when the National LANDFIRE Mapping Project is complete.

Added Options

In 2006, FEIS began to provide a new kind of review, the research project summary (RPS). An RPS summarizes research on preburn vegetation, fire weather, fire behavior, and fire effects. It summarizes fire effects on *all* species covered by the study and is linked to—and from—every relevant species review in FEIS. For example, an RPS that describes fire effects on plants in a rough fescue prairie (Gucker 2005, summarizing information from Archibold and others 2003) provides information on nine species reviewed in FEIS and an additional three "non-FEIS" species (see table). An RPS describing restoration treatments in ponderosa pine-Douglas-fir forests (Metlen and others 2006) describes fire effects on 76 FEIS species and 121 non-FEIS species.

How can readers find an RPS? In several ways:

1. From within species reviews. The "fire effects" section links to every relevant RPS.
2. Through the FEIS search engine. When FEIS is searched by species name, it produces a list containing the species review (if there is one) and all relevant RPSs. The search engine also locates RPSs for species not reviewed in FEIS. For instance, Virginia strawberry is not reviewed in FEIS, but a search on this species retrieves five RPSs, each containing a little information on the species' response to fire.
3. From the FEIS list of fire studies, available through the homepage (fig. 2B). This list can be searched for a location, species, or plant community of interest. The list includes not only RPSs but also fire case studies (embedded within FEIS reviews) and downloadable research papers linked from FEIS reviews.

FEIS has served wildland fire managers for more than 20 years and continues to adapt and respond to managers' needs and requests. Please send your comments, suggestions, and corrections to <fmi@fs.fed.us>.

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Representation of Ecological Systems within the Protected Areas Network of the Continental United States

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Abstract

If conservation of biodiversity is the goal, then the protected areas network of the continental US may be one of our best conservation tools for safeguarding ecological systems (i.e., vegetation communities). We evaluated representation of ecological systems in the current protected areas network and found insufficient representation at three vegetation community levels within lower elevations and moderate to high productivity soils. We used national-level data for ecological systems and a protected areas database to explore alternative ways we might be able to increase representation of ecological systems within the continental US. By following one or more of these alternatives it may be possible to increase the representation of ecological systems in the protected areas network both quantitatively (from 10% up to 39%) and geographically and come closer to meeting the suggested Convention on Biological Diversity target of 17% for terrestrial areas. We used the Landscape Conservation Cooperative framework for regional analysis and found that increased conservation on some private and public lands may be important to the conservation of ecological systems in Western US, while increased public-private partnerships may be important in the conservation of ecological systems in Eastern US. We have not assessed the pros and cons of following the national or regional alternatives, but rather present them as possibilities that may be considered and evaluated as decisions are made to increase the representation of ecological systems in the protected areas network across their range of ecological, geographical, and geophysical occurrence in the continental US into the future.

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Introduction

Traditionally, a mix of opportunity, available resources, and agency-specific conservation priorities are the foundation upon which networks of protected areas are developed over time [1–4]. This has led to a protected areas network in the continental US cultivated for multiple purposes including protecting biological resources, such as vegetation communities [5–8]. Often, to respond to conservation issues, such as habitat loss, the protected areas network is expanded by establishing new protected areas or enlarging existing ones [9–13]. However, with increasing land-use intensification the opportunities for expanding such networks are dwindling [4,14]. Furthermore, with the imminence of climate change along with increased loss and fragmentation of vegetation communities, the exigency of protecting areas that represent the full suite of vegetation communities and therefore the species found therein, has increased [15–17].

The conservation community has increasingly focused on landscape levels for national decision making, but the lack of relevant and consistent data at a national scale has been an impediment [18–20]. Most public land management agencies, even those with the broadest authorities to protect natural resources have yet to implement ecosystem-scale approaches, perhaps due to lack of relevant data [21,22]. However, the impediment that once prevented a national-scale approach to protected areas management in the continental US has recently been overcome with the availability of national-level data for vegetation communities, classified to ecological systems [23], and a protected areas database for the US [24]. Ecological systems are groups of vegetation communities that occur together within similar physical environments and are influenced by similar ecological processes (e.g., fire or flooding), substrates (e.g., peatlands), and environmental gradients (e.g., montane, alpine or subalpine zones) [23,25]. Ecological systems represent vegetation communities with spatial scales of tens to thousands of hectares and temporal scales of 50–100 years. They represent the habitat upon which vertebrate species rely for survival. The Protected Areas Database of the US (PAD-US) represents public land ownership and conservation lands (e.g., federal and state lands), including privately protected areas that are voluntarily provided (e.g. The Nature Conservancy) [24]. Each land parcel within PAD-US is assigned a protection status that denotes both the intended level of biodiversity protection and indicates other natural, recreational and cultural uses (Table 1) [24]. Together, these databases provide the foundation for assessing the representation of vegetation communities in the continental US within the protected areas network and thereby informing decision making at the national level.

The protected areas network within the continental US is often viewed as one of our best conservation tools for securing vegetation communities and the species they support into the future [26–29]. An inherent assumption behind a network of protected areas is that protection of vegetation communities will also protect the species that rely on them, including invertebrate and vertebrate species, many of which little is known of their life history or habitat requirements [11,30,31]. For our analysis, we narrowly defined a protected area as an area of land having permanent protection from conversion of natural land cover and a mandated management plan in operation to maintain a natural state within which disturbance events may or may not be allowed to proceed without interference and/or be mimicked through management (Table 1) [24]. Furthermore, we defined a protected areas network as a system of protected areas that increase the effectiveness of *in situ* biodiversity conservation [32]. Lastly, we defined biodiversity as a hierarchy from genes to communities encompassing the interdependent structural, functional, and compositional aspects of nature [33].

The questions of how much of a vegetation community to protect and what approach is best for systematically protecting vegetation communities have been discussed at length [34,35]. No single solution or specific amount of area has been established to meet both policy targets and biological conservation needs [35]. Most recently the Convention on Biological Diversity set a target of 17% for terrestrial areas in the Aichi Biodiversity Targets described within the Strategic Plan 2011–2020 [36]. The Aichi Biodiversity Targets also attempt to address biological needs by stating that areas protected should be ecologically representative [36]. Representation of vegetation communities is often put forth as a goal of conservation planning because the aim is to protect something of everything in order to conserve the evolutionary potential of the entire protected areas network [34,37,38]. The US has not explicitly addressed the representation of vegetation communities within the protected areas network; however, Canada has used representation targets to structure their protected areas network [39–41]. Even though climate change will likely alter what is represented within Canada's protected areas network, starting from a representative group of protected vegetation communities provides a foundation for climate change adaptation [40,41].

Numerous assessments of the US protected areas network and its effectiveness at conserving vegetation communities have all concluded the network is falling short [15,20,42–48]. Each assessment used the best data available at the time, but in all cases, extent, resolution, and consistency of the data were limited. Shelford [42] conducted the first assessment of protected areas in the US in 1926. His aim was to study the native biota of North America, which started with inventorying the existing protected areas and how their vegetation communities had been modified from pre-settlement conditions. Later, Scott et al. [15] found that 302 of 499 (~60%) mapped vegetation communities within the US had <10% representation within protected areas. Dietz and Czech [20] found the median percentage of area protected within the continental US was 4% for the ecological analysis units they defined.

We recently have had the opportunity to evaluate the representation (i.e., saving some of everything) and redundancy (i.e., saving more than one of everything) of ecological systems within the existing protected areas network for the continental US. This opportunity was possible because of the availability of a complete ecological systems database for the continental US and a comprehensive database of the current protected areas network. Hence, we can now assess how well the protected areas network

Table 1. Description of protection status categories in the Protected Areas Database for US [24].

Protection status	Description	Example
Lands managed to maintain biodiversity (i.e., protected areas network)	An area of land having permanent protection from conversion of natural land cover and a mandated management plan in operation to maintain a natural state within which disturbance events may or may not be allowed to proceed without interference and/or be mimicked through management.	Yellowstone National Park, Wyoming
Lands managed for multiple-use, including conservation	An area having permanent protection from conversion of natural land cover for the majority of the area, but subject to extractive uses of either a broad low-intensity type (e.g., logging) or localized intense type (e.g., mining). Protection of federally listed endangered and threatened species throughout the area may be conferred.	Kaibab National Forest, Arizona
Lands with no permanent protection from conversion, but may be managed for conservation	An area with no known public or private institution mandates or legally recognized easements or deed restrictions held by the managing entity to prevent conversion of natural habitats to anthropogenic habitat types. Conversion to unnatural land cover throughout is generally allowed and management intent is unknown.	Fort Irwin, California

Protection status denotes the intended level of biodiversity protection and indicates other natural, recreational, and cultural uses. These designations emphasize the managing entity rather than the land owner because the focus is on long-term management intent. Therefore an area gets a designation of permanently protected because that is the long-term management intent.

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encompasses the ecological and evolutionary patterns and processes that maintain ecological systems and thereby the species that depend on them [37]. Additionally, based on the Aichi Biodiversity Targets within the Strategic Plan 2011–2020 of the Convention on Biological Diversity, we can evaluate the current protected areas network in the continental US in context of meeting the suggested 17% target for terrestrial areas [36].

If the current protected areas network is falling short of conserving vegetation communities then what potential alternatives might be available to address those shortfalls? One such alternative is to replace protected areas that contribute minimally to conservation of vegetation communities with those with greater conservation value [49]. The goal would be to increase the overall biodiversity protection of the entire protected areas network. This approach proposed by Fuller et al. [49] could be attractive because the sale of protected areas with less conservation value could go towards acquiring new ones. Fuller et al. [49] proposed this approach in Australia where a protected areas network has been systematically designed with broad representation of Australia's vegetation types [49]. The protected areas network in the continental US has not been systematically designed [2,4]. Would this approach be feasible if the criteria for determining the contribution to conservation (i.e., cost-effectiveness analysis) could be agreed upon consistently across the continental US?

Another alternative to address the current protected areas network's shortfall could be to expand the network in area and number of protected areas [9,11,13]. A national assessment would be needed to identify vegetation communities not represented or under-represented within the existing protected areas network and a national conservation plan would be developed to prioritize acquisition of these vegetation communities to increase their representation on protected lands [50,51]. There are approximately 300 million hectares of public and private lands with no permanent protection on which native vegetation communities occur [23,24]. Could the representation of vegetation communities within the protected areas network be increased by prioritizing acquisition within these lands with no permanent protection?

A third alternative for addressing the protected areas network's shortcomings might be to increase the emphasis of maintaining biodiversity on some public and private lands currently managed for multiple-use (Table 1). Swaty et al. [52] found that in addition to the 29% of the continental US land area that has been converted by human use; there were an additional 23% of non-converted lands with altered vegetation structure and composition, which likely are lands managed for multiple-use. The protected areas network is comprised of approximately 50 million hectares in the continental US, while there are about 140 million hectares of public and private lands managed for multiple-use [24]. Vegetation communities that are currently not represented or underrepresented within the current protected areas network may have representation on the approximately 140 million hectares of land managed for multiple-use [20,24]. Could, therefore, an emphasis on maintaining biodiversity on a strategically targeted subset of lands managed for multiple-use be used to effectively expand the representation of vegetation communities within the entire protected areas network?

From a conservation management perspective for the US, the Department of Interior (DOI) has established a framework of Landscape Conservation Cooperatives (LCC) with the mission of landscape-level planning and management [53]. This national framework further supports the need for nationally consistent databases and analyses. We focused our analysis on alternative ways to potentially increase the representation of ecological systems in the protected areas network of the continental US. Specifically we asked (1) how well are ecological systems represented in the protected areas network relative to their occurrence in the continental US, including with regards to soil productivity and elevation, (2) how alternative approaches may potentially increase the representation of ecological systems in the protected areas network, and (3) how Landscape Conservation Cooperatives (LCC), the new landscape unit for conservation initiatives, can be used to regionally assess conservation status of ecological systems.

Materials and Methods

Data Description

We used the National Gap Analysis Program (GAP) Land Cover [23] and US Geological Survey GAP's (USGS-GAP) Protected Areas Database of the US (PAD-US 1.0) [24] as the national datasets for our analyses. The land cover data contains 3 nested hierarchical levels of vegetation communities. Level I contains 8 groupings, based on generalized vegetative physiognomy (e.g., grassland, shrubland, forest), while Level II has 43 groupings representing general groups of ecological systems based on physiognomy and abiotic factors (e.g., lowland grassland and prairie, alpine sparse and barren). The third hierarchical level contains 551 map classes, including 518 ecological systems. We focused on the non-modified, non-aquatic classes at each level (Level I: 5 classes, Level II: 37 classes, and Level III: 518 ecological systems).

The National GAP Land Cover was compiled from the Southwest, Southeast, Northwest, and California GAP land cover data completed during 2004–2009 [23]. We incorporated data from LANDFIRE (www.landfire.gov) for the Midwest and Northeast. These national land cover data were based on consistent satellite imagery (Landsat Thematic Mapper (TM) and Enhanced Thematic Mapper (ETM)) acquired between 1999 and 2001 in conjunction with digital elevation model (DEM) derived datasets (e.g., elevation, landform) and a common classification system (i.e., ecological systems) to model natural and semi-natural vegetation [54–56]. The resolution is 30-m and typically the minimum mapping unit is 1 ha. Regional accuracy assessments and validations have been conducted and, based on those, in general, forest and some shrub ecological systems typically had higher accuracies than rare and small patch ecological systems, such as wetlands [57,58].

PAD-US (Version 1.0) consists of federal, state, and voluntarily provided privately protected area boundaries and information including ownership, management, and protection status [24]. Protection status is assigned to denote the intended level of biodiversity protection and indicate other natural, recreational, and cultural uses (Table 1) [24]. In assigning protection status, the emphasis is on the managing entity rather than the owner and focuses on long-term management intent instead of short-term processes [11]. The criteria for assigning protection status includes perceived permanence of biodiversity protection, amount of area protected with a 5% allowance of total area for intensive human use, protection of single vs. multiple features, and the type of management and degree to which it is mandated [59]. The protection status ranges from lands managed to maintain biodiversity to lands with little or no biodiversity protection (Table 1). Lands managed for multiple-use, including conservation, are permanently protected, but allow for extractive uses, such as mining and logging. In the continental US, lands with no permanent protection are considered any land parcel not designated either of the other protection status categories. We included only lands permanently protected and managed to maintain biodiversity in our definition of the protected areas network.

We also used elevation data obtained from the National Elevation Dataset (NED) [60] and soil productivity. The National Elevation Dataset, a seamless dataset with a resolution of approximately 30 m, was the best available raster elevation data for the continental US [60]. We divided the National Elevation Dataset into 8 classes ranging from 0 to 4500 meters at 500-meter intervals. Soil productivity classes for the continental US were based on STATSGO data (http://soils.usda.gov/survey/geography/statsgo/). These data were reclassified into 8 soil productivity classes based on land capability classes (http://soils.usda.gov/technical/handbook) and ranged from very high to very low productivity.

To apply our analysis and results to current conservation management in the continental US, we used the LCC framework [53]. LCCs represent large area conservation-science partnerships between DOI and other federal agencies, states, tribes, non-governmental organizations (NGOs), universities, and other public and private stakeholders. Their intent is to inform resource management decisions to address landscape-level stressors, such as land use change, invasive species, and climate change [53].

Data Analysis

The PAD-US 1.0 [24] and LCC data [53] were converted to grids (i.e., 30×30 m cells) and combined with the National GAP Land Cover [23] using ArcGIS 9.3.1 (ESRI, Redlands, CA). To assess the protection of ecological systems relative to their occurrence, we calculated a frequency distribution of protected area sizes within the existing protected areas network. To evaluate how the size range of protected areas would change with the inclusion of land managed for multiple-use, we calculated a frequency distribution of the protected areas network with lands managed for multiple-use added in (Table 1). We also calculated the amount of area of land managed for multiple-use needed to meet the 17% Aichi Biodiversity Target. To assess least protected or most endangered ecosystems, we summarized within each hierarchical level of the National GAP Land Cover (i.e., Levels I, II, and ecological systems) the number, size, protection status, and ownership of land parcels within PAD-US, as well as their distribution among LCCs. At the broadest level (Level I), we calculated percent availability versus percent protected to gain insight into the representation of each system in the protected areas network. We used a comparison index line (i.e., 1:1 line) to indicate the relationship between percent availability and percent protected [61]. Similarly, we calculated the percent area of ecological systems protected (i.e., managed to maintain biodiversity), managed for multiple-use, and not permanently protected for soil productivity and elevation ranges by combining these data with PAD-US [24] using ERDAS Imagine 9.3 (Table 1).

The diversity of ecological systems across and redundancy within LCCs was calculated by counting the number of ecological systems occurring within each LCC. Diversity was defined as the number of ecological systems within each LCC, while redundancy was defined as the number of LCCs in which a single ecological system occurred [37]. For example, if an ecological system occurred in 2 LCCs, its redundancy value was 2. Unique ecological systems were those that occurred in a single LCC. Furthermore, we calculated the number and percent area protected of ecological systems by each protection status within each LCC. To assess whether lands were being protected at the same rate as those converted to human dominated classes, such as developed areas, cultivated croplands, orchards, vineyards, quarries, mines, gravel pits, oil wells, and pastures, we calculated the conservation risk index (CRI) for each LCC by dividing percent area converted by percent area managed to maintain biodiversity or percent area managed to maintain biodiversity and for multiple-use [23,62]. Finally, we summarized CRI values by protection status.

Results

The current protected areas network in the continental US covers approximately 10% of the total area in which ecological systems occur. Across about 30,000 protected areas, the mean size of an individual protected area was 1942 ha with a size range of approximately 25–2,500,000 hectares over all protected areas. The analysis of representation of the network shows that the distribution of ecological systems managed to maintain biodiversity (i.e., the distribution of the protected areas network) is skewed towards high elevation and low productivity soils (Figure 1A). Overall 68% of all 518 ecological systems have <17% of their area protected, which is a target suggested by the Aichi Biodiversity Target of the Convention of Biological Diversity [36] and most of the ecological systems with <17% protected occur at low elevation and in areas with moderate to high productivity soils (Figures 1B and 1C, Table S1).

In examining the percent available versus percent protected for lands managed to maintain biodiversity, only two of the five Level I land cover groups (sparse and barren; riparian and wetland) occurred above the 1:1 line indicating a greater percentage of these groups are protected in relation to their availability (Figure 2). Representation of Level II land cover groups was lowest for lowland grassland and prairie (xeric-mesic), but most groups had <17% protected (Figure 3). Out of 37 Level II groups, 11 fell at or above the 17% Aichi Biodiversity Target [36].

Ecological systems on lands managed for multiple-use and on lands with no permanent protection comprised 29% and 61%, respectively, of the total area of the continental US in which ecological systems occur. When lands managed for multiple-use were included as part of the protected areas network, the overall number of protected areas increased to about 88,000 with a size range of approximately 25–117,757,000 hectares.

When both lands managed to maintain biodiversity and for multiple-use were included all five Level I land cover groups occurred above the 1:1 line and all five occurred at or above the suggested 17% Aichi Biodiversity Target (Figure 2) [36]. The largest increases were within the shrubland, steppe, and savanna group, forest and woodland group, and sparse and barren group. The percent area of Level II land cover groups increased for all 37 groups when lands managed for multiple-use were added to lands managed to maintain biodiversity (Figure 3). The largest increases in percent area occurred within the lowland grassland and prairie (xeric-mesic) and sagebrush dominated shrubland. Out of 37 Level II groups, 33 fell at or above the 17% Aichi Biodiversity Target [36] when both lands managed to maintain biodiversity and multiple-use were included (Figure 3).

To meet the suggested 17% Aichi Biodiversity Target [36], approximately 9 million hectares (6.4%) of the 140 million hectares of public and private lands managed for multiple-use or 34 million hectares (11.3%) of the 300 million hectares of lands with no permanent protection would need to emphasize maintaining biodiversity or be acquired as part of the protected areas network (Table S1). Including lands managed for multiple-use with lands managed to maintain biodiversity, 98% of all ecological systems increased their percent area protected (Table S1). Using the suggested 17% Aichi Biodiversity Target [36], we found 32%

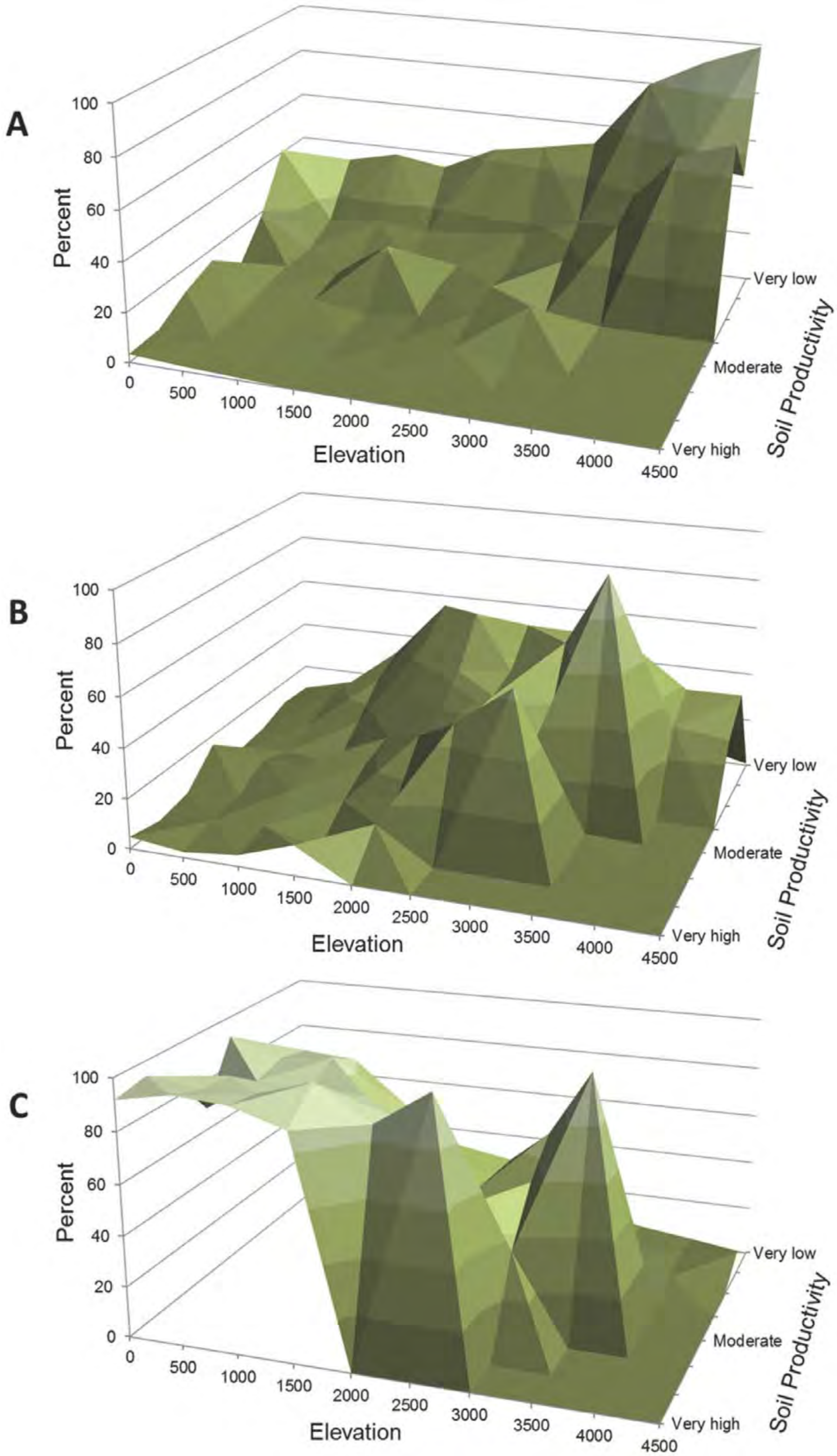

Figure 1. Percent area of ecological systems in relation to elevation, soil productivity, and protection status. Protection status designations include lands managed to maintain biodiversity (A), lands managed for multiple-use (B), and lands that have no permanent protection (C). See Table 1 for protection status descriptions. Percent area of ecological systems determined by combining data for elevation (meters) and soil productivity (http://soils.usda.gov/technical/handbook) with ecological systems grouped by protection status [23,24,60].
doi:10.1371/journal.pone.0054689.g001

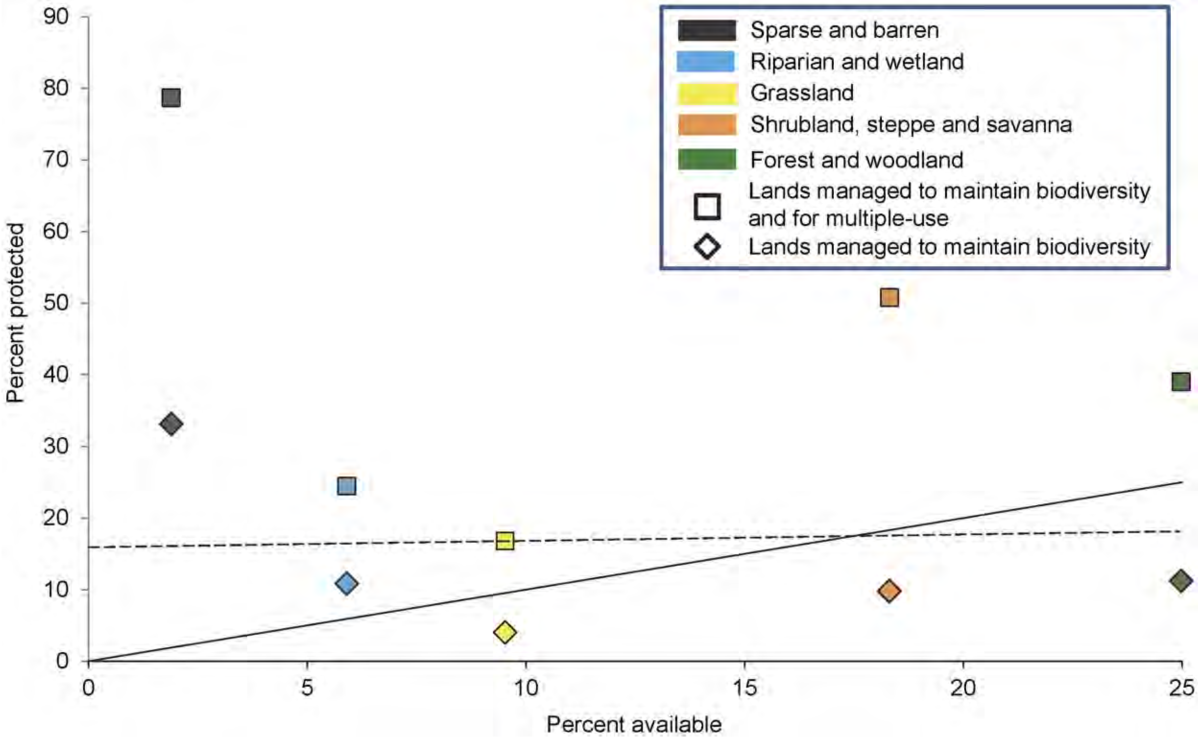

Figure 2. Percent protected and available for each Level I land cover group by protection status. Lands managed to maintain biodiversity (diamonds) are shown relative to lands managed to maintain biodiversity and for multiple-use (squares). See Table 1 for protection status descriptions. A comparison index line is shown, which indicates a 1:1 relation between percent availability and percent protected [61]. A value below the 1:1 line represents a Level I land cover group under-represented in the protected areas network, a value above represents a Level I land cover group well represented in the protected areas network, while a value on the line indicates a Level I land cover group available and protected equally [61]. For example, grassland, a Level I land cover group, has about 4% of its area managed to maintain biodiversity, but that increased to about 17% when lands managed for multiple-use were included [23,24]. A dashed line representing the 17% Aichi Biodiversity Target of the Convention on Biological Diversity is shown [36].
doi:10.1371/journal.pone.0054689.g002

of all ecological systems met that target, but that increased to 68% when lands managed for multiple-use were included (Table S1).

Including lands managed for multiple-use in the protected areas network would result in dramatic geographic changes in the western US, but noticeable changes were also evident in northeastern US, Florida, the Appalachian mountains, and around the Great Lakes (Figure 4). Federal, state, and local governments as well as private entities manage lands to maintain biodiversity and for multiple-use (Figure 5). There are approximately 50 million hectares of lands managed to maintain biodiversity with Bureau of Land Management (BLM) and US Forest Service (USFS) managing about 29 million hectares, which is more than US Fish and Wildlife Service (USFWS), National Park Service (NPS), and all other federal land combined (Figure 5). Approximately 140 million hectares is managed for multiple-use in the continental US with BLM and USFS managing about 100 million hectares (Figure 5, Table S1).

Redundancy values for ecological systems occurring in LCCs ranged from 1–8, with redundancy values higher in LCCs in the west (Figure 6A). Ecological systems were highly diverse in 4 LCCs (Great Northern, Great Basin, Desert, and Gulf Coast Plain and Ozarks); however, only 1 had numerous unique ecological systems (Gulf Coast Plains and Ozarks; Figure 6B and Table 2). When including lands managed for multiple-use in the protected areas network, 7 out of the 16 LCCs in the continental US more than doubled the percent area protected (Table 2). Lands managed to maintain biodiversity represented between 0.6–17.0% of the area of LCCs, adding lands managed for multiple-use increased that to 1.2–62.9% (Table 2). Eight out of 16 LCCs contained ecological systems that occurred only on lands managed for multiple-use or had no permanent protection (e.g., Great Plains, North Atlantic; Figure 7). The CRI values varied across LCCs with the Eastern Tallgrass Prairie and Big Rivers having the highest value (126.4) because almost 80% of its area was converted to human use (i.e., cultivated cropland) and the Desert and Southern Rockies having the lowest (0.2) because >10% of their area contained lands managed to maintain biodiversity (Figure 8). Including lands managed for multiple-use lowered the CRI for all LCCs and increased the number of LCCs meeting the suggested Aichi Biodiversity Target of 17% target from 1 to 7 (Figure 8) [36].

Discussion

Protection of Ecological Systems Relative to their Occurrence in the Continental US

The existing protected areas network in the continental US would need to capture a more representative complement of ecological systems if the US aims to meet the suggested Aichi Biodiversity Target of 17% for ecologically representative terrestrial areas [36]. The 518 ecological systems mapped in the continental US are disproportionately distributed by number, size, and protection status relative to elevation and soil productivity, which translates to an uneven representation of ecological systems within the protected areas network (Figure 1A) [15,63]. Soils with

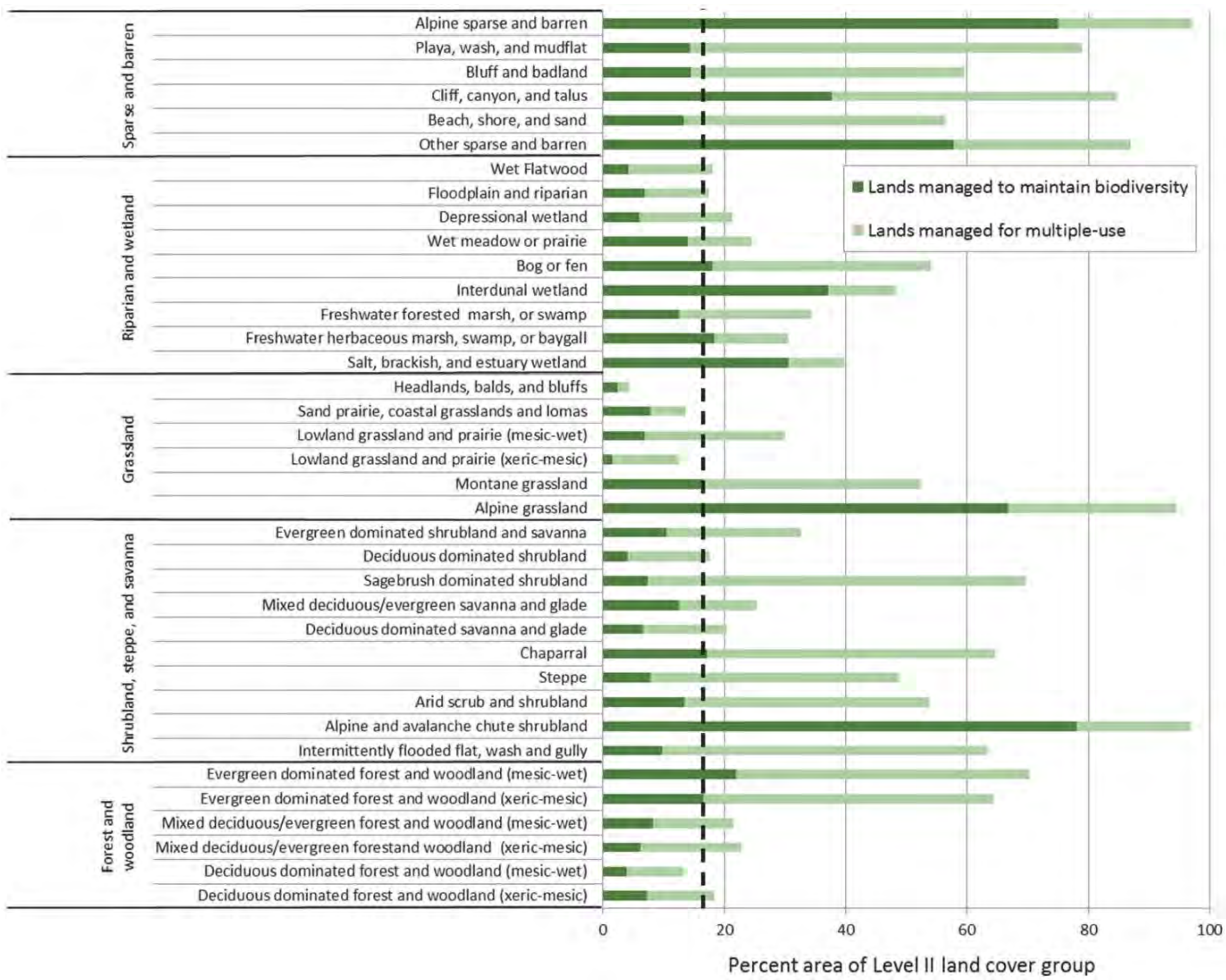

Figure 3. Percent area of Level II land cover groups by protection status. The Level II land cover groups are arranged by Level I land cover groups (see Table S1) [23]. Percent area for both lands managed to maintain biodiversity and lands managed for multiple-use are shown [24]. See Table 1 for protection status descriptions. A dashed line representing the 17% Aichi Biodiversity Target of the Convention on Biological Diversity is shown [36].

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low productivity at high elevation are more likely to be found within the protected areas network; therefore ecological systems that occur in those areas are disproportionally represented in the network. Typically, low soil productivity at high elevations occurs in sparse and barren areas and these areas are well represented within the protected areas network (Figure 2) [15]. Capturing a broader range of elevation could be important to spatial patterns of biodiversity because ecological systems might shift with climate change, but the patterns of biodiversity will likely endure with geophysical features, such as elevation range [64]. How can the representation of ecological systems increase within the protected areas network of the continental US?

Alternatives for Increasing Representation and Conservation of Ecological Systems

Many alternatives exist for conserving ecological systems and successful conservation will likely come from employing one or more of them. One approach, presented earlier in the paper, would be to replace protected areas that are minimally contributing to conservation and have a high cost associated with protecting ecological systems within a specific protected area (i.e., least cost effective) with those having greater conservation value (i.e., more cost effective) to increase the overall biodiversity protection of the entire network [49]. Applying this approach could be challenging because public support for existing protected areas may make it difficult to convince those supporters to relinquish a protected area for the benefit of the entire network [8,65]. This approach, even though controversial because of the concept of giving up protected areas, could play a prominent role in addressing the impacts of climate change because of the potential opportunity to shift the distribution of ecological systems on current protected areas in response to shifts in temperature and precipitation [66,67].

Protected areas have long been downgraded, downsized, delisted, and degazetted and these practices are currently widespread [68,69]. Approximately 60 National Parks have been delisted and downgraded since the establishment of the National Park System in 1916 [68,70,71]. One of the major drivers of protected area degazettement, which is loss of legal protection for

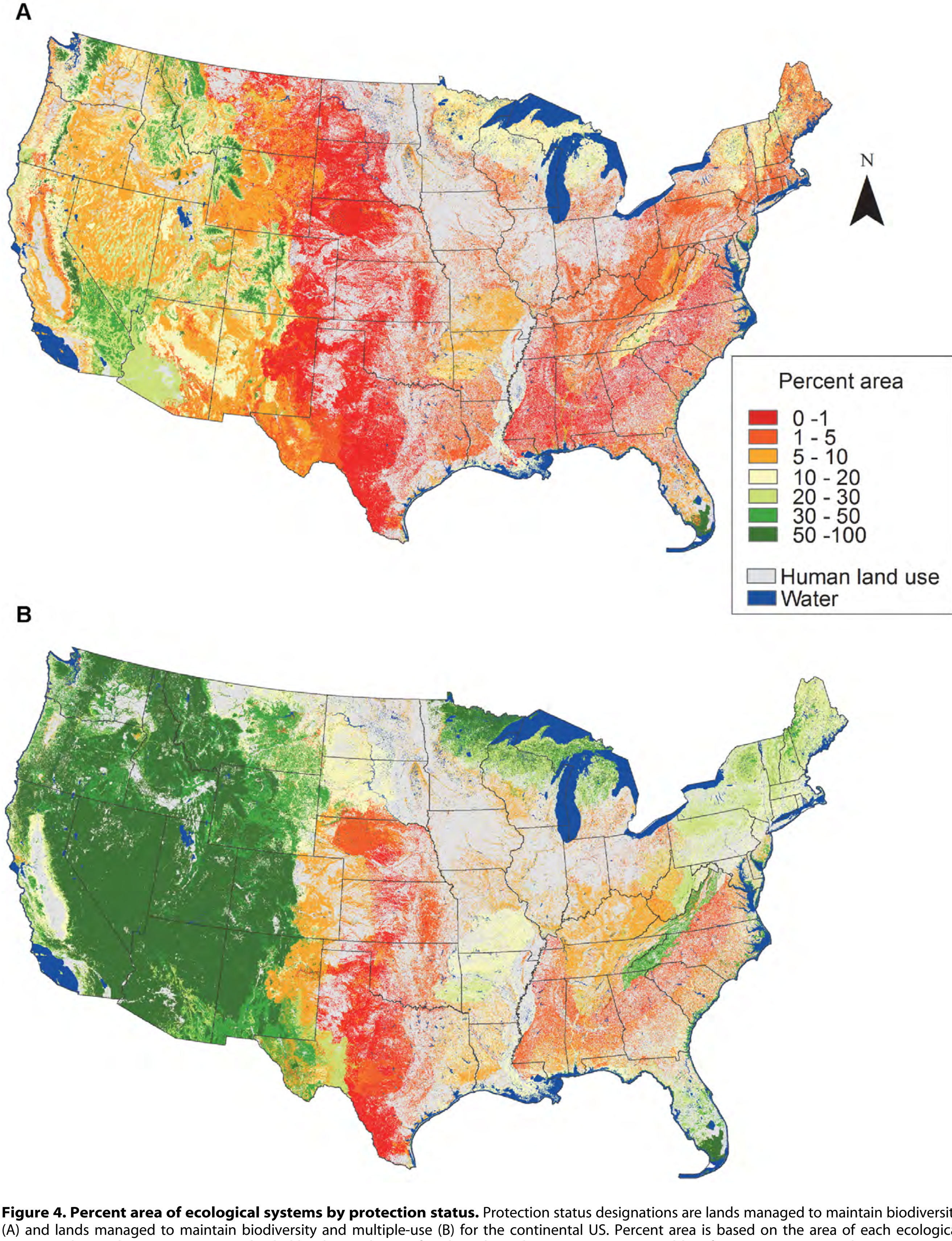

Figure 4. Percent area of ecological systems by protection status. Protection status designations are lands managed to maintain biodiversity (A) and lands managed to maintain biodiversity and multiple-use (B) for the continental US. Percent area is based on the area of each ecological system within each protection status divided by the total area of each ecological system [23,24]. See Table 1 for protection status descriptions. Only non-modified, non-aquatic ecological systems were included (n = 518; Table S1).
doi:10.1371/journal.pone.0054689.g004

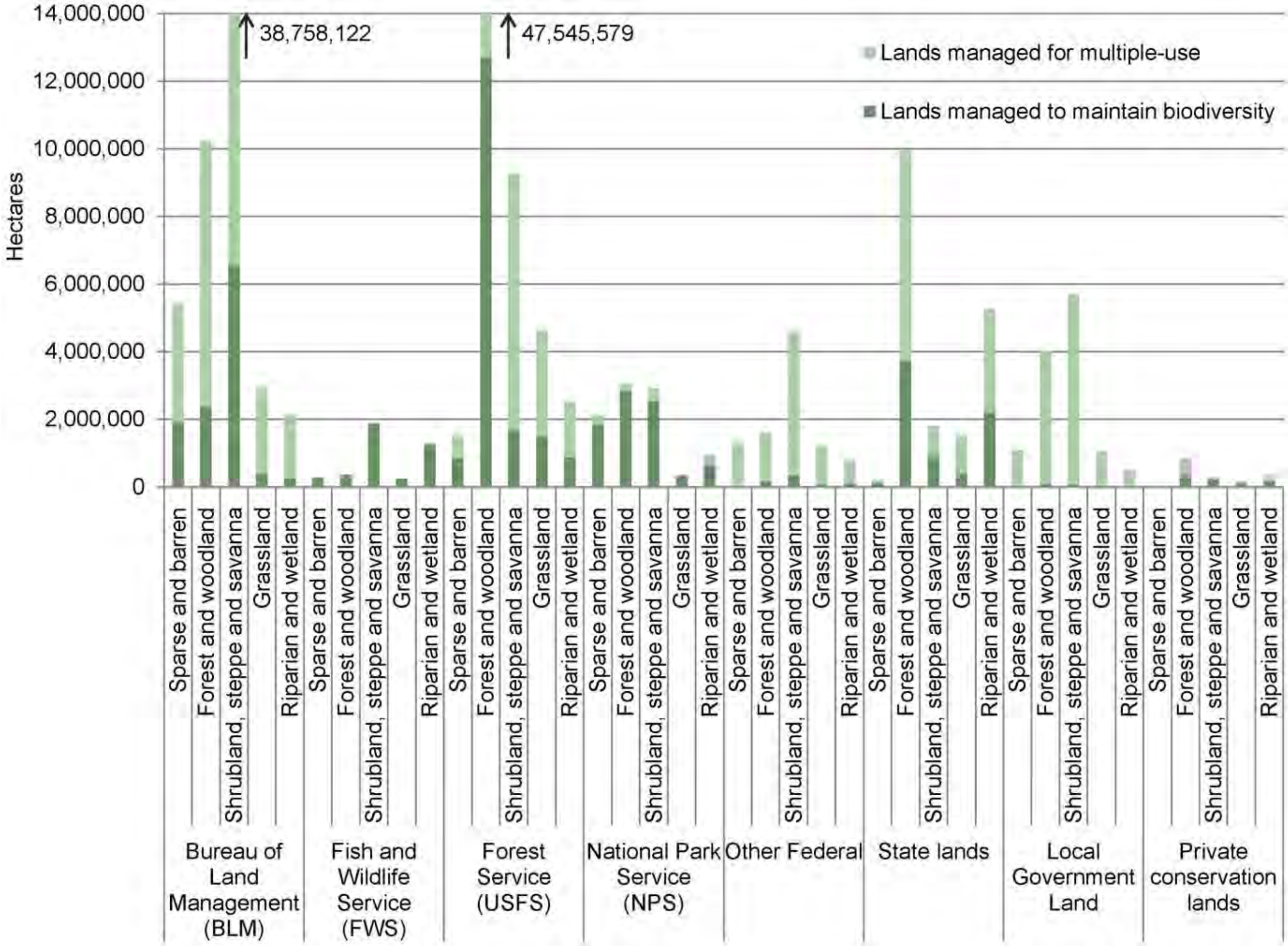

Figure 5. Area (ha) of Level I land cover groups by ownership and protection status. Ownership includes federal, state, and local governments as well as private conservation lands. See Table 1 for protection status descriptions. These values were for the continental US. Both BLM and USFS have areas of Level I land cover groups that fall outside the scale on this graph [23,24]. Values for those Level I land cover groups are shown.
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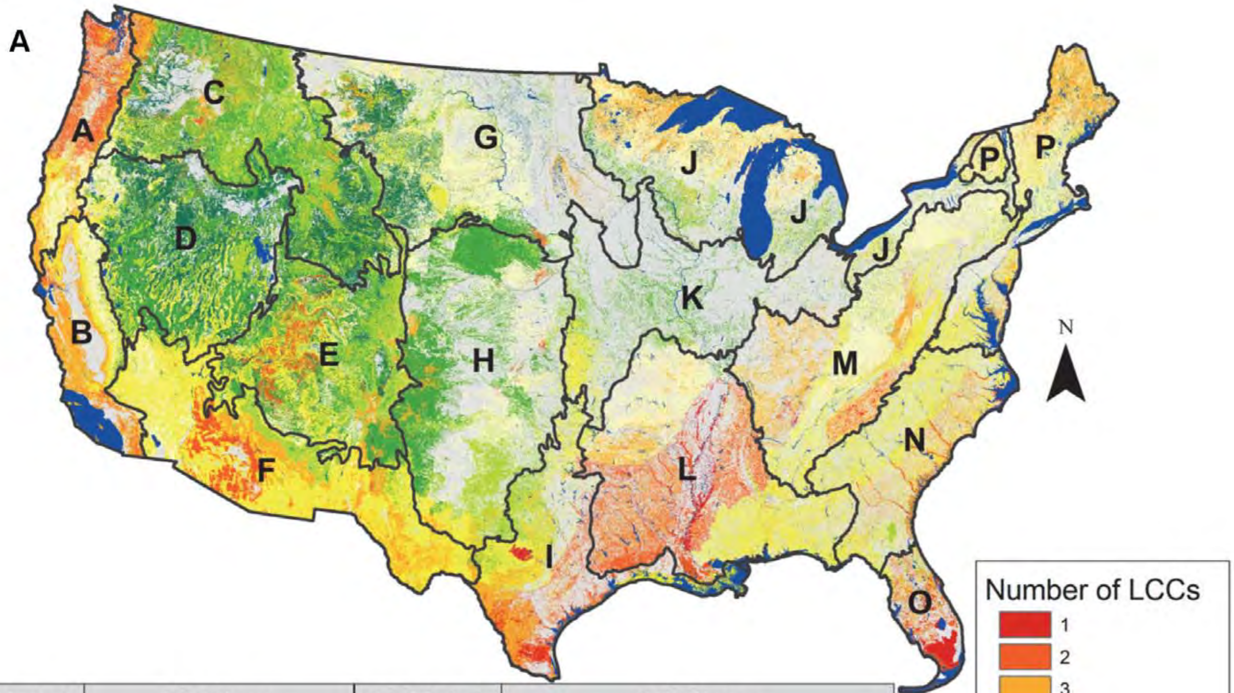
an entire protected area, is access to and use of natural resources (e.g., commodity extraction) [69]. The impact on biodiversity protection because of access and use of natural resources is evident in Midwestern US where a low percent area of land is managed to maintain biodiversity and many areas are mapped as human land use (Figure 4). LCC's in the Midwest (i.e., Plains and Prairie Potholes, Great Plains, and Eastern Tallgrass Prairie and Big Rivers) have low diversity and few unique ecological systems (Figure 6B). A large percent of their area has been converted to human land use, which is reflected in high CRI values (Figure 8). To date, the ecological consequences of degazettement are unclear [69]. Both Fuller et al. [49] and Kareiva [8] believe degazettement would lead to a more dynamic and flexible approach to maintaining the current protected areas network, however it could depend on the level of systematic design used to establish the protected areas network.

Even though we did not specifically assess cost effectiveness of protected areas, our analysis could help inform the approach proposed by Fuller et al. [49]. A cost effectiveness analysis could be based on land ownership, protection status, and percent area converted to human modified systems. For example, the Great Basin LCC has potential for including some of the most cost effective protected areas because it has a low CRI value and $<10\%$ of its area is converted. There is the potential to lower its CRI value and meet the suggested 17% Aichi Biodiversity Target [36] by increasing the percent of area managed to maintain biodiversity by 60% through emphasizing protection of biodiversity (Figure 8). The Great Basin LCC also contains ecological systems that occur only on lands managed for biodiversity (Figure 7) and has a high diversity of ecological systems even though only 1 is unique (Figure 6B). Other factors beyond land ownership, protection status, and percent area converted to human modified systems could be considered in efforts to assess the cost effectiveness of protected areas, such as representation of ecological systems and transaction costs. However, our analysis could help inform a conservation strategy for the continental US if the approach described by Fuller et al. [49] were implemented.

The second alternative for improving the conservation and representation of ecological systems described previously would be to increase the size (i.e., area or number) of our existing protected areas network through acquisition for the least protected, most endangered, or high priority ecological systems [50,51]. If a systematic approach for choosing new protected areas could increase the representation of elevation and soil productivity and thereby ecological systems then the network's ability to respond to varying conditions and future change could be strengthened



LCC Label	LCC Name	LCC Label	LCC Name
A	North Pacific	I	Gulf Coast Prairie
B	California	J	Upper Midwest and Great Lakes
C	Great Northern	K	Eastern Tallgrass Prairie and Big Rivers
D	Great Basin	L	Gulf Coast Plains and Ozarks
E	Southern Rockies	M	Appalachian
F	Desert	N	South Atlantic
G	Plains and Prairie Potholes	O	Peninsular Florida
H	Great Plains	P	North Atlantic

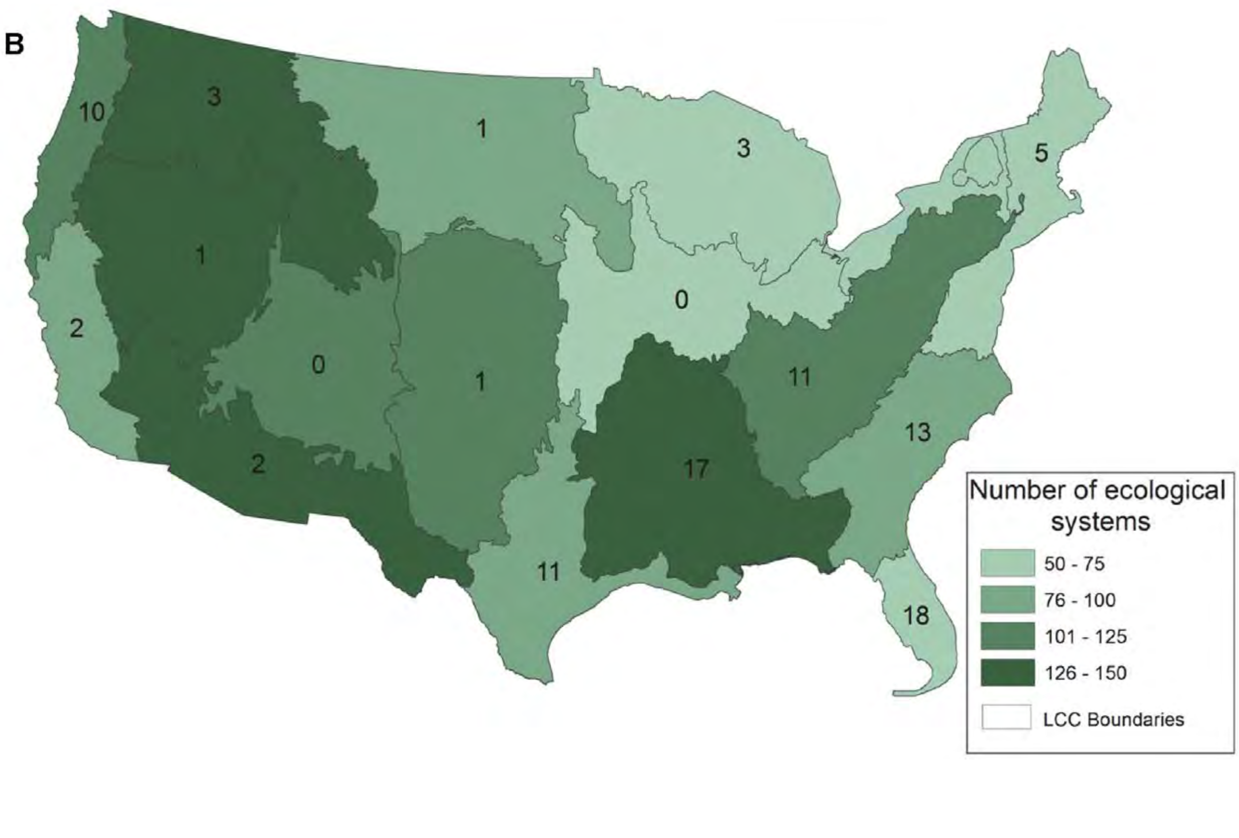


Figure 6. Redundancy, diversity, and uniqueness of ecological systems within Landscape Conservation Cooperatives (LCC). Redundancy measures the number of LCC's in which a single ecological system occurs (A) [23]. The higher the number of LCC's in which an ecological systems occurs the more redundancy displayed by that ecological system. For example, if an ecological system occurs in 2 LCCs, it has a redundancy value of 2. Diversity is the total number of ecological systems occurring with an LCC, which is shown by color shading of LCCs (B). Uniqueness is the number of ecological systems that occur in a single LCC, which is indicated by the number within each LCC (B). For example, the Great Northern LCC encompasses 126–150 ecological systems total, most of these occur in a total of 7 or 8 LCCs, but 3 are unique and only found in this LCC. Only non-modified, non-aquatic ecological systems were included (n = 518; Table S1). Each LCC is assigned a letter, which indicates the name of the LCC.
doi:10.1371/journal.pone.0054689.g006

(Figure 1) [15,63]. Our results were similar to Scott et al. [15] because we found that ecological systems at lower elevations and higher soil productivity were under-represented within the current protected areas network (Figure 1). These areas could be prioritized if acquisition of new protected areas was employed for increasing protection of ecological systems. The least protected ecological systems and potentially most endangered (see Figure 8) are within all the Level I land cover groups except sparse and barren (Figures 2, 3, and 5, Table S1) and are located mostly in the Midwestern US (Figure 4). Prioritizing acquisition of the Level I land cover groups within the Midwestern US would increase the overall representation of ecological systems in the continental US. However, the feasibility of land acquisition for conservation is continually a challenge as resources for obtaining new protected areas are dwindling and competition for undeveloped private land is limiting expansion opportunities [4,14]. Furthermore, the support of policy makers for creating new protected areas could be perceived as ephemeral [72]. The idea of increasing the amount of protected land is attractive in part because of the perceived permanence associated with that protection. In other words, expanding the protected areas network reduces the risk of more land being converted to a state from which it might not recover (i.e., urban development), even though the immediate benefit to conservation is dependent upon management strategies employed.

A third alternative for improving the current protected areas network might be to take stock of our management within the current protected areas network and to evaluate the potential role of lands managed for multiple-use in conserving ecological systems. Our analysis found that increasing the emphasis on maintaining biodiversity on lands currently managed for multiple-use, which are permanently protected, but allow for extractive uses (e.g., mining and logging), offers an alternative for increasing the representation of ecological systems. However, much of the land managed for multiple-use has undergone ecosystem alteration and increased management or restoration may be needed to recover existing ecological systems [52]. If we increased the emphasis on maintaining biodiversity on some public and private lands managed for multiple-use, the total percent area of ecological systems protected could increase up to 39% in the continental US (lands managed to maintain biodiversity: 10%; lands managed for multiple-use: 29%). Geographically, the greatest potential for increased emphasis on maintaining biodiversity on lands managed for multiple-use is in the West, but also in the Northeast, South, and Midwest (Figure 4). To meet the suggested Aichi Biodiversity Target of 17% [36] increased emphasis on maintaining biodiversity would need to occur on 6.4% of the lands managed for multiple-use (Table S1). Even though lands managed for multiple-use occur on both public (i.e., federal, state, and local government) and private (i.e., non-governmental organization) lands, the potential for conservation efforts to increase the protection of

Table 2. Total number and unique number of ecological systems as well as percent area of ecological systems on lands managed to maintain biodiversity and for multiple-use within each Landscape Conservation Cooperative (LCC) in the continental US.

Landscape Conservation Cooperative (LCC)	Number of ecological systems	Number of unique ecological systems	Percent area of lands managed to maintain biodiversity	Percent area of lands managed for multiple-use
Appalachian	103	11	3.5	8.3
California	88	2	10.7	16.3
Desert	133	2	17.0	40.0
Eastern Tallgrass Prairie & Big Rivers	75	0	1.2	1.2
Great Basin	143	1	11.2	62.9
Great Northern	143	3	14.8	39.3
Great Plains	102	1	0.6	2.5
Gulf Coast Plains & Ozarks	148	17	3.5	4.9
Gulf Coast Prairie	95	11	1.3	1.4
North Atlantic	63	5	6.6	8.7
North Pacific	123	10	15.1	25.5
Plains & Prairie Potholes	95	1	2.4	10.6
Peninsular Florida	56	18	8.8	13.1
South Atlantic	97	13	2.8	4.0
Southern Rockies	116	0	14.1	50.6
Upper Midwest & Great Lakes	60	3	5.7	8.3

See Figure for location of LCC. See Table 1 for protection status descriptions. LCCs are listed alphabetically.
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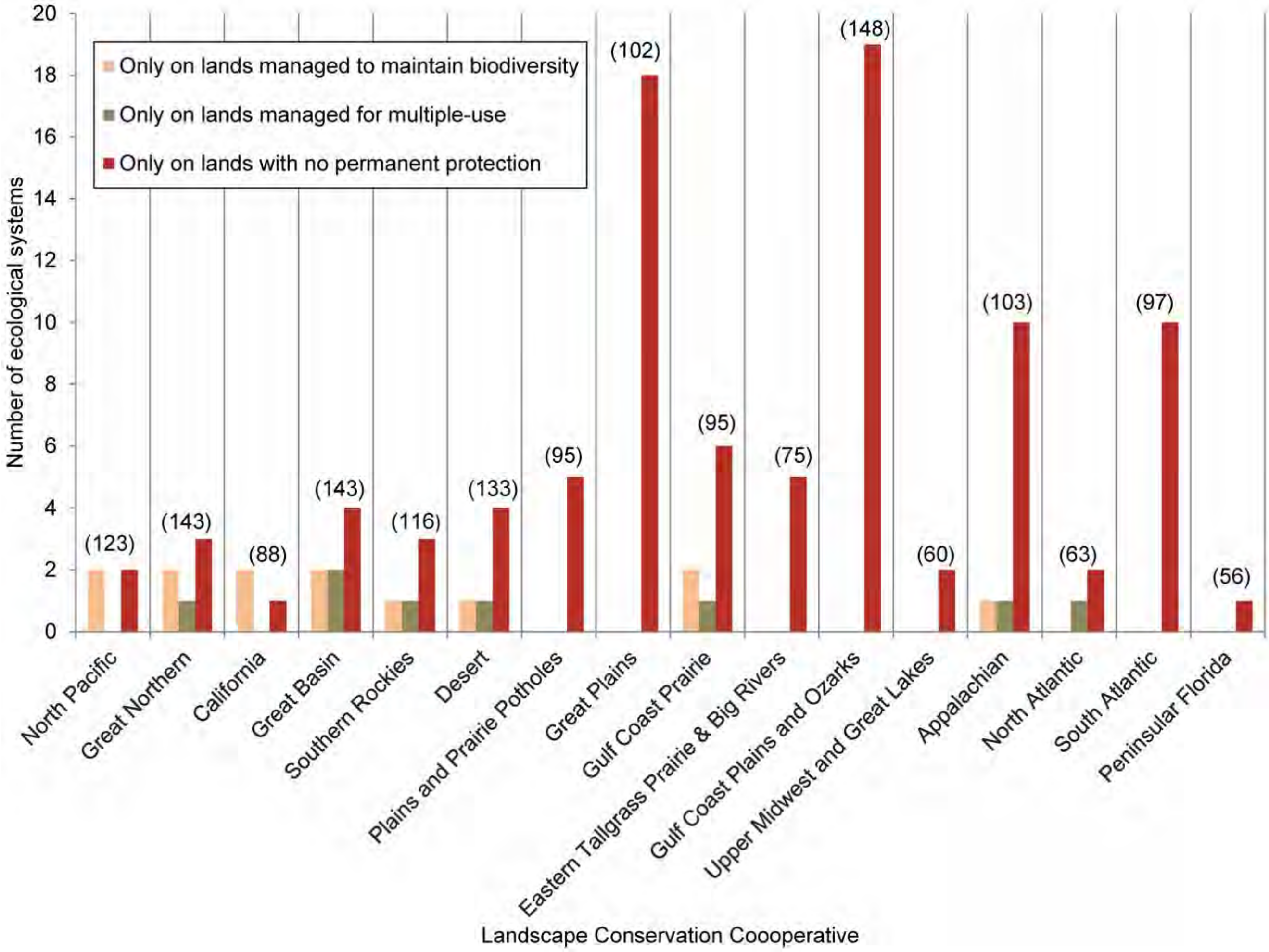

Figure 7. Number of ecological systems occurring only within each protection status by Landscape Conservation Cooperative (LCC). Ecological systems included occur only within the specified protection status [23,24]. The total number of ecological systems within each LCC is shown parenthetically. For example, the Great Plains LCC contains 102 ecological systems with 18 occurring only on lands with no permanent protection and none occurring on lands managed to maintain biodiversity or for multiple-use. See Table 1 for protection status descriptions. Only non-modified, non-aquatic ecological systems are included (n = 518; Table S1).
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ecological systems on public lands is greater (i.e., quantitatively and geographically) (Figure 5).

To protect a broad representation of ecological systems within the continental US, opportunities within public land management agencies fall largely on lands managed by BLM and USFS (Figure 5). Both manage lands that maintain biodiversity, but the majority of the lands they manage are for multiple-use (Figure 5). However, if the US is to become less dependent on foreign energy sources and meet its own resource needs within its boundaries, then shifting management focus on even a small portion of lands currently managed for multiple-use could become a public lands dilemma. Lands managed for multiple-use provide multiple public benefits, including domestic energy production. [17,73,74].

In addition to the lands BLM manages for multiple-use, it has also designated 11 million hectares to the National Landscape Conservation System (NLCS), which is a network of conservation areas specifically aimed at conserving biodiversity [75]. The USFS manages over 17 million hectares of land managed to maintain biodiversity, which is more than USFWS, NPS, and other federal land management agencies combined (Figure 5). With BLM and USFS managing millions of hectares of land for maintaining biodiversity, their role in protecting ecological systems is well established, and there may be potential to expand the protection and representation of ecological systems, for example, through the expansion of NLCS. In the past, administrative jurisdictional land transfers have occurred between land management agencies (e.g., BLM, USFWS, NPS, and USFS) [76–78]. Some of these land transfers have led to more emphasis on maintaining biodiversity.

Landscape Conservation Cooperatives Setting Priorities for Conservation of Ecological Systems

The framework and partnerships of the LCCs informs conservation at the landscape level, which will be needed to implement conservation across jurisdictional boundaries. Our analysis indicates that ecological systems in the East are less redundant and at more risk of conversion than those in the West (Figures 6 and 8). Because of this East-West dichotomy, increased conservation on some public and private lands may be important to the representation of ecological systems in the West, whereas increased public-private partnerships may play an important role in the East to increase the representation of ecological systems (Figures 4, 5, 6, 7, 8).

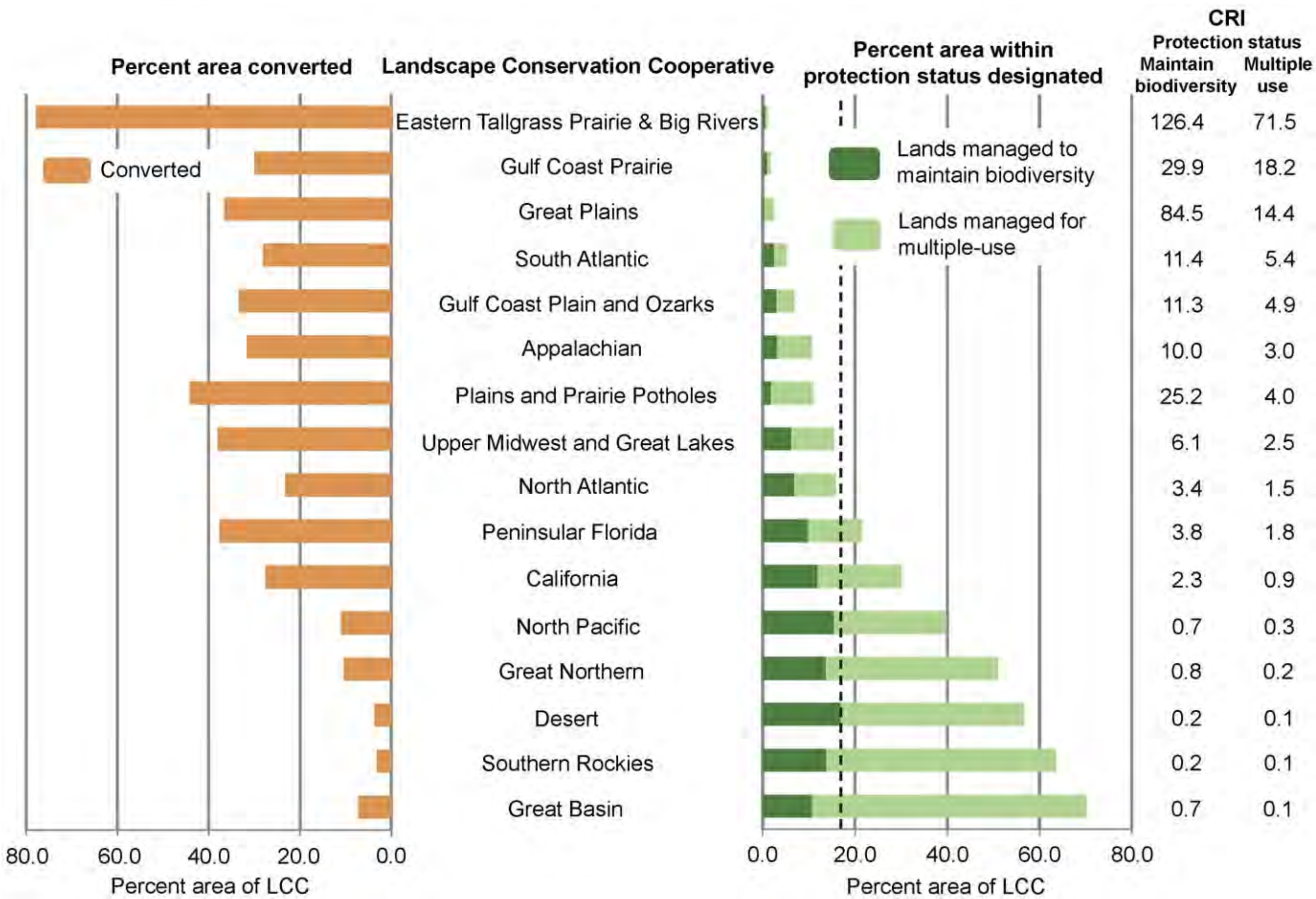

Figure 8. Percent area of Landscape Conservation Cooperative (LCC) protected or converted and its conversion risk index (CRI). CRI for each LCC is calculated by dividing percent area converted by percent area protected [62]. The CRI index is shown for lands managed to maintain biodiversity (i.e., labeled maintain biodiversity) as well as for lands managed to maintain biodiversity and multiple-use (i.e., labeled multiple-use) [23]. The LCCs are ordered by percent area within each protection status. See Table 1 for protection status descriptions. A dashed line representing the 17% Aichi Biodiversity Target of the Convention on Biological Diversity is shown [36].
doi:10.1371/journal.pone.0054689.g008

Our research results highlighting low redundancy and unique ecological systems corroborate results from other studies [13,18]. In particular, the eastern US was identified as an ecoregion with high threats and irreplaceability value with regards to identifying conservation priorities [13,18]. For example, the Gulf Coast Plain and Ozarks LCC in southeastern US has high diversity and uniqueness, but low redundancy and a high conservation risk index (Figures 6 and 8). Within this LCC, there are few opportunities for increasing the representation of ecological systems on lands managed for multiple-use (Table 2, percent protected changes from 3.5% to 4.9%). An initial practical approach for conservation of ecological systems in this LCC, which contains many diverse and unique ecological systems, would be to engage both public and private conservation partners. In this case, our research results could serve as a catalyst for building public and private conservation partnerships. The larger scale perspective of LCCs provides a unique forum that previously did not exist for putting nationwide conservation planning at a scale that allows strategic emphasis on ecological systems that are in most need of added representation and protection.

There are numerous benefits to exploring alternatives for increasing the conservation and representation of ecological systems in the protected areas network. First, we can increase the number and area of ecological systems protected. Ecological systems represent a range of the habitats upon which many species rely; therefore we are increasing the protection of numerous species, including threatened, endangered, and species of concern. Second, we can increase the adaptability of ecological systems and the protected areas network to climate change impacts [79]. A wider range of environmental variables will enable ecological systems and the vertebrate species that rely on them to have room to shift their ranges in response to changes in climate. Third, we can increase the buffer area for all ecological systems and thereby reduce edge effects and increase the integrity of existing ecological systems. Lastly, we are more likely to capture the ecological processes that drive the pattern of ecological systems that we observe and allow for a more fully functional and robust protected areas network.

The current protected areas network for the continental US does not capture the full range of ecological systems or geophysical features (i.e., elevation and soil productivity). As a consequence, the species that rely on these ecological systems and geophysical features have fewer opportunities to adjust to changing environmental conditions. We have not assessed the pros and cons of using our alternatives for increasing the representation of ecological systems, but rather we have presented them as possibilities that may be considered and evaluated as decisions are made to conserve biodiversity. Each alternative may increase the representation of ecological systems, which can lead to protecting and securing habitat across a broader range of ecological, geographical, and geophysical occurrence of species. And may provide the greatest opportunity for evolutionary processes to persist regardless of imminent changes in the near, intermediate, and long term.

Supporting Information

Table S1 Area (ha) and percent area of ecological systems by protection status nested into Level I and II land cover groups [23,24]. All 5 Level I groups, 37 Level II groups, and 518 ecological systems are listed. See Table 1 for protection status descriptions. Only non-modified, non-aquatic ecological systems are included (n = 518).
(XLSX)

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Author Contributions

Conceived and designed the experiments: JMS LS JA AD KG. Analyzed the data: AD JA. Wrote the paper: JLA LS AM AD JS KG.

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Variation in abundance across a species' range predicts climate change responses in the range interior will exceed those at the edge: a case study with North American beaver

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Abstract

The absence of information about how abundance varies across species' ranges restricts most modeling and monitoring of climate change responses to the range edge. We examine spatial variation in abundance across the northeastern range of North American beaver (*Castor canadensis*), evaluate the extent to which climate and nonclimate variables explain this variation, and use a species–climate envelope model that includes spatial variation in abundance to predict beaver abundance responses to projected climate change. The density of beaver colonies across Québec follows a roughly logistic pattern, with high but variable density across the southern portion of the province, a sharp decline in density at about 49°N, and a long tail of low density extending as far as 58°N. Several climate and nonclimate variables were strong predictors of variation in beaver density, but 97% of the variation explained by nonclimate variables could be accounted for by climate variables. Because of the peak and tail density pattern, beaver climate sensitivity (change in density per unit change in climate) was greatest in the interior and lowest at the edge of the range. Combining our best density–climate models with projections from general circulation models (GCM) predicts a relatively modest expansion of the species' northern range limit by 2055, but density increases in the range interior that far exceed those at the range edge. Thus, some of the most dramatic responses to climate change may be occurring in the core of species' ranges, far away from the edge-of-the-range focus of most current modeling and monitoring efforts.

Keywords: abundant center hypothesis, climate change, climate envelope modeling, furbearer, mammal, quantile regression, relative abundance, rodent, spatial ecology, wildlife

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Introduction

Climate is a major determinant of the distribution and abundance of species (Andrewartha & Birch, 1954; Jeffree & Jeffree, 1994; Lomolino *et al*., 2006). Global average surface temperatures have increased by 0.6 ± 0.2 °C since the late 19th century and are expected to rise from 1.4 to 5.8 °C over the next century (Houghton *et al*., 2001). Thus, there is a need to develop models linking species distributions to climate change scenarios in order to anticipate the effects of global warming on plant and animal populations (Ludwig *et al*., 2001; Lawler *et al*., 2006). Species–climate envelope approaches are being used extensively to predict how climate change will alter species distributions (Box, 1981; Sutherst & Maywald, 1985; Austin, 1992; Huntley *et al*., 1995; Iverson & Prasad, 1998; Peterson *et al*., 2002; Thuiller, 2003; Skov & Svenning, 2004; Thomas *et al*., 2004; Araújo & Rahbek, 2006; Elith *et al*., 2006; Botkin *et al*., 2007). Essentially, this method attempts to relate current species distributions with current climatic conditions, then uses predicted future climate scenarios usually derived from general circulation models (GCM), to predict associated shifts in species' geographic distributions (Davis *et al*., 1998; Lawler *et al*., 2006).

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Species–climate envelope models rely on occurrence data to predict the impacts of climate change on species distributions and regional biodiversity (Erasmus *et al*., 2002; Huntley *et al*., 2004; Araújo *et al*., 2005). Although presence/absence range maps provide a useful indication of the broad regional occurrence of a given species, they exclude information about how local abundance varies across the range. As a result, species–climate envelope approaches are capable of predicting range shifts, but not changes in abundance across the range. Although some climate envelope models assume a ramp of suitability or occurrence probability near the edge of the range, the absence of data regarding how abundance actually varies across the range limits predictions of species responses to climate change to the periphery of species range. Similarly, monitoring of species responses to recent climate change has primarily focused on species range expansions and contractions, with little attention paid to changes in abundance between range boundaries (Parmesan & Yohe, 2003; Root *et al*., 2003; Martinez-Meyer *et al*., 2004; Araújo *et al*., 2005). Thus, at present, we have little idea whether climate change modeling and monitoring efforts focused on the periphery of species' ranges are over- or underestimating the impacts of climate change. Our ability to provide more sensitive and/or representative assessment of climate change impacts thus rests on our understanding of geographical abundance patterns.

Numerous ecological and evolutionary hypotheses are based on the assumption that the local abundance of a species is highest in the center of its geographical range, and declines gradually into a tail of low abundance as its range edge is approached (Andrewartha & Birch, 1954; Whittaker, 1956; Hengeveld & Haeck, 1982; Rapoport, 1982; Brown, 1984; Brussard, 1984; Gaston, 1990, 2003; Brown *et al*., 1995). However, there is a paucity of rigorous empirical tests for this assumption and, among the few species that have been examined thoroughly, there is extensive variability in the location of peak abundance within the range (Sagarin & Gaines, 2002a, b; Sorte & Hofmann, 2004). This is particularly the case among mammals, where only a few studies have documented geographical abundance patterns across large spatial extents (Caughley *et al*., 1988; Rodriguez & Delibes, 2002; Williams *et al*., 2003). Hence, although data are sparse and support for population density peaking in the geographical center of the range is weak, there are theoretical and empirical reasons to expect that many species will be characterized by some pattern of systematic variation in local abundance across their range. This pattern will frequently include a tail of low abundance near the periphery of at least some portions of their range boundaries.

An important consequence of a tail of low abundance near a range edge is that the change in abundance per unit distance will tend to decrease as the range edge is approached. Because most climate variables will tend to vary more linearly across the same gradient, the change in abundance per unit change in climate (i.e., the species' local climate sensitivity) should decrease as the range edge is approached. Consequently, for species with a tail of low abundance at the periphery of their range, species–climate envelope models incorporating variation in abundance across the range should predict weak impacts of climate change at the edge of the range, and stronger impacts where the tail ramps upwards to higher abundance closer to the range interior. Predictions of climate change impacts focused on presence–absence data at the edge of species' ranges may therefore underestimate the magnitude of species' responses to climate change in the range interior.

In the present study, we incorporate spatial variation in relative abundance into a climate envelope model to test the hypothesis that predicted species responses to climate change will be larger near the interior of the range than at the edge of the range. We test this hypothesis using a unique, previously unpublished dataset involving 161 surveys of the regional abundance of North American beaver (*Castor canadensis*), covering 74% of their 1.1 million km^2 range in Québec, Canada. Beaver are well-suited to examining abundance patterns because their local abundance can be accurately assessed via aerial surveys of dams, lodges, and autumn food caches (Bergerud & Miller, 1977; Novak, 1987), their general habitat requirements (deciduous and shrubby vegetation along stable waterways; Slough & Sadleir, 1977; Allen, 1983; Howard & Larson, 1985) can be identified from land cover classifications, and they have been extensively surveyed in some regions. Despite better-than-typical survey efforts, equivalent estimates of local abundance are not available across their entire range, which encompasses most of North America. However, the volume and extent of the data available across Québec provides a unique opportunity to examine how beaver abundance varies across more than 1 million km^2 from the northeastern interior of their range to the northeastern edge of their range, and how this variation might influence predictions of climate change impacts on beaver density. The main objectives of this study were to (a) examine the spatial variation in beaver abundance across the northeastern portion of their range, (b) evaluate the extent to which climate and nonclimate variables predict this variation, and (c) use a species–climate envelope model that includes spatial variation in abundance to predict beaver density responses to projected climate change. We predicted that beaver abundance will decline in a logistic fashion from the interior to the edge of their range and will be strongly correlated with climate variables that decline roughly linearly across the same gradient. Thus, we hypothesized that the beaver climate sensitivity (change in abundance per unit change in climate) will be highest in the midrange and lowest at the core and edge of the range.

Materials and methods

Beaver density surveys

Beaver density estimates were derived from reports obtained from the Direction de l'Aménagement de la Faune de l'Outaouais (Gatineau, Québec), the Direction de l'Aménagement de la Faune de Mauricie (Trois-Rivières, Québec), and documentation centers at the Québec Ministry of Environment (Québec, Québec) and at Hydro Québec (Montréal, Québec). Results from aerial surveys conducted in the far north of Québec by SIJ (see Jarema, 2006) were also included in the dataset. We included only helicopter surveys in our analysis because plane surveys can overlook a majority of beaver structures (correction factor up to 75%; Payne, 1981; Potvin & Breton, 1982). The majority of study areas were surveyed in autumn, after deciduous leaf fall and before waterway freeze up, when beavers were completing their food caches. Survey teams consisted of a pilot and a minimum of one observer/navigator in a helicopter flying at low altitude (<100 m) and speed ($<140\,\mathrm{km\,h^{-1}}$). Both active and abandoned sites were recorded, with three active categories: (1) lodge with fresh food cache, (2) fresh food cache without the presence of a lodge, and (3) other obvious signs of beaver presence (e.g., peeled sticks, well-maintained dams, runways and burrows, beaver).

Areas were surveyed using total coverage or subsampling. Total coverage was the methodology used for 77% of surveys included in our study and involves surveying all waterways within the study area. Subsampling was used for the remaining 23% and involved dividing the study zone into equally sized quadrats (4, 9, 25 or 50 $\mathrm{km^2}$), randomly selecting 9–23% of these quadrats, and surveying all the waterways within selected quadrats. Whether the entire study area was surveyed, or a subsample of quadrats was surveyed, the total number of active beaver colonies observed was divided by the total area surveyed, to yield the average number of beaver colonies per $\mathrm{km^2}$. Survey years ranged from 1966 to 2004, but most surveys (80%) were initiated between 1980 and 1995. If a study region was surveyed in more than 1 year, and the survey coverage was within 20% of the maximum survey coverage, beaver densities were averaged. Otherwise, only the beaver density estimated from the most extensive survey was included in the analysis.

To render beaver survey data compatible with GIS, we obtained the vector data for recreational and protected areas in Québec (e.g., controlled harvesting zones, wildlife reserves, outfitting operations, national parks, and ecological reserves) (Limites des territoires récréatifs et protégés 1 : 250 000) and used digital maps imported from Lafond *et al*. (2003). The area, perimeter, and midpoint coordinates were then calculated for each of the 161 study polygons included in our analysis.

Climate and nonclimate explanatory variables

Point estimates of trimonthly temperature minima and maxima, precipitation totals, and agroclimatic indices (Table 1) were obtained for each study polygon centroid from Selected Modeled Climate Data for Point Locations created by The Landscape Analysis and Application Section (LAAS), Great Lakes Forestry Centre (GLFC), Canadian Forest Service (CFS), and Natural Resources Canada (NRCan). The originators used a software package called ANUCLIM (Centre for Resource and Environment Studies, Canberra, Australia) to obtain estimates of monthly mean climate variables, bioclimatic parameters, and indices relating to crop growth (McKenney, 2006). For average trimonthly temperatures, we used the Canadian Gridded Climate Data (50 km grid; Hopkinson, 2001). Once the gridded values were imported into ARCVIEW 8.2 (ESRI, Redlands, CA, USA), they were projected to NAD 1983 Québec Lambert, interpolated to a raster image using Inverse Distance Weighted in 3D Analyst, reclassified at intervals of 1.0 °C, and finally converted from a raster image to a feature. The final product was intersected with all study area polygons.

Potential nonclimate predictors of beaver density were selected based on previous beaver habitat studies (reviewed in Jarema, 2006) and included the nature and extent of waterways, shorelines, vegetation cover, soil composition, slope, beaver harvest intensity, and predator abundance (Table 1). The length, area, and perimeter of waterways (rivers, lakes, and wetlands) within each study polygon were estimated from 92 National Topographic Digital maps (1 : 250 000). Buffers around all waterways, 200 m in width to include the maximum inland foraging distance of beaver (Allen, 1983; Müller-Schwarze & Sun, 2003), were constructed using BUFFERWIZARD in ARCVIEW 8.2 (ESRI). Land cover within the 200 m buffer zones was estimated, for study polygons north of the 52nd parallel, from the Mosaïque du Québec (Photocartothèque Québécoise, 1 : 2 500 000 scale, 15 land cover classes; see Jarema, 2006) and, for study polygons south of the 52nd parallel, from the Spatiocarte Portrait du Québec Forestier Méridional (Direction des Inventaires Forestiers, 1 : 1 250 000

Table 1 Climate and nonclimate variables evaluated as potential predictors of beaver density across Québec

Climate variable	Definition	Nonclimate variable	Definition
PET	Potential evapotranspiration (mm) over growing season*	Smalllakes	Number of lakes $<1\,\text{km}^2$ within the polygon†
GDD	Growing degree days (°C days) above base temperature for the entire growing season*	Largelakes	Number of lakes $>1\,\text{km}^2$ within the polygon†
T_{avgann}	Average annual temperature (°C)‡	Rivershoreline	Proportion of total shoreline (rivers, lakes, wetlands) in the polygon that is along rivers†
T_{avgdjf}	Average December–January–February temperature (°C)§	Lakeshoreline	Proportion of total shoreline (rivers, lakes, wetlands) in the polygon that is along lakes†
T_{avgmam}	Average March–April–May temperature (°C)§	Wetlandshoreline	Proportion of total shoreline (rivers, lakes, wetlands) in the polygon that is along wetlands†
T_{avgjja}	Average June–July–August temperature (°C)§	Riverbuffer	Proportion of polygon area within 200 m buffers around all rivers†
T_{avgson}	Average September–October–November temperature (°C)§	Lakebuffer	Proportion of polygon area within 200 m buffers around all lakes†
T_{maxdjf}	Average maximum December–January–Februrary temperature (°C)‡	Wetlandbuffer	Proportion of polygon area within 200 m buffers around all wetlands†
T_{maxmam}	Average maximum March–April–May temperature (°C)‡	CdecidB	Proportion of 200 m buffer around all rivers, lakes, and wetlands in polygon covered by deciduous forest (including deciduous regrowth)†
T_{maxjja}	Average maximum June–July–August temperature (°C)‡	CmixedB	Proportion of 200 m buffer around all rivers, lakes, and wetlands in polygon covered by mixed forest (including mixed regrowth, mixed dominated by young coniferous, and mixed dominated by young deciduous)†
T_{maxson}	Average maximum September–October–November temperature (°C)‡	CconiferB	Proportion of 200 m buffer around all rivers, lakes, and wetlands in polygon covered by coniferous forest (including coniferous regrowth)†
T_{mindjf}	Average minimum December–January–February temperature (°C)‡	CshrubB	Proportion of 200 m buffer around all rivers, lakes, and wetlands in polygon covered by shrubs and lichens or shrubs and mosses†
T_{minmam}	Average minimum March–April–May temperature (°C)‡	CmossrockB	Proportion of 200 m buffer around all rivers, lakes, and wetlands in polygon covered by moss and rock†
T_{minjja}	Average minimum June–July–August temperature (°C)‡	CrockB	Proportion of 200 m buffer around all rivers, lakes, and wetlands in polygon covered by rocks†
T_{minson}	Average minimum September–October–November temperature (°C)‡	CagricB	Proportion of 200 m buffer around all rivers, lakes, and wetlands in polygon used by agriculture†
T_{iso}	Mean diurnal temperature range divided by the annual temperature range‡	CurbanB	Proportion of 200 m buffer around all rivers, lakes, and wetlands in polygon occupied by populated areas†
T_{seas}	Temperature seasonality expressed as the coefficient of variation of monthly mean temperatures‡	CbuiltupB	Proportion of 200 m buffer around all rivers, lakes, and wetlands in polygon occupied by populated zones where buildings are so close together that, for cartographic purpose, they are represented by a built-up area outline†
P_{avgann}	Sum of all monthly average precipitation estimates (mm)‡	Slope < 2°	Proportion of polygon area occupied by a slope < 2°†
P_{avgdjf}	Average December–January–February precipitation (mm)‡	Slope < 10°	Proportion of polygon area occupied by a slope < 10°†
P_{avgmam}	Average March–April–May precipitation (mm)‡	Slope < 30°	Proportion of polygon area occupied by a slope < 30°†
P_{avgjja}	Average June–July–August precipitation (mm)‡	Slope > 30°	Proportion of polygon area occupied by a slope > 30°†
P_{avgson}		Slope < 2°B	

Continued

Table 1 Continued

Climate variable	Definition	Nonclimate variable	Definition
	Average September–October–November precipitation (mm)‡		Proportion of 200 m buffer around all rivers, lakes, and wetlands occupied by a slope $<2°$†
P_{seas}	Precipitation seasonality expressed as the coefficent of variation of monthly average precipitation‡	Slope$<$10°B	Proportion of 200 m buffer around all rivers, lakes, and wetlands occupied by a slope $<10°$†
		Slope$<$30°B	Proportion of 200 m buffer around all rivers, lakes, and wetlands occupied by a slope $<30°$†
		Slope$>$30°B	Proportion of 200 m buffer around all rivers, lakes, and wetlands occupied by a slope $>30°$†
		Smineral	Proportion of polygon containing surface material made up of predominantly mineral particles containing $<30\%$ organic matter by weight¶
		Sorganic	Proportion of polygon containing surface material made up of $>30\%$ organic matter by weight¶
		Ssoftrock	Proportion of polygon containing surface material made up of rock that can be dug with a shovel (i.e., undifferentiated shales, upper Cretaceous and Tertiary materials)¶
		Sgranite	Proportion of polygon containing surface material made up of granite¶
		Slimestone	Proportion of polygon containing surface material made up of limestone¶
		Shardrock	Proportion of polygon containing surface material made up of hard rock of unspecified origin and undifferentiated properties¶
		Beaverharvest	Average beaver harvest per unit area for 'Structured' or 'Free Zones' in the Administrative Regions of Québec‖
		Beardensity	Estimated black bear density (individuals km^{-2}) by trapping zones**
		Wolfdensity	Estimated wolf density (individuals km^{-2}) in administrative regions or wildlife reserves††,‡‡
		Limitedroads	Kilometers of roads that vary seasonally in condition or to which public access is denied divided by polygon area†
		Roads	Kilometers of roads for the movement of motor vehicles divided by polygon area†

*Bootsma & McKenney (2005).
†Natural Resources Canada. 2006. Centre for Topographic Information: Glossary for NTBD data 1 : 250 000. http://www.cits.rncan.gc.ca/cit/servlet/CIT/site_id=01&page_id=1-002-001.html#b.
‡Landscape Analysis and Application Section (LAAS), Great Lakes Forestry Centre (GLFC), Canadian Forest Service (CFS), Natural Resources Canada (NRCan). Resources Canada 2006. Selected Modeled Climate Data for Point Locations. Sault Ste. Marie. LAAS.
§Hopkinson (2001).
¶Centre for Land and Biological Resources Research. 1996. Soil Landscapes of Canada, v.2.2, Research Branch, Agriculture and Agri-Food Canada. Ottawa.
‖Pierre Canac-Marquis Coordonnateur Piégeage Faune et Parcs Québec.
**Jolicoeur (2006).
††Jolicoeur & Heneault (2002).
‡‡Lariviere *et al*. (1998).

scale, 22 land cover classes; see Jarema, 2006). Slopes within the 200 m buffer zones were calculated from the same National Topographic Digital Maps using ARCVIEW 8.2 (ESRI) 3D Analyst to create a TIN from contour lines and the SLOPE function in Surface Analysis to derive slopes in degrees. The image was then reclassified using the following defined intervals: 0–2.0°, 2.1–10.0°, 10.0–30.0°, >30.1°, which provided, for each polygon, the area within the 200 m buffer represented by the different slope categories. The dominant value for soil composition within each study polygon (KINDMAT field) was obtained from Canadian Soil Information (CanSIS) website. For each study polygon, the area covered by built-up regions (populated zones where buildings are so close together that, for cartographic purpose, they are represented by a built-up area outline) and the length of roads were derived again from the National Topographic Digital Maps. To incorporate the abundance of potential beaver predators in the analysis, wolf (*Canis lupus*) densities were calculated by dividing the estimated number of wolves in each administrative region by the area of that administrative region (Lariviere *et al*., 1998; Jolicoeur & Heneault, 2002), whereas black bear (*Ursus americanus*) densities were calculated by dividing the estimated number of black bears within each trapping zone by the area of that trapping zone (Lamontagne *et al*., 2006). The average number of beaver harvested per km^2 was calculated by dividing the average number of beaver harvested in regions referred to as 'libre' (private lands and certain crown lands where trapping is carried out with no particular constraints) and 'structurée' (crown lands subdivided into trapping territories where exclusive trapping rights are leased to certain trappers) by the area of these zones within each administrative region (P. Canac-Marquis, Coordinator, Societé de la Faune et des Parcs Québec, personal communication 2004).

Model selection

Our modeling objective is to identify highly predictive but parsimonious models of beaver density based on variables that are commonly forecasted by GCM's. As a result, our model selection approach is biased towards climate over nonclimate variables, and univariate over multivariate models. However, in addition to identifying the strongest climate predictors of beaver density, we seek to quantify the opportunity cost of excluding nonclimate variables and multivariate climate models. Thus, we first compare the independent and combined explanatory power of climate vs. nonclimate variables, then examine the relative explanatory power of multivariate vs. univariate climate models, then focus on the strength and the form of the best univariate climate–density models.

The data were heteroscedastic and beaver density appeared to have a nonlinear relationship with most variables. Accordingly, beaver density was square-root transformed, which is a commonly used transformation for abundance data that is similar in effect to the log transform but that works on zeros. All proportional explanatory variables were arcsine-transformed before the analysis.

The role of climate vs. nonclimate variables was evaluated using regression on principal components. Specifically, a principal component analysis (PCA) was calculated on all 24 climate variables and the scores of each site on the first two axes were retained. Similarly, a PCA was performed on all 36 nonclimate variables and the scores of each site on the first two axes were retained. A Borcard partition (partial regression analysis) was performed to evaluate the proportion of variance explained by the scores on first two principal components solely for climate, solely for nonclimate, and jointly for climate and nonclimate.

To compare multivariate species–climate envelope models with univariate models, we (1) performed a multivariate linear regression with all 24 climate variables, (2) used two common multivariate selection models (stepwise regression and regression trees) to identify how many variables would be chosen and the predictive power (r^2) of these sets of variables, and (3) examined the predictive power of each climate and nonclimate variable as a univariate predictor of beaver density. The top 10 univariate climate variables with the highest r^2-values were selected to model their relationship with beaver density. The nontransformed data strongly suggested an envelope relationship rather than a simple curvilinear relationship so quantile regression was used (Cade & Noon, 2003). We examined the 10th, 50th, and 90th quantiles using linear, quadratic, and normal (Gaussian) curves,

Linear	$\text{Density} = a + b \times (z)$
Quadratic	$\text{Density} = a + b(z) + c \times (z)^2$
Normal	$\text{Density} = c \times e^{-(z-b)^2/a^2}$

where z is the best predictor, and a, b, and c are free parameters estimated using the interior point algorithm (Koenker & Park, 1996) adapted for MATLAB version 7.3 (Mathworks, Natick, MA, USA) by David Hunter (http://www.stat.psu.edu/~ dhunter/code/qrmatlab/). We compared distribution models using a quantile regression analog to the OLS coefficient of determination derived by Koenker & Machado (1999) that provides pseudo-r^2 for any quantile (traditional r^2 can be used on the 50th percentile, but not on other percentiles).

Climate sensitivity, climate change, and beaver density change

The top three univariate climate models for the 10th, 50th, and 90th percentiles were selected and used to predict present and future beaver densities across Québec. Gridded climate data from 1961 to 1990 and scenarios from 2040 to 2069 were used, respectively, for present and future periods (Bootsma & McKenney, 2005). These present and future climate data included monthly maximum and minimum air temperature, average annual air temperature, precipitation, growing degree-days, and potential evapotranspiration (PET). The climate model and emission scenario (CGCM1 GA1) used to predict future climate were described by Flato *et al*. (2000) and Boer *et al*. (2000a, b). To evaluate the generality of this model and scenario combination, we compared it with four other combinations involving two additional models with two emission scenarios each (CGCM2 A2, B2, Flato & Boer, 2001; HADCM3 A2, B2, Gordon *et al*., 2000; Pope *et al*., 2000). Climate sensitivity is expressed as the change in beaver density resulting from a 10% change in a given climate variable, calculated with parameters from the best-fit climate–density model. Similarly, beaver density change was

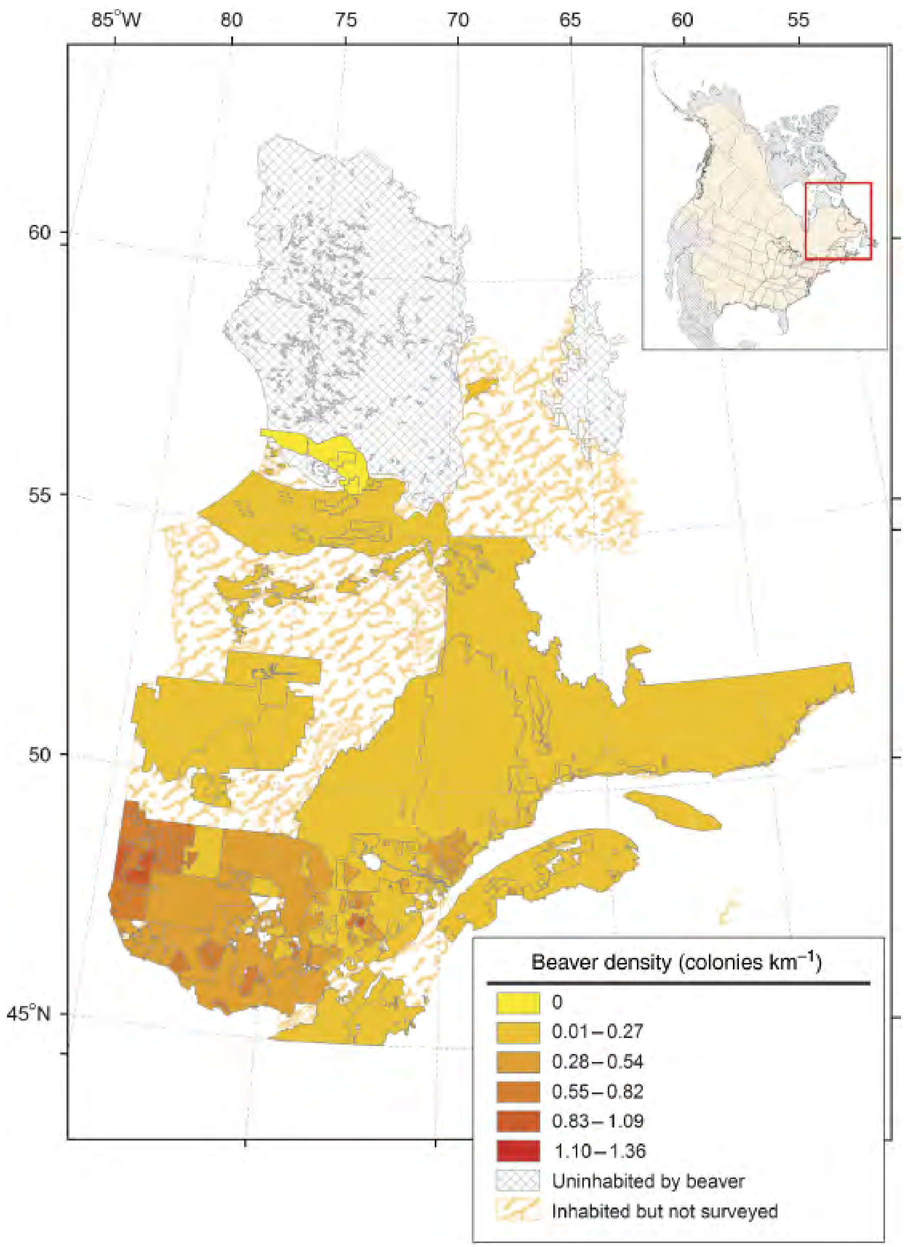

Fig. 1 Local abundance of North American beaver (*Castor canadensis*) across the province of Québec. Densities were derived from 161 helicopter surveys conducted between 1976 and 2004. The average number of active beaver colonies per km^2 for each survey area was calculated by locating active colonies along watercourses and dividing this number by the total area. Inset: the approximate North American range of *C. canadensis*.

calculated by comparing beaver densities predicted by best-fit models applied to current climate normals and GCM-predicted climate futures.

Results

The highest beaver densities in Québec are found in the southwest; in other southern portions of the province, beaver densities are variable, but generally declining from west to east (Fig. 1). Moving northwards, beaver densities decline sharply around 49°N and then form a long tail of low densities spanning more than 9° of latitude (Fig. 2a).

Partial regressions on two principal component axes (scaled PCA) derived from all climate and nonclimate variables revealed that climate variables alone explained 17.4% of the variation in beaver density, nonclimate variables alone explained 1.5%, and climate and nonclimate variables jointly explained 33.3% (leaving 47.7% unexplained). In other words, climate variables explained 97.1% of the variation that could be explained by a combination of climate and nonclimate variables (Fig. 3).

Both stepwise multivariate linear regression and regression trees selected a model with only two of the possible 24 climate variables (T_{avgann} and T_{maxmam} for stepwise, PET and T_{mindjf} for regression tree). The multivariate regression on all 24 climate variables had an r^2 of 0.67, the chosen stepwise model (with two climate variables) had an r^2 of 0.57, and the regression against the top two climatic PCA axes had an r^2 of 0.51. The selection of only two variables using both model selection techniques is presumably due to the high collinearity of climate variables (the first principal component accounted for 80% of all variation in the 24 climate variables and the first two principal components accounted for 92% of all variation).

Exploring univariate predictive power of climate and nonclimate variables, the majority of variation in square-root transformed beaver density across Québec can be accounted for by several univariate relationships (Table 2). The top univariate climate predictors include agroclimatic indices [e.g., PET and growing degree days (GDD)] and temperature variables (e.g., maximum, minimum, and average seasonal air temperatures) (Table 2). The top nonclimate predictors include

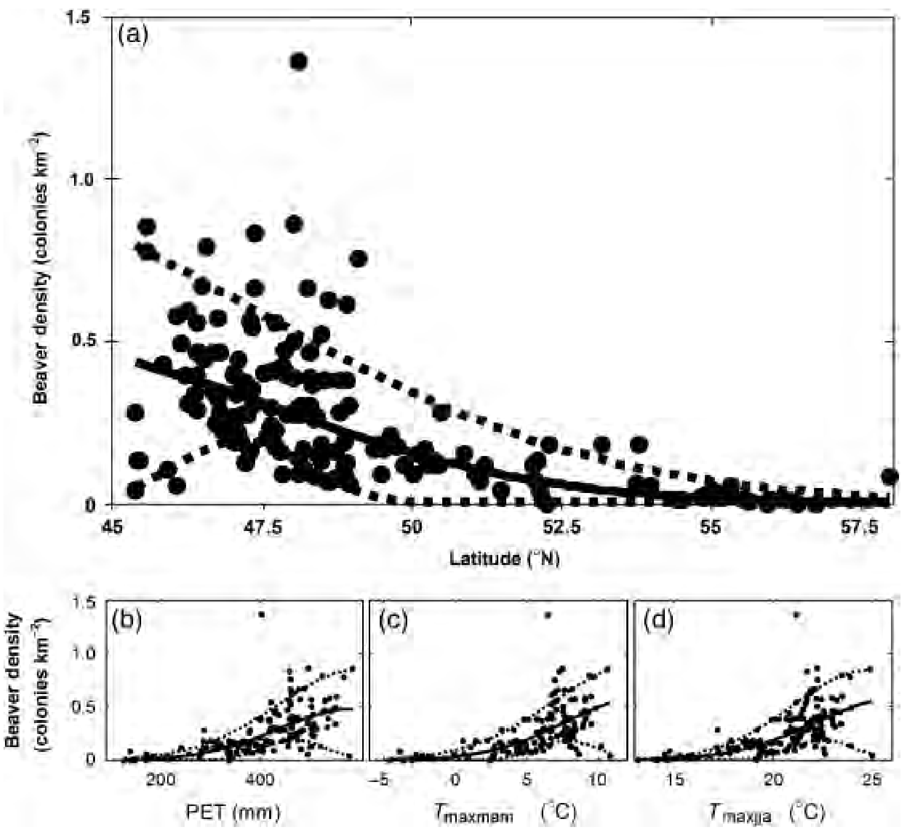

Fig. 2 Variation in local beaver density across Québec as a function of (a) latitude and the top-three univariate climate predictors, including (b) potential evapotranspiration (PET), (c) average maximum March–April–May temperature (T_{maxmam}), and (d) average maximum June–July–August temperature (T_{maxjja}). Lines represent the normal equations that best describes the 10th (dashed line), 50th (solid line), and 90th (dashed lines) percentiles of beaver density.

latitude, black bear density, and deciduous and shrub land covers (Table 2).

Using the top 10 climate variables to predict the 10th, 50th, and 90th percentiles of untransformed beaver densities, a normal model provided a better fit (based on pseudo-r^2 values appropriate for quantile regression) than a linear or quadratic model in 27 of 30 instances (Table 3). The fit of the quadratic model was frequently only marginally weaker than the normal model, but when this was the case, the quadratic curve was invariably concave (i.e., c was always positive), meaning that, similar to the normal model, the slope of the relationship between climate and abundance accelerated from low to high beaver density (i.e., from the edge to the interior of the range).

Overall, the best three predictors of the 10th, 50th, and 90th percentiles collectively and the 50th percentile in particular, are PET, average maximum March–April–May temperature (T_{maxmam}), and average maximum June–July–August temperature (T_{maxjja}) (Table 3). Each of these three climate variables assumes a normal relationship with percentiles of beaver density, with the slope of the curve peaking at intermediate climate values corresponding to the approximate midpoint of beaver's distribution in Québec, then flattening to varying extents at warmer climate values corresponding with southern Québec (Figs 2b–d).

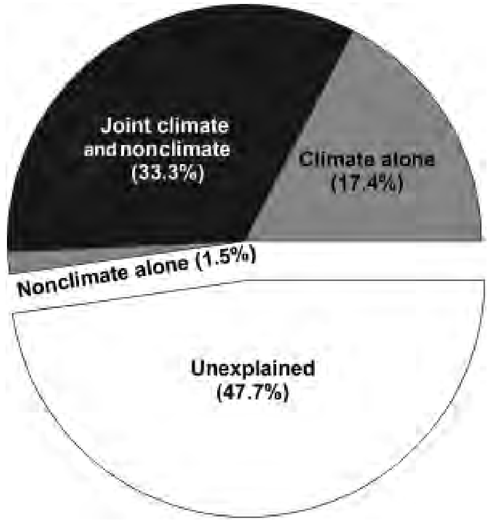

Fig. 3 Partial regression analysis estimating the variation in beaver density explained by climate and nonclimate variables. PCA was calculated on all 24 climate variables and the scores of each site on the first two axes were retained. Similarly, a principal component analysis (PCA) was performed on all 36 nonclimate variables and the scores of each site on the first two axes were retained. Each group uniquely accounts for only a small amount of variation in beaver density, whereas a much larger proportion is explained jointly by climate and nonclimate variables. Thus, a model including climate variables alone can account for >95% of the total variation explained by climate and nonclimate variables in combination.

For the top three climate variables, beaver climate sensitivity (predicted change in density per unit change in climate) is highest in the southern half of Québec and declines northward as the present day range limit is approached (Fig. 4 1a–c). GCM-projected change between now and 2055 in these climate variables peaks at high latitudes and generally diminishes southward (Fig. 4 2a–c). These climate projections differ marginally from other GCM and emission scenarios on a regional basis, but, in general, tend to be intermediate or conservative relative to other model and scenario combinations (Fig. 5). Combining projected climate change and beaver climate sensitivity, the largest absolute changes in density (future density–present density) are consistently predicted to occur in the southern half of Québec (Fig. 4 3a–c). Considering the present northern range limit of beaver distribution in Québec (Fig. 4 1a–c), relatively small and spatially restricted range expansion is predicted to occur (Fig. 4 3a–c). Thus, beavers are presently restricted to regions in Québec with average annual temperature above −5.1 °C, maximum summer temperature above 15.2 °C, maximum spring temperature above −1.4 °C, and PET above 200 mm. By 2055, these conditions are expected to expand northwards and be associated with a northern range expansion of <100 km in most regions of northern Québec (Fig. 4 3a–c), with the exception of the westcentral portion of the range limit where a ~200 km expansion is predicted.

Discussion

Beaver density across Québec follows a roughly logistic envelope pattern, with high but variable density across the southern portion of the province, a sharp decline in density at about 49°N, and a long tail of low density extending as far as 58°N. Although several climate and nonclimate variables were strong univariate predictors of variation in beaver abundance, 97% of the variation explained by nonclimate variables could be accounted for by climate variables. Furthermore, four PCA axes that included all climate and nonclimate variables (two axes derived from 24 climate variables and two derived from 36 nonclimate variables) explained less variation in beaver density ($r^2 = 0.51$) than the three top climate univariate models, each based on a single climate variable ($r^2 = 0.55$–0.56). Although stepwise regression and regression tree procedures both selected multivariate models over univariate models, in both cases the selected models contained only two climate variables, had only marginally higher explanatory power than the top univariate models ($r^2 = 0.57$ vs. 0.55–0.56 for top univariate climate models despite the positive r^2-bias

Table 2 Results from univariate regression of square-root transformed beaver density as a linear function of climate and nonclimate variables

Climate				Nonclimate			
Variable	Sign	R^2	P	Variable	Sign	R^2	P
PET	+	0.562	0.000	Latitude	–	0.495	0.000
T_{maxmam}	+	0.559	0.000	Beardensity	+	0.399	0.000
T_{maxjja}	+	0.546	0.000	CdecidB	+	0.375	0.000
T_{avgjja}	+	0.503	0.000	CshrubsB	–	0.359	0.000
GDD	+	0.502	0.000	CmixedB	+	0.352	0.000
T_{maxson}	+	0.489	0.000	CconiferB	–	0.309	0.000
T_{avgmam}	+	0.486	0.000	CmossrockB	–	0.251	0.000
T_{avgann}	+	0.466	0.000	Limitedroads	+	0.239	0.000
T_{iso}	+	0.448	0.000	Beaverharvest	+	0.199	0.000
T_{minmam}	+	0.432	0.000	Roads	+	0.166	0.000
T_{minjja}	+	0.431	0.000	Longitude	–	0.063	0.001
T_{maxdjf}	+	0.426	0.000	Riverbuffer	+	0.055	0.003
T_{avgson}	+	0.421	0.000	Lakebuffer	–	0.049	0.005
T_{minson}	+	0.360	0.000	Lakeshoreline	–	0.048	0.005
P_{seas}	–	0.315	0.000	Rivershoreline	+	0.032	0.023
T_{avgdjf}	+	0.303	0.000	Wolfdensity	+	0.027	0.039
T_{mindjf}	+	0.262	0.000	CrockB	–	0.021	0.069
P_{avgmam}	+	0.220	0.000	Largelakes	–	0.013	0.153
T_{seas}	–	0.158	0.000	Slope $\leq 2°$	–	0.011	0.176
P_{avgann}	+	0.146	0.000	Slope$>30°$	–	0.011	0.178
P_{avgdjf}	+	0.141	0.000	Slope $\leq 30°$	+	0.011	0.179
P_{avgjja}	+	0.121	0.000	Smalllakes	–	0.010	0.214
P_{avgson}	+	0.021	0.069	Shardrock	–	0.007	0.296
				Slope $\leq 30°$B	+	0.007	0.307
				Slope$>30°$B	–	0.007	0.307
				Ssoftrock	–	0.006	0.329
				Wetlandbuffer	+	0.006	0.337
				Wetlandshoreline	+	0.006	0.347
				Sgranite	+	0.004	0.400
				Smineral	–	0.004	0.451
				Slope $\leq 2°$B	–	0.002	0.598
				Slimestone	+	0.001	0.642
				Slope $\leq 10°$B	+	0.001	0.732
				Slope $\leq 10°$	–	0.001	0.752
				Sorganic	–	0.001	0.765
				CbuiltupB	–	0.000	0.811
				CurbanB	+	0.000	0.825
				CagricB	+	0.000	0.916

inherent in stepwise and regression tree procedures; Freedman, 1983), and included climate variables that were highly ranked as univariate predictors. Accordingly, we used univariate climate models because of their high predictive power in this application (in both absolute terms and relative to the alternatives), their parsimony, their ability to inform about potential mechanisms, and their compatibility with quantile regression.

Univariate climate–abundance relationships formed a logistic envelope pattern, with a long tail of low beaver density at low climate values, ramping up to high but

Table 3 Quantile regression pseudo-r^2-values explaining the variation in the 10th, 50th, and 90th percentile of beaver densities using the top 10 univariate climate predictors and three different models (linear, quadratic, and normal)

Climate variable	Linear (%)			Quadratic (%)			Normal (%)		
	10	50	90	10	50	90	10	50	90
PET	0.207	0.323	0.285	0.208	0.366	0.328	*0.239*	*0.366*	*0.340*
T_{maxmam}	0.205	0.297	0.262	0.208	0.356	0.328	*0.254*	*0.360*	*0.334*
T_{maxjja}	0.192	0.306	0.286	0.195	0.346	0.337	*0.247*	*0.346*	*0.357*
T_{avgjja}	0.187	0.258	0.214	0.189	0.306	*0.239*	*0.220*	*0.309*	0.239
GDD	0.177	0.314	0.294	0.192	0.336	0.295	*0.241*	*0.346*	*0.314*
T_{maxson}	0.171	0.294	0.278	0.186	0.311	0.279	*0.217*	*0.315*	*0.295*
T_{avgmam}	0.191	0.249	0.193	0.194	0.281	0.205	*0.207*	*0.285*	*0.207*
$T_{avgaann}$	0.173	0.263	0.237	0.189	0.283	*0.243*	*0.212*	*0.290*	0.241
T_{iso}	0.231	0.244	0.170	0.238	0.245	0.183	*0.273*	*0.299*	*0.280*
T_{minmam}	0.172	0.233	0.207	0.188	0.243	*0.211*	*0.219*	*0.252*	0.203

Values in italics indicate the highest pseudo-r^2, with the normal model performing best in 27 of 30 cases (90%). Note that the quantile-adapted pseudo-r^2 presented here is not comparable with the traditional r^2 presented in Table 2; both are valid for comparisons of relative explanatory power within but not across tables.

PET, potential evapotranspiration; GDD, growing degree days.

variable densities at high climate values. Thus, relatively warm climates appear necessary, but not sufficient for beavers to attain high densities in Québec. Presumably, beavers often occur at low densities in warm regions because not all localities within these regions provide the types of habitats, watercourses, and topography that beavers also require. On the other hand, beaver can clearly survive and reproduce in the extreme climatic and habitat conditions that prevail in far northern Québec (where average annual temperature is −5 °C, lakes are free of ice for only 4 months per year, and the only trees present are riparian shrubs; Lenormand *et al*., 2002), but appear to be unable to attain high densities in these regions.

Relatively few studies have examined correlations between climate and abundance across species' ranges because typically only presence/absence data are available (Scott *et al*., 1993; Guisan & Zimmerman, 2000; but see Lichstein *et al*., 2002). However, the range limits of many plants and animals appear to coincide with climatic isotherms (Root, 1988) and climatic predictors of range limits often outperform nonclimate predictors (Thuiller *et al*., 2004), regardless of the trophic level under consideration (Huntley *et al*., 2004). We selected climate variables for modeling purposes because they were slightly better predictors of beaver density and are more commonly and consistently projected in climate change scenarios than nonclimate variables. However, we could have explained nearly as much variation in

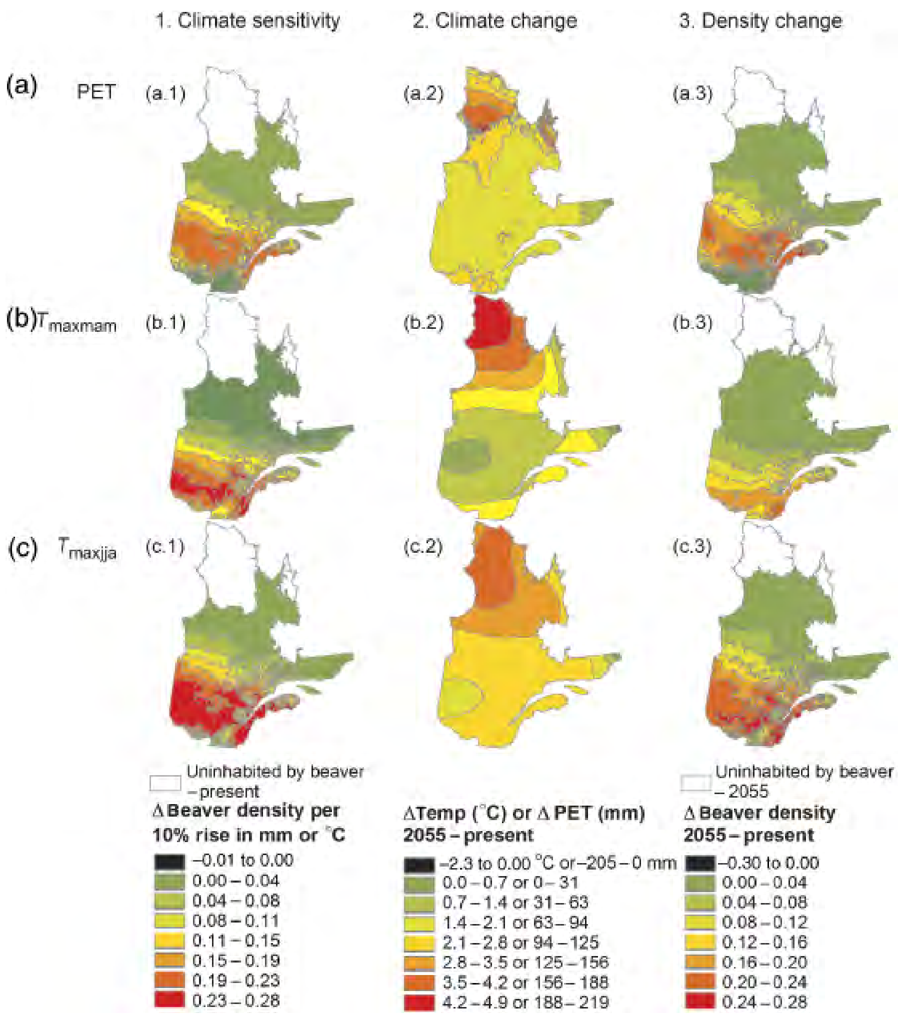

Fig. 4 Predicted changes in (1) beaver density (colonies km^{-2}) with a 10% increase in climate variables (climate sensitivity), (2) climate from present to the year 2055 (climate change) based on the CGCMI GA1 model, and (3) beaver density change across Québec from present to the year 2055 (density change) based on three climatic variables with best-fit models: (a) potential evapotranspiration (PET), (b) average maximum March–April–May temperature (T_{maxmam}), and (c) average maximum June–July–August temperature (T_{maxjja}). White areas indicate regions not inhabited by beavers at present (column 1; climate sensitivity) and in the future (column 3; density change). Projection of future range limits is based on matching the current isotherm delineating the northern most location of beaver at present, then using the GCM projection of the location of this isotherm in 2055 [(a.3) PET = 200 mm, (b.3) $T_{\text{maxmam}} = -1.4\,^{\circ}\text{C}$, (c.3) $T_{\text{maxjja}} = 15.2\,^{\circ}\text{C}$].

beaver density with several land cover variables and, based on results from our partial regression analysis, the variation explained would have overlapped extensively with that explained by climate variables. In other words, the independent effect of climate on beaver density (i.e., variation in climate not correlated with variation in nonclimate variables) was relatively weak. These results emphasize that (1) climate variables can serve as an effective proxy for the suite of climatic and nonclimatic factors that determine animal abundance and distribution but (2) the validity of using climate proxies to project animal responses to climate change hinges critically on the persistence of current correlations between climate, habitat, and other environmental features (Pearson & Dawson, 2003; Lawler *et al*., 2006).

We found general support for our hypothesis that the climate sensitivity of beaver abundance (change in abundance per unit change in climate) peaked in the interior of the range. The high variability of beaver densities in southern Québec, combined with our lack of data from jurisdictions south of Québec, prevented us from clearly differentiating the fit of normal models (with accelerating then decelerating slope from the edge to the interior) from quadratic models (with continuously accelerating slope from the edge to the interior; Table 3). However, this distinction is less important to

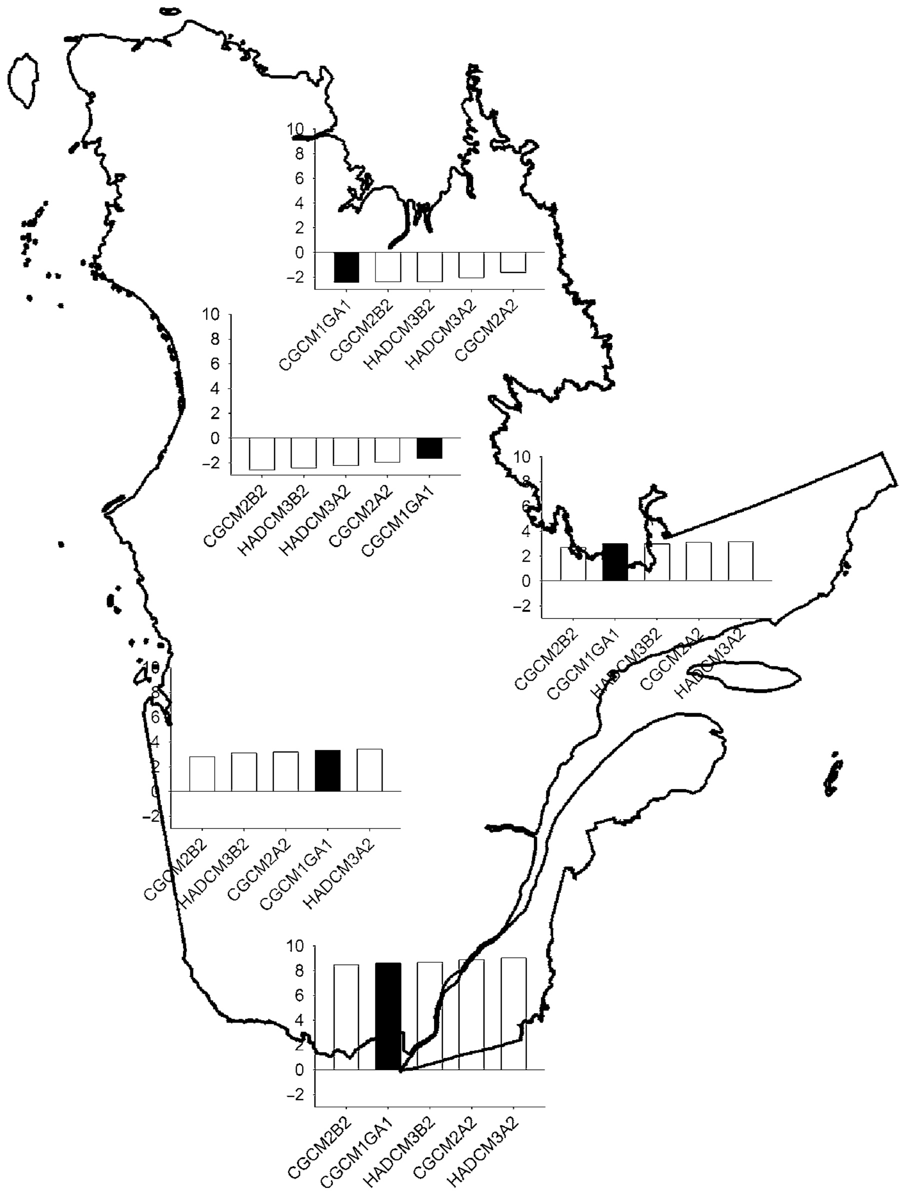

Fig. 5 Climate model and emissions scenario comparison for projected 2055 average annual temperature (T_{avgann}; °C) for different regions of Québec. The climate model and emission scenario combination used in this study (black bar; CGCM1 GA1) generated similar predictions as two other models each with two different emission scenarios (white bars; CGCM2 A2, B2 and HADCM3 A2, B2). Results are similar if other climate variables are used as the basis of comparison (e.g., T_{maxmam}, T_{maxjja}; not shown here), except CGCM1 GA1 projections of T_{maxjja} increases are consistently conservative across Québec relative to the other climate model and emission scenario combinations.

the general conclusion of the study than the agreement of both models that climate sensitivity is low at the northern edge of the range and accelerates towards the range interior.

Combining our best climate envelope models of beaver density with current GCM projections of future climate change, beavers are predicted to be characterized by only modest range expansion, but substantial increases in density within the interior of their range. We acknowledge the numerous limitations in using a correlative climate envelope approach, including the fact that we fail to directly account for biotic interactions, evolutionary change, or dispersal (Pearson & Dawson, 2003), and that the present relationships among abundance, distribution, and climate may not remain the same in the future (Lawler *et al.*, 2006).

Consequently, in using this approach, we assume that the relationships among climate, beaver abundance, and beaver distribution reflect some direct or indirect form of causality, that this causality will remain the same in the face of climate change, and that beaver responses and climate change will occur at a similar pace. Based on beaver's well-studied ecology (Slough & Sadleir, 1977; Allen, 1983; Howard & Larson, 1985; Novak, 1987), we expect that this will be the case only if there is a concomitant increase in abundance and/or productivity of their primary food sources (deciduous shrubs and trees) adjacent to waterways, and if other forms of environmental and anthropogenic changes (e.g., fire frequency, conversion of forests into agricultural and developed lands, trapping intensity) do not override the effects of climate change in this region. The pattern of dispersal and settlement of the reintroduced European beavers (*Castor fiber*) in Scandinavia provides a useful precedent for predicting how beavers colonize new habitats and alter their abundance in currently occupied habitats. This example indicates an important role of long distance dispersal within watersheds, followed by back-filling of suitable habitats between the dispersal front and the established population core (Hartman, 1995), as well as persistent influences of initial territory establishment on long-term patterns of beaver distribution and abundance (Campbell *et al*., 2005). The present pattern of North American beaver abundance across Quebec, as reflected in our dataset, will also be strongly influenced by historical recolonization events, following repeated large-scale overharvest, population depletion/extirpation episodes that have occurred in northeastern North America as recently as the 1930s (Müller-Schwarze & Sun, 2003). Thus, although patterns of individual movement and territory settlement may account for some of the unexplained variation in large-scale patterns of beaver abundance, they do not appear to preclude the emergence and persistence of strong climate–abundance associations.

Conclusions

Our central conclusion is that there is much to be gained by incorporating information about how abundance varies across species ranges when using spatial climate variability as a basis for predicting the impacts of climate change. Species–climate envelope models relying on presence/absence data can predict expected range shifts in the face of climate change, but cannot predict where the largest changes in abundance will occur. The associated emphasis on monitoring range boundaries to detect expansions or contractions has led to the discovery of sensitive bioindicators of the impacts of climate change and has improved our understanding of the ecological niche, threshold responses to environmental change, the nature of adaptation, speciation and co-evolution, species interactions, and invasion dynamics (Parmesan & Yohe, 2003; Holt & Keitt, 2005; Perry *et al*., 2005; Wilson *et al*., 2005). However, the current importance placed on monitoring range edges may cause the largest impacts of climate change to go undetected if tails of low abundance near species' range limits combined with linear variation in climate render relationships between climate and abundance weakest at the periphery of the range. Because changes in relative abundance are less frequently monitored by researchers and less easily perceived by the general public than changes in species presence or absence, some of the most dramatic responses to climate change in the interior of species range are likely being overlooked.

Achieving good measures of relative abundance across adequate spatial scales is difficult, in particular for species that are widely distributed, highly mobile, and difficult to observe directly. Population ecologists have overcome these difficulties to generate excellent abundance estimates for many populations, but due to research priorities and constraints, have tended to conduct these estimates year after year in one or very few localities. To adequately answer the questions posed by climate change, we need to add a spatial component to population–climate research that encompasses the range of climate variability projected by GCMs. Given the current paucity of data on how the abundance of most species varies with spatial climate variability, progress in this important area of research requires capitalizing on currently available coarse indices of abundance, as well as generation of new and better data on variation in abundance across the range.

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Do Habitat Corridors Provide Connectivity?

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Abstract: *Skeptics have questioned the empirical evidence that corridors provide landscape connectivity. Some also have suggested dangers of corridors. We reviewed published studies that empirically addressed whether corridors enhance or diminish the population viability of species in habitat patches connected by corridors. A randomized and replicated experimental design has not been used—and we argue is not required—to make inferences about the conservation value of corridors. Rather, studies can use observational or experimental analyses of parameters of target populations or movements of individual animals. Two of these approaches hold the greatest promise for progress, especially if the shortcomings of previous studies are remedied. First, experiments using demographic parameters as dependent variables—even if unreplicated—can demonstrate the demographic effects of particular corridors in particular landscapes. Such studies should measure demographic traits before and after treatment in both the treated area (corridor created or destroyed) and an untreated area (habitat patches isolated from one another). This approach is superior to observing the demographic conditions in various landscapes because of the tendency for corridor presence to be correlated with other variables, such as patch size, that can confound the analysis. Second, observations of movements by naturally dispersing animals in fragmented landscapes can demonstrate the conservation value of corridors more convincingly than can controlled experiments on animal movement. Such field observations relate directly to the type of animals (e.g., dispersing juveniles of target species) and the real landscapes that are the subject of decisions about corridor preservation. Future observational studies of animal movements should attempt to detect extra-corridor movements and focus on fragmentation-sensitive species for which corridors are likely to be proposed. Fewer than half of the 32 studies we reviewed provided persuasive data regarding the utility of corridors; other studies were inconclusive, largely due to design flaws. The evidence from well-designed studies suggests that corridors are valuable conservation tools. Those who would destroy the last remnants of natural connectivity should bear the burden of proving that corridor destruction will not harm target populations.*

Proveen Conectividad los Corredores de Hábitat?

Resumen: *Algunos escépticos han cuestionado la evidencia empírica de que los corredores proveen conectividad al paisaje. Otros han sugerido los peligros de los corredores. Revisamos estudios publicados que abordaron empíricamente si los corredores fomentan o disminuyen la viabilidad de poblaciones de especies en parches de hábitat conectados por corredores. A la fecha no se ha llevado a cabo un diseño experimental randomizado y con réplicas para realizar inferencias sobre el valor de los corresdores en la conservación—y nosotros argüimos que no es necesario. En cambio, los estudios pueden emplear análisis observacional o experimental de parámetros de poblaciones de interés o movimientos individuales de animales. Dos de estas aproximaciones son muy prometedoras y pueden progresar, especialmente si las limitantes de los estudios previos son remediadas. Primero, los experimentos que usan parámetros demográficos como variables dependientes—aún si no son replicados—pueden demostrar efectos demográficos de corredores en paisajes particulares. Estos estudios deberán medir características demográficas antes y después del tratamiento, tanto en el área tratada (corredor creado o destruído) como en un área no tratada (parches de hábitat aislados unos de otros). Esta aproximación es superior a observar las condiciones demográficas en varios paisajes puesto que la presencia de un corredor tiende a estar correlacionada con otras variables, como lo es el tamaño del*

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parche lo que puede confundir el análisis. Segundo, las observaciones de movimientos de animales que se desplazan normalmente en paisajes fragmentados puede demostrar el valor de los corredores en la conservación de manera mas convincente que los experimentos controlados sobre animales en movimiento. Este tipo de observaciones de campo están directamente relacionades con el tipo de animal (e.g., juveniles de la especie de interés dispersándose) y con el tipo de paisajes que están sujetos a las decisiones de preservación de corredores. Los estudios observacionales de movimientos de animales a futuro deberán tratar de detectar movimientos extra-corredores y enfocarse a especies sensitivas a la fragmentación y para las cuales los corredores son factibles a ser propuestos. Menos de la mitad de los 32 estudios revisados provee datos persuasivos referentes a la utilidad de los corredores; otros estudios fueron inconclusos, mayormente debido a diseños defectuosos. Las evidencias de estudios bien diseñados sugieren que los corredores son herramientas valiosas de conservación. Aquellos que intentan destruir los últimos remanentes de conectividad natural deberían sustentarse demostrando que la destrucción de los corredores no afectará a poblaciones de interés.

Introduction

Conservation biologists generally agree that landscape connectivity enhances population viability for many species and that, until recently, most species lived in well-connected landscapes (Gilpin & Soulé 1986; Noss 1987; Primack 1993; Noss & Cooperrider 1994; Hunter 1996; Meffe & Carroll 1997). Because urbanization and other human activities often sever natural connections among landscapes, many conservationists have advocated the retention of habitat corridors. In part, this approach has been justified by theoretical population models (e.g., metapopulation models, Gilpin & Hanski 1991). Such models demonstrate the utility only of habitat connectivity, however, which benefits population viability via the rescue effect (Brown & Kodric-Brown 1977) or other mechanisms. Conservation value accrues to corridors only if animals in real landscapes use corridors to bring about connectivity. Simberloff et al. (1992) argued that such evidence is lacking. Simberloff and Cox (1987), Simberloff et al. (1992), and Hess (1994) also argued that corridors might promote the spread of diseases, catastrophic disturbances (such as wildfires), or exotic species into the areas connected by corridors or might lure animals into areas—including the corridors themselves—where they experience high mortality (for a review see Hobbs 1992). A central concern is that funds spent acquiring corridors of questionable or unproven value might be better spent acquiring habitat areas for imperiled species, even if such areas are isolated (Simberloff et al. 1992).

We reviewed empirical papers that appeared relevant to the question, "Do corridors enhance or diminish the population viability of target species in the habitat patches connected by corridors?" Our goals were to make suggestions for future research on these issues and to evaluate scientific evidence that corridors serve as conduits for movement in a way that justifies their use as a conservation tool or that corridors have negative effects on target species.

Methods

We gathered papers on corridors (excluding modeling exercises) by searching for the word *corridor* in titles, abstracts, and keywords in all 1980–1997 volumes of *Auk, Biological Conservation, Condor, Conservation Biology, Ecological Applications, Ecology, Journal of Mammalogy, Journal of Wildlife Management, Wildlife Society Bulletin, Wilson Bulletin*, and recent monographs (e.g., Saunders & Hobbs 1991). We gleaned additional citations from relevant papers.

We define *corridor* as a linear habitat, embedded in a dissimilar matrix, that connects two or more larger blocks of habitat and that is proposed for conservation on the grounds that it will enhance or maintain the viability of specific wildlife populations in the habitat blocks. We define *passage* as travel via a corridor by individual animals from one habitat patch to another. Our definition of *corridor* explicitly excludes those linear habitats—such as riparian areas in agricultural landscapes—that support breeding populations of many species but do not connect larger habitat patches (e.g., Spackman & Hughes 1995). There are important conservation issues regarding nonconnective linear habitats, but we restricted our attention to linear patches of land whose conservation value is to allow passage between more significant habitat patches.

Nicholls and Margules (1991) and Inglis and Underwood (1992) discussed the formidable difficulties involved in designing a randomized and replicated experiment to test whether corridors enhance recolonization of habitat patches after local extinction. For such an experiment to be realistic, each experimental unit is an entire landscape, and there must be several replicate landscapes for each combination of treatments. Furthermore, we suggest that the species studied must be those that require connectivity on a landscape scale—fragmentation-sensitive species such as mammals with large home ranges—and that each species be studied individually. These requirements present staggering logistical and financial obstacles. Furthermore, to preclude confounding of corridor effects with other landscape effects, simple observation of various natural and anthropogenic landscapes is insufficient; the treatments must be applied randomly to those landscapes. The two essential "treatments" of the experiment, however, are creating and destroying corridors and causing local extinctions so that recolonization can occur. Randomly applying these treatments to replicate landscapes would be ethically questionable. Although one might argue that such an approach may be ethically acceptable for some abundant species, these are not the species for which conservation biologists design corridors, so the results would be of limited value.

Similar logistical, financial, and ethical problems would also bedevil any randomized and replicated experiment to determine the utility of corridors in enhancing population viability. Thus, it is not surprising we did not find a single paper that used a randomized and replicated experimental design and measured either recolonization rate or population viability as a dependent variable. Such a rigorous experiment may be unnecessary (cf. Hurlbert 1984), however. Even the most demanding critics of corridors concede that any habitat configuration that promotes immigration among patches will enhance population viability and likelihood of recolonization; the real issue is whether corridors allow such immigration in landscapes that would otherwise be fragmented (Simberloff et al. 1992). Thus, a researcher can shed light on the debate by conducting either experimental or observational analyses of parameters of target populations or parameters related to the movement of individual animals.

Parameters of target populations, such as immigration or individual survival rates, can be compared between habitat patches connected and unconnected by corridors or between landscapes where corridors are present or absent. Such studies should attempt to show that patch occupancy, abundance, colonization rate, immigration rate, disease rates, individual survival rate, frequency or intensity of disturbance, species richness, or occurrence of deleterious exotics increase or decrease in the presence of corridors relative to a landscape without corridors. Results can be meaningful only if they include a comparison to a landscape without corridors. Several widely cited papers (most notably MacClintock et al. 1977) are not helpful because they describe only a single landscape with corridors.

Because there is general agreement that landscape connectivity has at least the potential to enhance population viability, a study can simply attempt to show that animals use corridors in a way that provides such connectivity. Studies of parameters related to the movement of individual animals should attempt to confirm that animals (or diseases, disturbances, or exotic species) use corridors to move from one patch to another often enough to influence the population viability of the target species and that without corridors such movements would occur too rarely to influence the population.

We categorized each paper by the types of parameters it measured (population parameters, movements of individual animals, or the putative hazards of corridors) and whether the study used an observational or experimental approach. We then evaluated how each paper answered our research question of whether corridors enhance or diminish the population viability of species in habitat patches. In fairness, we note that the conservation value of corridors may not have been the research question of the investigations we reviewed.

Results and Discussion

Observational Studies Measuring Demographic Parameters

Seven studies (Table 1) measured either demographic parameters in relation to corridors or claimed to do so—six on birds and one on kangaroos. Five reported that corridors were beneficial for birds, one that corridors were not important for birds, and the seventh that corridors were not important for kangaroos. The main problem with this approach is severe risk of confounding; in addition, the dependent variable (especially in studies on birds) often was not closely tied to population viability.

Because each study in this group simply made observations in landscapes that were not designed to test the utility of corridors, all such studies risk the confounding of corridor effects with the effects of other factors that are highly correlated with corridors. For example, habitat patches lacking riparian corridors usually are more xeric, smaller, and further from large source populations than patches that abut such corridors. Patches without corridors may also be closer to homes, farms, cities, and human-subsidized predators. If patches with corridors are "better," it is difficult to determine whether the benefits are due to corridors or some other factors. Confounding is an inherent risk in any observational study because the treatments (corridors) are not randomly allocated to the experimental units. In studies for which randomization and true replication are impossible, investigators can minimize confounding in three ways. First, they should carefully select sites with and without corridors which are as similar as possible with respect to patch size, vegetation, moisture, distance to source populations, and proximity to disturbance. Second, they should forthrightly acknowledge and discuss plausible types of confounding. Third (and optionally), the investigator can collect ancillary data on movement routes of individual animals, especially on actual or potential extra-corridor movements. Such data can suggest whether observed differences are due to corridors or other factors correlated with corridors.

Table 1. Observational studies that compare patch occupancy, abundance, or other demographic parameters in habitat patches (or landscapes) with and without corridors.

Study[a]	*Dependent variable*	*Result*	*Treatment of confounding factors*	*Replication*	*Data on individual travel paths*
Arnold et al. 1991. Distribution and abundance of kangaroo in remnants of native vegetation in the central wheatbelt of Western Australia and the role of native vegetation along road verges and fencelines as linkages	patch occupancy and abundance	corridors not important	corridors, patch size, and proximity to next patch all highly correlated	yes	no
Date et al. 1991. Frugivorous pigeons, stepping stones, and weeds in northern New South Wales	patch occupancy and abundance	corridors not important for 5 spp of pigeons	elevation, corridors, patch size, and proximity to next patch all highly correlated	yes	no
Dmowski & Kozakiewicz 1990. Influence of a shrub corridor on movements of passerine birds to a lake littoral zone	numbers of forest birds visiting littoral zone near or away from a corridor[b]	increased number of birds in vicinity of corridor and in patch with a corridor	factors not discussed; only one corridor in the study	no	no
Dunning et al. 1995. Patch isolation, corridor effects, and colonization by a resident sparrow in a managed pine woodland	colonization rate	increased short-term colonization rates in landscape with corridors	sites well matched for landscape configuration and proximity to source patch; potential confounding factors discussed at length	no	no
Haas 1995. Dispersal and use of corridors by birds in wooded patches on an agricultural landscape	immigration rate (occupancy rates not reported)	immigration 15 times greater into patches connected by corridors (two wooded creeks); immigrants did nest in recipient patch	connected and unconnected pairs were separated by similar distances; no discussion of patch size, but map suggests that size and corridors are uncorrelated	yes	no
MacClintock et al. 1977. Evidence for the value of corridors and minimization of isolation in preservation of biotic diversity[c]	occupancy and species diversity	single 35-acre parcel connected by a short corridor was similar to "mainland"	no isolated fragment was studied	no	no
Saunders & de Rebeira 1991. Values of corridors to avian populations in a fragmented landscape	"immigration" rate (actually numbers of movements among patches by banded birds)	more "migration" between patches connected by corridors than between isolated patches	corridors, patch size, and proximity to next patch all highly correlated	yes	no

[a]*Abbreviated title; see literature cited for complete citation.*
[b]*Although not a demographic parameter, the inferred "visitation rate" might be correlated with dispersal or immigration rates, so this study (which did not assess animal travel in the single corridor) is included here.*
[c]*Although this study did not compare the single connected fragment to any corridorless fragment, it is widely cited as supporting the value of corridors as conduits.*

Because of how corridors and other factors are correlated in most fragmented landscapes, confounding is a less serious problem for studies that find corridors unimportant. Of the five studies claiming to show demographic benefits of corridors, only two (Dunning et al. 1995; Haas 1995) attempted to match the landscapes or patches with and without corridors with respect to potentially confounding factors and then discussed such confounding at some length. Although observational studies can never completely exorcise themselves of confounding, the careful treatment of these issues in these two papers greatly increased the credibility of the results.

Demographic parameters such as patch occupancy, abundance, and reproductive success influence the viability of populations in patches. Many observational studies, however, measured parameters less closely associated with the ability of the habitat patch to support animals. For instance, Haas (1995) reported that American Robins (*Turdus migratorius*) making a second nest attempt within a breeding season more often moved between patches connected by corridors than between unconnected sites. These data, however, do not suggest that the isolated patches had fewer robin nests or fewer second nest attempts than patches connected by corridors. Occupancy rates or nest density would have been a more direct measure of robin viability in the patches and probably could have been obtained with little or no extra field effort. In general, studies using short-term immigration rate must be interpreted with caution because, even if corridors help animals find suitable patches more rapidly, patches with and without corridors (if otherwise similar in size, vegetation, etc.) may have similar occupancy rates over the long term. An exception occurs in the case of species specializing in ephemeral habitat patches, such as the clearcuts used by the Bachman's Sparrows (*Aimophila aestivalis*) studied by Dunning et al. (1995). Because the clearcuts are suitable for only 4–7 years after creation, the colonization rate during the first 2 years after clearcutting was plausibly linked to viability in this case.

Although some bird species are reluctant to cross forest gaps (Bierregaard et al. 1992; Lens & Dhondt 1994; Desrochers & Hannon 1997), patch occupancy for birds is probably rarely influenced by the presence or lack of corridors a few hundred meters long. Bellamy et al. (1996) concluded similarly that, for birds, small gaps (mean 2.4 km, range 0.1–10 km) in forested landscapes did not "seriously hinder dispersal and recolonization opportunities," and Schmiegelow et al. (1997) found that 200-m wide clearcut barriers had less impact than expected on patch occupancy by forest birds. (This latter interpretation is ours; Schmiegelow et al. felt that 200-m barriers could isolate birds and attributed the small impact to counteracting factors.)

About half of these studies were unreplicated, consisting of one landscape with corridors and one without corridors. This fact was reported by the authors, allowing readers to make their own inferences. Although replication is desirable, it cannot ameliorate the more serious problem of confounding inherent in observational studies. As long as authors carefully address potentially confounding factors, observational studies can be valuable without replication.

Experiments Measuring Demographic Parameters in Different Landscapes

We found only four experimental studies that measured demographic parameters. Three studies (Mansergh & Scotts 1989; Machtans et al. 1996; Schmiegelow et al. 1997) destroyed or created corridors in real landscapes and collected pre- and post-manipulation data on both manipulated and unmanipulated areas. A third experiment (La Polla & Barrett 1993) measured animal abundance in highly artificial 20 × 20-m patches with and without corridors.

Perhaps the most defensible study was by Mansergh and Scotts (1989), who studied two subpopulations of a rare species, the mountain pygmy-possum (*Burramys parvus*). One subpopulation inhabited an intact landscape, whereas the formerly contiguous habitat of the second subpopulation had been fragmented by a ski development and an associated road. The fragmented area exhibited skewed sex ratios and lower survival rates than the intact area. After construction of a corridor, the population structure and survival rates in the ski resort changed to those observed in the undisturbed area. The study was not replicated, consisting of a single treated and a single control landscape. Nonetheless, Stewart-Oaten et al. (1986) demonstrate that if data are collected on both treatment and control areas before and after manipulation—as was the case here—investigators can make strong inferences regarding the effects of a particular unreplicated perturbation. Thus, although Mansergh and Scotts (1989) cannot make inferences about the utility of corridors in general, their study amply demonstrates the benefits of this particular corridor. We strongly encourage future studies to take the same vein as Mansergh and Scotts (1989) because, as such well-designed—albeit unreplicated—studies accumulate, each documenting local corridor effects, a more general pattern will gradually emerge.

The study of Machtans et al. (1996) similarly collected pretreatment and post-treatment data on both control and treatment areas, but it illustrates an important design limitation. It began with two intact landscapes, and the treatment consisted of creating a corridor out of formerly intact habitat and comparing bird movement rates across a control (intact) landscape to the landscape with a corridor. Because the study did not include a landscape without corridors, it is impossible to infer how readily birds would move through matrix habitat in the absence of a corridor (although the observations of Machtans et al. indicate that when a corridor was available, practically no forest birds were seen crossing the clearcut). Future experiments should contrast landscapes containing corridors with fragmented rather than intact landscapes. This can be achieved by either creating or destroying a corridor between two otherwise distinct patches.

In another experiment on bird response to forest fragmentation, Schmiegelow et al. (1997) reported two small but statistically significant benefits of 100-m wide riparian corridors: species turnover rate was higher in totally isolated fragments than in connected patches or in control areas, and diversity depended on fragment size only for the totally isolated fragments. This study was the most rigorous of the four in that pretreatment observations helped control for confounding (all fragments with corridors—but no isolated fragments—were adjacent to riparian areas) and because power analyses were used in the design phase to ensure adequate replication for statistical inference. Schmiegelow et al. (1997) noted, however, that the apparent benefits of corridors may have been an artifact of results from their smallest (1-ha) fragments because the effective size of each 1-ha fragment with corridors was doubled by the adjacent corridor habitat. Furthermore, the study was limited to short-term responses by the temporary nature of fragmentation (>1.5 m height growth in the first 2 years; Schmiegelow et al. 1997). This experimental design would be improved and made more relevant to conservation issues by altering it so that the area of habitat in the corridor has minimal influence on the dependent variable measured in the smallest habitat patch, by making longer-term observations (necessarily involving more-permanent fragmentation, such as by urban or agricultural activities), and by use of nonvolant focal species.

The more artificial experiment of La Polla and Barrett (1993) did not address the utility of corridors as a conservation tool. Through seeding they created uniform but artificial 20 × 20-m habitat patches that were connected or unconnected by 10-m-long corridors. They found higher numbers of voles in patches connected by corridors and attributed this difference in abundance to corridors. Nevertheless, rates of movement through their putative barriers (among "isolated" treatments and even among replicate sites) were comparable to those via corridors. In any event, the species (vole), corridor length (10 m), patch size (20 × 20 m), and matrix habitat (strips maintained in a mowed and tilled condition) suggest little relevance to real conservation problems and decisions. We see little prospect for elucidating the conservation value of corridors from experiments in settings so dissimilar to landscapes of conservation interest.

Observational Studies Measuring Movement of Individual Animals in Real Landscapes

If proponents and skeptics of corridors can agree on the value of connectivity in at least some situations, then it is not necessary to demonstrate the demographic effects of corridors. Instead, the issue is simply to document that animals will use corridors in a way that provides connectivity and that connectivity would be insufficient without the corridor. We found several studies (e.g., Catteral et al. 1991; Prevett 1991; Desrochers & Hannon 1997) that describe animal movements with respect to habitat edges, roads, suburbs, and domestic dogs, and other studies (Garrett & Franklin 1988) that anecdotally describe animal use of linear habitats. Some of these authors attempted to infer from these observations how animals might move through matrix or corridor lands. Although these studies can provide valuable understanding of the mechanisms underlying the use and avoidance of corridors and matrix, we excluded such studies as being too indirect to our question. We similarly excluded studies (e.g., Forys & Humphrey 1996) that document dispersal movements between habitat patches in fragmented landscapes but do not relate such movements to habitat corridors.

We considered in detail 17 observational studies (Table 2) that documented the presence or movements of nondisplaced animals (except for Reufenacht & Knight 1995) in landscapes that included corridors. Four of the 17 studies (Table 2, numbers 2, 4, 10, and 11) simply documented animal presence in corridors or the presence of individual animals in both habitat patches and corridors, without addressing the issue of whether animals made passages via the corridor from one habitat patch to another. Another six studies (Table 2, numbers 3, 5, 6, 9, 16, and 17) documented both presence and residence (i.e., probable breeding individuals) in the corridor. Of these, Vermeulen (1994) also documented movement rates, and Downes et al. (1997*a*) also compared corridor residents to forest-patch residents with respect to sex ratio, body mass, and reproductive potential. The occurrence of a resident population in a corridor—especially if residence occurs throughout its entire length—suggests that such corridors also would facilitate passage between patches. Maintaining resident populations of animals in wide corridors might be especially important when the distance between core populations is long, as is the case with grizzly bears (*Ursus arctos horribilis*) in much of the Rocky Mountains (Noss et al. 1996). Although territorial interactions between corridor residents and potential dispersers could inhibit dispersal by an individual from an adjacent patch, the corridor would still provide demographic benefits to the patches if there were modest immigration to and emigration from the corridor.

Reufenacht and Knight (1995) used a novel measure of corridor use—the number of midpoint crossings by displaced mice released in the corridor. They did not, however, report lengths of the corridors (aspen stringers), whether the stringers connected to any larger patches, where mice were released relative to the midpoints, or mouse travel distances. Hence, valid inferences from this study are limited.

Only 6 of the 17 studies (Table 2, numbers 1, 7, 8, 13, 14, and 15) provided strong evidence for passages by individual animals via corridors. Although all 6 suggested that such passages occur often enough to benefit the populations that interact via the corridor, only Suckling (1984) and Beier (1995) specifically reported on corridor passages by dispersing juveniles; both of these also reported the number of corridor transitions and the fraction of dispersers using corridors. Beier (1993, 1995) explicitly related this to the number of corridor passages

Table 2. Observations of animal movements with respect to potential corridors in landscapes not under control of the investigator.

Study[a]	Type of corridor use documented; measure of use	Documentation for (lack of) movement through matrix
1. Beier 1995. Dispersal of juvenile cougars in fragmented habitat	juvenile dispersal; fraction of dispersers making passages[b] and number of passages per corridor	radio-tagged animals never crossed urban matrix
2. Bennett et al. 1994. Corridor use and the elements of corridor quality: chipmunks and fencerows in a farmland mosaic	presence; number of captures in fencerows	not addressed
3. Bennett 1990. Habitat corridors and the conservation of small mammals in a fragmented forest environment	presence, residence, and movements between patch and corridor; number of marked animals caught in both patch and corridor	not addressed, but deemed improbable
4. Bentley & Catteral 1997. Use of brushland corridors and linear remnants by birds in southeastern Queensland Australia	presence; number of birds detected in corridor and in intact habitat	not addressed
5. Downes et al. 1997*a*. Use of corridors by mammals in fragmented Australian eucalypt forests	presence and residence; relative abundance, sex ratio, and body mass in corridor, patches, and matrix	nine native species did not use matrix (pasture), based on same sampling procedure used for corridor and patch
6. Henderson et al. 1985. Patchy environments and species survival: chipmunks in an agricultural mosaic	presence and residence; number of marked animals caught in both patch and corridor	not addressed; some animals moved via matrix
7. Heuer 1995. Wildlife corridors around developed areas in Banff National Park (wolf, lynx, and cougar; winter only)	passages[b] via corridor to other patches; number of passages per corridor (winter only)	deep snows and cliffs probably preclude movement outside of corridors
8. Johnsingh et al. 1990. Conservation status of the Chila-Motichur corridor for elephant movement in India	passage[b] via corridor to other patches; not quantified (implied that passage was frequent)	not addressed
9. Lindenmayer et al. 1993. Presence and abundance of arboreal marsupials in wildlife corridors within logged forest[c]	presence and residence; abundance of animals in linear habitats	not addressed
10. Lindenmayer et al. 1994. Patterns of use and microhabitat requirements of mountain brushtail possum in wildlife corridors	presence; number of detections in corridor	not addressed
11. Mock et al. 1992. Baldwin Otay Ranch wildlife corridor studies (deer, bobcat, and cougar)	presence; number of detections in corridor	not addressed; urban matrix likely impenetrable to bobcat and cougar
12. Ruefenacht & Knight 1995. Influences of corridor continuity and width on survival and movement of deermice	travel across midpoint of corridor (aspen stringers in sagebrush matrix) by displaced mouse; number of midpoint crossings	not addressed
13. Suckling 1984. Population ecology of the sugar glider in a system of habitat fragments	juvenile dispersal; fraction of dispersers using corridor for at least part of dispersal	at least 5 of 15 dispersals involved extra-corridor movement
14. Sutcliffe & Thomas 1996. Open corridors appear to facilitate dispersal by ringlet butterflies between woodland clearings	passage[b] via corridor to other patches; number of marked insects caught in both patch and corridor	indirect evidence suggests that less than 2% of movement occurs outside corridors
15. Tewes 1994. Habitat connectivity: importance to ocelot management and conservation	passage[b] via corridor to other patches; not quantified (implied that passage was frequent)	not addressed
16. Vermeulen 1994. Corridor function of a road verge for dispersal of stenotopic heathland ground beetles (nonvolant)	residence and movement; numbers of recaptures at various distances	apparently no movement via matrix, using same procedures as in corridor
17. Wegner & Merriam 1979. Movements by birds and small mammals between a wood and adjoining farmland habitats	presence and residence; number of marked animals caught in both patch and corridor	not addressed; some animals necessarily moved via matrix

[a] *Abbreviated title; see literature cited for complete citation; focal species listed if not in the title.*
[b] *A passage is when an animal enters a corridor from a habitat patch and travels to a habitat patch at other end of the corridor.*
[c] *This study focused on the value of linear strips as habitat, not as conduits for movement, but it has been cited as supporting the value of corridors as conduits.*

needed to enhance population viability. The greatest deficiency in such studies is that few attempted to document movements between patches via matrix land. In several studies (e.g., Wegner & Merriam 1979; Suckling 1984; Henderson et al. 1985), such extra-corridor movements clearly occurred, but the potential for such movements to connect habitat patches was not discussed or explicitly compared to corridor movements. Although several studies argued that extra-corridor movements were unlikely due to habitat unsuitability, only Beier (1995) documented this. Based on 181 overnight tracking sessions, Beier showed that the urban matrix land in his study was impermeable to the interpatch movements of cougars (*Puma concolor*).

Seven studies (Table 2, numbers 2, 4, 8, 9, 10, 12, and 15) did not attempt to document or even discuss the possibility of movements through a supposedly "hostile matrix." Other studies explicitly acknowledged the possibility of such movements but did not discuss the implications for population viability. For instance, Sutcliffe and Thomas (1996) showed that marked butterflies moved more often among habitat patches connected by corridors than among unconnected patches, and they presented indirect evidence that about 98% of movements are via these corridors. Nonetheless, might the 2% of butterfly movements through hostile habitat be sufficient to ensure the survival of isolated populations? And, if there were no corridors, might not some of the 98% find extra-corridor routes? Finally, several of the studies documented the movements of eastern chipmunks (*Tamias striatus*; Wegner & Merriam 1979; Henderson et al. 1985; Bennett et al. 1994) or other species that are unlikely to be the focus for corridor design—or even reasonable surrogates for species that are the focus—because they are relatively adaptable to anthropogenic habitats and tolerant of fragmentation.

Despite the shortcomings of many of these observational studies, the preponderance of evidence is that corridors almost certainly facilitate travel by many species. In the future this line of investigation can provide strong evidence for the utility of corridors. These studies should be improved in two ways. First, strong effort should be put into documenting actual travel paths, with equal emphasis on documenting both intra- and extra-corridor movement between patches. If extra-corridor movements do occur, their frequency relative to passages via corridors should be described quantitatively, and the implications for population viability should be discussed explicitly. Second, study species should be those most relevant to the design and implementation of corridors on real landscapes. Generally speaking, these are species that are area-dependent or fragmentation-sensitive, because they either have limited mobility or suffer high mortality moving between patches of suitable habitat.

Although lack of randomization—with its attendant potential for confounding—was a major drawback for observational studies of demographic parameters, this is not a serious issue in observational studies of animal movements because the experimental units are either individual animals or individual corridors. It is difficult to imagine that the selection of a travel path through a corridor or matrix would be correlated with an extraneous and potentially confounding factor.

Experiments on Movements of Individual Animals

We found four studies in which movements of individual animals were measured in landscapes under experimental control (Table 3). For several reasons, the results of these experiments have little or no relevance to the conservation value of corridors. First, the voles, fruit flies, mice, and salamanders in these experiments are neither the sort of species for which corridors are designed nor are they appropriate surrogates for such species. Second, all four studies used displaced animals as "simulated dispersers," usually by releasing them either directly into a corridor or into minuscule "patches" (3×3-m patch in Rosenberg 1994; a half-pint bottle in Forney & Gilpin 1989). These displaced animals and the environments in which they are released are at best poor indicators of how real dispersers would behave. The artificial corridors available to the animals have scant resemblance to the real landscapes across which animals must disperse. Finally, the lengths of corridors studied were 1 mm (Forney & Gilpin 1989), 40 m (Rosenberg 1994), and 300 m (Andreassen et al. 1996), and unstated (but clearly several hundred meters; Merriam & Lenoue 1990). Only Andreassen et al. (1996) explicitly compared the corridor length to the home-range diameter of the focal species (30 m), thus making the case that this distance may be relevant to dispersal movements.

We are skeptical of the arguments for "experimental model systems" (Ims et al. 1993; Wolff et al. 1997), especially when the results of studies are likely to be interpreted as lessons for conservation and land-use planning. In particular, experiments in highly controlled landscapes do not yield meaningful inferences about the conservation value of corridors in real landscapes. Nevertheless, elements of these experiments could be included in observational studies. For instance, Andreassen et al. (1996) found that movement was not inhibited by simulated competitors (caged voles) and predators (fox scats) in the corridors. Such treatments could be applied in real landscapes as well, either with true replication or in a before-after-control-impact-pair design (Stewart-Oaten et al. 1986), to yield valuable suggestions about the utility of corridors.

Studies Relevant to Negative Impacts of Corridors

Several authors have speculated on the negative impacts and other disadvantages of corridors (Noss 1987; Simberloff & Cox 1987; Simberloff et al. 1992; Hess 1994). We found ouly three studies with relevant results. Downes et al. (1997*b*) conducted the only study explicity designed to examine this issue: they found that exotic black rats (*Rattus rattus*) were abundant in corridors and that their abundance might affect the utility of the corridor for the native bush rat (*Rattus fuscipes*). The authors noted that black rats were matrix residents and

Table 3. Observations of animal movements with respect to potential corridors in landscapes under the experimental control of the investigator.

Study[a]	*Type of corridor use documented; measure of use*	*Documentation for (lack of) movement through matrix*
Andreassen et al. 1996. Optimal width of movement corridors for root voles	optimal width travel through 300-m-long artificial corridor by displaced voles; maximum distance, speed, and number of complete corridor transits	not addressed
Forney & Gilpin 1989. Spatial structure and population extinction: a study with *Drosophila*	transits via pinholes allowing movement between half-pint plastic bottles ("patches"); not quantified (flies not individually marked)	not addressed (no matrix available)[b]
Merriam & Lenoue 1990. Corridor use by small mammals: field measurements for three types of *Peromyscus leucopus*	presence in fencerow corridors by displaced radio-tagged mice released in farm fencerows; percentage of time traveling for 48 h, and total distance traveled in 48 h	not addressed; no corridorless landscape studied
Rosenberg 1994. Efficacy of biological corridors (for immigration movements by salamander)	travel through 40-m-long artificial corridor by displaced salamanders (released into 3×3-m patch); number of complete corridor transits	as many passages via matrix as via corridor

[a] *Abbreviated title; see literature cited for complete citation.*
[b] *Thus, this study demonstrated only that connectivity—not necessarily via corridors—enhances population persistence.*

did not use the corridors for inter-patch movement and that the bush rat would have essentially no prospect for inter-patch dispersal in the absence of corridors. Stoner (1996) found that mantled howling monkeys (*Alouatta palliata*) confined to linear habitats did have higher parasite loads than monkeys in large habitat blocks. The "corridor" site, however, was an area where the linear habitat was the only suitable habitat available, and Stoner wisely avoided making any inferences about the risks of movement corridors. Seabrook and Dettmann (1996) documented that exotic and poisonous cane toads (*Bufo marinus*) were more dense on "corridors" (roads and vehicle tracks) and probably used them to disperse. The corridors in this study (dirt roads) are certainly not the sort of wildlife movement routes that conservationists are trying to create. It has been widely observed that many pest species, including exotics and pathogens, disperse along disturbed habitats such as roads and roadsides (Noss & Cooperrider 1994). Furthermore, as was the case for most studies in Table 2, Seabrook and Dettmann (1996) provided no evidence of how fast the toad might spread through the matrix lands. In this regard, Bennett (1990) found that the exotic rodents in his study area were least influenced by lack of connectivity, being more abundant than the six native species in the smallest and most isolated patches. Hence, empirical evidence of negative impacts from corridors designed or preserved for conservation purposes has not yet emerged.

Conclusions

Generalizations about the biological value of corridors will remain elusive because of the species-specific nature of the problem. Indeed, there is no general answer to the question "Do corridors provide connectivity?" The question only makes sense in terms of a particular focal species and landscape. Nonetheless, we conclude that evidence from well-designed studies generally supports the utility of corridors as a conservation tool. Almost all studies on corridors suggest that they provide benefits to or are used by animals in real landscapes. Because most studies suffer from design limitations, only about 12 studies allow meaningful inferences of conservation value, 10 of which offer persuasive evidence that corridors provide sufficient connectivity to improve the viability of populations in habitats connected by corridors. No study has yet demonstrated negative impacts from conservation corridors. We are encouraged that the number and rigor of studies on these issues are increasing.

In comparing the approaches considered in this paper—experimental or observational analyses of target populations or individual animals—we suggest that progress will most rapidly proceed with one or both of two approaches. First, experiments using demographic parameters as dependent variables—even if unreplicated—can demonstrate the demographic effects of particular corridors in particular landscapes. Such studies should measure demographic traits before and after treatment in both the treated area (where a corridor was created or destroyed) and an untreated area (where habitat patches apparently are isolated from each other). Second, observations of movements by naturally dispersing animals in already fragmented landscapes can demonstrate the conservation value of corridors if efforts are made to document actual travel routes in both corridors and matrix land. Because corridor presence tends to be correlated and confounded with other variables, such as patch size and presence of riparian habitat, observations of demographic conditions in various landscapes is problematic, but careful selection of sites can reduce this risk.

We were surprised that most studies using birds as a focal species involved corridors and barriers that were small relative to their movement ability. We suspect that birds were selected at least in part because they are relatively easy to census, and we recognize that landscape scale is often beyond the control of the investigator. We urge greater attention to species with limited mobility and low population density, and, whenever possible, we urge observation on landscape scales relevant both to the focal species and to real conservation decisions.

The two approaches we advocate also can be used to evaluate proposed alternatives to corridors, such as "stepping stones" or managing "the entire landscape. . .as a matrix supporting the entire biotic community" (Simberloff et al. 1992). Controlled and replicated experiments on animal movement in artificial corridors have scant utility because they have little relevance to the kinds of landscapes and species for which decisions on conservation corridors will be made. Extrapolation across dissimilar species and spatial scales is generally unfounded. On the other hand, greatly lacking in the literature are studies of the community- or ecosystem-level effects of corridors. For example, rigorous studies of the effects of corridors on disturbance risk and spread, exotic species invasions, predation rates, and species richness or composition are absent.

Corridor skeptics have objected to the financial cost of corridors (Simberloff & Cox 1987; Simberloff et al. 1992). Because conservation funds are limited, each project should be considered carefully in terms of costs and benefits, including the alternative uses for the dollars that might be spent on corridors. There are certainly cases in which conservation dollars would be better spent acquiring high-quality but isolated patches of habitat for imperiled species, rather than acquiring corridors of dubious value. Many conservation projects are expensive, however, so this criticism has no unique relevance to corridor projects, which can be far cheaper than some alternatives. Furthermore, the more costly corridors are expensive precisely because they occur near large and growing human populations; the additional cost should be considered in light of proximity of the benefits—semblances of intact ecosystems, recreation—to those who ultimately pay for them.

Skeptics have correctly pointed out that the scientific evidence in support of corridors as a conservation tool has been weak. Developers and other opponents of conservation, however, frequently misrepresent this healthy spirit of inquiry and scientific self-criticism as a "disagreement among the experts." Thus, they are able to persuade planning agencies that habitat loss and fragmentation should proceed unhindered and that conservationists must bear the burden of proof for preserving each remaining corridor. Our review has shown that evidence from well-designed studies supports the utility of corridors as a conservation tool. All else being equal, and in the absence of complete information, it is safe to assume that a connected landscape is preferable to a fragmented landscape. Natural landscapes are generally more connected than landscapes altered by humans, and corridors are essentially a strategy to retain or enhance some of this natural connectivity (Noss 1987). Therefore, those who would destroy the last remnants of natural connectivity should bear the burden of proving that corridor destruction will not harm target populations.

Acknowledgments

We thank the investigators who led each of the studies we reviewed; we sincerely tried to treat each of you with respect and fairness. We recognize that some of the studies were not designed to test our hypotheses, and, despite our criticisms, we found useful information in each paper. C. S. Machtans reviewed an earlier draft of the manuscript.

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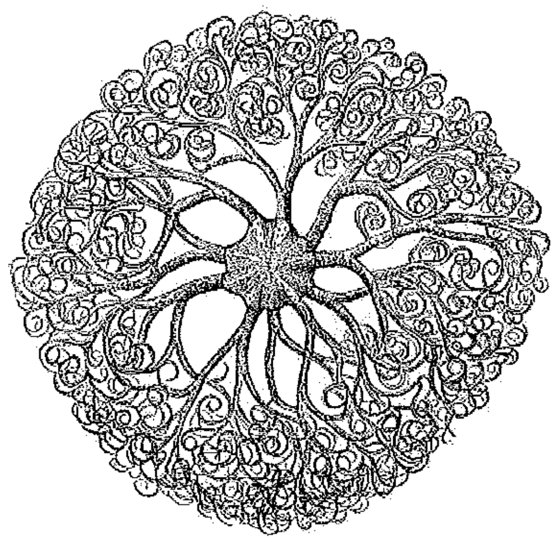

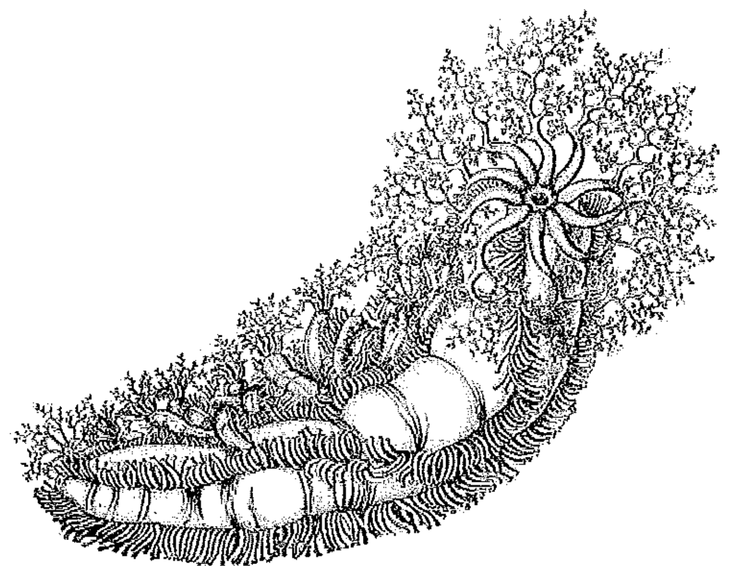

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Landscape connectivity: A conservation application of graph theory

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We use focal-species analysis to apply a graph-theoretic approach to landscape connectivity in the Coastal Plain of North Carolina. In doing so we demonstrate the utility of a mathematical graph as an ecological construct with respect to habitat connectivity. Graph theory is a well established mainstay of information technology and is concerned with highly efficient network flow. It employs fast algorithms and compact data structures that are easily adapted to landscape-level focal species analysis. American mink (Mustela vison) *and prothonotary warblers* (Protonotaria citrea) *share the same habitat but have different dispersal capabilities, and therefore provide interesting comparisons on connections in the landscape. We built graphs using GIS coverages to define habitat patches and determined the functional distance between the patches with least-cost path modeling. Using graph operations concerned with edge and node removal we found that the landscape is fundamentally connected for mink and fundamentally unconnected for prothonotary warblers. The advantage of a graph-theoretic approach over other modeling techniques is that it is a heuristic framework which can be applied with very little data and improved from the initial results. We demonstrate the use of graph theory in a metapopulation context, and suggest that graph theory as applied to conservation biology can provide leverage on applications concerned with landscape connectivity.*

Keywords: landscape ecology, graph theory, connectivity, modeling, metapopulations, focal species, American mink, *Mustela vison*, prothonotary warblers, *Protonotaria citrea*.

Introduction

The current trend in ecological research and land management is to focus on large biogeographic areas, which leaves the researcher and manager searching for landscape-scale data (Christensen *et al.*, 1996; Noss, 1996). Indeed, the interpretation of large spatial data, conceptually and technologically, can be the limiting factor in making conservation biology and ecosystem management a tangible goal. Because the internal heterogeneity of landscapes makes habitat-conservation planning a formidable challenge, modeling the spatial aspects of landscapes is a critical key to understanding. Until now, the varied approaches to building these models have focused primarily on two types of spatial data, coverages of vectors (polygons) or raster grids. We demonstrate the utility of a less familiar type of lattice, the graph (Harary, 1969), in determining landscape connectivity using focal-species analysis in an island model. A graph represents a binary landscape of habitat and non-habitat, where patches are described as nodes and the connections between them as edges.

Graph theory is a widely applied framework in geography, information technology and computer science. It is primarily concerned with maximally efficient flow or connectivity in networks (Gross and Yellen, 1999). To this end, graph-theoretic approaches can provide powerful leverage on ecological processes concerned with connectivity as defined by dispersal. In particular, graph theory has great potential for use in applications in a metapopulation context. Urban and Keitt (2000) have introdnced landscape-level graph-theory to ecologists, and here we build on that work by examining habitat connectivity for two species that share the same habitat but have different dispersal capabilities. Specifically, we ask how American mink (*Mustela vison*) and prothonotary warblers (*Protonotaria citrea*)

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perceive the same landscape. We explore the sensitivity of landscape connectivity through graph operations concerned with edge definition. We also examine each habitat patch's role in maintaining landscape connectivity in terms of source strength (Pnlliam, 1988) and long-distance traversability (den Boer, 1968; Levins, 1969) using graph operations concerned with node removal. This type of analysis is done very efficiently with graph theory. We also present an ecologically appealing way to calculate the functional distance between habitat patches using least-cost path modeling. Graph theory as applied to landscapes represents an important advance in spatially explicit modeling techniques because it is an additive framework: analysis of a simple, preliminary graph can prioritize further data collection to improve the graph model.

Study area and methods

Study area

Our study focuses on the Alligator River National Wildlife Refuge (NWR) and surrounding counties on the Coastal Plain of North Carolina (35°50′N; 75°55′W). It is a riverine and estuarine ecosystem with an area of almost 580 000 ha and over 1400 km of shoreline. The area is rich in wildlife habitat, dominated by the Alligator River NWR, the Pocosin Lakes NWR, Lake Mattamuskeet NWR, Swanquarter NWR, and a variety of other federal, state, and private wildlands (Figure 1).

The vegetation is characterized by the Southern Mixed Hardwoods forest community. The area has many diverse vegetation types, including fresh water swamps, pine woods and coastal vegetation. In the upland commnnity, dominant species include many types of oak (*Quercus* spp.), American Beech (*Fagus grandifolia*), and evergreen magnolia (*Magnolia grandiflora*). Mature stands may have five to nine codominants. The wet lowlands are dominated by bald cypress (*Taxomodium distichum*). The pine woods are dominated by longleaf pine (*Pinus palustris*), but loblolly pine (*P. taeda*) and slash pine (*P. elliottii*) are also important (Vankat, 1979).

Focal species

Because connectivity occurs at multiple scales and multiple functional levels (Noss, 1991), we have chosen two focal species to apply a graph-theoretic approach to connectivity. Focal species analysis is an essential tool for examining connectivity in a real landscape, as individual species have different spatial perceptions (O'Neill *et al.*, 1988). The American mink and the prothonotary warbler are appropriate candidates for focal species analysis as they share very similar habitat but have different ecological requirements, and fall into different categories as focal species. Both species are wetland dependent and indicators of wetland quality and abundance in a landscape. Both are charismatic. Furthermore, as meso-predators mink have small but important roles as a keystone species (Miller *et al.*, 1998/1999).

American mink are meso-level, semiaquatic carnivores that occur in riverine, lacustrine and palustrine environments (Gerell, 1970). In chief, they are nocturnal and their behavior largely depends on prey availability. They have a great deal of variation in their diet according to habitat type, season and prey availability (Dunstone and Birks, 1987). Muskrats (*Odantra zibethicus*) are a preferred prey item (Hamilton, 1940; Wilson, 1954), but mink diets in North Carolina are composed of aquatic and terrestrial animals, as well as semiaquatic elements (e.g. waterfowl; Wilson, 1954). In the southeast they have home ranges on the order of 1 ha and a dispersal range of roughly 25 km (Nowak, 1999).

Prothonotary warblers are neotropical migrants that breed in flooded or swampy mature woodlands. They have two very unusnal traits in common with wood warblers in that they are cavity nesters and prefer nest sites over water. They are forest interior birds that experience heavy to severe parasitism by brown-headed cowbirds (*Molothrus ater*); (Petit, 1999). They are primarily insectivorous, occasionally feeding on fruits or seed (Curson *et al.*, 1994). Preliminary data indicate that natal dispersal ranges from less than 1 km to greater than 12 km (Petit, 1999). Although this is formulated from a small sample, it is on the same order as other song bird dispersal (e.g. Nice, 1933; Sutherland *et al.*, 2000). Here, we posit warbler dispersal

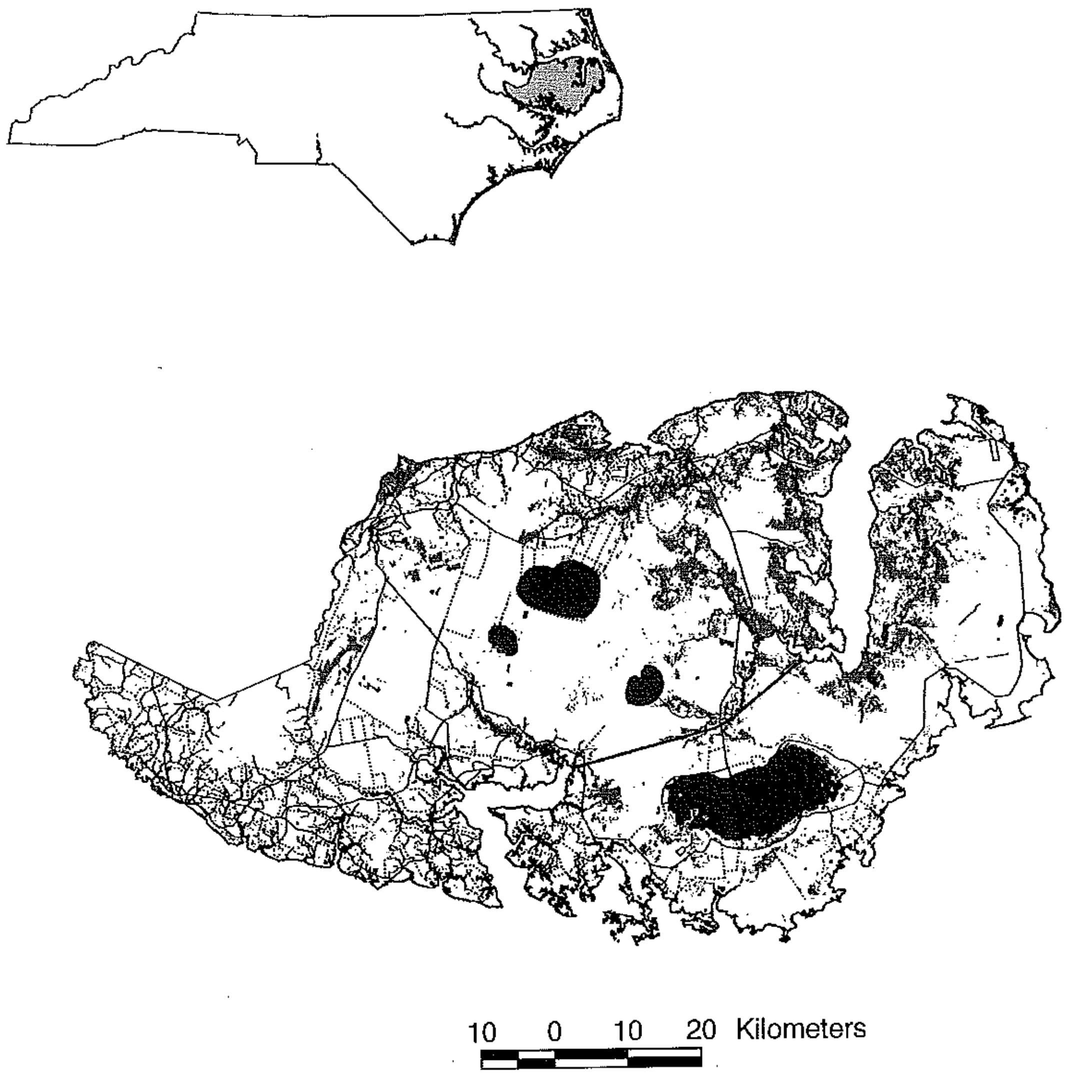

Figure 1. Study area in North Carolina with major roads and streams shown along with bottomland hardwood forest (focal species habitat) identified using GIS analysis.

to be 5 km and return to the uncertainty of this statement later.

Geospatial data

To our knowledge there are no current data on the spatial distribution of the focal species in our study area. The decline in trapping of the mink has perversely led to a decline in good biological information on the species. We are unaware of any work done with mink in the study area since Wilson's (1954) study. The Breeding Bird Survey indicates that this study area contains one of the highest concentrations of prothonotary warblers in the Southeast (Price *et al.*, 1995). Finer scale spatial information is not readily available.

Mink and prothonotary warblers are habitat specialists that use the same habitat. To identify habitat patches in the landscape we combined data from the US Fish and Wildlife Service's National Wetlands Inventory (http://wetlands.fws.gov) and a 1996 land-use coverage from the National Center for Geographic Information and Analysis (http://www.negia.ucsb.edu). Both were derived from Landsat 7 Thematic Mapper imagery with 30-m cells. Cells that were defined as being bottomland hardwood swamp or oak gum cypress swamp, aud riverine, lacustrine, or palustrine forested wetland were selected as habitat. These cells were then aggregated into regions using an eight-neighbor rule, and the intervening matrix was described as non-habitat. We also used zonal averaging techniques in an attempt to account for functional scaling in the habitat and found that the patch definition was robust. Transportation and hydrography digital line graphs were obtained for the study area from the US Geological Survey (http://www.usgs.gov) at 1:100 000 resolution.

Graph theory

Urban and Keitt (2000) give a general description of ecological applications of graph theory and readers should refer to any number of excellent texts on graphs as a primer (e.g. Gross and Yellen, 1999). However, this section describes the graph operations and definitions used in this study. While there are numerous excellent texts on the formalisms of graph theory (e.g. Gross and Yellen, 1999), the following largely conforms to Harary's (1969) classic text. A graph G is a set of nodes or vertices $V(G)$ and edges $E(G)$ such that each edge $e_{ij}=v_iv_j$ connects nodes v_i and v_j. A path in a graph is a unique sequence of nodes. The distance of a path from v_l to v_n is measured by the length of the unique set of edges implicitly defined by the path. A path is closed if $n_l=v_n$. Three or more nodes in a closed path is called a cycle. A path with no cycles is a tree. A tree that includes all the vertices in the graph is a spanning tree. The minimum spanning tree is the spanning tree in the graph with the shortest total length. The minimum spanning tree in effect represents the parsimoniously connected backbone of the graph.

A graph is connected if a path exists between each pair of nodes. An unconnected graph may include several connected components or subgraphs. A graph's diameter, $d(G)$, is the longest path between any two nodes in the graph, where the path length between those nodes is itself the shortest possible length. If nodes i and j are not adjacent, then the shortest path between them cannot be the distance between them but must use stepping-stones. Here, we use graph diameter (or diameter of the largest component) as an index to overall traversability of the habitat mosaic.

A graph is defined by two data structures: one that describes its nodes and one that describes its edges. We defined the nodes (habitat patches) by their spatial centroid and size (x, y, s). We defined the edges by a distance matrix $\mathbf{D}$ whose elements d_{ij} are the functional distances between patches i and j. For n patches $\mathbf{D}$ is n by n but because $d_{ij}=d_{ji}$ and $d_{ii}=d_{jj}=0$, it is sufficient to compute the lower triangle of the matrix.

Although the spatial array of nodes is simple to produce from a GIS, the other matrices are not as easy to define. Distance between patches can be measured in several different ways: edge to edge, centroid to centroid, centroid to edge, etc. However, measuring these as Euclidean distance makes little sense when the variance in mortality cost associated with traversal of the intervening habitats is large, and cost associated with traversal of the intervening habitats is large and spatially heterogeneous. Few organisms or even ecosystem processes, such as groundwater movement or wildfire spread, move in this way. To differing extents they are all constrained by the landscape. Good multidimensional models exist to predict some ecosystem processes (e.g. pollution plumes; Bear and Verruit, 1987) but not others. Spatially explicit models that simulate the dispersal of animals have been explored in some depth but the process is still poorly understood (Gaines and Bertness, 1993). Most are complex parametric models which are data-hungry. They require specific information and are hard to parameterize (Gustafson and Gardner, 1996).

For this reason, we have computed $\mathbf{D}$ not as Euclidean distances but as a series of least-cost paths on a cost surface appropriate to the organisms in question. These paths are designed to approximate the actual distance the focal species (or any other landscape agent, e.g. fire) covers moving from one patch to the next. For instance, in this riverine ecosystem, the path a mink might take from one side of a river delta to the other would likely involve traversing the shore for 10 km under cover, rather than a 5-km swim across open water. This allows the animal to use stepping-stones of other habitat (low cost) along the way rather than set off into an unknown habitat matrix (high cost). The least-cost modeling combines habitat quality and Euclidean distance in determining d_{ij}.

Cost was defined in 90-m cells (aggregated up from 30-m cells to improve processing time) by a surface comprised of x, y and z, where z was a uniform impedance that represented the cost of moving through that cell, i.e. its resistance to dispersal. Weights were approximated, based on perceived traversability. Cells corresponding to areas of habitat were given a weight of 0·5, all other forest types were given a weight of one. Cells classified as riverine/estuarine herbaceons were given a weight of two. Shrubland was given a weight of three. Sparsely vegetated cells (cultivated, managed herbaceous) were given a weight of four. Areas of development and large water bodies were given a weight of five. Streams were defined with a weight of one. We used grid functions inside a macro in ArcInfo 7.2.1 (ESRI, 1998) to iteratively loop through the array of patches and compnte d_{ij} for each unique pair of nodes in the array. The macro uses area-weighted distance functions to calculate least-cost paths. These functions are similar to Euclidean distance functions, but instead of working in geographical units they work in cost units.

We explored alternative methods for constructing **D**, including Euclidean distance and resistance-weighted distance between nodes. We found that the topology of the graph is robnst and not sensitive to the difference between least-cost path distance and Euclidean distance except at the scale of large obstacles in the landscape. For instance, least-cost paths in our model did not cross the 5-km mouth of the Alligator river when moving from the eastern side of the study area but chose a route through habitat instead. In this case Euclidean distance and least-cost path distance were quite different. The least-cost path technique is useful to land managers as the surface can be parameterized based on best available data. Thus, the surface can be tailored to features in the landscape for which the manager has knowledge. The surface can be refined as data becomes available, e.g. in the form of radio tracking.

Gustafson and Gardner (1996) found that dispersal routes are difficult to predict in even slightly heterogeneous landscapes. We have kept that in mind by building a simple cost surface that avoids committing the animals to movement patterns that are not readily possible to predict at 90-m resolution. We are not suggesting that the organisms modeled move purely according to least-cost paths. We use the framework because the distance of the least-cost path is a better approximation of the actual distance covered than a straight line between patches. Our goal has been to get a better estimate of distance traveled using least-cost and not to predict corridors. This modeling technique can be applied in a GIS, with limited spatial data, making it accessible to land managers and conservation practitioners. Despite these advantages, cost-surface analysis has been only occasionally used by ecologists (Krist and Brown, 1994; Walker and Craighead, 1997), bnt widely used in computer science which is concerned with optimal route planning (e.g. McGeoch, 1995; Bander and White, 1998). This type of analysis is also common in applications of artificial intelligence (e.g. Xia *et al.*, 1997).

To focus on scaling between the two focal species we chose to explicitly incorporate only patches greater than 100 ha in our analyses, as prothonotary warblers are not likely to persist in forest patches less than 100 ha (Petit, 1999). Using habitat patches greater than 100 ha results in 83 patches, roughly 83% of the 53 392 ha of possible habitat. Because all habitat patches, regardless of size, are given the lowest value on the cost surface, they are implicitly included in all analyses in that the species can traverse them easily as stepping-stones, accruing minimal cost.

We further defined edges by a dispersal probability matrix **P** that expresses the probability that an individual in patch i will disperse at least the distance between patch i and j. We computed the elements of **P** as negative exponential decay:

$$p_{ij} = -e^{(\theta \cdot d_{ij})} \qquad (1)$$

where θ is an extinction coefficient greater than 0. This way dispersal functions can be indexed by noting the tail distance corresponding to $P=0{\cdot}05$ is $-\ln(0{\cdot}05){\cdot}\theta^{-1}$. The tail distance for mink and prothonotary warbler are indexed as 25 km and 5 km, respectively. The tail distance is the distance to a selected point on the flat tail of the dispersal-distance function. Other curves are possible and Clark *et al.* (1999) provide a discussion of alternate dispersal kernels.

The graphs are described most succinctly by an adjacency matrix **A** in which $a_{ij}=1$ if nodes i and j are connected and 0 if not. We set $a_{ij}=1$ if $d_{ij}\leq 25$ km for mink and $d_{ij}\leq 5$ km for prothonotary warbler.

We can also define the graph's edges in terms of dispersal fluxes. Combining **P** and **s** allows us to compute dispersal flux from i to j:

$$f_{ij}=\frac{s_i}{s_{tot}}\cdot p'_{ij} \tag{2}$$

where s_i is relativized as the proportion of the total habitat area s_{tot} in i and p'_{ij} is p_{ij} normalized by the row sum of i in **P**. Because dispersal flux is asymmetrical ($f_{ij}\neq f_{ji}$) when $s_i\neq s_j$ we average the directions between nodes to give area-weighted dispersal flux w_{ij}:

$$w_{ij}=w_{ji}=1-\left(\frac{f_{ij}+f_{ji}}{2}\right) \tag{3}$$

Subtracting from 1 allows the flux value to have the smaller fluxes at greater distances. The area-weighted dispersal flux matrix allows us to compute a version of a minimum spanning tree with more dispersal biology incorporated.

Graph operations

With graph construction complete we performed two types of graph operations relating to connectivity: edge thresholding and node removal. Edge thresholding allows us to determine connectivity for mink and prothonotary warblers based on their tail dispersal distances. It also allows us to gauge the importance of variation of the tail distance. We removed edges from the graph iteratively with a edge distauce thresholded at 100 m to 50 000 m in 100 m increments. At each iteration the number of graph components, the number of nodes in the largest component and the diameter of the largest component were recorded.

Node removal is a way to examine the relative importance of habitat area and connectivity in the landscape. We used node removal to tell us about the dynamics of the entire landscape under different habitat-loss scenarios. Nodes were removed from the graph iteratively. We began with the entire graph and removed nodes randomly (with 100 repetitions), by minimum area, and by endnodes with the smallest area (Urban and Keitt, 2000). An endnode in a graph is a leaf in the spanning tree (here based on area-weighted flux) that is adjacent to only one other node. All edges incident to the removed node were also removed. At each iteration of the removal process the graph was analyzed to determine the importance of the patch to the graph's area-weighted dispersal flux (F), and traversability (T). Area-weighted dispersal flux was indexed as:

$$F=\sum_{i}^{n}\sum_{j,i\neq j}^{n}p_{ij}s_i \tag{4}$$

where s_i is the size of node i and p_{ij} is from Equation (1) above.

Traversability was indexed as the diameter of the largest component in the graph formed by the removal of the node:

$$T=d(G') \tag{5}$$

where G' is the largest component of G. We use F as an index of a patch's source strength, after Pulliam (1988). We use T in the sense of spreading-of-risk or rescue from catastrophe, after den Boer (1968) and Levins (1969).

Finally, we determined the importance of individual nodes to the entire landscape by assessing their individual contribution to area-weighted dispersal and traversability in the graph by computing F and T for the entire landscape, and then recomputing each with a single node removed from the graph. That node's impact is the difference between the intact metric and the metric that its removal elicited. Furthermore, we sought to determine the landscape's overall sensitivity to scale by repeating this process with edge definition thresholds from 2·5 km to 25 km, increasing in 2·5-km increments. We assessed the robustness of the patches' sensitivity rankings on F and T with Spearman's rank correlation, using the middle edge distance of 12·5 km to as the reference case.

Results

The mean distance between patches in the 83×83 matrix is 62·7 km. The study area and habitat patches are illustrated in Figure 1.

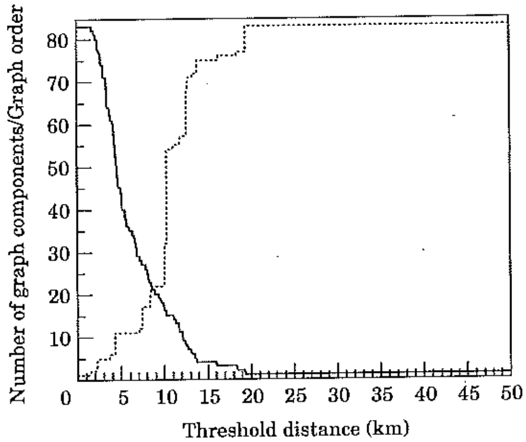

Figure 2. Number of graph components (-----) and graph order (—.—) as a function of effective edge distance.

Edge thresholding

The graph begins to disconnect and fragment into subgraphs at a 19 km edge distance, and quickly fragments into numerous components containing only a few nodes (Figure 2). The diameter of the largest component increases quickly with threshold distance, peaking at 20 km and decliniug slightly at greater thresholding distances (Figure 3). The edges are drawn as straight lines between patch centroids with 5, 10, 15 and 20 km thresholding distances in Figure 4, even though the actual paths are computed by least-cost aud are circuitous.

The distinct threshold at a 19-km functional edge distance (Figures 2–4) implies that the landscape as it stands now is perceived as being connected for species with

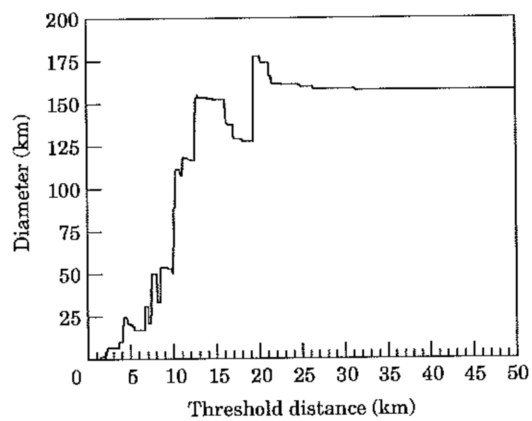

Figure 4. Diameter of the largest component remaining in the graph with increasing thresholding distance.

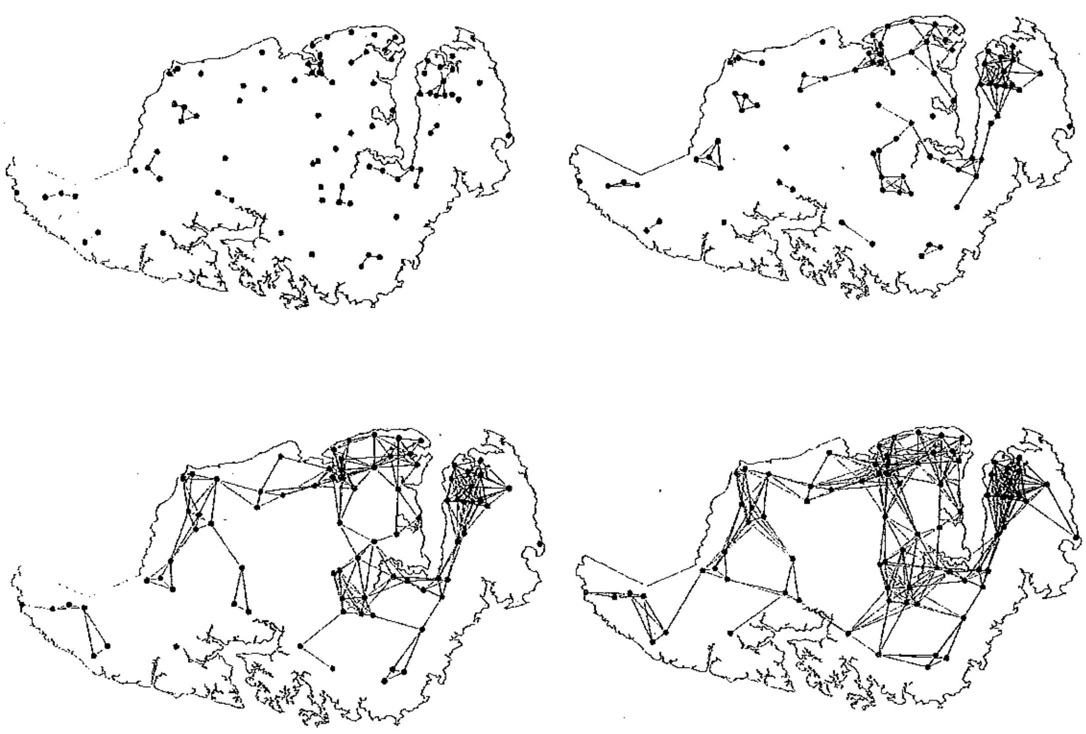

Figure 3. All graphs edges with increasing thresholded distances from 5 to 20 km.

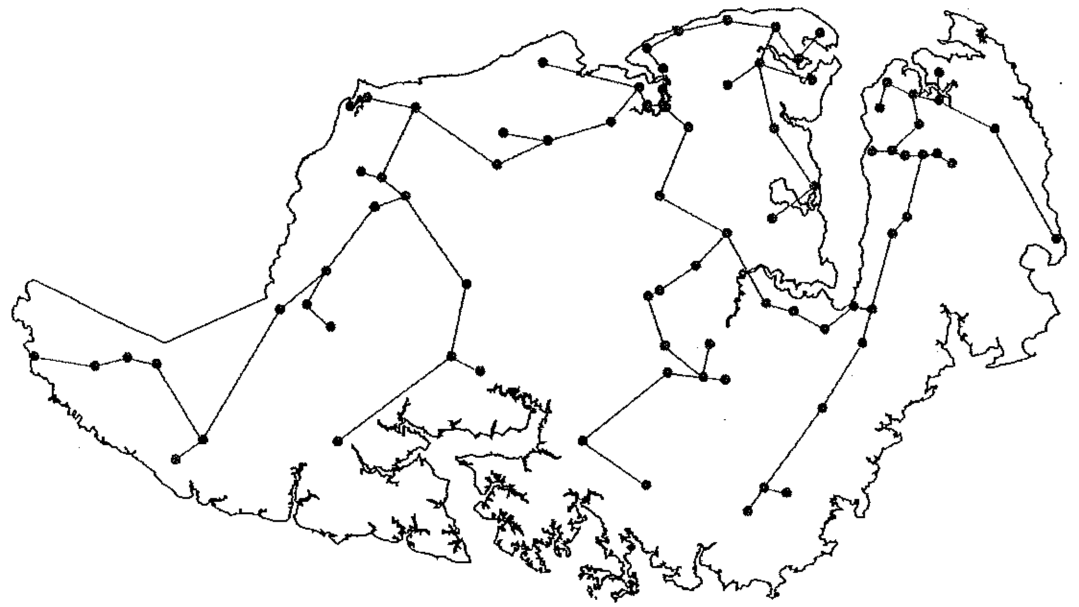

Figure 5. Minimum spanning tree for mink and prothonotary warbler based on distance.

a dispersal range of at least 20 km, and unconnected for species with a dispersal range of less than 20 km. Using this edge-thresholding scenario, and the language of percolation theory, the landscape percolates for mink but not for prothonotary warblers (Gardner *et al.*, 1987, 1992). Another way to envision this landscape is that prothonotary warblers may have a tendency to act as many discrete populations, while the robust connectivity of the landscape indicates that mink will act as one patchy population (Harrison, 1994).

For organisms with a 5-km dispersal distance, like the prothonotary warbler, the landscape graph divides into subgraphs. The implications from edge-thresholding operations are that some portions of the landscape have natnral units to partition for management. Edge thresholding also indicates nodes that are easily isolated. This can serve as an early blueprint for decisions regarding habitat acquisition or enhancement. For instance, this analysis indicates useful areas for patch creation via wetland restoration.

This preliminary exploration of edge thresholding can provide some idea of landscape connectivity relative to the dispersal capabilities (however nncertain) of mink and warblers. Using this framework it is easy to highlight important nodes and edges under different dispersal distances.

For mink, the minimum spanning tree on distance (Figure 5) is an excellent first look at habitat-specific connectivity in the landscape. The minimum spanning tree represents the backbone of the habitat in the matrix. The minimum spanning tree based on area-weighted dispersal flux (Figure 6) is very different. Couched in the mainland-island model of Harrison (1994), the tree is now weighted by larger patches which are expected to produce a larger number of propagules. The largest patch now radiates spokes which illustrates the spatial effect on dispersal under these kernels.

Node removal

Node removal is habitat removal. We measured the effects of node removal in two ways which can indicate a landscape's potential to provide conditions that foster metapopulations. Flux (F), as governed by area and dispersal potential, measures a node's influence to a landscape-level metapopulation. Flux can measure the patch's potential to act as a source in a source-sink metapopulation model (Pulliam, 1988). Traversability (T) is a function of the graph's diameter. In this light it can be thought of as a proxy for spreading-of-risk or long distance rescue (den Boer, 1968; Levins, 1969). T has the possibility to point

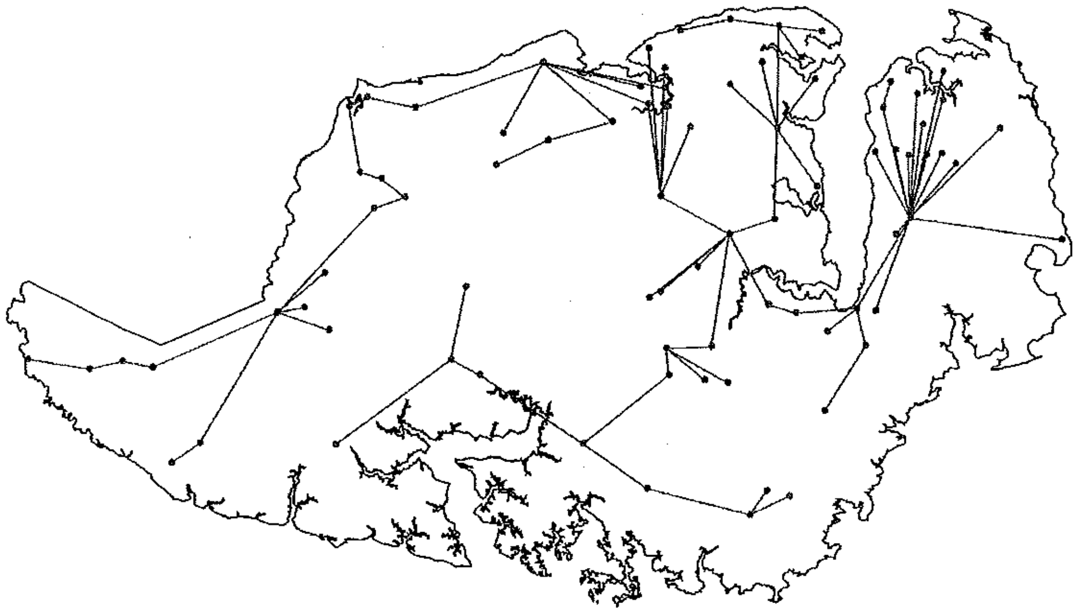

Figure 6. Area-weighted minimum spanning tree for mink with 25-km tail dispersal.

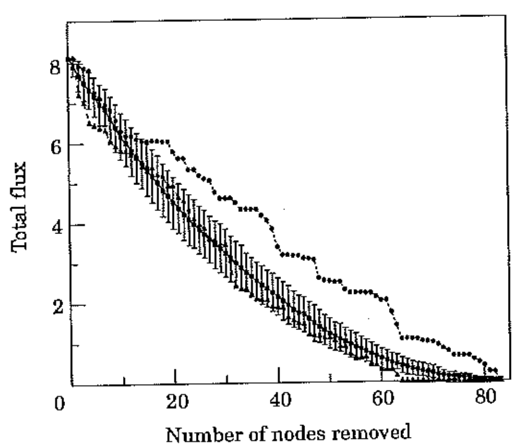

Figure 7. Area-weighted dispersal flux (F) as a function of three different node pruning scenarios. (-■-), random; (··▲··), minnode; (··●··), endnode. Graph defined with 25 km adjancency threshold.

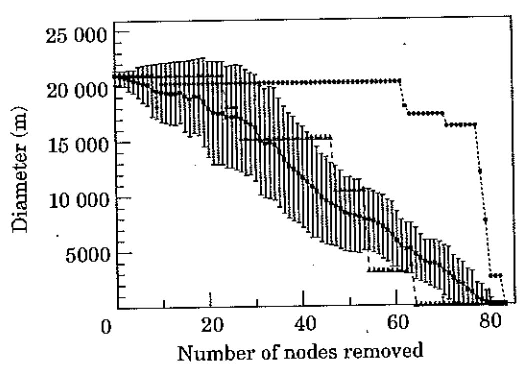

Figure 8. Traversability (T) of the largest graph component as a function of three different node prunning scenarios. (-■-), random; (··▲··), minnode; (··●··), endnode. Graph defined with 5 km adjancency threshold.

out important stepping-stone patches in the landscape. Source strength and long-distance rescue are well established in conservation biology. F and T are codified versions of those that fit into the graph context.

The different node removal scenarios give different pictures of the landscape. The better performance of endnode pruning over random or minimum area pruning for F indicates the tendency for endnodes to be less connected to the landscape (Figure 7). The advantage of endnode pruning is clear in its effect on T in the graph (Figure 8). Traversability of the graph is maintained with a majority of the graph nodes removed. The effect of endnode pruning on this landscape may indicate that this riverine ecosystem has a high degree of natural connectivity that an ecosystem not comprised of linearly connected features may not posses.

Area-weighted dispersal flux relies on **P** and **s** and is functionally similar for mink and prothonotary warblers under random, endnode, and minimum area pruning. The three thinning procedures produce similar results, although endnode pruning resulted in slightly higher flux values (Figure 7). The effect of different types of patch removal on traversability is markedly different for mink and prothonotary warblers. For mink, with a 25-km functional adjacency threshold, the three removal methods produce very similar results. For prothonotary warblers, with 5 km functional adjacency threshold, the random and minimum area pruning produce similar linear results but the effect of endnode pruning is substantially different. Traversability of the graph is not effected until ~75% of the nodes are removed (Figure 8).

Node sensitivity

The spatial arrangement of habitat patches in a landscape in combination with scale can influence measures of connectivity (Keitt *et al.*, 1997). Our two main metrics for connectivity, F and T, show differing responses to scale. Traversability, T, is indexed independently of patch area and is quite scale-dependent, showing little to no rank correlation between scales (Figure 9). Conversely, F is calculated explicitly with patch area and is very robust across scales. This is likely to be a function of patch area, and illuminates interesting management and ecological aspects of the landscape. In a Levins metapopulation model, T is analogons to spreading-of-risk and is sensitive to scale.

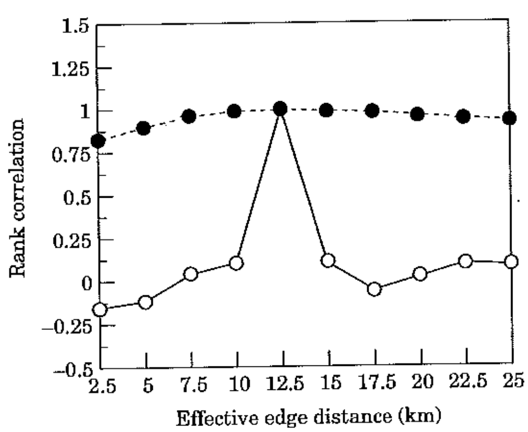

Figure 9. Correlation length of the vectors for area-weighted dispersal (F; --●--) and traversability (T; -○-) at varying scales. Reference variables are respective vectors at 12·5 km edge distance. Filled symbols are significant at $P<0{\cdot}005$.

In the more commonly used Pulliam model, F is analogous to source-sink strength and is not sensitive to scale because it is influenced most by close patches (short distance).

We have chosen two focal species defined by extremes in dispersal. In our model, mink can disperse five times farther than warblers through the landscape. Our results indicate that the high degree of connectivity for the mink and low connectivity for the warbler do not cause meaningful interpretation of node sensitivity at that scale. However, the great flexibility of the graph approach is the ability to instantly posit other degrees of dispersal based on edge distance. Figures 2–4 illustrate that the landscape begins to fragment seriously with a functional distance threshold between 10 and 15 km. These distances become important if we are concerned with issues of connectivity, as this is the scale that the landscape begins to meaningfully connect. Figure 10 shows a false-color composite of patch sensitivity at 12·5-km effective edge-distance that displays each patch's sensitivity to flux and traversability. We separated the metric F used above into recruitment potential (R) and dispersal flux (F'). Here, F' is a dispersal flux coefficient not influenced by area and computed only with $\mathbf{P}$ ($F=\sum p_{ij}$) so as to separate it from area. R is a neutral model of connectivity that is computed as a function of patch size alone ($R=\sum s_{ij}$). Each patch in the landscape was tested for sensitivity, and scored for the three metrics. This allows us to send R, F', and T to the red, green and blue color guns respectively. When the patches are displayed in a false-color composite (Figure 10), some interesting patterns emerge. In this image, patches that register high on metrics R, F', and T, saturate on all the colors and show up as white. Conversely, patches that show up as a dark color have registered low on every metric. Varions other shades are readily interpretable for each patch. Thus, node sensitivity analysis can illuminate nodes that have contextual importance. For instance, the blue patch indicated by the arrow in Figure 10 contributes to T but could be easily dismissed by a land manager as being unimportant because it is small and somewhat isolated. This type of view on the landscape can indicate crucial linkages or bottlenecks to conuectivity.

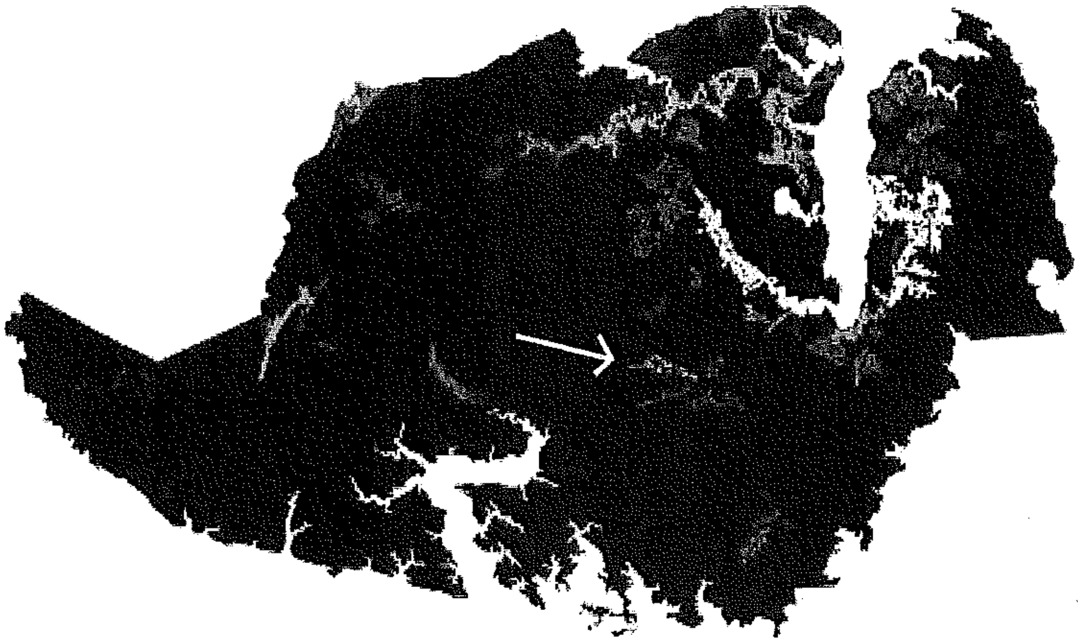

Figure 10. False-color composite of node sensitivity with 12·5 km functional edge distance. Red, recruitment potential (*R*); green, dispersal flux (*F'*); and blue, Traversability (*T*). For instance, the white arrow points to a patch with a high score for *T* and low scores for *R* and *F*.

Discussion

We found that mink and warblers perceived this landscape differently, as a function of their dispersal capabilities. For instance, Figures 2–4 show that the landscape has a great variability in connectivity depending on dispersal distance. According to the dispersal estimates used, we found that mink perceive this landscape as connected, while prothonotary warblers do not. Given a single patchy environment for mink we are able to exercise the graph and highlight the minimum spanning trees based on distance and area-weighted distance (Figures 5 and 6). These represent the parsimoniously connected backbone of the landscape. For the warbler, which experiences this landscape as fragmented components, these graph structures are not as meaningful, but minimum spanning trees based on connected subgraphs can provide utility in examining connectivity on a finer scale (not shown). The edge-thresholding operations serve as a continuous picture of connectivity in the landscape, and can be applied to other focal species.

In contrast to edge-thresholding procedures, which highlight mink and warbler spatial perceptions, the various habitat-removal scenarios allow us to determine patch function in reference to both species. Specifically, they allow us to envision, and then prioritize, habitat loss in the landscape. Habitat fragmentation and loss is one of the greatest threats to biodiversity and a great deal of management decisions focus around minimizing the impact of habitat reduction (Burgess and Sharp, 1981; Harris, 1984). In this landscape, like many others, habitat is managed by many different agencies and private landowners. Conflicting management paradigms virtually guarantee habitat alteration and loss. Given this, node removal also allows us to quickly gauge the tendency of a species to act like a metapopulation. For instance, we found that individual patches have different functions based on their size and position in the landscape. In the node removal graph perturbations, we found a tendency for endnodes to be poorly connected and therefore contribute weakly to dispersal flux and traversability. These results held for mink (Figure 7) and prothonotary warblers (Figure 8).

The node-sensitivity results show that patches also have contextual importance. Figure 10 is a powerful depiction of each patch's contribution to landscape connectivity, describing the landscape based on an edge threshold of 12·5 km. Given our dispersal estimates, this is an intermediate dispersal threshold not of direct importance to mink or warblers. However, this distance serves two pertinent functions in this landscape. First, it is approximately the distance at which the landscape begins to meaningfully connect. Second, it highlights the important versatility of the graph-theoretic approach and lets us instantly posit a gradient of dispersal thresholds. Given the overwhelming complexity of dispersal biology, the node sensitivity analysis provides an initial estimate of the relative importance of individual patches in the landscape. These preliminary analyses can also marshall further study by identifying those patches where field studies should be concentrated. For example, the blue patch highlighted in Figure 10, and surrounding green patches, offer themselves as likely candidates to determine the effectiveness of T and F.

Challenges persist in developing macroscopic landscape models. Although focal species analysis can enrich macroscopic approaches by producing a species-specific perspective to the analyses (O'Neill *et al.*, 1988; Pearson *et al.*, 1996), reliable habitat definition from relatively coarse spatial data (e.g. 30-m cells) is challenging for many species, and limited to habitat specialists. The use of the intervening non-habitat matrix is especially important, as this affects the functional scale at which patches are defined. Edge definition in a graph calls for dispersal biology that is often difficult to parameterize. However, well chosen focal species in a landscape can provide ecological and political effectiveness in issues of connectivity.

When appropriate species such as mink and warblers are available in a landscape, then focal species analysis is particularly well-suited to graphic representation, because ecological flux is a primary concern. The graph-theoretic approach differs from most focal species analyses as it allows one to use surrogates as a rapid assessment tool without long-term population data, although population data can (and should) be incorporated as knowledge of the system improves. It is a heuristic framework which is a robust way to represent connectivity in the landscape. The utility of applying graph theory to landscapes is that it allows managers and researchers to take an initial, but thorough, look at the spatial configuration of a landscape. It is applicable at any scale.

Another benefit of a graph-theoretic approach is that dispersal biology does not need to be fully understood for the graphs to be interpretable. To give context to the graph framework, we have postulated that dispersal for mink is 25 km and for prothonotary warblers, 5 km. We used a conservative negative exponential decay curve for dispersal probability, matrix $\mathbf{P}$. An advantage of the graph-theoretic approach is that gaming with alternative kernels is easy, and will affect dispersal in the landscape based on the spatial arrangement of patches. Dispersal biology is incredibly complex, and precise distances are virtually always unknown. Here our results and their interpretation are largely interpretable despite this uncertainty and can be immediately tailored to different dispersal estimates. Edge thresholding and node removal, as well as node sensitivity, are graph descriptors that are useful macroscopic metrics when dispersal can only be estimated (see Keitt *et al.*, 1997 for an additional example). From a management perspective, the graph can provide a powerful visualization of connectivity when used in conjunction with dispersal estimates such as those based on allometric relationship to body mass (see Sutherland *et al.*, 2000).

Prospectus

Land and conservation management is increasingly concerned with regional-scale habitat analyses. The development of graph theory in an ecological framework represents a promising step forward in that regard. Graph theory rests on a foundation of intensive study for computer networks which must be efficient. Therefore, the theory and algorithms are well developed; many are computationally optimal. Like metapopulation theory, the graph can merge landscape configuration and focal species biology to arrive at process-based measures of connection (Hanski, 1998; Urban and Keitt, 2000). The advantage of graph-theoretic approaches to conservation planners and researchers is that, while reasonable quality spatial data are required, long-term population data are not.

The conservation potential of graph theory is far from realized. The existing body of ecological work that considers landscape graphs is slim (Cantwell and Forman, 1993; Keitt *et al.*, 1997; Urban and Keitt, 2000). The most appealing feature of graph theory as applied to ecology is that it is a heuristic framework for management which is necessarily perpetual. With very little data, one can construct a graph of loosely-defined habitat patches and then explore the structure of the graph by considering a range of threshold distances to define edges. It is important that as more ecological information is collected it can be infused into the graph and consequently add more precision and confidence to the analyses. Graph theory can provide initial processing of landscape data and can serve as a guide to help develop and marshall landscape-scale plans, including the identification of sensitive areas across scales. This does not mean that graph theory should displace alternative approaches. We suggest graph theory as a computationally powerful adjunct to these other approaches. The simplicity and flexibility of graph-theoretic approaches to landscape connectivity offers much to land practitioners and can increase the scope and effectiveness of resource management.

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Conservation Biology

Contributed Paper

Use of Linkage Mapping and Centrality Analysis Across Habitat Gradients to Conserve Connectivity of Gray Wolf Populations in Western North America

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Abstract: *Centrality metrics evaluate paths between all possible pairwise combinations of sites on a landscape to rank the contribution of each site to facilitating ecological flows across the network of sites. Computational advances now allow application of centrality metrics to landscapes represented as continuous gradients of habitat quality. This avoids the binary classification of landscapes into patch and matrix required by patch-based graph analyses of connectivity. It also avoids the focus on delineating paths between individual pairs of core areas characteristic of most corridor- or linkage-mapping methods of connectivity analysis. Conservation of regional habitat connectivity has the potential to facilitate recovery of the gray wolf* (Canis lupus), *a species currently recolonizing portions of its historic range in the western United States. We applied 3 contrasting linkage-mapping methods (shortest path, current flow, and minimum-cost-maximum-flow) to spatial data representing wolf habitat to analyze connectivity between wolf populations in central Idaho and Yellowstone National Park (Wyoming). We then applied 3 analogous betweenness centrality metrics to analyze connectivity of wolf habitat throughout the northwestern United States and southwestern Canada to determine where it might be possible to facilitate range expansion and interpopulation dispersal. We developed software to facilitate application of centrality metrics. Shortest-path betweenness centrality identified a minimal network of linkages analogous to those identified by least-cost-path corridor mapping. Current flow and minimum-cost-maximum-flow betweenness centrality identified diffuse networks that included alternative linkages, which will allow greater flexibility in planning. Minimum-cost-maximum-flow betweenness centrality, by integrating both land cost and habitat capacity, allows connectivity to be considered within planning processes that seek to maximize species protection at minimum cost. Centrality analysis is relevant to conservation and landscape genetics at a range of spatial extents, but it may be most broadly applicable within single- and multispecies planning efforts to conserve regional habitat connectivity.*

Keywords: *Canis lupus*, centrality, circuit theory, corridor, graph theory, least cost path, network flow

Utilización del Mapeo de Vínculos y el Análisis de Centralidad en un Gradiente de Hábitats para Conservar la Conectividad de Poblaciones de Lobo Gris en el Occidente de Norte América

Resumen: *Las medidas de centralidad evalúan las vías entre todas las combinaciones pareadas posibles de sitios en un paisaje para clasificar la contribución de cada sitio en la facilitación de los flujos ecológicos en una red de sitios. Los avances de la computación permiten la aplicación de medidas de centralidad en paisajes representados como gradientes continuos de calidad de hábitat. Esto evita la clasificación binaria de paisajes en parches y matriz como lo requiere el análisis de grafos de conectividad basado en parches. Esto también evita el enfoque en la delineación de vías entre pares individuales de áreas núcleo característico*

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de la mayoría de los métodos de mapeo de corredores o de vínculos en el análisis de conectividad. La conservación de la conectividad de hábitat regional tiene el potencial de facilitar la recuperación del lobo gris (Canis lupus), *una especie que actualmente esta recolonizando porciones de su rango de distribución histórica en el occidente de Estados Unidos. Aplicamos 3 métodos de mapeo de vínculos contrastantes (vía más corta, flujo de corriente y costo mínimo-flujo máximo) a datos espaciales representando el hábitat de lobos para analizar la conectividad entre poblaciones de lobo en Idaho centra y el Parque Nacional Yellowstone (Wyoming). Posteriormente aplicamos 3 medidas de centralidad análogas para analizar la conectividad de hábitat de lobos en el noroeste de Estados Unidos y el suroeste de Canadá para determinar si sería posible facilitar la expansión del rango y la dispersión interpoblacional. Desarrollamos software para facilitar la aplicación de las medidas de centralidad. La centralidad de la vía más corta identificó una red mínima de vínculos análogos a los identificados por mapeo de corredores con la vía de menor costo. La centralidad de flujo actual y de costo mínimo-flujo máximo identificó redes difusas que incluyeron vínculos alternativos, que permitirán una mayor flexibilidad en la planificación. La centralidad de costo mínimo-flujo máximo, mediante la integración de costo de la tierra y la capacidad del hábitat, permite considerar a la conectividad en los procesos de planificación que buscan maximizar la protección de especies al menor costo. El análisis de centralidad es relevante para la conservación y la genética de paisaje en un rango de extensiones espaciales, pero puede ser ampliamente aplicable en esfuerzos de planificación de la conservación de la conectividad del hábitat de una o múltiples especies.*

Palabras Clave: *Canis lupus*, centralidad, corredor, flujo de redes, teoría de circuitos, teoría de grafos, vía de menor costo

Introduction

Consideration of landscape connectivity in conservation planning has increasingly shifted from a focus on preserving static landscape elements such as corridors to facilitating functional connectivity. Functional connectivity is defined as ecological processes such as demographic and genetic flows that support persistence of peripheral populations and long-term maintenance of a species' evolutionary potential (Taylor et al. 2006; Pressey et al. 2007). Due in part to computational limitations, most current reserve-design efforts remain focused on landscape pattern (e.g., selection of areas that capture species occurrences) (Cabeza & Moilanen 2001; Pressey et al. 2007). However, effective conservation of connectivity requires evaluation of how landscape composition and structure influence ecological and evolutionary processes at multiple levels of biological organization (Rayfield et al. 2011).

Here, we describe 3 contrasting methods of connectivity analysis that employ alternative assumptions concerning the relation between habitat and movement and offer complementary information for both corridor design and regional conservation planning. Graph theory provides a common conceptual framework that underlies all 3 methods. In graph theory, a graph (Fig. 1) is a set of nodes in which pairs of nodes may be connected by edges that represent functional connections (e.g., dispersal) between nodes (Urban et al. 2009). Edges may be assigned weights that represent an attribute such as habitat quality. A sequence of nodes connected by edges forms a path. Although they are highly abstracted depictions of landscape pattern, graphs may reveal emergent aspects of landscape structure that are not otherwise discernible.

Graph theory has been widely applied in landscape ecology and conservation planning (Urban et al. 2009). Such applications include analyses that represent continuous habitat gradients as a binary patch-matrix structure, with patches (nodes) linked by edges whose attributes (e.g., weight) are defined on the basis of geographic distance or attributes of the intervening matrix (Bodin et al. 2006; Urban et al. 2009). This patch-based approach contrasts with methods used within geographic information systems (GIS) to delineate corridors between pairs of habitat patches in raster grids (Beier et al. 2008). Although seldom transparent to the user, graph algorithms also underlie these latter methods, which analyze continuous habitat gradients by representing each raster cell (pixel) as a node in a regular lattice (an arrangement of points in a regular pattern). Edges in such graphs connect only a node and its immediately adjacent neighbors. We term these types of graphs *landscape lattices* (Supporting Information) in contrast to graphs that delineate discrete patches within a landscape matrix (Supporting Information).

Corridor-delineation methods available in GIS software analyze raster data by representing cost (e.g., energetic cost or mortality risk) of movement through different habitat types as distance (points in less permeable habitat are conceived as farther apart). Such methods then use computationally efficient algorithms to identify the route between 2 predetermined endpoints that has the shortest total distance (least total cost) (Supporting Information; Newman 2010). We use the term *shortest path* (Supporting Information) in place of *least-cost path* to avoid confusing the cost of moving between patches with monetary cost (e.g., of land purchase) (Newman 2010). Recent applications of shortest-path methods have

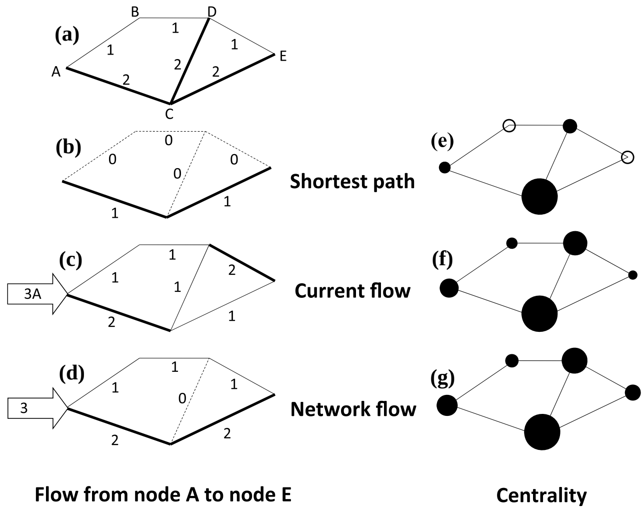

Figure 1. A simple graph with 5 nodes and 6 edges demonstrates contrasts between graph analyses with shortest- or least-cost-path, current-flow, and maximum-flow methods. (a) Edge values shown may be derived from models of habitat quality. Edge values are proportional to conductance (current flow) and flow capacity (minimum-cost-maximum-flow) and inversely proportional to distance (shortest path). (b–d) Pairwise flow between nodes A and E, with line widths proportional to flow ([b] shortest-path analysis; [c] current-flow analysis with a 3-ampere source at A; [d] maximum-flow analysis with a 3-unit flow source at A and a 3-unit flow sink at E). (e–g) Centrality analysis of flow between all node pairs in the graph, with node sizes proportional to centrality values (open circles indicate zero values; [e] shortest-path betweenness centrality; [f] current-flow betweenness centrality; and [g] maximum-flow betweenness centrality).

broadened their focus from identifying a single path or corridor to identifying a set of near-optimal paths that may be termed a habitat linkage or landscape linkage (Chetkiewicz et al. 2006; Beier et al. 2008).

We compared shortest-path analysis with 2 alternate connectivity-analysis methods, current flow and network flow. Current-flow methods examine probabilistic flow across all possible paths, whereas network-flow methods identify optimal flow that could use but may not use all possible paths. Current-flow models use algorithms from electrical-circuit theory to evaluate connectivity (McRae et al. 2008; Supporting Information). These methods treat landscapes as conductive surfaces (i.e., networks of nodes connected by resistors). When current is injected into a source node and allowed to flow across a network until it reaches a target node, the amount of current flowing through each intermediate node reflects the likelihood that a "random walker" leaving the source node and moving along edges with probabilities proportional to edge weights will pass through the intermediate node on its way to the target node. By modeling the movement of random walkers, current-flow models integrate the contributions of all possible pathways across a landscape or network (Fig. 1c). As in electrical circuits, the addition of new pathways increases connectivity by distributing flow across more routes (McRae et al. 2008).

Network-flow models frame connectivity analysis as an optimization problem rather than as probabilistic movement (Supporting Information; Phillips et al. 2008). Network flow is analogous to the behavior of water in a pipe, in that it has constrained capacity (the amount of flow on an edge cannot exceed its capacity) and flow is conserved (the amount of flow into a node equals the amount of flow out of it, except when the node is a source or sink). There are several types of network-flow analyses. In a maximum-flow analysis, each edge is assigned a flow less than or equal to its capacity, which maximizes total flow between a source and a sink node. Although there may be many alternative sets of paths in a network that allow the maximum flow, computationally efficient maximum-flow algorithms tend to identify maximum flows with low total number of edges (Ahuja et al. 1993). Minimum-cost-maximum-flow algorithms, in contrast, identify which of the alternative maximum-flow sets has minimum total cost (here, monetary cost of land acquisition or management). Minimum-cost-maximum-flow may be more informative than maximum-flow analyses on landscape lattices, particularly when edge capacities are relatively similar, because a large number of equivalent maximum-flow solutions exist on such lattices.

Centrality and Regional Connectivity Analysis

Shortest path, current flow, and network flow have largely been applied to evaluate options for linking predetermined endpoints rather than analyzing habitat connectivity across the landscape (but see Phillips et al. 2008). However, a group of analogous graph-theory metrics are based on the concept of centrality (Supporting Information). These metrics consider paths between all possible pairs of nodes in order to evaluate the role of each node in mediating ecological flows (Bunn et al. 2000; Borgatti 2005). The loss of a node that lies on a large proportion of the paths in the network would disproportionately lengthen distances or transit times between nodes (Brandes 2001). A wide variety of centrality metrics have been proposed (Newman 2010). Many have been applied to analyze patch-based representations of landscapes (Bodin & Norberg 2007; Estrada & Bodin 2008). We did not attempt to comprehensively review centrality metrics; rather, we focused on 3 metrics that are analogous to the 3 major methods of linkage mapping described above (Chetiewicz et al. 2006; McRae et al. 2008; Phillips et al. 2008).

Centrality calculations increase in computational complexity at a polynomial rate (typically quadratic to cubic) as the number of nodes increases (Ahuja et al. 1993). Although centrality analysis has been applied to patch-based representations of landscapes, networks were typically limited to hundreds of nodes or less (Estrada & Bodin 2008). Computationally efficient algorithms for analysis of large networks, which have recently been developed for purposes such as ranking web pages on the internet, allow analysis of landscape connectivity at a resolution that makes simplifying assumptions less necessary (Hagberg et al. 2008). This facilitates application of centrality metrics to contexts in which a continuous habitat gradient is more ecologically realistic than a binary patch-matrix framework (Chetkiewicz et al. 2006).

Because centrality analysis produces a continuous surface of values, it facilitates integration of the 3 connectivity-analysis methods into commonly used reserve design algorithms along with inputs representing species distribution or other conservation criteria (e.g., Possingham et al. 2000; Moilanen et al. 2009). The methods we developed thus avoid 2 key simplifications of landscape complexity. Because centrality metrics analyze paths between all node pairs, we avoided the a priori identification of endpoints necessary in current methods for delineating habitat linkages. By applying centrality analysis to graphs that represent landscapes as regular lattices, we avoided the binary classification of landscapes into patch and matrix required by patch-based graph analyses.

We used shortest path, current flow, and minimum-cost-maximum-flow (Supporting Information) to delineate habitat linkages between a single source and target patch and contrasted the results. We then developed 3 analogous centrality metrics that analyze connectivity across a landscape without reference to specific source and target patches. We contrasted results from the centrality metrics and assessed their relevance to regional conservation planning in a case study of a gray wolf (*Canis lupus*) metapopulation in the northwestern United States and southwestern Canada.

Methods

Linkage Analysis Methods and Their Analogous Centrality Metrics

Assumptions underlying the 3 methods of habitat-connectivity analysis affect conclusions about the contributions of different edges to connectivity (Fig. 1). In a simple example graph, shortest-path analysis assigns all priority to a single path with the least cumulative distance (Fig. 1b). Current-flow analysis identifies 2 edges with highest current (used most frequently by random walkers). All other edges have lower but nonzero current levels that indicate the degree to which the other edges provide alternative pathways for random walkers moving from the source node to the target node (Fig. 1c) (Newman 2005; McRae et al. 2008). Maximum-flow analysis between source A and sink E (Fig. 1d) identifies a path with relatively high flow and a path with relatively low flow. Maximum-flow analysis assigns zero flow to edges not on these paths because these edges cannot contribute to increasing the total flow. Because there is only one maximum-flow solution for flow from A to E in Figure 1, minimum-cost-maximum-flow would be identical to maximum-flow.

Centrality analyses extend these methods from single pairs of source-target nodes to all pairs of nodes in a graph (Newman 2010). The 3 centrality metrics considered here are variants of betweenness centrality (BC), in that they measure to what extent a node contributes to paths or flows between all other nodes (Borgatti & Everett 2006; Newman 2010). Shortest-path BC identifies the one or several shortest (geodesic) paths that connect each pair of nodes on a graph and counts the number of such shortest paths in which a node is included (Borgatti & Everett 2006). Current-flow BC assesses the centrality of a node on the basis of how often, summed over all node pairs, the node is traversed by a random walk between 2 other nodes (Newman 2005). Minimum-cost-maximum-flow BC evaluates a node's contribution to connectivity on the basis of portion of the minimum-cost-maximum-flow that must pass through that node, summed over all node pairs (Freeman et al. 1991).

In Figure 1 shortest-path BC (Fig. 1e; Supporting Information) resembles shortest-path results between a node pair (Fig. 1b) because it assigns high centrality to node C, which lies on the shortest path between many node pairs, and zero centrality to nodes (B, E), which do not lie on the shortest paths between any pair of nodes. Current-flow BC (Fig. 1f; Supporting Information) ranks the importance (for facilitating flow) of nodes similarly as does shortest-path BC, but centrality values are more evenly distributed among nodes and there are no nodes of zero centrality due to the model's random-walk behavior. Maximum-flow BC ranks nodes similarly to current flow BC, but values are distributed more evenly (Fig. 1g). If all edges have equal cost, results of minimum-cost-maximum-flow BC (not shown) resemble maximum-flow BC.

Case Study

The gray wolf was extirpated from the northwestern United States by the 1940s, but it remained extant through much of southwestern Canada (Boyd & Pletscher 1999; Wayne & Hedrick 2011). Natal dispersal of wolves averages 100 km (Boyd & Pletscher 1999). Natural

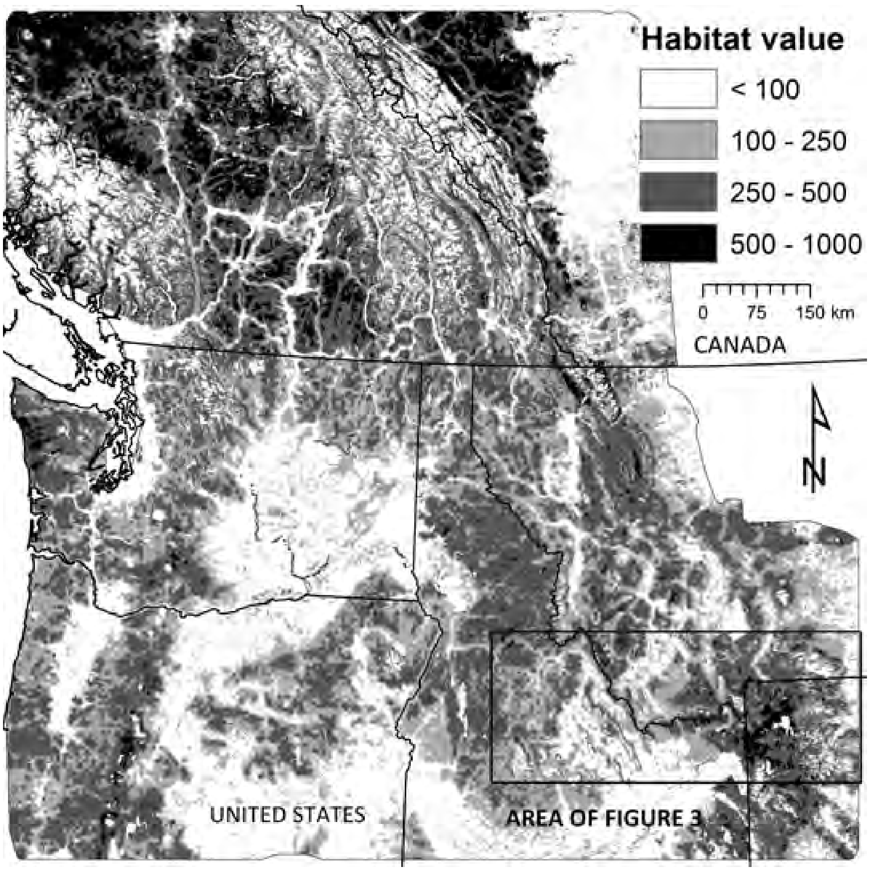

Figure 2. Model of habitat quality for gray wolf in the northwestern United States and southwestern Canada on the basis of land cover, slope, roads, and human population data. Edge weights in the connectivity analyses (Figs. 3–4) are derived from this habitat model.

recolonization via dispersal from Canada reestablished wolves in northwestern Montana in the 1980s and in northern Washington in 2008 (Wayne & Hedrick 2011). Reintroduction of wolves to central Idaho and northwestern Wyoming in 1995–1996 resulted in populations of >1000 in those areas and subsequent dispersal into Oregon, Utah, and Colorado (Wayne and Hedrick 2011). However, ongoing litigation has focused attention on whether habitat connectivity in the U.S. northern Rocky Mountains is sufficient to ensure continued genetic exchange between the region's 3 major wolf populations (Vonholdt et al. 2011; Wayne & Hedrick 2010). Analysis of habitat connectivity for the wolf may identify likely sources of natural dispersal from extant populations into currently unoccupied habitat and evaluate what areas have the greatest probability of facilitating continuing exchange among existing populations.

In developing a habitat model over this region (the U.S. states of Washington, Oregon, Montana, Idaho, and Wyoming, and the southern portions of the Canadian provinces of Alberta and British Columbia), we were constrained by the limited set of habitat variables for which data are available in all jurisdictions. Although empirical models of wolf habitat have been developed for the U.S. northern Rocky Mountains (Oakleaf et al. 2006), data are not available to allow their extrapolation across the entire region. We sought to demonstrate application of new methods of connectivity analysis rather than developing new habitat models. We therefore used a previously published habitat model (Fig. 2) that predicted wolf habitat quality from data on land cover, primary productivity, slope, road density, and human population density (Carroll et al. 2006). Details of the habitat model are in Supporting Information. We used a metric combining road density and human population density to represent factors negatively associated with wolf survival (Fuller et al. 2003). Because estimates of ungulate abundance are inconsistent across jurisdictional boundaries, we used land cover and tasseled-cap greenness, a satellite-imagery-derived metric, as a surrogate for prey density. Because wolves have reduced hunting success on steep terrain, we incorporated a negative effect of slope (Carroll et al. 2006). Because the above habitat variables may affect selection of dispersal habitat differently than selection of habitat for permanent occupancy, a subsequent refinement of the analysis with a model that is based on dispersal data would improve its accuracy (see Discussion).

Graph Analyses at Multiple Resolutions and Extents

We developed and contrasted analyses of wolf habitat connectivity at 2 spatial extents. First, we applied 3 linkage-mapping methods (shortest path, current flow, and minimum-cost-maximum-flow) at the local extent to analyze connectivity between 2 areas occupied by source populations of wolves in central Idaho and Yellowstone National Park, Wyoming (Fig. 2). In this analysis, we divided the region into a lattice of hexagons, each with an area of 5 km^2. Each hexagon's centroid became a graph node (total 21,889 nodes) that was connected to the 6 hexagons that were its immediate neighbors. Linkage mapping is a special case of centrality analysis termed subset centrality. In contrast with the application of centrality to analyze all pairs of nodes in a graph, subset centrality considers paths between the nodes of the graph that fall within the source and target patches (Hagberg et al. 2008). To illustrate application of the local minimum-cost-maximum-flow analysis, we simply assigned private lands 2 times the management or acquisition cost of public lands.

We then applied 3 centrality metrics (shortest-path, current flow, and minimum-cost-maximum-flow BC) analogous to the linkage-mapping metrics to assess connectivity across the northwestern United States and southwestern Canada (Fig. 2) with 2 lattices, one of hexagons with areas of 50 km^2 ($n = 23{,}831$ nodes) and one of hexagons with areas of 100 km^2 ($n = 9601$ nodes). Use of 2 resolutions was necessary because calculation of minimum-cost-maximum-flow BC was computationally infeasible on the higher-resolution graph of 23,831 nodes. For the regional minimum-cost-maximum-flow analysis, we assigned each node a cost of 1. Minimum-cost-maximum-flow analysis with uniform cost values on all nodes results in identification of the maximum-flow solution of minimum total area (minimum number of nodes).

In all analyses, we used either undirected or symmetric directed graphs (Supporting Information) in which the weight of edge i-j (from node i to j) equaled the weight of the edge j-i (Newman 2010). Edge weights were derived from the mean habitat-quality value of the edge's 2 end nodes. We used untransformed habitat-quality values from the conceptual model, which ranged from 1 to 1000, to derive conductance (current flow) and capacity (minimum-cost-maximum-flow) (Supporting Information). We used the reciprocal of the mean habitat-quality value to represent distance in calculating the shortest-path metrics. Each of the 3 methods thus assigned different attributes to the graph edges (distance, conductance, and capacity for shortest-path, current flow, and network flow, respectively) that in effect represent alternative assumptions of how habitat quality affects dispersal (Supporting Information).

Comparison of Graph Metrics

We contrasted results of the different metrics by deriving a Spearman rank correlation matrix of node-centrality values. We hypothesized that metrics might show stronger relations at their extreme rather than mean values. Therefore, we also used quantile–quantile regression to assess whether higher quantiles (e.g., 99th percentiles) of the shortest-path BC metric were significantly correlated with current flow and minimum-cost-maximum-flow BC (Cade & Noon 2003), as might be expected if shortest paths were subsets of the multiple paths identified by the latter 2 methods. To assess the degree to which priority areas for connectivity conservation differed from priority areas for other potential conservation features, we determined the proportion of areas with highest quantile of centrality values that fell within source (lambda, or intrinsic population growth rate > 1) or core (probability of occupancy > 50%) habitat. Population growth rates and occupancy were predicted by a spatially explicit population model that was based on the same habitat model inputs but was limited to the U.S. portion of the analysis region (Carroll et al. 2006).

We calculated shortest-path and current-flow BC with the NetworkX library (version 1.3) in Python (version 2.6) (van Rossum & Drake 2006; Hagberg et al. 2008). Network flow metrics were derived with the C++ library LEMON (Library for Efficient Modeling and Optimization in Networks, version 1.2) (EGRES 2010). We used Hexsim software (Schumaker 2011) to import and export files from a GIS. We developed a program, the Connectivity Analysis Toolkit (freely available at www.connectivitytools.org), which has a graphical user interface that allows generation of centrality metrics from habitat data without the need to learn Python or C++ (Carroll 2010).

Results

Computational feasibility varied widely among the different metrics, due to the complexity of the underlying algorithms (Newman 2010) and the specifics of the implementation in the software (Carroll 2010). In the regional-extent analysis, shortest-path BC showed low requirements for both memory and computational time (<1 GB and <1 h on a 3 GHz desktop system), whereas current-flow BC required large amounts of memory (>10 GB) (Carroll 2010). Minimum-cost-maximum-flow BC required low amounts of memory (<1 GB) but very long computational times (>1000 h) for the regional-extent analysis, but it was completed in <3 h for the local-extent analysis, which considered source and target patches encompassing approximately 100 hexagons (Fig. 3c).

Shortest-path analysis identified the single best (least cost) path between each pair of source and target hexagons (Fig. 3a). Current-flow analysis identified areas of high current flow along a more diffuse area surrounding the shortest path, as well as along alternate paths (Fig. 3b). Minimum-cost-maximum-flow analysis identified a set of paths that was diffuse in the western portion of the linkage, but constricted in the eastern portion due to the lower proportion of public lands in that

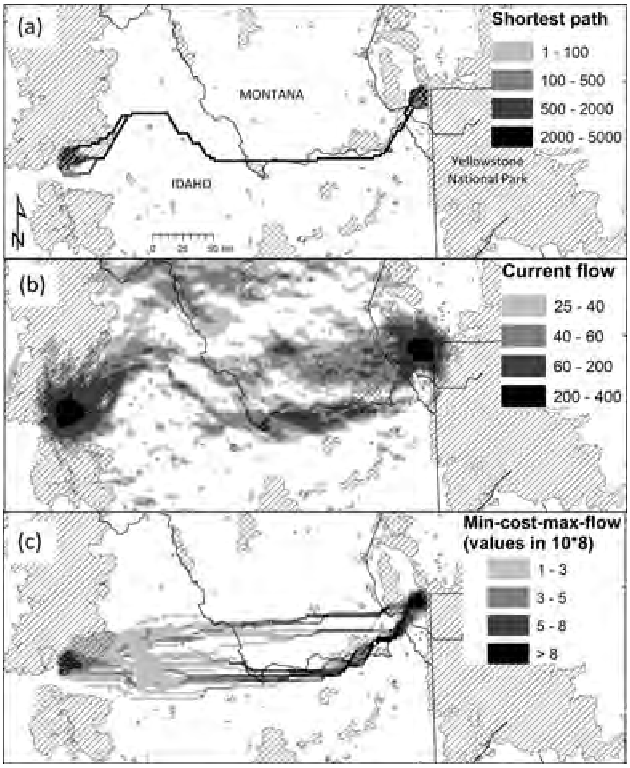

Figure 3. Graph-based analysis of habitat connectivity for gray wolf between central Idaho and Yellowstone National Park. The local-extent analysis compares 3 linkage mapping methods or subset-centrality metrics that are based on (a) shortest- or least-cost path, (b) current flow, and (c) minimum-cost-maximum-flow (min-cost-max-flow). Parks and wilderness areas are crosshatched.

area (Fig. 3c). In the regional-extent analysis, shortest-path BC identified a minimal network connecting the regions of high habitat value (Fig. 4a). Current-flow BC identified areas that encompassed the linkages derived from the shortest-path betweenness analysis, but these areas were more diffusely distributed (Fig. 4b) than were the shortest-path priority areas. Minimum-cost-maximum-flow BC results resembled current-flow results, but were only available at coarser resolution due to their greater computational complexity (Fig. 4c).

At the resolution of 50 km^2 hexagons ($n = 23{,}831$), shortest-path and current-flow BC values from the regional analysis were weakly correlated with habitat-quality value (0.55 and 0.58, respectively) and with each other (0.58). At the resolution of 100 km^2 hexagons ($n = 9601$), correlations were similar (0.59, 0.55, and 0.61, respectively). Additionally, minimum-cost-maximum-flow BC at this resolution was highly correlated with current-flow BC (0.85) but weakly correlated with shortest-path BC (0.45) and habitat-quality value (0.36). Although shortest-path BC showed low correlation with other centrality metrics in the Spearman correlation tests, quantile–quantile regression results showed a significant relation ($p < 0.001$) of shortest-path BC with the higher percentiles of both current-flow and minimum-cost-maximum-flow BC (Supporting Information). Source habitat (12.8% of the U.S. portion of the region) held 20.8%, 20.2%, and 21.7%, respectively, of the areas with highest centrality values (top 20%) for the shortest-path, current-flow, and minimum-cost-maximum-flow BC metrics, whereas core or frequently occupied habitat (25.3% of the region) held 36.6%, 35.6%, and 42.9%, respectively, of the areas with highest centrality values for the 3 metrics.

Discussion

Because centrality analysis simultaneously considers the relations between all areas on a landscape, it provides a means to quantitatively incorporate connectivity within the planning process by ranking the contribution of those areas to facilitating ecological flows. Application of centrality metrics to lattices (graphs with nodes arranged in a regular pattern) avoids both the binary classification of landscapes into patch and matrix required by patch-based graph analyses and the focus on paths between a single pair of patches characteristic of corridor-mapping methods. Rather than addressing connectivity by adding linkages to a system of preidentified core areas, it is possible to compare the relative conservation priority of all linkages in a region and incorporate this information within the multicriteria optimization framework of most conservation-planning software (Possingham et al. 2000; Moilanen et al. 2009).

Although centrality metrics from exploratory analyses such as ours may be used to inform regional planning, input data (Carroll et al. 2006) and key assumptions of the methodology should be tested and revised on the basis of observed connectivity data and results from more-detailed population models. Connectivity models are often based on data on species distribution and rarely test the assumption that dispersal habitat resembles habitat that can be occupied. Habitat variables, such as vegetation structure, influence selection of both dispersal and permanently occupied habitat (Chetkiewicz et al. 2006), but short-term dispersal can occur through habitat that lacks resources for long-term occupancy. It is increasingly possible to rigorously build and test connectivity models from observed levels of dispersal and gene flow derived from genetic and telemetry data (Lee-Yaw et al. 2009; Schwartz et al. 2009; Richard & Armstrong 2010). Our goal was not to contrast these 2 approaches, but rather to describe and compare 3 alternative graph-based connectivity methods that are relevant to analysis of either habitat or dispersal data.

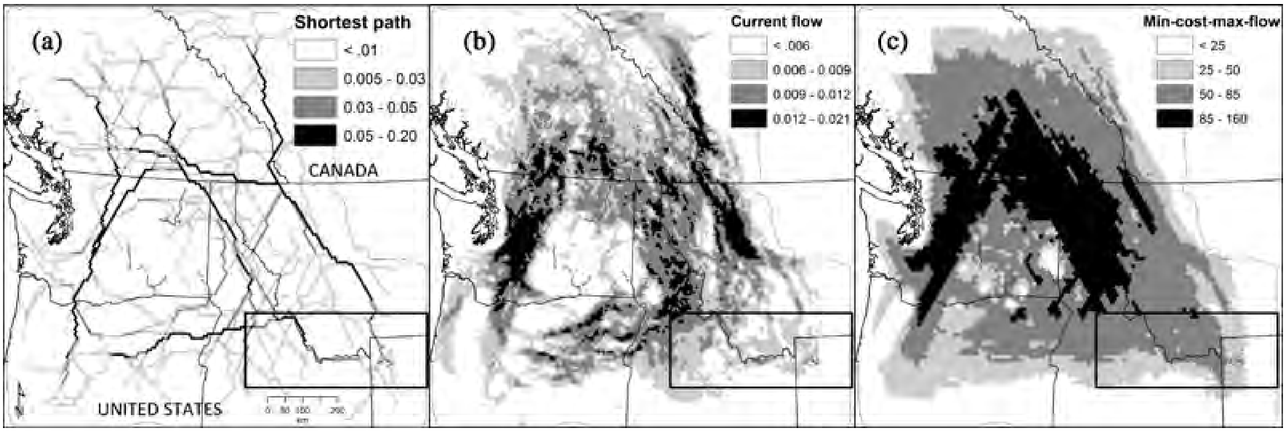

Figure 4. Analysis of habitat connectivity for gray wolf in the northwestern United States and southwestern Canada. The regional-extent connectivity analysis compares results from 3 metrics on the basis of (a) shortest-path, (b) current flow, and (c) minimum-cost-maximum-flow betweenness centrality. The area of the local-extent analysis (Fig. 2) is outlined (rectangle). The units in which the 3 centrality metrics (Figs. 4a–c) are expressed are not directly comparable.

Building and testing connectivity models with empirical dispersal data can help identify the ecologically appropriate spatial resolution and extent for conserving connectivity. Depending on the species of interest, regional habitat linkages may be designed to facilitate individual dispersal events or multigenerational genetic exchange via occupied stepping-stone habitat. Additionally, the degree to which such functional connectivity influences population viability (i.e., how much connectivity is enough to maintain a population) depends on factors such as population size and may be evaluated with more complex population models that simulate both demographic and dispersal processes (Carroll et al. 2006).

Whereas shortest-path models implicitly assume dispersers have perfect knowledge of the landscape, current flow assumes dispersers have no knowledge of the path more than one step ahead (Newman 2005). Real-world behavior of dispersers may fall somewhere between these extremes (McRae et al. 2008; Richard & Armstrong 2010). Shortest-path methods have been used to develop empirical multivariate models of habitat connectivity (Schwartz et al. 2009; Richard & Armstrong 2010). Predictions from current flow-based models are also highly correlated with observed genetic distance in several plant and animal populations (McRae et al. 2008; Lee-Yaw et al. 2009). A comprehensive evaluation of the relative accuracy of these 2 methods in a range of species would be informative. However, given that all graph-based methods are simplified representations of complex dispersal behavior, we advocate use of contrasting metrics as complementary sources of information rather than focusing on a single best metric.

We recommend that planning efforts focused on connecting a single pair of core areas (Fig. 3) compare results from the 3 methods to identify primary and alternative linkage options. In our case study, the comparison suggests it would be informative to evaluate 2 alternative or complementary linkage zones (Figs. 3b–c). In minimum-cost-maximum-flow sensitivity analyses, the southern linkage zone for wolves, which is longer than the northern linkage zone but contains less private land, received increasing priority as the difference in cost between public and private land increased (not shown). Unlike shortest-path analyses, which may combine land cost and habitat quality into a single aggregate index, minimum-cost-maximum-flow incorporates the 2 as distinct criteria, facilitating such sensitivity analyses.

Given that it is computationally challenging to derive minimum-cost-maximum-flow BC over regional extents (Fig. 4c), we suggest regional planning efforts compare results from shortest-path and current-flow BC analyses (Figs. 4a–b). Higher-resolution, local extent analysis of individual linkages (Fig. 3) can be placed in context using the priority assigned to the linkage area in regional analyses (Fig. 4). Although resolution of the landscape lattice remains limited by computational feasibility, it may often be possible to approximate resolutions relevant to habitat associations of the species of interest. In some cases, however, a graph derived from a patch-based representation of a landscape may be more informative than a lattice-based graph (e.g., if the coarse resolution of the lattice obscures key habitats such as riparian forest patches within an upland matrix). The software we developed can also be applied to such nonlattice graphs (Carroll 2010).

Our quantile–quantile regression results suggest that areas with high values of shortest-path BC are a subset of areas with high current flow and minimum-cost-maximum-flow BC values. Areas prioritized by shortest-path BC, which were either central to zones of high-quality habitat or formed shortest paths between them, identify the minimal set of linkages whose loss would greatly reduce regional connectivity (Fig. 4a). In contrast, the zones identified by current-flow BC assist in incorporating redundancy within a linkage network, which may be important for designing networks that are resilient to changing climate and land-use patterns or environmental catastrophes (Fig. 4b) (McRae et al. 2008).

Because nodes near the study-area boundary inherently receive low centrality values (Fig. 4), the analysis area should typically extend beyond the area of interest if data permits. When the scaling of habitat-quality value is not derived from a statistical model that is based on dispersal data, sensitivity analysis with alternate scalings of habitat-quality values (e.g., transforming values by squaring them) can help assess the relative influence on centrality results of a node's habitat-quality value and location in relation to the edge of the analysis area.

Although predictions from network-flow models have not yet been compared with empirical data on dispersal, these algorithms' ability to address flow conservation (Supporting Information) and to consider both cost and capacity suggests they may offer models of connectivity that can be integrated within processes that seek to maximize species protection at minimum cost (Phillips et al. 2008). The minimum-cost-maximum-flow BC metric we used also resembles more complex spatial population models in that it effectively weights the importance of each pairwise relation by the habitat-quality value (and hence ability to produce dispersers) of the source node. Analyses such as ours that prioritize areas with high centrality on the present-day landscape provide a heuristic approach to incorporating connectivity into multicriteria reserve-selection algorithms (Possingham et al. 2000; Moilanen et al. 2009). Full integration of centrality analysis within such algorithms, which requires comparison of the centrality of reserves within many alternate reserve designs, remains computationally challenging.

We focused our case study on informing conservation planning for a single species, the gray wolf. Facilitating dispersal between wolf populations within the western United States and Canada has been proposed as a method to enhance the long-term genetic diversity and viability of the regional wolf metapopulation (Vonholdt et al. 2010). Areas of high centrality were often associated with source or core habitats (Carroll et al. 2006), but they also were found outside those areas. This suggests that conserving connectivity of wolf metapopulations may require different strategies than conserving core populations. Results from our analysis may aid planning to enhance connectivity via habitat protection or reduction of mortality for dispersing wolves within linkage zones. Similar analyses may have broad relevance to conservation planning at a variety of spatial scales appropriate to metapopulations of other species. Centrality analyses may also inform the increasing number of multispecies planning efforts by agencies and nongovernmental organizations that seek to conserve regional habitat connectivity (Western Governors Association 2008).

Acknowledgments

We are grateful to the participants in a series of workshops on connectivity at the National Center for Ecological Analysis and Synthesis, especially P. Beier, E. Fleishman, S. Phillips, R. Pressey, N. Schumaker, and D. Theobald, and to V. Shah for comments that improved the manuscript. K. Djang and the developers of the NetworkX and LEMON graph-analysis modules provided assistance with software development.

Supporting Information

Glossary of terms (Appendix S1), description of conceptual habitat model for gray wolf (Appendix S2), and quantile–quantile regression plots (Appendix S3) are available online. The authors are responsible for the content and functionality of these materials. Queries (other than absence of the material) should be directed to the corresponding author.

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Estimated Hiking Use on Colorado's 14ers

Total Hiker Use Days: 260,000 (2015 Data)

Front Range	Best Est: 72,000
Longs Peak	7,000-10,000
Pikes Peak	7,000-10,000
Torreys Peak	20,000-25,000*
Grays Peak	
Mount Evans	10,000-15,000
Mount Bierstadt	20,000-25,000

Tenmile Range	Best Est: 18,000
Quandary Peak	15,000-20,000*

Sawatch Range	Best Est: 95,000
Mount Elbert	20,000-25-000*
Mount Massive	7,000-10,000
Mount Harvard	3,000-5,000
La Plata Peak	5,000-7,000
Mount Antero	3,000-5,000
Mount Shavano	5,000-7,000
Tabegauche Peak	
Mount Belford	7,000-10,000
Mount Oxford	
Mount Princeton	5,000-7,000
Mount Yale	7,000-10,000
Mount Columbia	3,000-5,000
Missouri Mountain	3,000-5,000
Mt. of the Holy Cross	3,000-5,000
Huron Peak	7,000-10,000

San Juan Mountains	Best Est: 20,000
Uncompahgre Peak	3,000-5,000
Mount Wilson	<1,000
El Diente Peak	<1,000
Mount Eolus	<1,000
Windom Peak	1,000-3,000
Sunlight Peak	
Handies Peak	3,000-5,000*
Mount Sneffels	1,000-3,000
Redcloud Peak	1,000-3,000*
Sunshine Peak	
Wilson Peak	<1,000
Wetterhorn Peak	1,000-3,000
San Luis Peak	1,000-3,000

Mosquito Range	Best Est: 33,000
Mount Lincoln	15,000-20,000
Mount Bross	
Mount Democrat	
Mount Sherman	10,000-15,000

Elk Mountains	Best Est: 7,000
Castle Peak	3,000-5,000*
Maroon Peak	<1,000
Capitol Peak	<1,000
Snowmass Mountain	<1,000
Pyramid Peak	<1,000

Sangre de Cristo Range	Best Est: 14,000
Blanca Peak	1,000-3,000
Ellingwood Point	
Crestone Peak	1,000-3,000
Crestone Needle	1,000-3,000
Kit Carson Peak	1,000-3,000
Challenger Point	
Humboldt Peak	3,000-5,000
Culebra Peak	<1,000
Mount Lindsey	1,000-3,000
Little Bear Peak	<1,000

Approximation Notes

*Indicates data drawn from CFI TRAFx recorders, with data gap and early/late season infills from either previous years or nearby peak data. All other estimates are interpolated from a correlation between TRAFx data and 14ers.com peak use statistics.

Range and overall totals are adjusted to account for the frequent practice of summiting multiple peaks in one day.

A Resistant-Kernel Model of Connectivity for Amphibians that Breed in Vernal Pools

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Abstract: *Pool-breeding amphibian populations operate at multiple scales, from the individual pool to surrounding upland habitat to clusters of pools. When metapopulation dynamics play a role in long-term viability, conservation efforts limited to the protection of individual pools or even pools with associated upland habitat may be ineffective over the long term if connectivity among pools is not maintained. Connectivity becomes especially important and difficult to assess in regions where suburban sprawl is rapidly increasing land development, road density, and traffic rates. We developed a model of connectivity among vernal pools for the four ambystomatid salamanders that occur in Massachusetts and applied it to the nearly 30,000 potential ephemeral wetlands across the state. The model was based on a modification of the kernel estimator (a density estimator commonly used in home range studies) that takes landscape resistance into account. The model was parameterized with empirical migration distances for spotted salamanders (*Ambystoma maculatum*), dispersal distances for marbled salamanders (*A. opacum*), and expert-derived estimates of landscape resistance. The model ranked vernal pools in Massachusetts by local, neighborhood, and regional connectivity and by an integrated measure of connectivity, both statewide and within ecoregions. The most functionally connected pool complexes occurred in southeastern and northeastern Massachusetts, areas with rapidly increasing suburban development. In a sensitivity analysis estimates of pool connectivity were relatively insensitive to uncertainty in parameter estimates, especially at the local and neighborhood scales. Our connectivity model could be used to prioritize conservation efforts for vernal-pool amphibian populations at broader scales than traditional pool-based approaches.*

Keywords: Ambystomatidae, *Ambystoma opacum*, *Ambystoma maculatum*, amphibian conservation, metapopulation, pond-breeding amphibian, resistant-kernel model, seasonal pond, vernal pool

Un Modelo de Núcleo Resistente de la Conectividad para Anfibios que se Reproducen en Charcos Vernales

Resumen: *Las poblaciones de anfibios que se reproducen en charcos operan en escalas múltiples, del charco individual al hábitat circundante al grupo de charcos. Cuando la dinámica de la metapoblación juega un papel en la viabilidad a largo plazo, los esfuerzos de conservación limitados a la protección de charcos individuales o aun charcos asociados con hábitat circundante pueden ser inefectivos a largo plazo si no se mantiene la conectividad entre charcos. La conectividad se vuelve especialmente importante y difícil de evaluar en regiones donde la expansión urbana esta incrementando rápidamente el desarrollo de tierras, la densidad de caminos y las tasas de tráfico. Desarrollamos un modelo de conectividad entre charcos vernales para las cuatro especies de salamandras ambystomoideas que ocurren en Massachussets y lo aplicamos a los casi 30,000 potenciales humedales efímeros en el estado. El modelo se basó en una modificación del estimador de núcleo (un estimador de densidad utilizado comúnmente en estudios de rango de hogar) que toma en consideración la resistencia del paisaje. El modelo fue parametrizado con distancias de migración empíricas para* Ambystoma maculatum, *distancias de dispersión para* A. opacum *y de la resistencia del paisaje derivada de estimaciones por expertos. El modelo clasificó los charcos vernales en Massachussets por la conectividad local,*

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vecinal y regional y por una medida integrada de la conectividad, tanto estatal como dentro de ecoregiones. Los complejos de charcos más conectados funcionalmente ocurrieron en el sureste y noreste de Massachussets, que son áreas con desarrollo suburbano en rápida expansión. Mediante análisis de sensibilidad, las estimaciones de la conectividad de charcos fueron relativamente insensibles a la incertidumbre en la estimación de los parámetros, especialmente en las escalas local y vecinal. Nuestro modelo de conectividad podría ser utilizado para priorizar los esfuerzos de conservación de poblaciones de anfibios de charcos vernales a escalas más amplias que las basadas tradicionalmente en charcos individuales.

Palabras Clave: Ambystomatidae, *Ambystoma opacum*, *Ambystoma maculatum*, anfibios que se reproducen en charcos, charcos vernales, conservación de anfibios, metapoblación, modelo de núcleo resistente

Introduction

Conservation of vernal-pool amphibians must account for the multiple spatial scales of population dynamics. Vernal-pool amphibians such as the ambystomatid salamanders typically exist in local populations associated with discrete breeding pools. With low dispersal rates and the potential for asynchronous dynamics among local populations, metapopulation dynamics may play an important role in long-term population persistence (Semlitsch 2003; Gamble 2004; Smith & Green 2005). Conservation efforts limited to the protection of individual pools or even pools with associated upland habitat may be ineffective over the long term if connectivity among pools is not maintained (e.g., due to the loss of individual wetlands or because of intervening roads or development; Gibbs 1993; Gibbs & Shriver 2005). Nevertheless, broad-scale efforts to address pool connectivity can be complicated because of the large number of ephemeral wetlands in a region and the difficulty of prioritizing pools and surrounding uplands for conservation.

Vernal pools in eastern North America support diverse faunal communities. These small fishless wetlands provide habitat for many obligate invertebrates and amphibians, including ambystomatid salamanders (Colburn 2004). Conservation of vernal pools has usually focused on protecting pool basins themselves, often with small terrestrial buffers. Although this strategy may accommodate flying or wind-dispersed invertebrates, it is inadequate for vernal-pool amphibians, which spend most of their lives in uplands and must disperse overland (Semlitsch 1998; Gamble et al. 2006). In Massachusetts conservation concern is focused on salamanders in the family Ambystomatidae, including marbled salamanders (*Ambystoma opacum*), spotted salamanders (*A. maculatum*), Jefferson's salamanders (*A. jeffersonianum*), blue-spotted salamanders (*A. laterale*), and a number of clonal lineages of *A. jeffersonianum* × *A. laterale* hybrids. At the state level the marbled salamander is listed as threatened and Jefferson's and blue-spotted salamanders are listed as special concern (Kenney & Burne 2000). All four of these species breed in vernal pools, which support the egg and larval life stages, but upland forests provide habitat for juveniles and adults.

Population dynamics of vernal-pool amphibians may be evaluated at four discernable ecological scales: (1) the breeding pool or basin, (2) the breeding pool with surrounding upland habitat, (3) neighboring pools and upland habitat, and (4) clusters (groups of groups) of pools in a broader regional framework. The pool itself is likely a primary determinant of population size and stability. Because adults exhibit high breeding-site fidelity (Whitford & Vinegar 1966; Pechmann et al. 1991; L.R.G., unpublished data), each vernal pool generally supports a distinct breeding population. Pools vary in habitat quality, supporting populations that vary widely among pools and across years (Pechmann et al. 1991; Skelly et al. 1999). Pool hydroperiod seems to be the most important variable structuring vernal-pool communities (Semlitsch et al. 1996; Skelly et al. 1999; Snodgrass et al. 2000; Colburn 2004).

The second scale is the pool with its surrounding upland habitat, or the "life zone" (Semlitsch 1998). Ambystomatids spend 90–95% of their lives in upland forests, up to several hundred meters from breeding pools (Semlitsch 1998), and upland habitat may overlap for several breeding pools. Clearly, protecting pools without this upland habitat does little for even the short-term persistence of populations. Although the details of upland habitat use is an area of active research (e.g., see Madison & Farrand 1998; Faccio 2003; Regosin et al. 2003; McDonough-Haughley & Paton 2007), a reasonable surrogate for the availability of upland habitat is simply the amount of forested area surrounding a pool that is accessible to individual salamanders (e.g., not across a major road; Guerry & Hunter 2002; Homan et al. 2004).

At a third scale, connectivity among populations represents the degree to which dispersal may support metapopulation processes. If dispersal (defined as demographic and genetic exchange among populations, as opposed to migration, which is annual upland movement within a population) among pool-centered populations is low but not zero, then pools and their surroundings represent discrete populations with the potential for occasional gene flow and demographic interactions (such as colonization and the rescue effect; Brown & Kodric-Brown 1977). If all populations have a high potential for extinction over time, and if these extinctions are neither synchronized nor deterministic, then populations show metapopulation structure (Hanski & Gilpin 1991). Recent research on ambystomatid salamanders provides evidence for metapopulation structure in at least some populations (Gamble 2004; Smith & Green 2005; but see references in Marsh & Trenham 2001). If ambystomatids do generally operate in metapopulations, conservation at the scales of pool and local upland habitat is insufficient to ensure persistence over the long term because even in the absence of anthropogenic stressors, many (or even all) populations are expected to become extinct due to stochastic fluctuations over decades or centuries. If connectivity among pools is interrupted, natural dispersal that enables recolonization, rescue effects, and gene flow will not support metapopulation processes. Over long time periods connectivity takes place at even broader spatial scales because the contribution of dispersers from neighboring pools depends in part on how connected these pools are to more distant pools. Metapopulations in broader connected clusters may be more likely to persist than those in smaller clusters. Thus, regional connectivity is structured by the connectivity among clusters of pools at multiple spatial scales. For the sake of convenience, we lump these poorly understood broader scales into a fourth, broadly defined, "regional scale."

A number of strategies have been used to assess the functional connectivity (organism based, see Calabrese & Fagan 2004) of amphibian populations at one or more of these scales. For example, Ray et al. (2002) used a least-cost path approach to evaluate migratory connectivity ("local" scale) for the common toad (*Bufo bufo*) and the alpine newt (*Triturus alpestris*) for 127 ponds in Geneva, Switzerland. Their model showed some success in predicting presence and absence of toads across their study ponds. Rustigian et al. (2003) developed spatially explicit population models integrating multiple scales for four common amphibians in two Iowa watersheds. This approach allowed comparison of the effects of alternative land-use scenarios on populations of these species. In a third approach Pyke (2005) used graph theory to model linkages ("neighborhood" scale) among 122 wetlands used by the California tiger salamander (*A. californiense*) as part of a fuzzy-logic-based decision-support system for conservation action.

We present a modeling framework for assessing the three broader scales of connectivity. These scales are the most intractable to assess in the field; in fact, empirically assessing connectivity is unlikely to be feasible for more than a handful of pools in any region due to the costs and time required for mark-recapture or genetic studies. We applied our model to all four Massachusetts ambystomatids because of their relatively similar breeding and upland habitat associations. A new metric, the resistant-kernel estimator, is used to assess functional interpool connectivity at the neighborhood and regional scales, and a modification of this metric is used to assess connectivity of pools to local upland habitat. We used empirically based migration and dispersal parameters, expert-derived landscape resistance values, and statewide land-use coverages to rank almost 30,000 photointerpreted potential vernal pools in Massachusetts by their modeled level of connectivity at each scale. The resulting rank scores can be used to help identify vernal pools that have intact upland habitat and are highly connected across the landscape for groundtruthing and focused conservation action.

Methods

The resistant-kernel estimator is a hybrid between two existing approaches, the kernel estimator and least-cost paths with resistant surfaces. The kernel estimator (Silverman 1986; Worton 1989) is a density estimator commonly used for home range analysis in radiotelemetry studies. Given two-dimensional data (e.g., x, y points) it produces a three-dimensional surface representing an estimate of the underlying probability distribution by summing across bivariate curves centered on each sampled point. Resistant surfaces are being increasingly used in landscape ecology, replacing the binary habitat/nonhabitat classifications of island biogeography and classic metapopulation models with a more nuanced approach that represents variation in habitat quality (Ricketts 2001). A resistance value is typically assigned to each cover type in a land-cover map, representing a divisor of the expected dispersal or migration distance of animals moving through that cover type. Least-cost path analysis is then used to find the shortest functional distance between two points. This least-cost path approach can be extended to a multidirectional approach that measures the functional distance from a focal cell to every other cell in the landscape within a maximum dispersal or migration distance. Such a least-cost "kernel" is a surface that can be scaled to represent the probability of an individual dispersing from the focal cell arriving at any other point in the landscape. The resistant kernel estimator is calculated by creating a least-cost kernel for each focal cell that represents a source of dispersers (i.e., each vernal pool) and summing across all kernels at each cell (Fig. 1).

The cost assigned to each cover type in the resistant surface represents an integration of the willingness of an animal to cross this cover type, the physiological cost of moving, and the reduction in survival for an organism moving across the landscape. Empirical data on these costs for ambystomatid salamanders are sparse. In a field experiment in which metamorphs were released in enclosed runs, Rothermel and Semlitsch (2002) recaptured spotted salamander (*A. maculatum*) metamorphs at twice the rate in forested runs than open fields, suggesting that survival rates in forests are approximately

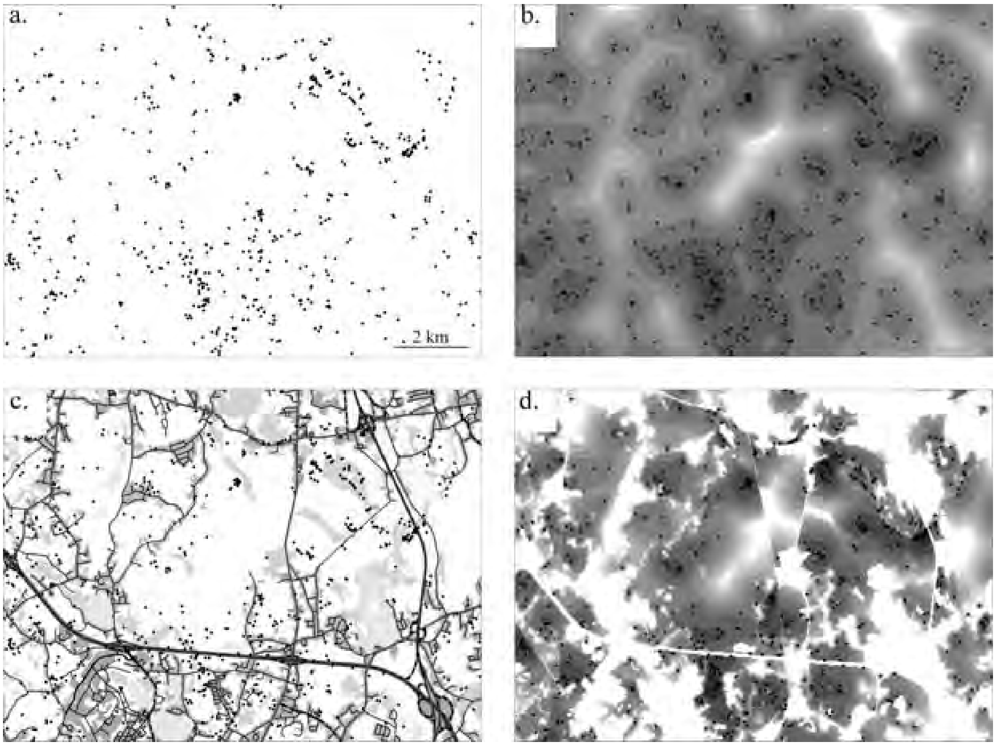

Figure 1. An example of standard versus resistant-kernel estimator applied to a number of potential vernal pools: (a) potential vernal pools represented as points on the landscape, (b) standard-kernel estimator ($h = 399.6$ *m) applied to these pools (darker shading represents higher probability of a dispersing salamander arriving at a particular point and thus higher connectivity), (c) pools with roads and land use included in representation, (d) resistant-kernel estimator (*$h = 399.6$ *m) applied to pools, taking roads and land use into account. Resistant-kernel values are reduced (in comparison with the standard-kernel estimator) by highly resistant land-cover types such as roads.*

double that in fields. McDonough-Haughley and Paton (2007) similarly found reduced survival rates in radio-tracked adult spotted salamanders on golf courses compared with forests. deMaynadier and Hunter (1999) experimentally released wood frog (*Rana sylvatica*) metamorphs in artificial pools along a forest-powerline edge; recapture rates (interpreted as the result of habitat selection) were positively associated with canopy and understory density.

Given the paucity of empirical data, we used expert opinion to parameterize resistance values. We met with a group of seven researchers with field experience on ambystomatid salamanders in southern New England. After discussing our land-cover types and the meaning of resistance values, each expert team member independently assigned a resistance value for each land-cover type for juvenile and adult marbled salamanders. The team then discussed how these values might differ for other ambystomatids in Massachusetts. For each cover type we took a trimmed mean (by dropping the lowest and highest value before taking the mean). These were the landscape resistance values we used in the model (Table 1). Resistance values for vernal pool and forest were fixed at 1.0, the optimal value, and all other values were relative to this optimum. Given our cell size, resistances > 40 act as an absolute barrier. When running the model the resistance value for each cell was multiplied by the three-dimensional Euclidean distance between cell centers to account for diagonally adjacent cells and slopes.

Local Connectivity

We modeled local connectivity (Fig. 2a) between breeding pools and upland habitat by setting the kernel bandwidth h (the standard deviation of a bivariate normal curve) to the expected upland migration distance, based on radiotelemetry data for spotted salamanders in Rhode Island (McDonough-Haughley & Paton 2007). We set h to the 66th percentile of maximum migratory distances from pools for 28 spotted salamanders tracked through forests, or 124 m. As a check on this parameter estimate, we compared percentiles of maximum migratory distances with those of eight spotted and eight Jefferson salamanders tracked in Vermont (Faccio 2003). Percentiles were generally similar; the 66th percentile was 97 m.

Table 1. Resistance values* (trimmed mean with range in parentheses) assigned by seven expert team members to each land-cover type for dispersing *Ambystoma opacum* juveniles and for migrating *A. maculatum* adults.

Cover type	*Dispersal*	*Migration*
Vernal pool	1.0 (1–1)	1.0 (1–1)
Forest	1.0 (1–1)	1.0 (1–1)
Old field	3.4 (2–5)	3.2 (2–5)
Powerline	3.2 (2–5)	3.0 (2–5)
Pasture	9.2 (5–20)	8.6 (5–20)
Row crop	10.2 (4–15)	9.7 (4–15)
Orchard	6.4 (3–15)	6.2 (2.3–15)
Nursery	6.8 (4–15)	6.6 (3–15)
Pond/lake	22.0 (10–40)	10.6 (5–20)
Salt marsh	absolute barrier	absolute barrier
Nonforested wetland	3.0 (2–5)	2.5 (2–5)
Low-density residential	6.8 (4–15)	6.4 (2–15)
High-density residential	12.6 (4–30)	9.8 (3–30)
Urban	26.0 (10–40)	24.0 (10–40)
Expressway	39.0 (30–40)	37.0 (30–40)
Major highway	32.6 (20–40)	30.6 (20–40)
Major road	16.4 (10–35)	14.9 (7.5–31.5)
Minor street or road	7.2 (2–20)	6.6 (1.5–20)
Unpaved road	4.8 (1–10)	4.4 (1–10)
Railroad	15.0 (4–40)	14.2 (3.8–40)
Stream: 1st order	1.3 (1–3)	1.3 (0.8–3)
Stream: 2nd order	2.8 (2–5)	2.6 (1.5–5)
Stream: 3rd order	12.6 (8–30)	12.0 (6–30)
Stream: 4th order	33.0 (15–40)	32.4 (11.3–40)

**Resistance values represent the estimated integrated costs of movement and survival through each cover type. A resistance value of 1 indicates minimal resistance (i.e., movement through preferred habitat, a resistance of 2 means that an individual would be expected to successfully move half as far as the preferred habitat, and a maximum resistance of 40 indicates a complete barrier.*

A single resistant kernel for each pool represented the expected probability distribution of terrestrial habitat use. We summed the cell values of each pool's kernel across forested and vernal-pool cells (rather than sum across all kernels at each cell, as in the kernel estimator) to give the proportion of upland habitat available relative to a kernel in intact optimal habitat (i.e., a pool surrounded by continuous forest). This quantity ranged from near 0 (for a pool with no accessible upland habitat) to 1 (for a pool with optimal upland habitat). This approach differs from simply counting the amount of forest in a circle around each pool in two ways. First, for each pool, forested cells were scaled by the distance from the pool to account for the distribution of expected migratory distances. Second, this approach accounted for differential survival and willingness to cross different land-cover types such as golf courses or roads.

Neighborhood Connectivity

We modeled neighborhood connectivity (the number of dispersers each pool was expected to receive directly from populations associated with neighboring pools; Fig. 2b) with the estimated dispersal distance of marbled salamanders as the kernel bandwidth *h*. Dispersal distances were fit to empirical data from a 7-year study of marbled salamander dispersal among 14 vernal pools in South Hadley, Massachusetts (L.R.G., unpublished data). Dispersal distances are typically fit to a negative exponential distribution (Berven & Grudzien 1990; Trenham et al. 2001) to represent both philopatric and dispersing individuals. We chose to fit dispersal distances to a normal curve for two reasons. First, kernel estimators require a rounded, rather than sharply peaked, distribution (Silverman 1986). Second, observed philopatry in our study population was so high (>90%; L.R.G., unpublished data) that a single exponential curve fit the data poorly. Therefore, we separated the philopatric and dispersing animals. For our purposes only the dispersing animals were of interest.

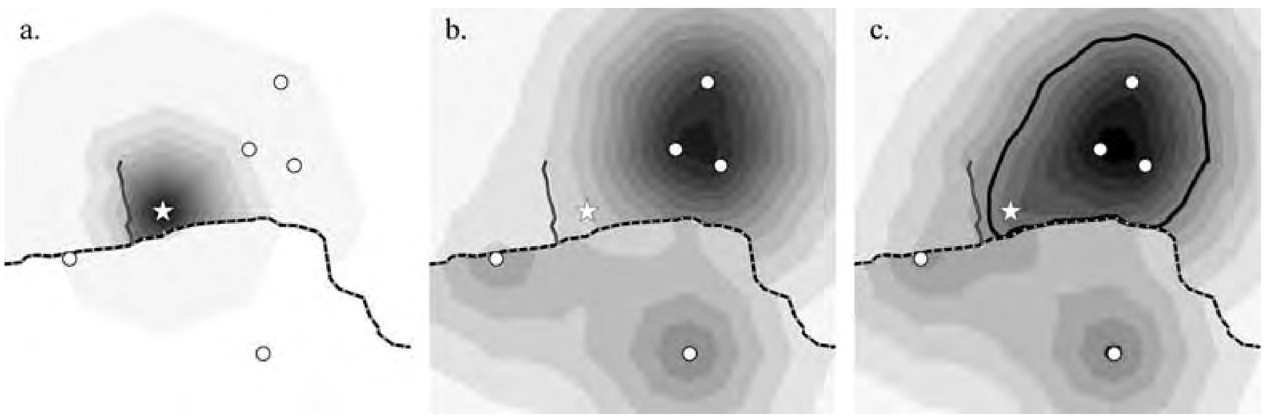

Figure 2. Examples of the resistant-kernel estimator at three scales in a landscape with a focal pool (star), five neighboring pools (circles), and two roads: (a) local scale, showing connectivity to upland habitat from the focal pool; (b) neighborhood scale, showing the probability of the focal pool receiving dispersing animals from each neighboring pool; and (c) regional scale, with dark outline indicating pools that are interconnected by a specified level of dispersal. Darker shading indicates greater connectivity at each scale.

Although we assumed that prebreeding juveniles are the primary dispersers, our empirical measures were of lifetime dispersal (individuals marked as juveniles breeding at non-natal pools as adults). Thus, the lifestage at which dispersal takes place did not have a major effect on the model. The standard deviation of the normal dispersal curve (corresponding to the kernel bandwidth h) was 399.6 m (L.R.G., unpublished data).

At the neighborhood scale connectivity represents the expected number of dispersing animals arriving at a pool from neighboring pools annually. We modeled neighborhood connectivity by applying a resistant kernel (scaled to sum to 1, thus representing the probability of a single individual dispersing to each point surrounding the pool) to each pool and summing across kernels, creating a cumulative kernel surface (as in a standard kernel estimator). The value at the center of each kernel was subtracted from each pool so that the model represented the contribution of dispersers from neighboring pools. We sampled this surface at each pool to yield the neighborhood connectivity metric.

Regional Connectivity

Connectivity at a regional scale measured the size of pool clusters with a specified level of dispersal among pools. This was simply a matter of slicing the cumulative kernel surface at a selected height and counting the number of pools in each cluster (Fig. 2c). If populations and expected numbers of dispersers were consistent among pools, a regional slice could be taken at, for instance, one arriving disperser per generation (Mills & Allendorf 1996). Nevertheless, breeding populations of ambystomatid salamanders vary considerably among pools, and many pools do not support populations at all. Without an estimate of pool-based populations (which would require at least some knowledge about individual pools), determination of regional-scale connectivity becomes somewhat arbitrary. Because our goal was to differentiate among pools for conservation prioritization, we selected the scale that best distinguished among the top 50% of pools. We did this by taking a number of slices, throwing out the 50% of pools with the worst scores, and selecting the scale that gave the largest number of distinct values.

Pool Scores

To score pools across the landscape, we took the geometric mean of the three metrics (local, neighborhood, and regional connectivity) for each pool. Each metric was first rescaled by percentiles to give a qualitative ranking. The geometric mean, often used to integrate limiting factors, was used because a pool that is poorly connected at any one scale will be less likely to contribute to a viable metapopulation. We then rescaled this geometric mean by percentiles across the state, to give a final score for each pool of between 0 and 0.99. A second score was calculated for each pool by rescaling these final scores by percentile within each of the 13 Environmental Protection Agency Level III ecoregions (epa.gov/bioindicators/html/usecoregions.html) that fall within Massachusetts, to give a measure of the most connected pools within each ecoregion. Thus, each pool had a percentile for local, neighborhood, and regional connectivity and for these three metrics combined at the statewide and ecoregional levels. These results can be used to select, for instance, the 5% of pools across the state with the highest scores to be used for conservation prioritization. Full results are available (both as a text file and GIS coverage) at the University of Massachusetts Landscape Ecology Program Web site (www.umass.edu/landeco).

Sensitivity Analysis

There was a high degree of uncertainty in model parameters, due both to the difficulty of obtaining empirical measures of migration and dispersal, and the nature of our expert-derived resistance values. We performed a sensitivity analysis designed to bracket likely parameter values to assess robustness of model results. The sensitivity analysis was conducted at the three scales by altering each parameter or set of parameters one at a time and comparing results with those from the standard model. At the local scale we altered migratory distance $\pm$ 50% (to 62 and 186 m) and used the lowest and the highest expert-supplied resistance values (Table 1; these extreme values were omitted in calculating the trimmed mean resistance for the standard model). At the neighborhood scale we altered dispersal distance $\pm$ 50% (to 200 and 600 m) and used lowest and highest resistance values. At the regional scale we altered dispersal and resistance as for the neighborhood scale and maximized differentiation among the top 25% and top 75% of pools. For each sensitivity run we calculated the coefficient of determination (r^2) between the results at the chosen scale (transformed to percentiles) with the results from the standard model. High values of r^2 indicated that the chosen parameter had little leverage on the ranking of pools, whereas low values indicated that the results were sensitive to the parameter in question. To address the question of whether results were affected by resistance values at all (as opposed to simply the arrangement of pools on the landscape), we also compared the standard model run to a run with all resistance values set to 1.0, thus removing the effect of landcover resistance from the model. Finally, to assess the effect of the geographic scope on pool scaling, we calculated the correlation between pools scored across the entire state and scores rescaled within each ecoregion.

GIS Data

The GIS data consisted of potential vernal pools, land use, roads, streams, and slope. Potential vernal pools were photointerpreted from 1:12,000 color infrared aerial photographs by the Massachusetts Natural Heritage and Endangered Species Program (Burne 2001). These data consist of point locations of nearly 30,000 potential vernal pools across the state and have not been extensively field validated. Known errors of omission include pools <40 m across, pools under conifer canopy and pools embedded in larger wetlands; errors of commission include tree shadows, small permanent ponds, and seeps and shallow pools with extremely short hydroperiods (Burne 2001). Land-use data were photointerpreted from 1999 aerial photographs by the University of Massachusetts Resource Mapping Unit and included 24 cover classes (Table 1). Road data were photointerpreted by the Massachusetts Highway Department and categorized into six classes. Streams were classified by order on the basis of stream center lines. All data layers were converted to a 30-m grid and combined into a comprehensive land cover with each potential vernal pool represented by a single cell. Source data are available from the Massachusetts Office of Geographic and Environmental Information (www.state.ma.us/mgis).

We completed GIS and statistical analyses with ArcInfo (version 9.1, Environmental Systems Research Institute, Redlands, California), JMPIN (version 3.0.2, SAS Institute, Cary, North Carolina), and programs written by B.W.C. in APL+Win (version 6.0, APLNow, Brielle, New Jersey) and by E. Ene in Visual C++ (version 6.0, Microsoft, Redmond, Washington, D.C.).

Results

Potential vernal pools across Massachusetts were ranked at each of the three scales of connectivity and given a combined score. Pools and their combined rankings were distributed unevenly across the state, with the highest concentrations of high-valued pools generally following the highest concentrations of potential vernal pools. These were located mostly in the coastal plain, particularly in Bristol, Middlesex, Essex, and Plymouth counties.

Values for local connectivity were distributed uniformly (values vs. ranks, $r^2 = 1.000$). Values for neighborhood connectivity were long tailed (reciprocal of values vs. ranks, $r^2 = 0.991$). The regional scale also had a long tail, with clumping at the upper end, because all pools in larger clusters had the same value (log of value vs. ranks, $r^2 = 0.974$). Values were rescaled by percentiles at each scale to yield uniform distributions. Each pool was assigned a combined score by taking the geometric mean across the three scales (Figs. 3 & 4).

Sensitivity Analysis

Results of the sensitivity analysis (Table 2) indicated that pool rankings were relatively insensitive to the parameter values we used, suggesting that the model was robust to modest estimation errors in migration and dispersal distances and to the expert-based estimates of resistance

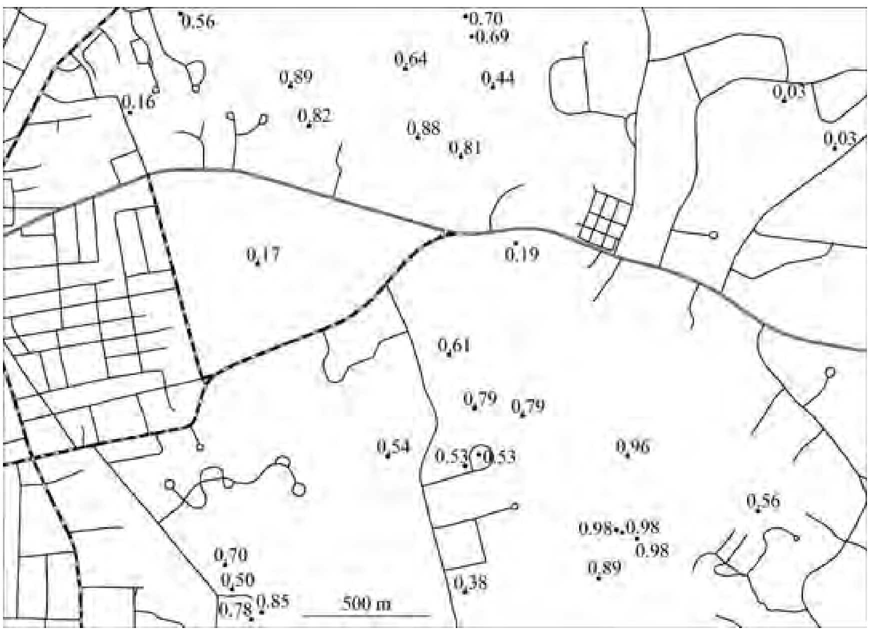

Figure 3. Combined pool scores (integrated level of vernal-pool connectivity across all three scales) for a small area, with roads for context. Scores represent the percentile for each pool based on all three scales of connectivity. A score of 0.99 represents the 1% most connected pools in the landscape (across scales).

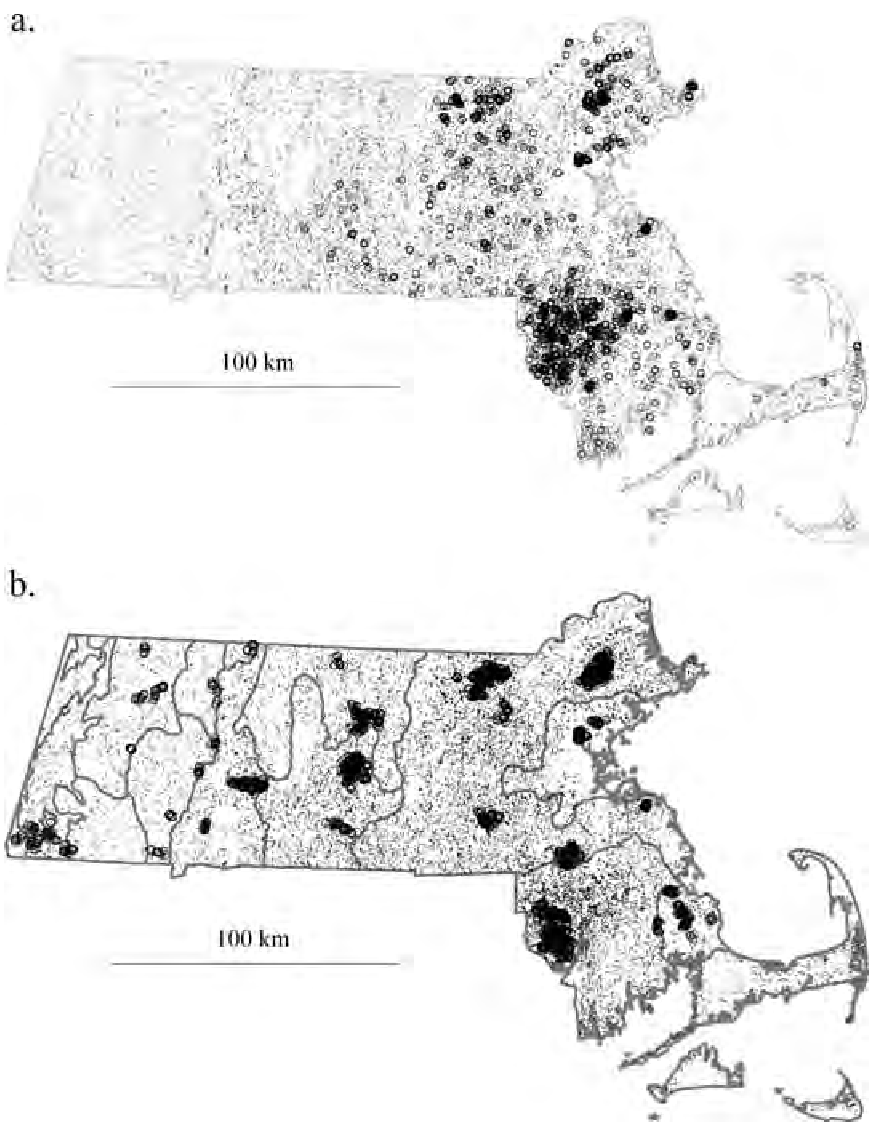

Figure 4. Vernal-pool connectivity scores (integrated across all three scales) for all pools across Massachusetts: (a) combined pool scores across Massachusetts and (b) pool scores by ecoregion (black circles, 10% most connected pools; small dots, 90% least connected pools; gray lines [in b], ecoregion boundaries).

values. Local and neighborhood rankings were quite stable (all $r^2 \geq 0.86$), whereas rankings at the regional scale were less so. The greater instability of regional rankings was not surprising because scores were assigned on the basis of the number of pools in a cluster, so changes in parameters that resulted in large clusters being split or joined could radically change the scores for many pools.

Results of the null resistance model (Table 2) were not highly correlated with results of the standard model at the local and regional scales; however, at the neighborhood scale, the standard and no-resistance runs were correlated ($r^2 = 0.79$). The median reduction in raw neighborhood pool scores in the standard versus the null resistance model was 5.5% (interquartile range = 1.4–13.5%).

Finally, the correlation between combined pool scores rescaled within each ecoregion to scores scaled across the entire state was relatively low ($r^2 = 0.24$). This was an expected result of changing the scope of the analysis; such a rescaling elevates scores of pools in ecoregions with relatively low scores overall at the expense of higher-scoring pools in ecoregions with generally higher scores (Fig. 4).

Discussion

The density of potential vernal pools was strongly reflected in the connectivity metrics. Although it was not possible to explicitly partition variance between landscape resistance and pool configuration, the null resistance model (Table 2) suggested that, at the neighborhood scale, pool scores reacted primarily to pool configuration rather than landscape resistance. Unfortunately, the densest groupings of potential vernal pools and thus the largest clusters of highest-ranked pools were in the coastal plains of Essex, Middlesex, Bristol, and Plymouth counties, on the leading edge of suburban sprawl from

Table 2. Correlations between pool scores (rankings of connectivity among pools for ambystomatids at each scale) from standard-model run and scores from sensitivity-analysis runs.[a]

		r^2		
Scale	*Parameter*	*lower*[c]	*upper*[d]	*null*[e]
Local	migratory distance (±50%)	0.90	0.96	
	resistance values[b]	0.96	0.99	0.02
Neighborhood	dispersal distance (±50%)	0.86	0.95	
	resistance values	0.96	0.97	0.79
Regional	dispersal distance (±50%)	0.55	0.72	
	resistance values	0.56	0.57	0.15
	top 25% / top 75%[f]	0.72	0.59	

[a] *High correlations indicate the model is insensitive to parameter values.*
[b] *Resistance values are estimates of costs of moving through each cover type set by expert opinion. The resistance values used in the standard model were the trimmed mean for each cover type. In the sensitivity analysis, results of the standard model were compared with those from a model based on the minimum resistance across experts for each cover type and with the maximum resistances.*
[c] *Correlation between results of standard model and those from runs with minimum parameter values.*
[d] *Correlation between results of standard model and those from runs with maximum parameter values.*
[e] *Correlation between results of standard model and those from run with all resistance values set to 1.0.*
[f] *For the standard model, the scale was chosen to maximize differentiation among the top 50% of pools. In this sensitivity analysis, scales were chosen to maximize the top 25% and top 75% of pools.*

the Boston metropolitan area (Fig. 4a). The model suggested that despite current levels of development, pools in these areas may still offer the most connected habitat for ambystomatids in Massachusetts and should be a priority for conservation action.

The resistant-kernel estimator we used was a functional measure of connectivity that realistically modeled movement across different cover types while avoiding the complexity and computational costs of an individual-based model. As a functional metric, the resistant-kernel estimator is parameterized on the basis of the biology of particular organisms, as opposed to structural metrics, which measure connectivity as a feature of the landscape (Calabrese & Fagan 2004). We used resistant kernels to explicitly model connectivity at multiple scales, thus allowing separate assessment of each scale, trade-offs among scales, and integration across scales.

Sensitivity Analysis

The sensitivity analysis suggested that the model results at the regional scale were less reliable for ranking pools than the local and neighborhood scales. Therefore, a user may choose to omit the regional scale when ranking pools for conservation action. The relative insensitivity of pool rankings to changes in resistance values at the local and neighborhood scales suggested that expert-based resistance values need not be precise (a clearly unattainable goal), but it did bring up the question of whether resistance values (and thus land cover) have any effect on model results. Were pool rankings primarily a reflection of the arrangement and density of pools on the landscape? A comparison of results of the null resistance model with the standard model indicated that at the local and regional scales landscape resistance played a large role in pool rankings. At the local scale the null model simply reflected the amount of upland habitat available to each population; the low agreement with the standard model suggested that habitat configuration (and thus landscape resistance) played a major role. Likewise, at the regional scale, the null model simply reflected the density and configuration of pool clusters; the low agreement with the standard model suggests that land-cover patterns in the intervening landscape between clusters of pools has the potential to significantly affect connectivity at these broader scales. However, at the neighborhood scale, there was fairly strong agreement between the standard and null resistance models in ranking of pools (Table 2). At this scale, the null model reflected the density and configuration of nearby pools; the agreement with the standard model suggested that landscape resistance between pools in a cluster had relatively little effect on connectivity at this scale.

Pool rankings were sensitive to the geographic scope of analysis. Rescaling by percentiles within each ecoregion provided an assessment of the most connected pools within each region. These geographically nested analyses allow targeting of both the most connected pools across the state (which were skewed heavily to eastern Massachusetts) and the most connected pools within each ecoregion.

Conservation Application

The large number of potential vernal pools across Massachusetts would preclude site visits to more than a small fraction. At the same time landscape-scale issues such as connectivity and, to some extent, availability of upland habitat, are difficult to assess objectively in the field. Current regulatory protection mechanisms focus on the pool basin and a small (31 m) buffer around each pool, leaving upland habitat and connectivity to be assessed on a case-by-case basis. As a result, at these broader scales, there is little effective protection from the cumulative effects of development.

We propose a strategic framework for conservation of vernal pools at multiple scales. Our approach is hierarchical, starting from a broad landscape scale, and allows for flexibility in matching efforts to available resources. The model of habitat connectivity presented here would be used in the initial step. Conservation planners could use the results from our model statewide or across a smaller region of interest (e.g., ecoregion, watershed, or town). Pools with high scores for connectivity would be identified. Such identification could take other variables into account, such as proximity to protected open space. Depending on the resources available, this could include the top 1%, 10%, or more—such use of qualitative metrics is to some extent a political, rather than a biological decision (e.g., What percentage of vernal pools need protection at all scales?). The result of this step would be the identification of hotspots of potential vernal pools with high connectivity to other pools and intact upland habitat.

Once clusters of high-ranking potential vernal pools are identified, field validation could target these subsets of pools. Such efforts could make use of volunteers, as has been previously done effectively in Massachusetts. Depending on available resources, field validation could range from confirming the existence of standing water during various seasons as an estimate of hydroperiod (e.g., from aerial photos), to biologically based pool certification, to more intensive work targeted at confirming the presence of rare species (such as marbled salamanders) and estimating populations sizes. This two-step process is a highly efficient way to identify vernal pools with high conservation value for ambystomatids. Such work must, obviously, be linked to efforts to protect high-ranking pools, their surrounding upland habitat, and connections among pools.

Our model results allow assessment of pools at each of the three scales independently, assessment of pools based on the integrated score, or exploration of trade-offs among the different scales. Although our integrated score is based on equal weighting of the three scales, the scores at each scale may be given weights reflecting the purported importance of each scale before integration. If surveys allow assignment of a value that reflects breeding habitat quality of each pool in an area, these values can be easily incorporated as a fourth scale in the integrated score.

The results of the model were not scaled and parameterized appropriately to cover other taxa that use or require vernal pools (such as obligate vernal-pool invertebrates or turtles that feed on vernal-pool species). To some extent, by targeting clusters of vernal pools in intact uplands, other vernal-pool species may also be protected. This is less likely to be true for invertebrates with extremely patchy distributions and dispersal that is either strongly limited or takes place at much broader scales than salamanders. A pool is not necessarily a low conservation priority simply because it is poorly connected. Many isolated pools or small clusters of pools may support rare species or genotypes, or may contain sufficiently robust salamander populations to persist over the long term despite their isolation. Isolated pools in urban areas can also provide important educational or "wilderness" values to humans.

When applying this model to individual ambystomatid species the output should first be clipped to the approximate range of the species within Massachusetts. The marbled salamander, for instance, apparently does not occur in north-central Massachusetts or in the higher elevations of western Massachusetts.

Assumptions and Limitations

A modeling effort such as ours carries a number of assumptions. We assumed that land use and road data were correct and that the categories assigned were meaningful. Roads, for example, were classified by size (Table 1), which is assumed to correspond to the more ecologically meaningful road width and traffic rate. In addition, land use does not correspond exactly to land cover. For instance, "low-density residential" includes both mowed lawns and small patches of forested areas. Finally, these data may carry positional errors. All of these potential sources of error may affect model results to some extent, but are unlikely to have a major effect. Gross misclassifications in land use, most likely caused by land-use changes since the coverage was created in 1999, are likely to have a larger effect.

The model relied heavily on the photointerpreted potential vernal pools coverage (Burne 2001), which has not been field validated extensively. Errors of omission and commission will affect our results. More important, each vernal pool was represented as a point in the landscape; thus, we assumed that all pools provide ecologically equal habitats (and essentially, equal population sizes) for the species under consideration. In reality the size, hydroperiod, water chemistry, and other features of vernal pools vary widely. These pool-scale factors are probably the primary determinants of local amphibian populations. In amphibian metapopulations there is a strong source-sink aspect to metapopulation dynamics among pools because pools vary in habitat quality. Representing these important pool-scale factors requires extensive local (and usually field based) information that is unavailable at the large extents we addressed. Thus, our model addressed connectivity among pools and to upland habitat, assuming that pools themselves are equal. We see this model as an important first step in estimating the relative conservation value of different pools that should be followed up with more intensive study of selected pools at the local scale.

Our model was static, based on a current snapshot of the landscape. Thus, it did not account for the effects of land history or future changes in land use. Land-use history may have an important effect on the distribution of vernal-pool amphibians because more than half of the forests in Massachusetts were converted to agriculture during the eighteenth and nineteenth centuries and much of this land has since become reforested (Hall et al. 2002). Thus, many amphibian populations may have been extirpated because of the loss of upland habitat and have yet to recolonize currently available habitat. There is also likely to be a time lag as upland habitat is developed and connections are lost among pools because metapopulation dynamics play out over many generations, which in long-lived species such as spotted salamanders (Flageole & Leclair 1992) may take several decades or longer. Our model represents the current connectivity among pools, whereas past connectivity is likely a more important determinant of current population distribution (Findlay & Bourdages 2000). Finally, future changes in land use and traffic levels will continue to affect connectivity among pools.

Our model depended on several poorly known parameters: dispersal and migration distances and the resistance of different land uses and road types. We obtained estimates of movement and life-history parameters from empirical field studies of spotted and marbled salamanders and thus assumed that these data were representative of ambystomatid salamanders across Massachusetts. Although some variation is likely in migration and dispersal distances and landscape resistance among these four species, field work has not yet demonstrated such differences. Our model assumed that dispersal is random and nondirectional; thus it focused on available upland habitat and among-pool connections rather than predicted actual movements. We assumed the shape of the dispersal curve is normal. Sufficient data do not currently exist for these species to allow confident distinction among different dispersal distribution models. Finally, we used expert opinion to obtain resistance values for each land-use type and road size. Empirical resistance values are poorly known, although recent and current field experiments are addressing this issue (Rothermel & Semlitsch 2002). Sensitivity analysis suggests, however, that the model responded more strongly to pool arrangement and land cover than to the particular values of migration, dispersal, and resistance values.

Another issue omitted from the model is an analysis of key pools (or sets of pools) that act as critical links or "stepping stones" to connect two or more clusters of pools. If these key pools (or linkages to and from these pools) are destroyed or degraded, a large complex of interconnected pools could be split into two smaller complexes, with potential implications for metapopulations they support (Semlitsch & Bodie 1998). Identifying pools that contribute disproportionally to connectivity would require an iterative "take-one-out" analysis (e.g., Keitt et al. 1997; Urban & Keitt 2001). At the scale of this analysis, such an approach would be computationally infeasible; perhaps future investigation along these lines will provide valuable insights on critical pools or groups of pools.

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Assessing the value of roadless areas in a conservation reserve strategy: biodiversity and landscape connectivity in the northern Rockies

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Summary

1. Roadless areas on United States Department of Agriculture (USDA) Forest Service lands hold significant potential for the conservation of native biodiversity and ecosystem processes, primarily because of their size and location. We examined the potential increase in land-cover types, elevation representation and landscape connectivity that inventoried roadless areas would provide in a northern Rockies (USA) conservation reserve strategy, if these roadless areas received full protection.

2. For the northern Rocky Mountain states of Montana, Wyoming and Idaho, USA, we obtained GIS data on land-cover types and a digital elevation model. We calculated the percentage of land-cover types and elevation ranges of current protected areas (wilderness, national parks and national wildlife refuges) and compared these with the percentages calculated for roadless and protected areas combined. Using five landscape metrics and corresponding statistics, we quantified how roadless areas, when assessed with current protected areas, affect three elements of landscape connectivity: area, isolation and aggregation.

3. Roadless areas, when added to existing federal-protected areas in the northern Rockies, increase the representation of virtually all land-cover types, some by more than 100%, and increase the protection of relatively undisturbed lower elevation lands, which are exceedingly rare in the northern Rockies. In fact, roadless areas protect more rare and declining land-cover types, such as aspen, whitebark pine, sagebrush and grassland communities, than existing protected areas.

4. *Synthesis and applications*. Landscape metric results for the three elements of landscape connectivity (area, isolation and aggregation) demonstrate how roadless areas adjacent to protected areas increase connectivity by creating larger and more cohesive protected area 'patches.' Roadless areas enhance overall landscape connectivity by reducing isolation among protected areas and creating a more dispersed conservation reserve network, important for maintaining wide-ranging species movements. We advocate that the USDA Forest Service should retain the Roadless Area Conservation Rule and manage roadless areas as an integral part of the conservation reserve network for the northern Rockies.

Key-words: conservation, elevation zones, land-cover types, landscape metrics, national forests, reserve design

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Introduction

A growing body of scientific evidence indicates that the current USA system of federal protected areas (designated wilderness areas, national parks and national wildlife refuges) may be too small and disconnected to protect against the decline and loss of native species diversity or to accommodate large natural ecosystem processes (Wright, Dixon & Thompson 1933; MacArthur & Wilson 1967; White 1987; Wilcove 1989; Baker 1992; Turner *et al*. 1993; Noss & Cooperrider 1994; Reice 1994; Newmark 1995; Sinclair *et al*. 1995; Soule & Terborgh 1999). Expanding road networks, human settlements, resource extraction and other encroachments on the landscape have increased the fragmentation and loss of natural areas. Such disturbances have isolated many protected areas, causing them to function as terrestrial 'islands' surrounded by a matrix of lower quality altered lands (Harris 1984; Pickett & White 1985; Wiens, Crawford & Gosz 1985; Turner 1989; Saunders, Hobbs & Margules 1991). The long-term persistence of many species within protected areas is dependent on the degree of human activities and land-use practices on lands adjacent to and near protected areas. There is a need to identify relatively undisturbed lands located outside protected areas that may increase the potential of protected areas in maintaining native biodiversity and certain ecological processes, and to include these lands within the conservation reserve system before they are lost or altered.

Inventoried roadless areas, large tracts of relatively undisturbed land on USA Forest Service lands, are often left out of landscape assessments for identifying functional conservation reserves. Only two studies (DeVelice & Martin 2001; Strittholt & DellaSala 2001) have analysed the contribution that roadless areas make to the current protected areas reserve network. However, more than one-third of inventoried roadless areas on national forests are adjacent to protected areas (DeVelice & Martin 2001). They hold the potential to increase the size and connectivity of designated wilderness areas, national parks and national wildlife refuges, thus increasing the ability of protected areas to maintain natural landscape dynamics and native species population viability over the long term. Smaller, isolated roadless areas are also important because they may contain rare species, capture more habitat variation, including underrepresented habitat types, and may function as 'stepping stones' that connect current protected areas across a landscape (Shafer 1995; Strittholt & DellaSala 2001).

There is a precedent for the protection of national forest roadless areas. The USA Congress has designated as wilderness more than half, 6 million ha, of roadless areas that the Forest Service inventoried in national forests in the 1970s. In 1998, the Forest Service began to devise regulations aimed at protection of roadless area characteristics in national forests. In May 2000, the agency released its proposed rule, familiarly known as the Roadless Rule, and draft environmental impact statement. Eight months later, the Forest Service adopted the rule. In July 2004, the Forest Service proposed to repeal the Roadless Rule and replace it with a state petition and rule-making process, which would offer less protection by presumably opening national roadless areas to all forest service activities and requiring state governors to 'opt in' Roadless Rule protections affirmatively for any roadless area.

Included in the Roadless Rule environmental impact statement was an evaluation of the potential contribution that protection of roadless areas could make to the conservation of biodiversity at a national scale (USDA Forest Service 2000b). In that evaluation, DeVelice & Martin (2001) found that the inclusion of roadless areas in the network of federal protected areas would expand representation of ecoregions in protected areas, increase the acreage of reserved areas at lower elevations, and increase the number of areas large enough to provide refuge for wide-ranging species.

Strittholt & DellaSala (2001) focused on similar questions at a regional scale for the Klamath-Sikiyou area in southern Oregon and northern California, USA. They found that roadless areas protect a wide range of ecological attributes, especially at mid- to lower elevations, important in this region. They also concluded that roadless areas increase the connectivity among ecoregions.

The northern Rocky Mountain states of Montana, Wyoming and Idaho comprise a region particularly rich in roadless areas, roughly 2·6 million ha, providing a unique opportunity to create a relatively intact reserve design that captures important elements of conservation for the northern Rockies. Using two key concepts in conservation biology, biodiversity representation and landscape connectivity, we investigated the potential contributions of national forest roadless areas to the protected areas reserve network across the northern Rocky Mountain region.

DIVERSITY REPRESENTATION

An important goal in the design and establishment of conservation reserves is to represent a full range of native biodiversity (Shelford 1926; Margules, Nicholls & Pressey 1988; Church, Stoms & Davis 1996; Possingham, Ball & Andelman 2000). Even though this goal has been articulated for some time, most protected areas are demarcated around areas with high scenic and recreational attributes (Davis *et al*. 1996). As a result, existing protected areas in the northern Rockies are, for the most part, concentrated at higher elevations, where other important elements of biodiversity are most likely to be poorly represented (Scott *et al*. 2001).

Representation of a full range of biodiversity in reserves requires an understanding of all species and ecosystem processes operating within a given landscape. However, many researchers have used ecological communities and elevation ranges as coarse-scale surrogates for native biodiversity in the design of conservation reserves (Scott *et al.* 1993; Host *et al.* 1996). This concept is based on the idea that if a full range of ecological communities and elevation ranges is protected, it is more likely that many ecological communities, wide-ranging species and ecosystem processes will be maintained in the reserves. In the northern Rockies, ecological communities are often associated with elevation gradients (Hansen & Rotella 1999). Hence, roadless areas situated at middle and lower elevations may make valuable contributions in protecting many elements of biodiversity that are currently not well represented in protected areas (DeVelice & Martin 2001).

LANDSCAPE CONNECTIVITY

Connectivity refers to the degree to which the structure of a landscape helps or hinders the movement of wildlife species or natural processes such as fire (Wiens, Crawford & Gosz 1985; Turner *et al.* 1993; Noss & Cooperrider 1994; Bascompte & Solé 1996; With 1999). A 'well-connected' area can sustain important elements of ecosystem integrity, namely the ability of species to move and natural processes to function, and is more likely to maintain its overall integrity compared with a highly fragmented area.

Roads are highlighted in the scientific literature as major causes of landscape fragmentation, and function as barriers to organism movements, resulting in a reduction of overall landscape connectivity for many native species. The effects of roads are broad and include mortality from collisions, modification of animal behaviour, disruption of the physical environment, alteration of chemical environments, spread of exotic and invasive species, habitat loss, increase in edge effects, interference with wildlife life-history functions and degradation of aquatic habitats through alteration of stream banks and increased sediment loads (Franklin & Forman 1987; Andrews 1990; Noss & Cooperrider 1994; Reice 1994; Reed, Johnson-Barnard & Baker 1996; Trombulak & Frissell 2000; McGarigal *et al.* 2001). Thus, the addition of roadless areas to existing protected areas reserve is likely to maintain or increase landscape connectivity, as well as increase the integrity of protected areas.

With the advent of landscape metrics, it is now possible to quantify connectivity for landscapes, land-cover types, species' habitats, species' movements and ecosystem processes across a given region (O'Neill *et al.* 1988; McGarigal & Marks 1995; Gustafson 1998; With 1999). Many different metrics that quantify spatial characteristics of patches or entire landscape mosaics have been described (Turner & Gardner 1991; McGarigal & Marks 1995; Ritters *et al.* 1995; Hargis, Bisonette & David 1998; Dale 2000; Jaeger 2000; McGarigal & Holmes 2002). We chose metrics that measure three elements of landscape connectivity: area, isolation and aggregation.

Area

It is known that larger areas (patches) generally contain more species, more individuals, more species with large home ranges and/or sensitive to human activity, and more intact ecosystem processes than smaller areas (Robbins, Dawson & Dowell 1989; Turner *et al.* 1993; Newmark 1995; Shafer 1995). Higher numbers of patches will usually contribute to greater resilience of populations and may also increase the utility of patches that act as 'stepping stones' or connectors across a landscape (Buechner 1989; Lamberson *et al.* 1992).

Isolation

The distance between patches plays an important role in many ecological processes. Studies have shown that patch isolation is the reason that fragmented habitats often contain fewer bird and mammal species than contiguous habitats (Murphy & Noon 1992; Reed, Johnson-Barnard & Baker 1996; Beauvais 2000; Hansen & Rotella 2000). As habitat is lost or fragmented, residual habitat patches become smaller and more isolated from each other, species movement is disrupted, and individual species and local populations become isolated (Shinneman & Baker 2000).

Aggregation

The spatial arrangement of patches may help to explain how certain species are found in patches located close together and are not found in patches that are more isolated, or vice versa (Ritters *et al.* 1995; He, DeZonia, & Mladenoff 2000). This concept generally follows the ideas developed in island biogeography theory (MacArthur & Wilson 1967) and metapopulation theory (Levins 1969, 1970).

For some species or natural processes, the isolation or aggregation of patches across the landscape may be more important, for others, area may be the key element. Together, these three elements offer a comprehensive assessment of the importance of roadless areas to the maintenance of overall landscape connectivity and ecosystem integrity of current protected areas in the northern Rockies.

In this study, we aimed to assess the extent to which roadless areas increase biodiversity representation and landscape connectivity when they are included in the protected areas reserve network for the northern Rockies.

Methods

STUDY AREA

Of the 84 million ha of land that stretch across Montana, Wyoming and Idaho in the USA, roadless areas cover 2·6 million ha and existing federal protected areas (wilderness areas, national parks, special management areas and national wildlife refuges) protect almost 8·7 million ha. Within this region, three large, relatively

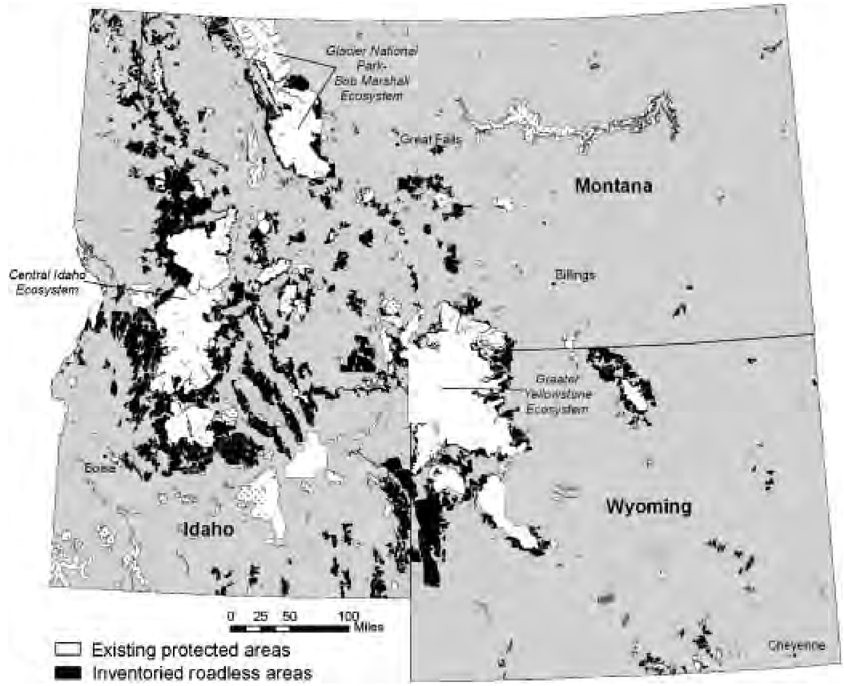

Fig. 1. Roadless areas and protected areas across the states of Idaho, Montana and Wyoming, USA.

undisturbed, mountain ecosystems are delineated around national parks and/or wilderness complexes. These are the Greater Yellowstone Ecosystem, Glacier National Park–Bob Marshall Ecosystem, and the Central Idaho Ecosystem (Fig. 1).

The topography of the northern Rocky Mountain states spans steep physical gradients in elevation, slope, aspect, temperature and precipitation that give rise to diverse vegetation types. Elevations range from 150 m to 4200 m. Average precipitation ranges from 28 cm to 51 cm (Franklin 1983). The northern Rockies comprise a variety of non-forested and coniferous forest types. Low-lying valleys are characterized by grasslands, sagebrush (*Artemisia* spp.) and desert shrublands, interspersed with juniper (*Juniperus* spp.) and riparian woodlands. Ponderosa pine *Pinus ponderosa* dominates lower elevation montane forests, while xeric coniferous forests of mainly Douglas fir *Psuedotsuga mensiezia*, ponderosa pine, grand fir *Abies grandis*, lodgepole pine *Pinus contorta* and aspen *Populus tremuloides* occur at mid-elevations. Mesic forests in the north and west largely contain western larch *Larix occidentalis*, grand fir, western red cedar *Thuja plicata* and mountain hemlock *Tsuga mertensiana*. Higher elevations are composed of Engelmann spruce *Picea engelmannii*, subalpine fir *Abies lasiocarpa*, alpine larch *Larix lyalli* and whitebark pine *Pinus albicaulis* intermixed with subalpine meadows. Herb lands, rock, alder *Alnus sinuata* shrubfields and snowfields/ice occur at the highest elevations.

DATA COLLECTION

We used a land management status GIS coverage and classification system developed by the USA Geological Survey's Biological Resources Division in its nationwide GAP Analysis Programme (Scott, Tear & Davis 1996) to delineate 'protected areas'. This programme devised a ranking scheme to represent various levels of protection, ranging from the least protected lands (category 4, e.g. private lands) to those with the highest level of protection (category 1, e.g. wilderness areas) for all public lands in the GIS spatial database. For this study, we assumed that categories 1 and 2 represent adequate protection as their primary management objective is conservation (Scott, Tear & Davis 1996), and selected these categories as our protected areas on all forest service lands located in the three states.

We used the federal inventoried roadless areas GIS database (USDA Forest Service 2000a). This includes areas that are greater than 2000 ha in size, where road building is prohibited under current National Forest Plan decisions and where road building is presently allowed. We recognize that our decision leaves out smaller roadless areas that were not considered during the inventory of federal roadless areas and that these areas serve important conservation goals (Strittholt & DellaSala 2001). For this study, the term 'roadless areas' refers to inventoried roadless areas.

We used three independently derived land cover maps for Montana, Wyoming and Idaho from the GAP Analysis Programme (Scott, Tear & Davis 1996). The Montana and Idaho GAP products were produced based on classification techniques by Redmond *et al.* (1998) for raw Landsat Thematic Mapper (TM) satellite imagery. Spatial resolution of the grid was 90 m for Montana and 30 m for Idaho. The Wyoming GAP Analysis Programme digitized land cover data in a vector format from Landsat TM satellite imagery at a scale of 1 : 100 000 (Gap Analysis Wyoming 1996). We converted Wyoming's vector map into a grid format and resampled the three data sets to 90-m resolution. Then we merged the three land cover maps into a single image and a common land cover classification scheme (Appendix 1).

Similar to most GIS databases, errors are associated with the land management status, inventoried roadless areas and land-cover grids. These grids represent a composite of data from many sources and include variations in mapping procedures and possible misclassifications that could potentially cause inconsistencies that are difficult to detect. However, we believe, based on professional judgement, that the error rate is not large enough to affect conclusions drawn from this large regional-scale analysis.

To investigate the representation of roadless areas at various elevation classes, we downloaded a digital elevation model from the 30-m National Elevation Dataset produced by the USA Geological Survey's EROS Data Center (Sioux Falls, SD). We reclassified the elevation range into 21 equal-interval classes ranging in 200-m increments from approximately 150 m to 4200 m.

DATA ANALYSIS

All data analyses were conducted in ARC/INFO and ArcView GIS software from Environmental Systems Research Institute (Redlands, CA).

Land cover representation

Using ARC/INFO, we combined the protected areas database with the land cover map. To calculate the percentage representation of each land-cover type, we divided the protected portion of each land-cover type by the total area of each land-cover type across the study area. Next, we appended the national forest inventoried roadless areas to the existing protected areas and repeated the same calculation described above to measure the additional representation of each land-cover type because of the inclusion of roadless areas. In addition, we calculated the percentage increase between each land cover percentage representation for protected areas alone and protected areas and roadless areas combined. This measure quantified the 'relative' ecological contribution from roadless areas for each land-cover type. We then ranked these land-cover types according to the level of representation within the existing protected areas.

Elevation representation

Using ARC/INFO, we combined the protected areas database with the 30-m digital elevation model. Similar to the procedure for land-cover types described above, we added the roadless areas to the existing protected areas, intersected this image with the elevation data, and calculated the change in representation for each elevation class provided by protection of roadless areas.

To examine the potential increase of landscape connectivity caused by roadless areas, we used ARC/INFO and FRAGSTATS (McGarigal & Marks 1995; McGarigal & Holmes 2002), a computer program developed to quantify heterogeneity of the landscape. We identified five landscape metrics available in FRAGSTATS to assess our three elements of landscape connectivity (McGarigal & Holmes 2002). To assess area, we used the metrics percentage land (PLAND), number of patches (NP) and patch size (AREA). We included the metrics NP and AREA to help explain the context of an increase in PLAND. For example, an increase in PLAND and AREA and a decrease in NP would indicate that the added roadless patches were located next to existing conservation patches, resulting in an increase in the size of patches and a decrease in the number of patches across the landscape. Conversely, a decrease in AREA and an increase in NP would indicate that the added patches were generally smaller and did not combine with existing patches.

To assess isolation we used nearest neighbour distance (ENN). A decrease or increase in ENN would indicate that patches are either located closer together or farther apart, respectively, across the landscape.

To assess aggregation, we used contagion (CONTAG). An increase in CONTAG would indicate that patches are, to a certain extent, aggregated together across the landscape.

Using FRAGSTATS, we selected and ran our five landscape metrics on the two grids described above (current protected areas only, and roadless areas and current protected areas combined). Each grid was a binary map where all grid cells that comprised the 'protected' and 'roadless' patches were classified as 1 and all other 'non-protected' grid cells were masked out as background (–99). For each landscape metric, we computed the mean, area-weighted mean and coefficient of variation where applicable. We then compared the differences in metrics between the two grids. In addition, differences in the mean, area-weighted mean and coefficient of variation helped to explain how the range of values for each metric were distributed when existing protected areas were compared with the conservation system including roadless areas.

Results

LAND COVER REPRESENTATION

In existing protected areas, burned forest and snowfields/ice had the highest land cover representation, 88% and 86%, respectively. Representation of other land-cover types, such as alpine meadows, whitebark pine, exposed rock/soil, subalpine meadows, wetlands, mixed subalpine forest and lodgepole pine, ranged from 31% to 71%.

The inclusion of roadless areas increased the representation of all land-cover types except for one, sand dunes (Table 1). Relative percentage increases ranged

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Table 1. Additional representation and percentage increase in representation of each land-cover type across the northern Rockies when national forest roadless areas are added to existing protected areas

Land-cover type	Existing level of representation (%)	Potential level of representation including roadless areas (%)	Percentage increase including roadless areas
Burned forest	88·12	93·09	5·65
Snowfields/ice	86·12	97·48	13·19
Alpine meadow	71·51	94·18	31·70
Mixed whitebark pine	59·62	84·94	42·46
Exposed rock/soil	44·67	59·92	34·12
Subalpine meadow	40·49	68·85	70·05
Wetlands	37·34	38·68	3·61
Mixed subalpine forest	32·20	68·63	113·11
Lodgepole pine	31·35	59·42	89·54
Mixed barren lands	21·66	22·61	4·37
Sand dunes	18·44	18·44	0·00
Mixed conifer	16·97	37·24	119·44
Mesic upland shrub	10·74	26·14	143·44
Shrub-dominated riparian	7·98	12·77	59·91
Forest-dominated riparian	7·18	12·14	69·11
Sagebrush	6·33	9·91	56·55
Juniper	5·87	6·80	15·95
Xeric upland shrub	5·85	7·97	36·33
Vegetated sand dunes	5·69	6·03	5·89
Western red cedar	5·57	22·00	295·08
Mud flats	5·33	7·39	38·79
Ponderosa pine	4·94	9·88	99·97
Aspen	4·48	25·99	479·80
Shrub–grassland associations	4·25	5·89	38·46
Western hemlock	3·36	23·62	602·54
Grasslands	2·49	3·64	46·31
Grass-dominated riparian	2·15	3·07	43·01
Salt-desert shrub flats	1·58	1·71	8·63
Bur oak woodland	0·00	2·40	NA

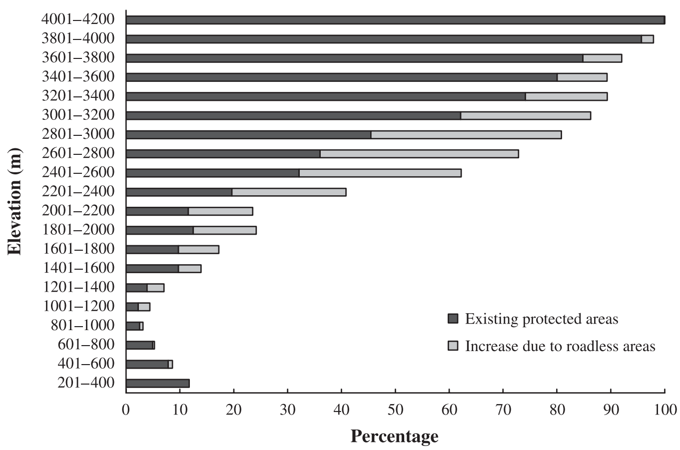

Fig. 2. Additional representation of elevation ranges resulting from the inclusion of roadless areas with protected areas for the northern Rockies. The *x*-axis represents elevation in 200-m increments and the *y*-axis shows absolute increase in percentage representation when roadless areas are added to protected areas. Black bars represent protected areas and grey bars represent roadless areas.

from 5% to 600%. Fifteen land-cover types increased by more than 40%, among them important ecological communities, western hemlock, aspen, ponderosa pine, western red cedar and sagebrush, each of which has less than 10% representation in current protected areas. Moreover, the addition of roadless areas represented one land-cover type, bur oak *Quercus macrocarpa* woodland, not present in protected areas.

ELEVATION REPRESENTATION

Our elevation analyses showed that elevations in the range of 2200–4200 m were well represented in protected areas (Fig. 2). The addition of roadless areas resulted in a large increase in representation of lands at elevations ranging from 1000 m to approximately 3400 m. For elevation ranges below 1000 m and above 3400 m, the

Table 2. Landscape metrics comparing the spatial pattern of protected areas alone with a scenario that includes protected areas and national forest roadless areas combined for the northern Rockies. + and – indicate an increase or decrease, respectively, in the metric value caused by the addition of roadless areas

Landscape Metrics	Protected areas	Protected and roadless areas	+/ –
Area			
Class area (ha)	8 814 900	15 673 600	+
Percentage land	9	16	+
Number of patches	770	722	–
Patch size (mean, ha)	11 447·92	21 708·59	+
Patch size (area-weighted mean)	1 105 055·78	2 505 909·11	+
Patch size (coefficient of variation)	977·39	1 069·74	+
Isolation			
Nearest neighbour (m)	7 013·72	5 353·11	–
Nearest neighbour (area-weighted mean)	3 153·73	2 518·75	–
Nearest neighbour (coefficient of variation)	122·47	134·16	+
Aggregation			
Contagion index	72·56	58·64	–

contribution of roadless areas was small. However, the proportion of area represented at lower elevations increased when we included roadless areas with protected areas.

Connectivity

Results from the landscape metrics showed that the addition of roadless areas increased regional connectivity for all three connectivity elements (Table 2). Area metrics demonstrated that the addition of roadless areas almost doubled the amount of area protected, rising from 9% to 16%, and the mean patch size in protected areas changed from 11448 ha to 21709 ha. The number of patches decreased from 770 to 722. Area-weighted mean patch size increases and the patch size coefficient of variation increased from 977 to 1070. Isolation metrics showed a decrease in the mean and area-weighted mean nearest-neighbour metrics when roadless areas were added. The mean distance between nearest protected patches decreased from 7014 m to 5353 m. The decrease in the area-weighted mean was less than the overall mean when patches of all sizes were considered. The coefficient of variation also increased for this metric. The aggregation metric (contagion) decreased from 72·56 to 58·64 when roadless areas were included, signifying more dispersion of patches across the landscape.

Discussion

Biodiversity representation

A review of the literature suggests that a given vegetation community is adequately represented when 12–25% of it is included in a conservation area (World Commission on Environment & Development 1987; Noss & Cooperrider 1994), although it is not certain that these thresholds are truly adequate to protect vegetation communities. Based on this range, we define land-cover types above 25% as adequately protected, land-cover types within the range of 12–25% as minimally protected, and those below 12% as underrepresented, similar to DeVelice & Martin (2001).

Our results show that roadless areas make a substantial contribution in maintaining regional biodiversity. One of our most important findings is that roadless areas would protect a wider range of land-cover types and elevation ranges than protected areas alone, especially those characteristic of mid- to low elevations that are underrepresented in protected areas. These lands are among the last remnants of biologically productive lands that have not been significantly altered through human settlements, resource extraction and road construction (Scott *et al*. 2001; Strittholt & DellaSala 2001). We also found that protected areas adequately represent land-cover types that are characteristic of higher elevations. This finding supports the generally accepted notion that wilderness areas and national parks mainly protect higher elevation ecological communities (Davis *et al*. 1996; Possingham, Ball & Andelman 2000). Contrary to DeVelice & Martin (2001), whose study found that roadless areas mainly occurred at mid- to lower elevations, but similar to Strittholt & DellaSala (2001), we found that roadless areas considerably increase the protection of higher elevations and corresponding cover types as well. The different results are probably because of the scale at which the studies were implemented. DeVelice & Martin's (2001) study included all roadless areas across the nation, incorporating a wide range of elevations from sea level to the highest peaks. Our study, and that of Strittholt & DellaSala (2001), focused on smaller regions at higher elevations.

Across the northern Rockies region (Montana, Wyoming and Idaho), protected areas adequately represent nine land-cover types, whereas five biologically important land-cover types, western hemlock, aspen, ponderosa pine, western red cedar and mesic upland shrub, are underrepresented in protected areas. However, the addition of roadless areas increases representation of two cover types (western hemlock and western red cedar) to the minimally protected threshold and two cover types (aspen and mesic upland shrub) to the adequately represented threshold (greater than 25%). Ponderosa pine, even though it increases by nearly 100%, remains underrepresented. Overall, the magnitude of the increased representation, from 100% to 600%, indicates that roadless areas can make substantial contributions to the protection of land-cover types that are not well represented in protected areas.

Increased representation of certain rare ecological communities is particularly important in a northern Rockies conservation strategy. Aspen, for example, is thought to be declining in the northern Rockies (Gallent *et al*. 1998). When roadless areas are added to protected areas, aspen moves up two full categories: from underrepresented to adequately represented, a 480% increase in representation for this forest type, on which many avian species depend upon (Hansen & Rotella 2000). Representation of whitebark pine changes from 60% to 85% when roadless areas are added. Whitebark pine is declining throughout North America due to blister rust *Cronartium ribicola*, an introduced disease, and is a 'keystone species' important for many higher elevation species (Keane, Morgan & Menakis 1994).

Elevation representation results demonstrate that protected areas are mainly located at higher elevations. We also found that roadless areas are generally concentrated at mid- to high elevations and represent a wider range of elevations, especially low- to mid elevations, than protected areas. However, our results show that protected areas encompass more lower elevation lands than roadless areas. This situation is somewhat deceiving. Representation of lower elevations in protected areas is largely a result of two well-placed low-elevation conservation areas: Hell's Canyon National Recreation Area and Missouri Breaks National Monument. In fact, low-elevation lands below 1000 m are not well represented in either protected areas or roadless areas. As a majority of lower elevation lands in the northern Rockies have been converted to other uses, it is of utmost importance to increase representation of lower elevation sites in protected areas (Strittholt & DellaSala 2001). Protection of these lower elevation roadless areas would contribute greatly to the conservation of lower elevation species and ecological communities that are poorly represented in protected areas.

LANDSCAPE CONNECTIVITY

Our analyses of three elements of connectivity show that roadless areas increase connectivity across the northern Rockies, and increase both the area and size of protected area patches. In addition, the number of protected area patches decreases with the addition of roadless areas because they combine with protected areas to form one larger patch. Larger patches will protect more species and more individuals, species with large home ranges, species sensitive to human activity, and more intact ecosystem processes than smaller areas (Askins, Philbrick & Sugeno 1987; Robbins, Dawson & Dowell 1989; Turner *et al*. 1993; Newmark 1995; Shafer 1995). Roadless areas also reduce the distance between protected areas and create a more evenly dispersed reserve system, critical for maintaining many species' movements and a large distribution of local populations (MacArthur & Wilson 1967; Murphy & Noon 1992; Reed, Johnson-Barnard & Baker 1996; Ritters *et al*. 1996; Beauvais 2000; Hansen & Rotella 2000; He, DeZonia, & Mladenoff 2000; Shinneman & Baker 2000).

Our results show an increase in the coefficient of variation for patch size and isolation metrics, which may be an important consideration in delineating conservation reserve systems capable of maintaining movements of various species and ecological processes (Wiens & Milne 1989; Wilcove & Murphy 1991; Noss 1992; Noss *et al*. 1996; O'Neill *et al*. 1996). Smaller patches may supplement larger reserves by protecting rare species that occur only in certain areas (Franklin & Forman 1987; Hansen *et al*. 1991; Shafer 1995). The dispersion of roadless areas may also contribute to greater resilience or survival of island populations by allowing a greater chance for species exchange, essentially maintaining a metapopulation or source–sink population structure (Wiens, Crawford & Gosz 1985; Pullium 1988; Gilpin & Hanski 1991; Murphy & Noon 1992). Many studies are investigating how species move through landscapes and their use of stepping-stone habitats, especially in fragmented landscapes (Freemark *et al*. 1993; With 1999; Beauvais 2000; Hansen & Rotella 2000; Holloway, Griffiths & Richardson 2003; Johnson, Seip & Boyce 2004). Being relatively undisturbed and well-distributed among protected areas, roadless areas are top candidates for the delineation of high-quality 'habitat connections' across the northern Rockies, particularly those that target rare or declining species. The loss or alteration of roadless areas may further reduce the movement of species among interdependent island populations located in protected areas and roadless areas, resulting in greater isolation.

Moreover, the addition of roadless areas increases the effective size of the three largest wilderness and national park complexes in the northern Rockies: the Greater Yellowstone Ecosystem, the Glacier National Park–Bob Marshall Ecosystem and the Central Idaho Ecosystem, where management challenges include maintaining large-scale ecological processes such as species' movements and natural fire across jurisdictional boundaries (Pickett & White 1985; Turner *et al*. 1993). Roadless areas not immediately adjacent to these complexes are dispersed in the surrounding landscape, which helps to decrease the degree of isolation between the complexes and possibly allows for species movement among these ecosystems.

MANAGEMENT IMPLICATIONS

Using research to guide reserve design and develop land protection policies is the strongest approach in conservation. The importance of intact, functioning natural ecosystems to the maintenance of native biodiversity and ecological processes is unquestioned (Wright, Dixon & Thompson 1933; MacArthur & Wilson 1967; Usher 1987; White 1987; Shafer 1995; Noss, O'Connell & Murphy 1997). The negative impacts of roads in natural areas are well known (Andrews 1990; Foreman & Wolke 1992; Reed, Johnson-Barnard & Baker 1996; Spellerberg 1998; Trombulak & Frissell 2000; McGarigal *et al*. 2001). Our landscape assessment demonstrates how roadless areas, the remaining relatively undisturbed forested lands in the northern Rockies, are essential for maintaining biodiversity and landscape connectivity in a conservation reserve strategy for this area. This has direct bearing on management decisions regarding the protection of roadless areas in this region. Our results, along with the findings of DeVelice & Martin (2001) and Strittholt & DellaSala (2001), highlight the important role of roadless areas in USA conservation efforts and contribute to the larger policy dialogue surrounding roadless areas.

The methods used in this study can help land managers determine appropriate guidelines to identify and assess roadless areas that are critical in maintaining regional biodiversity, ecosystem processes, landscape connectivity and overall intact ecosystem integrity. Land managers should avoid activities such as road building, logging, spread of exotic species, off-road vehicle use and exurban development in roadless areas that would result in their degradation or loss. If roadless areas are not protected from these activities as a matter of priority, it is possible that their potential contribution to conservation effort in the future will be diminished and existing protected areas surrounded by or in close proximity to roadless areas will be negatively affected as well. We recommend that roadless areas receive full protection and are managed responsibly, so that they can function as an important part of the current conservation reserve system in the USA.

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Supplementary material

The following material is available from http://www.blackwellpublishing.com/products/journals/suppmat/JPE/JPE996/JPE996sm.htm.

Appendix 1. Land-cover types across the northern Rocky Mountain region reclassified from USA Geological Survey's Biological GAP Analysis Programme (Scott, Tear & Davis 1996).

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ERRATA

The preceding paper unfortunately was published with several minor errors that are corrected below:

Page 181: Gregory H. Aplet's address is The Wilderness Society, 1660 Wynkoop Street, Ste. 850, Denver, CO 80202, USA.

"Forest Service" should be capitalized throughout.

Page 183, col. 2: The first sentence of the Study Area section should read: Of the 84 million ha of land that stretch across Montana, Wyoming and Idaho in the USA, roadless areas cover 6.8 million ha, and existing federal protected areas (wilderness areas, national parks, special management areas and national wildlife refuges) protect 8.8 million ha.

Page 184, par. 1, line 13: The correct spelling is: Douglas-fir *Pseudotsuga menziesii*.

Page 185, col. 2: The top of the column should begin with the heading *Landscape connectivity.*

Page 187, col. 2, par. 1.: The last sentence should read: Our study, and that of Strittholt and DellaSala (2001), focused on smaller regions, where national forests are concentrated at higher elevations.

Page 188, col. 1, par. 1: The third sentence should read: When roadless areas are added to protected areas, aspen moves up two full categories: from underrepresented to adequately represented, a 480% increase in representation for the forest type, upon which many avian species depend (Hansen and Rotella 2000).

Page 188, col. 1, par. 2: The first two sentences should read: Elevation representation results demonstrate that higher elevations are well represented in existing protected areas. We also found that roadless areas would add substantially to protected areas at mid- to high elevations.

Page 189, col. 2: The reference to Beauvais (2000) should refer to F.W. Smith.

Page 191, col. 2, last line: The manuscript was received 30 December 2003.

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FEATURE

Roadless areas and clean water

Dominick A. DellaSala, James R. Karr, and David M. Olson

Clean water, like biodiversity, is most closely linked to undisturbed natural ecosystems. When undisturbed watersheds in roadless and protected areas (e.g., national parks, state parks, wilderness areas, national monuments) are fragmented by roads, logging, and intensive recreation development, both water quality and biodiversity decline as hydrological integrity is lost (USFS 1972, 1979, 2001; Alexander and Gorte 2008; Anderson 2008). In the United States, inventoried roadless areas (IRAs) are lands without roads exceeding 2,000 ha (5,000 ac) that have been inventoried by the USDA Forest Service. IRAs collectively amount to approximately one third of the 77 million ha (193 million ac) of the 155 national forests but are disproportionately concentrated in western states (figure 1) (Trout Unlimited 2004; Anderson 2008). The roaded, intensively managed landscapes of the other national forest lands have been closely correlated with heavily sediment-laden streams and dramatic changes in flow regimes (Espinosa et al. 1997; Trombulak and Frissell 2000; CBD et al. 2001; Coffin 2007; Frissell and Carnefix 2007). While the biodiversity benefits of IRAs are well documented (DeVelice and Martin 2001; Strittholt and DellaSala 2001; Loucks et al. 2003; Strittholt et al. 2004; Gelbardi and Harrison 2005), little has been made of the importance of IRA water for downstream users and wildlife.

In this paper, we assess the importance of IRAs from a water quality perspective, including the likely water quality effects of developing IRAs. We provide conservative estimates of the economic impact of intact unroaded watersheds on national forests for clean water and associated water resource benefits. In particular, rising demand and shrinking water supply associated with changing climate will likely make intact areas in drought-prone regions of the West even more valuable and crucial to protect. Thus, our findings are especially relevant to drought-prone states considering development of IRAs. The state of Colorado, for example, with approximately 1.7 million ha (4.2 million ac) of IRAs, has been seeking federal permission to develop its IRAs for logging, expanding ski areas, coal-bed mining, and producing oil and gas (figure 2) (Anderson 2008; Colorado Division of Wildlife 2010; Colorado, State of 2010; Straub 2010, USFS 2011). Although we focus on IRAs throughout the western United States, we also emphasize the importance of uninventoried roadless areas (unroaded) <2,000 ha (Henjum et al. 1994; Greenwald 1998; Beschta et al. 2004) that collectively cover an area roughly 1.5 times that of the total IRA network (USFS 2000; Strittholt et al. 2004). Those smaller unroaded areas also play a strategic role in maintaining reliable supplies of high-quality water and protecting aquatic ecosystems.

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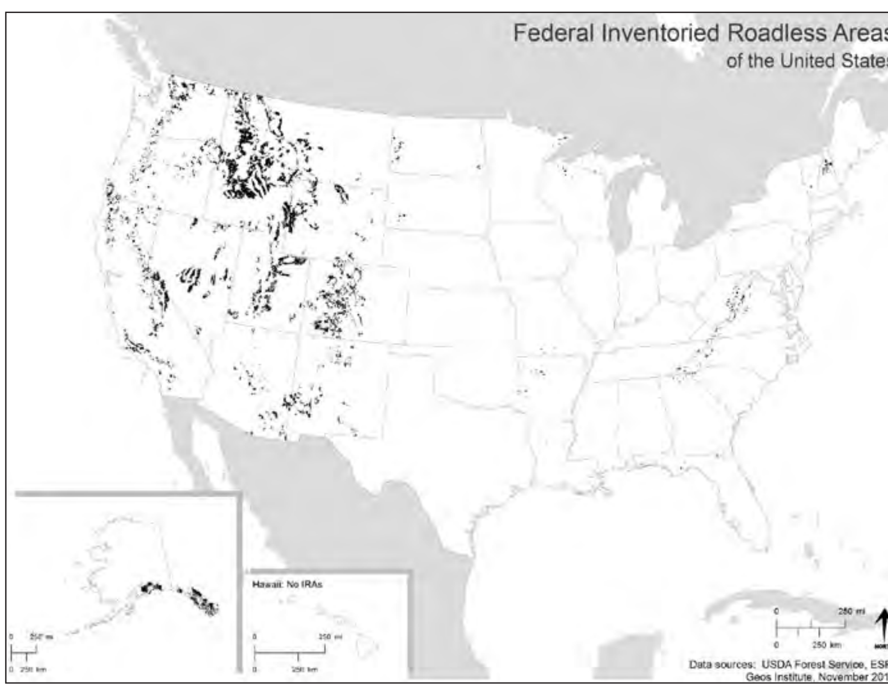
ROADLESS AREAS PROVIDE SUBSTANTIAL WATER RESOURCE BENEFITS

IRAs benefit society in many ways, including providing a valuable and increasingly rare natural supply of abundant, clean, and naturally reliable water (Sedell et al. 2000); affordable drinking water for municipal and rural communities; water for agricultural and industrial uses; flood control; in-stream aquatic recreation; aquifer recharge; flood protection; reliable water supply; diverse and productive fisheries; healthy aquatic ecosystems; resident and migratory waterfowl habitat; recovery of endangered species; and, increasingly, the vitality and sustainability of local economies (table 1). These benefits accrue nationally and at the local and regional levels.

National Benefits of Clean Roadless-Area Water. At least 124 million Americans directly benefit from water

Figure 1

Federal inventoried roadless areas (IRAs) of the United States (Source: USDA Forest Service).



originating from national forests (Sedell et al. 2000). In fact, national forests provide about 15% of the nation's runoff with an estimated net value of $3.7 (Sedell et al. 2000) to $27 billion (Krieger 2001). The water treatment value alone of National Forests ranges from $490 million (Loomis 2005) to $18 billion (Krieger 2001).

Because IRAs represent roughly a third of national forestland, by inference they contribute significantly to the overall run-off volume and value (Anderson 1997, 2008) estimated in billions of dollars annually (Loomis and Richardson 2001; Sechhi et al. 2005). For instance, using Forest Service data (USFS 2000), IRAs make up 661 of the 914 national forest watersheds, with 55% of the 914 watersheds acting as source areas for facilities that treat and distribute drinking water to the public. The cost-savings to water treatment plants and highway departments from avoiding sedimentation caused by logging in IRA watersheds is estimated at up to $18 billion annually (Loomis 1988). IRAs provide $490 million annually in waste treat-

Figure 2

Colorado's 2001 inventoried roadless areas (IRAs) are shown in light gray, the 2011 proposed Colorado roadless areas (CRAs) are shown in gray, and overlap between CRAs and IRAs is shown in black. Water quality will be most impacted by changes of allowable activities within existing IRAs relative to changes in designated areas (USFS 2011).

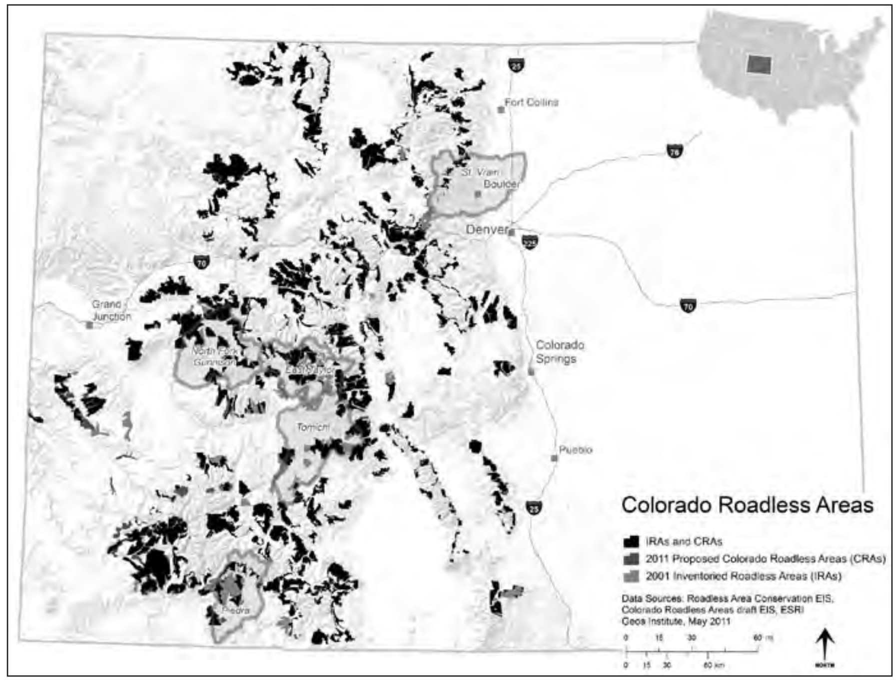


Table 1

General ecosystem services and benefits related to water that are provided by undisturbed IRAs and watersheds (derived from Greenway 1996; Costanza et al. 1997; Talberth and Moskowitz 1999; GAO 2000;Heal 2000, Loomis and Richardson 2001; Sedell et al. 2000; Krieger 2001; Dombeck 2003; Berrens et al. 2006).

	Benefits
Off-stream benefits	Low treatment costs for water for all beneficiaries
	Low price per unit volume costs for water for all beneficiaries
	High-quality and abundant drinking water for rural communities and municipal water supplies
	High-quality water for agricultural and industrial purposes
	High-quality water for downstream livestock production
	High-quality water for reduced health care and epidemic control
	Reduced costs of flood damage and flood control; enhanced local economies and property values
	Community benefits, including jobs, income, favorable trends for key economic indicators, and economic sustainability and stability
	Recharging of groundwater aquifers
	Healthy terrestrial and riparian ecosystems and their component species, sustained ecological and evolutionary processes, and resilient ecosystems
In-stream benefits	Healthy aquatic ecosystems
	Recovery of endangered species and protection of refugia
	Diverse and productive fisheries
	High-quality habitat for wildlife, including migratory waterfowl and game and nongame species
	Aquatic recreation such as swimming, rafting, and boating; enhancement of hiking and camping
	The inherent value of wild rivers and wilderness (including passive use benefits such as option, bequest, and existence values)
	Moderation of runoff and streamflows (e.g., lower peak flows, higher low flows, year-round water)
	Soil stabilization and erosion control
	Scientific value (intact watersheds are very rare today)
	Maintaining sediment production to streams at normal background rates
	Reducing potential for damage to downstream properties and water users during periods of high flow
	Breakdown and containment of waste and toxins (e.g., atmospheric, prior use)

ment services through recovering mobile nutrients and cleansing the environment, both processes that involve water flow through intact watersheds (Loomis and Richardson 2001).

Regional Benefits of Clean Roadless-Area Water. In the US Rocky Mountains, roughly one third of utilized streamflow is derived directly from IRAs (which cover a quarter of Colorado's headwaters), with cities like Denver receiving about 30% of their water supply from IRA watersheds. Annually, IRAs in Colorado are estimated to provide an equivalent of nearly 2.5 times Denver's annual water use (Doyle and Gardner 2010; Denver Water 2010). Similarly, IRAs in New Mexico provide an estimated water quality benefit up to $42 million annually (Berrens et al. 2006).

Flood Control Protection and Inventoried Roadless Areas. The intact watersheds of IRAs are especially important for ameliorating the frequency and intensity of flooding, saving millions of dollars annually from averted floods and associated sedimentation, a service that will only increase in value as climate change drives more floods (Seeds 2010). Dredging reservoirs to increase capacity and channels to enable navigation costs cities, states, and ultimately taxpayers millions annually. Salem, Oregon, spent approximately $100 million on new treatment facilities after logging in upper watersheds created conditions leading to mass sedimentation in its watershed following storms in 1996 (Schwickert and Mauldin 1997; Talberth and Moskowitz 1999). In addition, Seattle, Washington, deferred a $150 million filtration plant expenditure through an intensive watershed rehabilitation program that will decommission 480 km (300 mi) of roads over a 10-year period, fix road erosion problems, and limit access and high-risk activities for fire and sedimentation within their watersheds (Seeds 2010).

Recreation Benefits and Strong Local Economies. IRA water benefits outdoor recreation and the people that either engage in or earn their living from outdoor recreation. The nation's IRAs generate $600 million annually from recreation (Loomis and Richardson 2001). Passive-use values (i.e., the intrinsic value of wilderness, wildlands, and benefits for the future) are estimated at an additional $280 million annually. At the regional scale, New Mexico IRA water provides an estimated $27 million active outdoor recreation benefit and a $14 million passive-use benefit annually (Berrens et al. 2006). For many visitors, much of the attraction to wildlands is associated with the presence of clean and abundant water—a dwindling resource as logging, grazing, and road-building continues across mountain landscapes and droughts from a changing climate intensify in much of the West (Saunders et al. 2008).

Freshwater Biodiversity and Healthy Fisheries. Clean water from IRAs also maintains healthy fisheries, such as salmon and trout fisheries, sustains viable aquatic ecosystems, and helps protect threatened species and ecosystems (Abell et al. 2000; Trout Unlimited 2004). Indeed, IRAs may act as important refugia for many salmon and trout populations, as well as for a diversity of endangered freshwater species (Henjum et al. 1994; Huntington 1998; NRC 1996; Trombulak and Frissell 2000; CBD et al. 2001; Strittholt and DellaSala 2001; Oechsli and Frissell 2002; Strittholt et al. 2004; Petersen 2005). Restoration of salmon and trout fisheries in places with high road densities will likely fail without the pivotal role provided by IRAs as fishery strongholds.

ROADLESS AREAS ARE IMPORTANT SOURCES FOR DRINKING WATER

The distribution of IRAs across prime hydrologic real estate—headwaters and upper watersheds—makes them particularly valuable for providing reliable supplies of clean water. In Colorado, IRAs occur in the headwaters of all major drainages, covering roughly a third of upper watersheds in the state. Indeed, most IRAs are located in mountainous terrain in western states, including Oregon, Idaho, New Mexico, Utah, Montana, California, and Washington. This extensive coverage of IRAs in headwaters, and because they are often the last minimally disturbed watersheds within larger landscapes of degraded lands, makes them hydrologic hotspots—areas with relatively small spatial extent that have a disproportionately important role in producing abundant and reliable clean water (Frissell and Carnefix 2007).

For many major drainages (entire watersheds of major rivers, such as the Columbia River Basin), IRAs and other wilderness areas represent the last few percentages (typically 1% to 5%) of the landscape with a minimally disturbed, or near natural, hydrology. As in many other ecological contexts, losing the last relatively natural systems typically results in major losses in water resource benefits, losses that can only be compensated by very expensive actions. The known relationship between watershed degradation and water quality decline deserves to be more rigorously incorporated as a central foundation for decisions on watershed management and protection.

Developing Roadless Areas Degrades Water Quality. In addition to their keystone location within watersheds, roadless areas typically encompass the most fragile of natural landscapes—montane forests and meadows. Road building and other intensive management in these otherwise intact areas damage their ability to provide clean water for downstream communities and biodiversity over both short and long terms (Beschta 1978; Forman and Alexander 1998; Lugo and Gucinski 2000; Trombulak and Frissell 2000; Gucinski et al. 2001; Coffin 2007). Logging, including post-disturbance, fire-risk reduction, forest health, and insect control; livestock grazing; mining; and road building are responsible for chronic and acute sedimentation of aquatic ecosystems, alter overland flow and stream structure, and change a range of physical and biological features by causing more frequent and intense floods, decreasing available water throughout the year, increasing stream and ambient temperatures, and elevating turbidity and nutrient levels (Beschta 1978; Fleischner 1994; Trombulak and Frissell 2000; DellaSala et al. 2006; Coffin 2007). Logging roads have been linked to great increases in erosion rates and sediment delivery to streams—up to 850% over rates in undisturbed habitat—with long-term and often catastrophic impacts on stream biota, aquatic ecosystems, and water quality (Fredricksen 1970; Megahan and Kidd 1972; Amaranthus et al. 1985; Bilby et al. 1989; King 1989, 1993; Haynes and Horne 1997; Jones et al. 2000; Wemple and Jones 2003).

Depending on severity and duration of impacts, disturbance can elevate average turbidity levels well above background levels (Seeds [2010] provides examples from Oregon), along with triggering more frequent and intense turbidity spikes that are a major source of excess costs to municipal water supply departments. Relative to roadless watersheds with intact natural vegetation, intensively managed watersheds also produce less available water (i.e., average monthly usable raw water) due to intensified high flows with very high turbidity and exacerbated low flow conditions (Seeds 2010). The monthly reliability of water is also diminished.

Even small disturbances in upper watersheds can result in significant, cumulative, and long-term impacts to downstream water and aquatic ecosystems (Platts and Nelson 1985; Boise National Forest 1993; McIntosh et al. 1994, 1995). In unstable terrain, for instance, small areas (e.g., less than 10% of a watershed's area) of low-intensity disturbance, including roads, may greatly increase the frequency and size of mass erosion events, with subsequent acute and chronic reduction in downstream water quality. Management activities that damage natural vegetation typically result in loads of suspended solids that exceed background levels and more frequent and intense spikes in suspended solids stemming from an increase in mass erosion events like landslides, debris flows, and bank failures. These impacts are strongly correlated with roads, as well as with logging and grazing (Amaranthus et al. 1985; Fleischner 1994; Trombulak and Frissell 2000; Coffin 2007).

Rising Demand and Climate Change Diminish Water Supply. Population in the West is projected to increase by 300% within just 30 years, with similar increases in demand for water (Sedell et al. 2000). Urban and exurban areas are growing exponentially, including communities adjacent to wilderness areas and IRAs (Theobald 2005). The demand for water in Colorado is expected to triple by 2050. Similarly, the number of people relying on national forest water has doubled in Oregon in the last 30 years, and 86% of the population of Washington rely on national forest water to some degree (Sedell et al. 2000).

The dramatic population growth in the West is concurrent with a warming and drying climate in many places. Temperatures are increasing, snow pack is declining and melting sooner, and drought and summer water deficits are more frequent and longer (Barnett et al. 2008; Mohammed and Tarboton 2008; Saunders et al. 2008; Miller et al. 2010). Streamflow reductions ranging from 10% to 35% are likely for the western states over the next half century as a consequence of climate change (Barnett and Pierce 2009). A 10% drop in streamflow is considered calamitous by municipal water districts. More frequent and intense flood events are also likely in places (Raff et al. 2009), despite drying conditions. Costs for flood control, repair and reconstruction, and insurance rates will also increase (GAO 2007). These events will worsen the severe and unprecedented droughts already afflicting much of the West (Drechsler et al. 2006; Saunders et al. 2008).

SOLUTION: A LIGHT HYDROLOGICAL FOOTPRINT IN ROADLESS AREAS

IRAs should be managed in the same way many municipalities manage their watersheds—sustaining a light ecological and hydrological footprint and hydrologic restoration through decommissioning or, even better, obliteration of roads (Barten et al. 1998; NRC 2000; Payne et al. 2004; Gallo et al. 2005; Postel and Thompson 2005; Seeds 2010). The most cost-effective and prudent approach to maintain water supplies and high-quality fresh water in the face of population growth and climate change is to manage upper watersheds in a roadless condition with undisturbed natural vegetation. The high, long-term economic cost of degrading clean water for millions of people, by itself, is argument strong enough to continue protection of the current roadless areas network either at national or state levels. Development of IRAs, as proposed in Colorado, would primarily provide opportunities for short-term gains, but the substantial and long-term impacts on water quality and availability will come at a time of increasing demand and shrinking supply. Managers should, therefore, treat IRAs as natural reservoirs of high quality water for downstream users before approving development projects. Cost-benefit analyses should include regionally and locally specific estimates of water quality to better inform project management decisions that may reduce the value of high-quality water in the short and long run.

CONCLUSIONS

Roadless areas and the relatively intact ecosystems they maintain provide many important biodiversity benefits, including acting as strongholds for threatened freshwater species. Beyond these important values, their role in producing clean and reliable water for people and economies is more likely to compel decision-makers to leave roadless areas undeveloped. We reviewed the importance of inventoried roadless areas on national forests in the United States to determine their importance in providing clean water for downstream users. We concluded that (1) many intact watersheds are in headwaters, (2) they supply downstream users with high-quality drinking water, and (3) developing these watersheds comes at significant costs associated with declining water quality and availability. Several case studies from the western United States, particularly Colorado, demonstrated the importance of assessing the diverse consequences of developing roadless areas. Managers should perform comprehensive cost-benefit analyses when weighing development options. A light-touch hydrological footprint is recommended to sustain the many values that derive from roadless areas, especially clean and abundant water.

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Decision Notice &
Finding of No Significant Impact
Pronghorn Migration Corridor Forest Plan Amendment

USDA Forest Service
Bridger-Teton National Forest
Wyoming

Decision and Reasons for the Decision

Background

The pronghorn (*Antilocarpa americana)* that summer in Jackson Hole migrate annually between there and wintering areas in the Green River basin. Documented round trip migration distances from 175 to 330 miles make this the longest known terrestrial animal migration in the 48 contiguous states. Typically, the pronghorn migrate through the corridor in April or May and again in October or November. These pronghorn are a part of the impressive panorama of free-ranging native Rocky Mountain mammals in northwest Wyoming. This landscape and its wildlife draw tourists from around the world and support a robust regional economy.

A significant portion of the full migration route of these pronghorn is within the Bridger-Teton National Forest. The Forest portion extends from the Forest boundary near the Green River Lakes Road north of Pinedale in Sublette County, Wyoming to the Forest boundary with Grand Teton National Park northeast of Kelly in Teton County, Wyoming. It includes approximately 47,000 acres within the Pinedale and Jackson Ranger Districts of the Bridger-Teton National Forest.

Managing this migration corridor to facilitate continued successful movement of pronghorn will help ensure protection of this herd and its migration. The purpose of this amendment to the Bridger-Teton National Forest Land and Resource Management Plan (Forest Plan) is to ensure that projects, activities, and facilities authorized by the Forest Service on National Forest System lands within the corridor allow for continued successful pronghorn migration.

It should be noted that the Forest Service by itself cannot guarantee continued successful migration of this herd over the entire migration route. There are numerous factors beyond Forest Service control such as activities on lands under other jurisdictions within the migration route.

Decision

Based upon my review of the Environmental Assessment (EA), I hereby amend the Bridger-Teton National Forest Land and Resource Management Plan by 1) designating a Pronghorn Migration Corridor as shown on the attached map; and 2) adding the following standard, "All projects, activities, and infrastructure authorized in the designated Pronghorn Migration Corridor will be designed, timed and/or located to allow continued successful migration of the pronghorn that summer in Jackson Hole and winter in the Green River basin." This amendment does not remove any current Forest Plan direction for the area encompassed by the corridor; it simply designates the corridor and adds the above standard. This amendment makes no decisions about the compatibility of specific uses with the pronghorn migration, but requires that all uses be found to allow continued migration before they are authorized.

Activities currently authorized by the Forest Service within this migration corridor, including livestock grazing operations, coexist with the currently successful pronghorn migrations, so changes to current activities and infrastructure are not required by this amendment.

Before future activities can be authorized, a determination must be made that the activity will allow continued successful migration.

It is important to note that, while the full length of the pronghorn migration route includes lands under various jurisdictions, this Forest Plan amendment applies only to National Forest System lands within that larger corridor. Furthermore, the amendment does not constrain activities on private land within the Forest boundary.

Reasons for the Decision

I have decided to create the Forest Plan amendment because it meets the purpose and need of ensuring that Forest Service authorized activities and infrastructure allow continued successful pronghorn migration in the corridor. Furthermore, I find that there are no unacceptable impacts from the amendment. As noted above, activities currently authorized by the Forest Service within the corridor coexist with successful migration, so changes to current activities will not be required by this amendment.

Other Alternatives Considered

In addition to the selected alternative, I considered the No Action alternative. Under the No Action alternative there would be no Forest Plan amendment and current management plans would continue to guide management of the area. This alternative does not meet the purpose and need of ensuring that Forest Service authorized activities in the corridor allow continued successful pronghorn migration.

Public Involvement

The proposal was provided to the public and other agencies for comment in a Scoping Statement dated March 6, 2008. The proposal was listed in the Bridger-Teton Schedule of Proposed Actions on April 1, 2008. Comments were received from government entities such as the Bureau of Land Management, Grand Teton National Park, and the Wyoming Game and Fish Department; from livestock associations and permittees; from conservation organizations; and from many private citizens. Using the comments received from scoping, the interdisciplinary team developed the issues that were addressed in the EA.

Approximately 19,400 emails were received supporting the proposed amendment. Several livestock interests were concerned that the proposal could negatively affect livestock grazing operations. Because current grazing operations coexist with successful migration, current grazing operations will not be affected by this amendment. Future grazing operations will need to be designed to allow continued successful migration. Some conservation organizations wanted specific restrictions added to the amendment such as a decision that no oil and gas leasing be authorized in the corridor. This amendment makes no decisions about the compatibility of specific future uses with the pronghorn migration, but requires that all future uses allow continued migration. I feel that this meets the purpose and need of the amendment.

Finding of No Significant Impact

After considering the effects described in the EA, I have determined that this amendment will not have a significant effect on the quality of the human environment considering the context and intensity of impacts (40 CFR 1508.27). Thus, an environmental impact statement will not be prepared. I base my finding on the following:

1. My finding of no significant impacts is not based on a belief that the benefiscial effects outweigh significant adverse impacts. Rather, it is my finding that there are no significant adverse impacts.

2. There will be no significant effects on public health and safety, because the amendment is limited in scope and does not authorize any specific activity on the ground that could affect public health or safety.

3. There will be no significant effects on unique characteristics of the area, because the amendment is limited in scope and does not authorize any specific activity on the ground that could impact the unique characteristics of the area.

4. The effects on the quality of the human environment are not likely to be highly controversial because there is no known scientific controversy over the impacts of the project.

5. The effects are not highly uncertain, and do not involve unique or unknown risk.

6. The action is not likely to establish a precedent for future actions with significant effects.

7. The cumulative impacts are not significant; this is addressed in the EA.

8. The action will have no significant adverse effect on districts, sites, highways, structures, or objects listed in or eligible for listing in the National Register of Historic Places and will not cause loss or destruction of significant scientific, cultural, or historical resources. This plan amendment authorizes no specific actions on the ground that could cause such effects. Future actions proposed within the migration corridor will still be subject to National Historic Preservation Act Section 106 review by the BTNF and the Wyoming State Historic Preservation Office.

9. As discussed in the Biological Assessment (BA) for this amendment, the action will not adversely affect any endangered or threatened species or its habitat that has been determined to be critical under the Endangered Species act of 1973. The BA documents a determination of “no effect” on the Canada lynx and on the Kendall warm springs dace, the only threatened or enangered species in the area.

10. The amendment does not threaten a violation of Federal, State, and local laws or requirements for the protection of the environment.

Findings Required by Other Laws and Regulations

This decision to amend the Forest Plan is consistent with the National Forest Management Act and its implementation regulations. Because the amendment does not result in significant changes to multiple-use goals and objectives for long-term land and resource management, the proposed amendment is considered to be "non-significant" according to the planning regulations at 36 CFR 219.14(2). Therefore, this amendment is authorized in this Decision Notice. The amendment is also consistent with the Forest Plan's goals, objectives, and specific management direction for the Forest, Management Areas, and Desired Future Conditions. As noted in the Decision section, this amendment does not remove any current Forest Plan direction for the area, it simply adds an additional standard to the corridor.

Implementation Date

This amendment will be implemented 7 days after the legal notice of this decision has been published in the Casper Star-Tribune and the appeal period has begun.

Administrative Review or Appeal Opportunities

This decision is subject to appeal pursuant to 36 CFR 217.3. Appeals must meet the content requirements of 36 CFR 217.9. A written appeal must be postmarked or received by the Appeal Reviewing Officer within 45 days of the date of publication of the legal notice of this decision in the Casper Star-Tribune. Appeals must be sent to: Regional Forester, Intermountain Region USFS, 324 25th Street, Ogden, Utah 84401; by fax to 801-625-5277; or by email to: appeals-intermtn-regional-office@fs.fed.us. Emailed appeals must be submitted in rich text (rtf) or Word (doc) and must include the project name in the subject line. Appeals may also be hand delivered to the above address, during regular business hours of 8:00 a.m. to 4:30 p.m. Monday through Friday.

Contact

For additional information concerning this decision or the Forest Service administrative appeal process, contact John Kuzloski by mail at the Bridger-Teton National Forest, P.O. Box 1888, Jackson, WY 83001; by email at jkuzloski@fs.fed.us or by phone at (307) 739-5568.

*_/s/ Kniffy Hamilton_*___________________ _May 31, 2008_
CAROLE 'KNIFFY' HAMILTON Date
Forest Supervisor
Bridger-Teton National Forest

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Assessing the value of roadless areas in a conservation reserve strategy: biodiversity and landscape connectivity in the northern Rockies

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Summary

1. Roadless areas on United States Department of Agriculture (USDA) Forest Service lands hold significant potential for the conservation of native biodiversity and ecosystem processes, primarily because of their size and location. We examined the potential increase in land-cover types, elevation representation and landscape connectivity that inventoried roadless areas would provide in a northern Rockies (USA) conservation reserve strategy, if these roadless areas received full protection.
2. For the northern Rocky Mountain states of Montana, Wyoming and Idaho, USA, we obtained GIS data on land-cover types and a digital elevation model. We calculated the percentage of land-cover types and elevation ranges of current protected areas (wilderness, national parks and national wildlife refuges) and compared these with the percentages calculated for roadless and protected areas combined. Using five landscape metrics and corresponding statistics, we quantified how roadless areas, when assessed with current protected areas, affect three elements of landscape connectivity: area, isolation and aggregation.
3. Roadless areas, when added to existing federal-protected areas in the northern Rockies, increase the representation of virtually all land-cover types, some by more than 100%, and increase the protection of relatively undisturbed lower elevation lands, which are exceedingly rare in the northern Rockies. In fact, roadless areas protect more rare and declining land-cover types, such as aspen, whitebark pine, sagebrush and grassland communities, than existing protected areas.
4. *Synthesis and applications*. Landscape metric results for the three elements of landscape connectivity (area, isolation and aggregation) demonstrate how roadless areas adjacent to protected areas increase connectivity by creating larger and more cohesive protected area 'patches.' Roadless areas enhance overall landscape connectivity by reducing isolation among protected areas and creating a more dispersed conservation reserve network, important for maintaining wide-ranging species movements. We advocate that the USDA Forest Service should retain the Roadless Area Conservation Rule and manage roadless areas as an integral part of the conservation reserve network for the northern Rockies.

Key-words: conservation, elevation zones, land-cover types, landscape metrics, national forests, reserve design

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Introduction

A growing body of scientific evidence indicates that the current USA system of federal protected areas (designated wilderness areas, national parks and national wildlife refuges) may be too small and disconnected to protect against the decline and loss of native species diversity or to accommodate large natural ecosystem processes (Wright, Dixon & Thompson 1933; MacArthur & Wilson 1967; White 1987; Wilcove 1989; Baker 1992; Turner *et al*. 1993; Noss & Cooperrider 1994; Reice 1994; Newmark 1995; Sinclair *et al*. 1995; Soule & Terborgh 1999). Expanding road networks, human settlements, resource extraction and other encroachments on the landscape have increased the fragmentation and loss of natural areas. Such disturbances have isolated many protected areas, causing them to function as terrestrial 'islands' surrounded by a matrix of lower quality altered lands (Harris 1984; Pickett & White 1985; Wiens, Crawford & Gosz 1985; Turner 1989; Saunders, Hobbs & Margules 1991). The long-term persistence of many species within protected areas is dependent on the degree of human activities and land-use practices on lands adjacent to and near protected areas. There is a need to identify relatively undisturbed lands located outside protected areas that may increase the potential of protected areas in maintaining native biodiversity and certain ecological processes, and to include these lands within the conservation reserve system before they are lost or altered.

Inventoried roadless areas, large tracts of relatively undisturbed land on USA Forest Service lands, are often left out of landscape assessments for identifying functional conservation reserves. Only two studies (DeVelice & Martin 2001; Strittholt & DellaSala 2001) have analysed the contribution that roadless areas make to the current protected areas reserve network. However, more than one-third of inventoried roadless areas on national forests are adjacent to protected areas (DeVelice & Martin 2001). They hold the potential to increase the size and connectivity of designated wilderness areas, national parks and national wildlife refuges, thus increasing the ability of protected areas to maintain natural landscape dynamics and native species population viability over the long term. Smaller, isolated roadless areas are also important because they may contain rare species, capture more habitat variation, including underrepresented habitat types, and may function as 'stepping stones' that connect current protected areas across a landscape (Shafer 1995; Strittholt & DellaSala 2001).

There is a precedent for the protection of national forest roadless areas. The USA Congress has designated as wilderness more than half, 6 million ha, of roadless areas that the Forest Service inventoried in national forests in the 1970s. In 1998, the Forest Service began to devise regulations aimed at protection of roadless area characteristics in national forests. In May 2000, the agency released its proposed rule, familiarly known as the Roadless Rule, and draft environmental impact statement. Eight months later, the Forest Service adopted the rule. In July 2004, the Forest Service proposed to repeal the Roadless Rule and replace it with a state petition and rule-making process, which would offer less protection by presumably opening national roadless areas to all forest service activities and requiring state governors to 'opt in' Roadless Rule protections affirmatively for any roadless area.

Included in the Roadless Rule environmental impact statement was an evaluation of the potential contribution that protection of roadless areas could make to the conservation of biodiversity at a national scale (USDA Forest Service 2000b). In that evaluation, DeVelice & Martin (2001) found that the inclusion of roadless areas in the network of federal protected areas would expand representation of ecoregions in protected areas, increase the acreage of reserved areas at lower elevations, and increase the number of areas large enough to provide refuge for wide-ranging species.

Strittholt & DellaSala (2001) focused on similar questions at a regional scale for the Klamath-Sikiyou area in southern Oregon and northern California, USA. They found that roadless areas protect a wide range of ecological attributes, especially at mid- to lower elevations, important in this region. They also concluded that roadless areas increase the connectivity among ecoregions.

The northern Rocky Mountain states of Montana, Wyoming and Idaho comprise a region particularly rich in roadless areas, roughly 2·6 million ha, providing a unique opportunity to create a relatively intact reserve design that captures important elements of conservation for the northern Rockies. Using two key concepts in conservation biology, biodiversity representation and landscape connectivity, we investigated the potential contributions of national forest roadless areas to the protected areas reserve network across the northern Rocky Mountain region.

DIVERSITY REPRESENTATION

An important goal in the design and establishment of conservation reserves is to represent a full range of native biodiversity (Shelford 1926; Margules, Nicholls & Pressey 1988; Church, Stoms & Davis 1996; Possingham, Ball & Andelman 2000). Even though this goal has been articulated for some time, most protected areas are demarcated around areas with high scenic and recreational attributes (Davis *et al*. 1996). As a result, existing protected areas in the northern Rockies are, for the most part, concentrated at higher elevations, where other important elements of biodiversity are most likely to be poorly represented (Scott *et al*. 2001).

Representation of a full range of biodiversity in reserves requires an understanding of all species and ecosystem processes operating within a given landscape. However, many researchers have used ecological communities and elevation ranges as coarse-scale surrogates for native biodiversity in the design of conservation reserves (Scott *et al*. 1993; Host *et al*. 1996). This concept is based on the idea that if a full range of ecological communities and elevation ranges is protected, it is more likely that many ecological communities, wide-ranging species and ecosystem processes will be maintained in the reserves. In the northern Rockies, ecological communities are often associated with elevation gradients (Hansen & Rotella 1999). Hence, roadless areas situated at middle and lower elevations may make valuable contributions in protecting many elements of biodiversity that are currently not well represented in protected areas (DeVelice & Martin 2001).

LANDSCAPE CONNECTIVITY

Connectivity refers to the degree to which the structure of a landscape helps or hinders the movement of wildlife species or natural processes such as fire (Wiens, Crawford & Gosz 1985; Turner *et al*. 1993; Noss & Cooperrider 1994; Bascompte & Solé 1996; With 1999). A 'well-connected' area can sustain important elements of ecosystem integrity, namely the ability of species to move and natural processes to function, and is more likely to maintain its overall integrity compared with a highly fragmented area.

Roads are highlighted in the scientific literature as major causes of landscape fragmentation, and function as barriers to organism movements, resulting in a reduction of overall landscape connectivity for many native species. The effects of roads are broad and include mortality from collisions, modification of animal behaviour, disruption of the physical environment, alteration of chemical environments, spread of exotic and invasive species, habitat loss, increase in edge effects, interference with wildlife life-history functions and degradation of aquatic habitats through alteration of stream banks and increased sediment loads (Franklin & Forman 1987; Andrews 1990; Noss & Cooperrider 1994; Reice 1994; Reed, Johnson-Barnard & Baker 1996; Trombulak & Frissell 2000; McGarigal *et al*. 2001). Thus, the addition of roadless areas to existing protected areas reserve is likely to maintain or increase landscape connectivity, as well as increase the integrity of protected areas.

With the advent of landscape metrics, it is now possible to quantify connectivity for landscapes, land-cover types, species' habitats, species' movements and ecosystem processes across a given region (O'Neill *et al*. 1988; McGarigal & Marks 1995; Gustafson 1998; With 1999). Many different metrics that quantify spatial characteristics of patches or entire landscape mosaics have been described (Turner & Gardner 1991; McGarigal & Marks 1995; Ritters *et al*. 1995; Hargis, Bisonette & David 1998; Dale 2000; Jaeger 2000; McGarigal & Holmes 2002). We chose metrics that measure three elements of landscape connectivity: area, isolation and aggregation.

Area

It is known that larger areas (patches) generally contain more species, more individuals, more species with large home ranges and/or sensitive to human activity, and more intact ecosystem processes than smaller areas (Robbins, Dawson & Dowell 1989; Turner *et al*. 1993; Newmark 1995; Shafer 1995). Higher numbers of patches will usually contribute to greater resilience of populations and may also increase the utility of patches that act as 'stepping stones' or connectors across a landscape (Buechner 1989; Lamberson *et al*. 1992).

Isolation

The distance between patches plays an important role in many ecological processes. Studies have shown that patch isolation is the reason that fragmented habitats often contain fewer bird and mammal species than contiguous habitats (Murphy & Noon 1992; Reed, Johnson-Barnard & Baker 1996; Beauvais 2000; Hansen & Rotella 2000). As habitat is lost or fragmented, residual habitat patches become smaller and more isolated from each other, species movement is disrupted, and individual species and local populations become isolated (Shinneman & Baker 2000).

Aggregation

The spatial arrangement of patches may help to explain how certain species are found in patches located close together and are not found in patches that are more isolated, or vice versa (Ritters *et al*. 1995; He, DeZonia, & Mladenoff 2000). This concept generally follows the ideas developed in island biogeography theory (MacArthur & Wilson 1967) and metapopulation theory (Levins 1969, 1970).

For some species or natural processes, the isolation or aggregation of patches across the landscape may be more important, for others, area may be the key element. Together, these three elements offer a comprehensive assessment of the importance of roadless areas to the maintenance of overall landscape connectivity and ecosystem integrity of current protected areas in the northern Rockies.

In this study, we aimed to assess the extent to which roadless areas increase biodiversity representation and landscape connectivity when they are included in the protected areas reserve network for the northern Rockies.

Methods

STUDY AREA

Of the 84 million ha of land that stretch across Montana, Wyoming and Idaho in the USA, roadless areas cover 2·6 million ha and existing federal protected areas (wilderness areas, national parks, special management areas and national wildlife refuges) protect almost 8·7 million ha. Within this region, three large, relatively

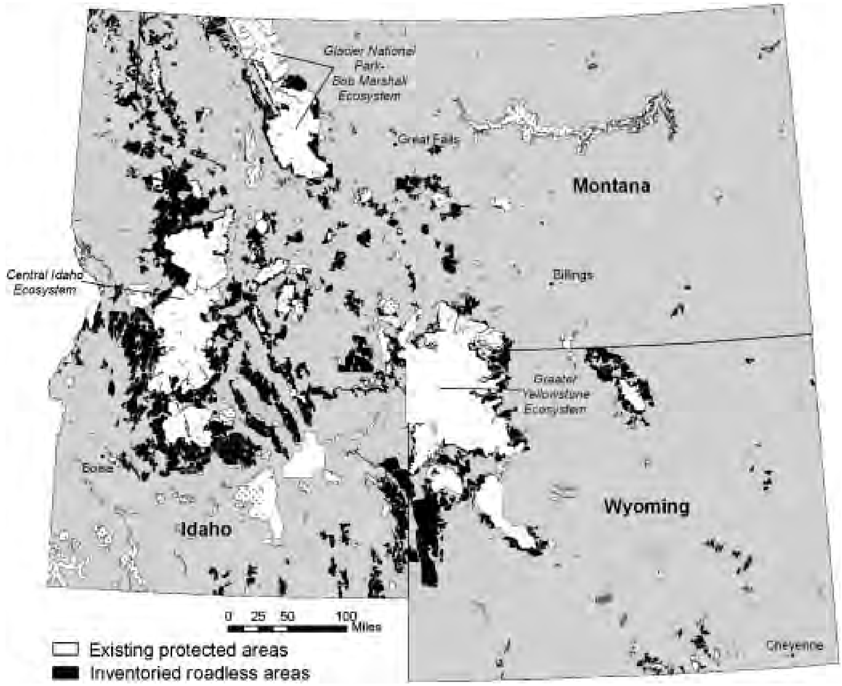

Fig. 1. Roadless areas and protected areas across the states of Idaho, Montana and Wyoming, USA.

undisturbed, mountain ecosystems are delineated around national parks and/or wilderness complexes. These are the Greater Yellowstone Ecosystem, Glacier National Park–Bob Marshall Ecosystem, and the Central Idaho Ecosystem (Fig. 1).

The topography of the northern Rocky Mountain states spans steep physical gradients in elevation, slope, aspect, temperature and precipitation that give rise to diverse vegetation types. Elevations range from 150 m to 4200 m. Average precipitation ranges from 28 cm to 51 cm (Franklin 1983). The northern Rockies comprise a variety of non-forested and coniferous forest types. Low-lying valleys are characterized by grasslands, sagebrush (*Artemisia* spp.) and desert shrublands, interspersed with juniper (*Juniperus* spp.) and riparian woodlands. Ponderosa pine *Pinus ponderosa* dominates lower elevation montane forests, while xeric coniferous forests of mainly Douglas fir *Psuedotsuga mensiezia*, ponderosa pine, grand fir *Abies grandis*, lodgepole pine *Pinus contorta* and aspen *Populus tremuloides* occur at mid-elevations. Mesic forests in the north and west largely contain western larch *Larix occidentalis*, grand fir, western red cedar *Thuja plicata* and mountain hemlock *Tsuga mertensiana*. Higher elevations are composed of Engelmann spruce *Picea engelmannii*, subalpine fir *Abies lasiocarpa*, alpine larch *Larix lyalli* and white-bark pine *Pinus albicaulis* intermixed with subalpine meadows. Herb lands, rock, alder *Alnus sinuata* shrub-fields and snowfields/ice occur at the highest elevations.

DATA COLLECTION

We used a land management status GIS coverage and classification system developed by the USA Geological Survey's Biological Resources Division in its nation-wide GAP Analysis Programme (Scott, Tear & Davis 1996) to delineate 'protected areas'. This programme devised a ranking scheme to represent various levels of protection, ranging from the least protected lands (category 4, e.g. private lands) to those with the highest level of protection (category 1, e.g. wilderness areas) for all public lands in the GIS spatial database. For this study, we assumed that categories 1 and 2 represent adequate protection as their primary management objective is conservation (Scott, Tear & Davis 1996), and selected these categories as our protected areas on all forest service lands located in the three states.

We used the federal inventoried roadless areas GIS database (USDA Forest Service 2000a). This includes areas that are greater than 2000 ha in size, where road building is prohibited under current National Forest Plan decisions and where road building is presently allowed. We recognize that our decision leaves out smaller roadless areas that were not considered during the inventory of federal roadless areas and that these areas serve important conservation goals (Strittholt & DellaSala 2001). For this study, the term 'roadless areas' refers to inventoried roadless areas.

We used three independently derived land cover maps for Montana, Wyoming and Idaho from the GAP Analysis Programme (Scott, Tear & Davis 1996). The Montana and Idaho GAP products were produced based on classification techniques by Redmond *et al.* (1998) for raw Landsat Thematic Mapper (TM) satellite imagery. Spatial resolution of the grid was 90 m for Montana and 30 m for Idaho. The Wyoming GAP Analysis Programme digitized land cover data in a vector format from Landsat TM satellite imagery at a scale of 1 : 100 000 (Gap Analysis Wyoming 1996). We converted Wyoming's vector map into a grid format and resampled the three data sets to 90-m resolution. Then we merged the three land cover maps into a single image and a common land cover classification scheme (Appendix 1).

Similar to most GIS databases, errors are associated with the land management status, inventoried roadless areas and land-cover grids. These grids represent a composite of data from many sources and include variations in mapping procedures and possible misclassifications that could potentially cause inconsistencies that are difficult to detect. However, we believe, based on professional judgement, that the error rate is not large enough to affect conclusions drawn from this large regional-scale analysis.

To investigate the representation of roadless areas at various elevation classes, we downloaded a digital elevation model from the 30-m National Elevation Dataset produced by the USA Geological Survey's EROS Data Center (Sioux Falls, SD). We reclassified the elevation range into 21 equal-interval classes ranging in 200-m increments from approximately 150 m to 4200 m.

DATA ANALYSIS

All data analyses were conducted in ARC/INFO and ArcView GIS software from Environmental Systems Research Institute (Redlands, CA).

Land cover representation

Using ARC/INFO, we combined the protected areas database with the land cover map. To calculate the percentage representation of each land-cover type, we divided the protected portion of each land-cover type by the total area of each land-cover type across the study area. Next, we appended the national forest inventoried roadless areas to the existing protected areas and repeated the same calculation described above to measure the additional representation of each land-cover type because of the inclusion of roadless areas. In addition, we calculated the percentage increase between each land cover percentage representation for protected areas alone and protected areas and roadless areas combined. This measure quantified the 'relative' ecological contribution from roadless areas for each land-cover type. We then ranked these land-cover types according to the level of representation within the existing protected areas.

Elevation representation

Using ARC/INFO, we combined the protected areas database with the 30-m digital elevation model. Similar to the procedure for land-cover types described above, we added the roadless areas to the existing protected areas, intersected this image with the elevation data, and calculated the change in representation for each elevation class provided by protection of roadless areas.

To examine the potential increase of landscape connectivity caused by roadless areas, we used ARC/INFO and FRAGSTATS (McGarigal & Marks 1995; McGarigal & Holmes 2002), a computer program developed to quantify heterogeneity of the landscape. We identified five landscape metrics available in FRAGSTATS to assess our three elements of landscape connectivity (McGarigal & Holmes 2002). To assess area, we used the metrics percentage land (PLAND), number of patches (NP) and patch size (AREA). We included the metrics NP and AREA to help explain the context of an increase in PLAND. For example, an increase in PLAND and AREA and a decrease in NP would indicate that the added roadless patches were located next to existing conservation patches, resulting in an increase in the size of patches and a decrease in the number of patches across the landscape. Conversely, a decrease in AREA and an increase in NP would indicate that the added patches were generally smaller and did not combine with existing patches.

To assess isolation we used nearest neighbour distance (ENN). A decrease or increase in ENN would indicate that patches are either located closer together or farther apart, respectively, across the landscape.

To assess aggregation, we used contagion (CONTAG). An increase in CONTAG would indicate that patches are, to a certain extent, aggregated together across the landscape.

Using FRAGSTATS, we selected and ran our five landscape metrics on the two grids described above (current protected areas only, and roadless areas and current protected areas combined). Each grid was a binary map where all grid cells that comprised the 'protected' and 'roadless' patches were classified as 1 and all other 'non-protected' grid cells were masked out as background (–99). For each landscape metric, we computed the mean, area-weighted mean and coefficient of variation where applicable. We then compared the differences in metrics between the two grids. In addition, differences in the mean, area-weighted mean and coefficient of variation helped to explain how the range of values for each metric were distributed when existing protected areas were compared with the conservation system including roadless areas.

Results

LAND COVER REPRESENTATION

In existing protected areas, burned forest and snowfields/ice had the highest land cover representation, 88% and 86%, respectively. Representation of other land-cover types, such as alpine meadows, whitebark pine, exposed rock/soil, subalpine meadows, wetlands, mixed subalpine forest and lodgepole pine, ranged from 31% to 71%.

The inclusion of roadless areas increased the representation of all land-cover types except for one, sand dunes (Table 1). Relative percentage increases ranged

Table 1. Additional representation and percentage increase in representation of each land-cover type across the northern Rockies when national forest roadless areas are added to existing protected areas

Land-cover type	Existing level of representation (%)	Potential level of representation including roadless areas (%)	Percentage increase including roadless areas
Burned forest	88·12	93·09	5·65
Snowfields/ice	86·12	97·48	13·19
Alpine meadow	71·51	94·18	31·70
Mixed whitebark pine	59·62	84·94	42·46
Exposed rock/soil	44·67	59·92	34·12
Subalpine meadow	40·49	68·85	70·05
Wetlands	37·34	38·68	3·61
Mixed subalpine forest	32·20	68·63	113·11
Lodgepole pine	31·35	59·42	89·54
Mixed barren lands	21·66	22·61	4·37
Sand dunes	18·44	18·44	0·00
Mixed conifer	16·97	37·24	119·44
Mesic upland shrub	10·74	26·14	143·44
Shrub-dominated riparian	7·98	12·77	59·91
Forest-dominated riparian	7·18	12·14	69·11
Sagebrush	6·33	9·91	56·55
Juniper	5·87	6·80	15·95
Xeric upland shrub	5·85	7·97	36·33
Vegetated sand dunes	5·69	6·03	5·89
Western red cedar	5·57	22·00	295·08
Mud flats	5·33	7·39	38·79
Ponderosa pine	4·94	9·88	99·97
Aspen	4·48	25·99	479·80
Shrub–grassland associations	4·25	5·89	38·46
Western hemlock	3·36	23·62	602·54
Grasslands	2·49	3·64	46·31
Grass-dominated riparian	2·15	3·07	43·01
Salt-desert shrub flats	1·58	1·71	8·63
Bur oak woodland	0·00	2·40	NA

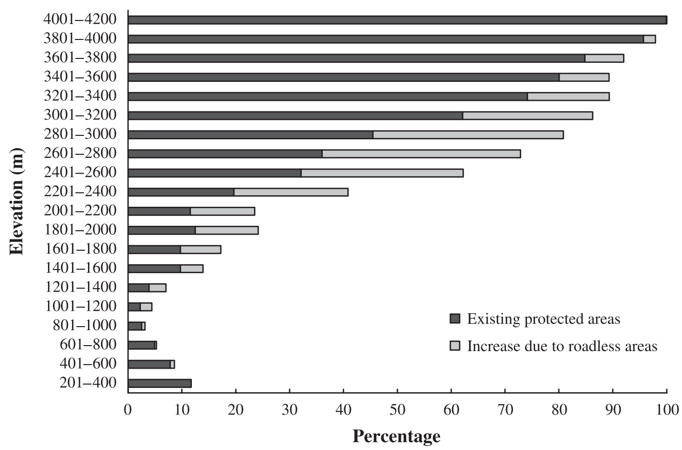

Fig. 2. Additional representation of elevation ranges resulting from the inclusion of roadless areas with protected areas for the northern Rockies. The *x*-axis represents elevation in 200-m increments and the *y*-axis shows absolute increase in percentage representation when roadless areas are added to protected areas. Black bars represent protected areas and grey bars represent roadless areas.

from 5% to 600%. Fifteen land-cover types increased by more than 40%, among them important ecological communities, western hemlock, aspen, ponderosa pine, western red cedar and sagebrush, each of which has less than 10% representation in current protected areas. Moreover, the addition of roadless areas represented one land-cover type, bur oak *Quercus macrocarpa* woodland, not present in protected areas.

ELEVATION REPRESENTATION

Our elevation analyses showed that elevations in the range of 2200–4200 m were well represented in protected areas (Fig. 2). The addition of roadless areas resulted in a large increase in representation of lands at elevations ranging from 1000 m to approximately 3400 m. For elevation ranges below 1000 m and above 3400 m, the

Table 2. Landscape metrics comparing the spatial pattern of protected areas alone with a scenario that includes protected areas and national forest roadless areas combined for the northern Rockies. + and – indicate an increase or decrease, respectively, in the metric value caused by the addition of roadless areas

Landscape Metrics	Protected areas	Protected and roadless areas	+/–
Area			
Class area (ha)	8 814 900	15 673 600	+
Percentage land	9	16	+
Number of patches	770	722	–
Patch size (mean, ha)	11 447·92	21 708·59	+
Patch size (area-weighted mean)	1 105 055·78	2 505 909·11	+
Patch size (coefficient of variation)	977·39	1 069·74	+
Isolation			
Nearest neighbour (m)	7 013·72	5 353·11	–
Nearest neighbour (area-weighted mean)	3 153·73	2 518·75	–
Nearest neighbour (coefficient of variation)	122·47	134·16	+
Aggregation			
Contagion index	72·56	58·64	–

contribution of roadless areas was small. However, the proportion of area represented at lower elevations increased when we included roadless areas with protected areas.

CONNECTIVITY

Results from the landscape metrics showed that the addition of roadless areas increased regional connectivity for all three connectivity elements (Table 2). Area metrics demonstrated that the addition of roadless areas almost doubled the amount of area protected, rising from 9% to 16%, and the mean patch size in protected areas changed from 11448 ha to 21709 ha. The number of patches decreased from 770 to 722. Area-weighted mean patch size increases and the patch size coefficient of variation increased from 977 to 1070. Isolation metrics showed a decrease in the mean and area-weighted mean nearest-neighbour metrics when roadless areas were added. The mean distance between nearest protected patches decreased from 7014 m to 5353 m. The decrease in the area-weighted mean was less than the overall mean when patches of all sizes were considered. The coefficient of variation also increased for this metric. The aggregation metric (contagion) decreased from 72·56 to 58·64 when roadless areas were included, signifying more dispersion of patches across the landscape.

Discussion

BIODIVERSITY REPRESENTATION

A review of the literature suggests that a given vegetation community is adequately represented when 12–25% of it is included in a conservation area (World Commission on Environment & Development 1987; Noss & Cooperrider 1994), although it is not certain that these thresholds are truly adequate to protect vegetation communities. Based on this range, we define land-cover types above 25% as adequately protected, land-cover types within the range of 12–25% as minimally protected, and those below 12% as underrepresented, similar to DeVelice & Martin (2001).

Our results show that roadless areas make a substantial contribution in maintaining regional biodiversity. One of our most important findings is that roadless areas would protect a wider range of land-cover types and elevation ranges than protected areas alone, especially those characteristic of mid- to low elevations that are underrepresented in protected areas. These lands are among the last remnants of biologically productive lands that have not been significantly altered through human settlements, resource extraction and road construction (Scott *et al*. 2001; Strittholt & DellaSala 2001). We also found that protected areas adequately represent land-cover types that are characteristic of higher elevations. This finding supports the generally accepted notion that wilderness areas and national parks mainly protect higher elevation ecological communities (Davis *et al*. 1996; Possingham, Ball & Andelman 2000). Contrary to DeVelice & Martin (2001), whose study found that roadless areas mainly occurred at mid- to lower elevations, but similar to Strittholt & DellaSala (2001), we found that roadless areas considerably increase the protection of higher elevations and corresponding cover types as well. The different results are probably because of the scale at which the studies were implemented. DeVelice & Martin's (2001) study included all roadless areas across the nation, incorporating a wide range of elevations from sea level to the highest peaks. Our study, and that of Strittholt & DellaSala (2001), focused on smaller regions at higher elevations.

Across the northern Rockies region (Montana, Wyoming and Idaho), protected areas adequately represent nine land-cover types, whereas five biologically important land-cover types, western hemlock, aspen, ponderosa pine, western red cedar and mesic upland shrub, are underrepresented in protected areas. However, the addition of roadless areas increases representation of two cover types (western hemlock and western red cedar) to the minimally protected threshold and two cover types (aspen and mesic upland shrub) to the adequately represented threshold (greater than 25%). Ponderosa pine, even though it increases by nearly 100%, remains underrepresented. Overall, the magnitude of the increased representation, from 100% to 600%, indicates that roadless areas can make substantial contributions to the protection of land-cover types that are not well represented in protected areas.

Increased representation of certain rare ecological communities is particularly important in a northern Rockies conservation strategy. Aspen, for example, is thought to be declining in the northern Rockies (Gallent *et al*. 1998). When roadless areas are added to protected areas, aspen moves up two full categories: from underrepresented to adequately represented, a 480% increase in representation for this forest type, on which many avian species depend upon (Hansen & Rotella 2000). Representation of whitebark pine changes from 60% to 85% when roadless areas are added. Whitebark pine is declining throughout North America due to blister rust *Cronartium ribicola*, an introduced disease, and is a 'keystone species' important for many higher elevation species (Keane, Morgan & Menakis 1994).

Elevation representation results demonstrate that protected areas are mainly located at higher elevations. We also found that roadless areas are generally concentrated at mid- to high elevations and represent a wider range of elevations, especially low- to mid elevations, than protected areas. However, our results show that protected areas encompass more lower elevation lands than roadless areas. This situation is somewhat deceiving. Representation of lower elevations in protected areas is largely a result of two well-placed low-elevation conservation areas: Hell's Canyon National Recreation Area and Missouri Breaks National Monument. In fact, low-elevation lands below 1000 m are not well represented in either protected areas or roadless areas. As a majority of lower elevation lands in the northern Rockies have been converted to other uses, it is of utmost importance to increase representation of lower elevation sites in protected areas (Strittholt & DellaSala 2001). Protection of these lower elevation roadless areas would contribute greatly to the conservation of lower elevation species and ecological communities that are poorly represented in protected areas.

LANDSCAPE CONNECTIVITY

Our analyses of three elements of connectivity show that roadless areas increase connectivity across the northern Rockies, and increase both the area and size of protected area patches. In addition, the number of protected area patches decreases with the addition of roadless areas because they combine with protected areas to form one larger patch. Larger patches will protect more species and more individuals, species with large home ranges, species sensitive to human activity, and more intact ecosystem processes than smaller areas (Askins, Philbrick & Sugeno 1987; Robbins, Dawson & Dowell 1989; Turner *et al*. 1993; Newmark 1995; Shafer 1995). Roadless areas also reduce the distance between protected areas and create a more evenly dispersed reserve system, critical for maintaining many species' movements and a large distribution of local populations (MacArthur & Wilson 1967; Murphy & Noon 1992; Reed, Johnson-Barnard & Baker 1996; Ritters *et al*. 1996; Beauvais 2000; Hansen & Rotella 2000; He, DeZonia, & Mladenoff 2000; Shinneman & Baker 2000).

Our results show an increase in the coefficient of variation for patch size and isolation metrics, which may be an important consideration in delineating conservation reserve systems capable of maintaining movements of various species and ecological processes (Wiens & Milne 1989; Wilcove & Murphy 1991; Noss 1992; Noss *et al*. 1996; O'Neill *et al*. 1996). Smaller patches may supplement larger reserves by protecting rare species that occur only in certain areas (Franklin & Forman 1987; Hansen *et al*. 1991; Shafer 1995). The dispersion of roadless areas may also contribute to greater resilience or survival of island populations by allowing a greater chance for species exchange, essentially maintaining a metapopulation or source–sink population structure (Wiens, Crawford & Gosz 1985; Pullium 1988; Gilpin & Hanski 1991; Murphy & Noon 1992). Many studies are investigating how species move through landscapes and their use of stepping-stone habitats, especially in fragmented landscapes (Freemark *et al*. 1993; With 1999; Beauvais 2000; Hansen & Rotella 2000; Holloway, Griffiths & Richardson 2003; Johnson, Seip & Boyce 2004). Being relatively undisturbed and well-distributed among protected areas, roadless areas are top candidates for the delineation of high-quality 'habitat connections' across the northern Rockies, particularly those that target rare or declining species. The loss or alteration of roadless areas may further reduce the movement of species among interdependent island populations located in protected areas and roadless areas, resulting in greater isolation.

Moreover, the addition of roadless areas increases the effective size of the three largest wilderness and national park complexes in the northern Rockies: the Greater Yellowstone Ecosystem, the Glacier National Park–Bob Marshall Ecosystem and the Central Idaho Ecosystem, where management challenges include maintaining large-scale ecological processes such as species' movements and natural fire across jurisdictional boundaries (Pickett & White 1985; Turner *et al*. 1993). Roadless areas not immediately adjacent to these complexes are dispersed in the surrounding landscape, which helps to decrease the degree of isolation between the complexes and possibly allows for species movement among these ecosystems.

MANAGEMENT IMPLICATIONS

Using research to guide reserve design and develop land protection policies is the strongest approach in conservation. The importance of intact, functioning natural ecosystems to the maintenance of native biodiversity and ecological processes is unquestioned (Wright, Dixon & Thompson 1933; MacArthur & Wilson 1967; Usher 1987; White 1987; Shafer 1995; Noss, O'Connell & Murphy 1997). The negative impacts of roads in natural areas are well known (Andrews 1990; Foreman & Wolke 1992; Reed, Johnson-Barnard & Baker 1996; Spellerberg 1998; Trombulak & Frissell 2000; McGarigal *et al*. 2001). Our landscape assessment demonstrates how roadless areas, the remaining relatively undisturbed forested lands in the northern Rockies, are essential for maintaining biodiversity and landscape connectivity in a conservation reserve strategy for this area. This has direct bearing on management decisions regarding the protection of roadless areas in this region. Our results, along with the findings of DeVelice & Martin (2001) and Strittholt & DellaSala (2001), highlight the important role of roadless areas in USA conservation efforts and contribute to the larger policy dialogue surrounding roadless areas.

The methods used in this study can help land managers determine appropriate guidelines to identify and assess roadless areas that are critical in maintaining regional biodiversity, ecosystem processes, landscape connectivity and overall intact ecosystem integrity. Land managers should avoid activities such as road building, logging, spread of exotic species, off-road vehicle use and exurban development in roadless areas that would result in their degradation or loss. If roadless areas are not protected from these activities as a matter of priority, it is possible that their potential contribution to conservation effort in the future will be diminished and existing protected areas surrounded by or in close proximity to roadless areas will be negatively affected as well. We recommend that roadless areas receive full protection and are managed responsibly, so that they can function as an important part of the current conservation reserve system in the USA.

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Supplementary material

The following material is available from http://www.blackwellpublishing.com/products/journals/suppmat/JPE/JPE996/JPE996sm.htm.

Appendix 1. Land-cover types across the northern Rocky Mountain region reclassified from USA Geological Survey's Biological GAP Analysis Programme (Scott, Tear & Davis 1996).

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CENTER FOR LARGE LANDSCAPE CONSERVATION

Federal Agency Planning for Wildlife Corridors and Habitat Connectivity

Language Used in Completed or Draft Documents

A COMPENDIUM

Compiled by:

Robert Ament and Katie Meiklejohn

June 4, 2009

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DEPARTMENT OF AGRICULTURE-FOREST SERVICE

FOREST PLANS

Beaverhead-Deerlodge National Forest
Draft Revised Land and Resource Management Plan
Draft EIS

Linkages: Maintain options for Forest Service's contributions to linkages between landscapes, unless such landscape isolation is determined to be beneficial. Linkage areas are those areas that have been identified for a federally listed species through a conservation strategy. Options may include, but are not limited to:

- Maintaining forest ownership at highway and road crossings
- Acquiring lands to consolidate ownership at highway and road crossings
- Providing adequate cover within linkage areas
- Minimizing open motorized roads and trails within linkage areas

Wildlife Secure Areas and Connectivity: Provide secure areas for ungulates, large carnivores and connectivity while recognizing the variety of recreational opportunities. Manage open motorized roads/trails density by landscape to [minimize impacts from motorized vehicles]

Urban expansion, both locally and regionally, also increases public concerns that National Forests also function as biological reserves and provide wildlife habitat connectivity at broad scales.

Connectivity or Linkage Areas
Connections to other public and private lands at this point have mostly been challenged by development of adjacent land. The forest is characterized by mountainous island landscapes separated by broad valleys in mixed private, State and BLM ownerships. State management and the Dillon Resource Area draft management plan are generally compatible with maintaining habitat linkage to the island landscapes and neighboring public lands. Development of private lands will present the greatest challenges to maintaining habitat linkages to public lands.

…habitat connectivity has not been fundamentally compromised by management actions.

Two interstate highways (I-15 and I-90) traverse the area with approximately only 13 miles of right of way on national forest land. State Highways 1, 12, 43, and 278 encompass an approximate total of 30 miles of right of way. Other than these paved highways and small utility corridors, the Forest remains largely intact compared to its original composition. All of the Alternatives maintain options to address wildlife crossing concerns as they develop.

…linkage can also develop challenges related to disease introductions and the spread of noxious weeds. The latter negative connotation for 'linkage' is addressed amongst the alternatives through restriction of motorized access.

Gallatin National Forest Travel Plan FEIS
Issue 3: Biological Diversity and Ecological Sustainability

Transportation systems of any kind across the landscape with linear trails and/or roads may affect vegetation, wildlife movement and habitat use; facilitate species invasion (native and nonnative plants and animals) and disrupt corridors.

The Travel Management Plan or any other Forest Service document or action must maintain viable populations of wildlife species. …The most likely threat to viability that the Forest Service transportation system could cause is damage to wildlife movement corridors in areas not currently covered by recovery plans and specific direction for threatened and endangered and other species.

Affected Environment –

Corridors:
Corridors are defined as "…avenues along which wide-ranging animals can travel, plants can propagate, genetic interchange can occur, populations can move in response to environmental changes and natural disasters, and threatened species can be replenished from other areas". The term corridor is often used synonymously with connectivity and linkage or linkage zone. Corridors help determine how and if an animal can move through the landscape. Confusion arises with whether or not the species in question just uses a corridor for travel or if it must be able to meet all of its needs for survival and reproduction there. The intention in this document is to define a corridor as a passageway, and not as meeting the full habitat requirements for the species of interest. A corridor need not provide all the life requirements for a species within the corridor (passage species), but some species will live entirely within a corridor (corridor dwellers).

Wildlife corridors may have several functions:

- Wide-ranging animals can move through these corridors
- Plants can propagate
- Genetic interchange can occur
- Populations can move in response to changes in the environment
- Areas can be recolonized where populations have been extirpated

Roads affect the connectivity of the landscape. "Landscape connectivity is the degree to which the landscape facilitates animal movement and other ecological flows." Good connectivity exists if there are no barriers in the landscape and the habitat types that exist are usable by the species of interest. Many species must move through the landscape to meet their habitat needs throughout their life, and some species must move large distances (e.g. large carnivores, migratory species). Barriers to movement can result in mortality, reduced reproduction, and a smaller, less viable population. Connectivity also allows areas to be repopulated if there have been local declines of some species. Roads can be barriers to animal movements. Forest interior species may be the most affected by roads. This is because roads provide openings in a forested area and the openings change both the abiotic and biotic factors in the habitat (light, snow depth, precipitation, facilitation of movement for some predators, etc.).

Roads may pose a threat to carnivore populations due to road mortality and the indirect effects of barriers. Populations of both small and large mammals may become effectively isolated by barriers. Barriers to wildlife movement are most often caused by wide roads that have high speeds and may have center barriers and/or medians. Roads that have adjacent power lines, frontage roads, and/or railroad tracks can be formidable barriers for many wildlife species. Secondary and unpaved roads seem to have little effect on most animal movement and can be fairly permeable to wildlife. However, for small animals, the width of the road can be an important variable. The relative permeability (ease of crossing) of a road and its adjacent edge habitat influences how animals may cross it. The hardness or abruptness of an edge seems to be important to some animals, especially forest dwelling species. Some animals may actually move parallel along the road.

Where habitat truly occurs between islands, connectivity between islands becomes important. Physically continuous corridors may be preferred by many species. Riparian corridors may be especially important due to the presence of water, nutrients and energy from the riparian system. Riparian systems are often dominated by hardwoods and host higher bird populations. Riparian strips are excellent means of connecting islands of habitat across elevations.

…Mid and large-size carnivores typically have large home ranges and they range widely in the environment. They may be more vulnerable than most species to habitat fragmentation on a landscape scale. Even for common species like elk, it is critical to maintain security areas and migration corridors.

…Key linkage areas are areas where habitat connectivity has been decreased…

…Highways and private lands are the elements that lead to the most risk to key linkage areas. Those areas with high priority for maintaining wildlife connectivity are:

- Four-lane highways
- Two-lane highways that may be upgraded
- Two-land highways with high traffic volume
- Roads with a high potential for improvement
- Highways that parallel railroads

…The large amount of private land surrounds the islands of mountainous National Forests. Once the private lands are developed it will be much more difficult for wildlife to move between protected islands of public land.

For linkages, Interstate highways that are typically four-lane and often have some type of center barrier and large clearings on either side as well as occasionally in the median, are the roads of most concern. On and around the Gallatin NF, the road of most concern is I-90…Most of the actual linkages identified are either not located on the Forest or are not roads the FS has jurisdiction over. Therefore, for the most part, the corridor issue is one of cumulative effects, but the parts of the NF that facilitate animals to get to the corridors of concern are part of direct or indirect effects analysis and several of these areas will be analyzed.

Gallatin Travel Plan – Alternatives Document

Goal E:
Wildlife Corridors. Provide for wildlife movement and genetic interaction (particularly grizzly bear and lynx) between mountain ranges at Bozeman Pass (linking the Gallatin Range to the Bridger/Bangtails); across highway 191 from Big Sky to its junction with highway 287 (linking the Gallatin and Madison Mountain Ranges); the Lionhead area (linking the Henry's Lake Mountains to the Gravelly Mountains and areas west); Yankee Jim Canyon (linking the Absoroka Mountains to the Gallatin Range); and at Cooke Pass (linking the Absoroka/Beartooth Range to areas south).

Kootenai-Idaho Panhandle National Forests Plan Revision
Corridors/Linkage Areas/Approach Areas Desired Condition – Forest wide

Compared to historical conditions portions of the forests have become more isolated as cover needed for travel between patches is disturbed by highways, cities, rural housing, reservoirs, or other barriers to migration. Species now often have to travel greater distances to find food and den sites. These changes are affecting large, mobile species such as grizzly bear, wolf, wolverine and fisher which have lost much of their historical range.

Corridors/linkage areas (including approach areas) are established (with completion of a forest wide management plan) that provide for wildlife movement (migration/dispersal corridors) and genetic interaction. Established corridors/linkage areas and approach areas provide secure habitat conditions for wildlife movement, especially across valley bottoms (termed approach areas). These corridors provided connectivity for wildlife such as lynx, grizzly bear, and wolverine. Suitable habitat and conditions within established corridors/linkage areas allow wildlife species movement between large blocks of habitat, and seasonal and special habitats on a localized and landscape scale. Corridors/linkage areas are most often in areas with established wildlife use, and in areas relatively free of development such as roads and developed campgrounds. These areas provide cover and often connect key habitat components for those species that use the area. Forest Service lands contribute to linkages between landscapes, unless such landscape isolation is determined to be beneficial. Mortality in these associated approach areas is reduced as safer crossings are provided in areas with high levels of human development through coordination and or cooperation with State Highway Departments, private landowners, and other entities.

Approach areas are defined and 24 have been identified on the Kootenai National Forest. See: Brundin, L. and W. Johnson. 2008. Kootenai National Forest Wildlife Approach Areas.

Desired Condition – Canada lynx
A forest wide linkage area management plan is complete, providing areas for connectivity of habitat and movement of animals within and between LAUs. The lynx and wolverine steering committee established coarse scale maps used to complete this plan. Established corridors//linkage areas provide suitable habitat conditions for cover and security.
Desired Condition – grizzly bear
Corridor/linkage areas are established providing for movement of bears within and between Bear Management Units and between recovery zones. The establishment of wildlife corridors/linkage

areas is directed by the Interagency Grizzly Bear Committee. Established corridors/linkage zones provide suitable habitat conditions for cover and security, based on the species needs that use the area, as determined during management planning.

Geographic Area Desired Condition
The Forest provides for movement and genetic exchange of wide ranging carnivores, through the Scotchman Peaks and the McArthur Lake wildlife management area.

Management activities within established corridors/linkage areas should:

- Minimize new permanent roads
- Maintain hiding cover based on the needs of those species that use the area
- Minimize new site developments such as campgrounds

Draft Proposed Land Management Plan
Shoshone National Forest
August 2008

http://www.fs.fed.us/r2/shoshone/projects/planning/revision/revision_documents/february_2009/2008_0820_plan.pdf

NOTE: Chapter 1 of the Draft Proposed Land Management Plan lists the various desired conditions for the Forest. On pages 40-41 is the section on habitat connectivity

HABITAT CONNECTIVITY

Background
Many species in this ecosystem move long distances between summer and winter ranges, specifically, bighorn sheep, elk, moose, mule deer, greater sage grouse, grizzly bear, Canada lynx, wolverine, and gray wolves. Many other species make shorter seasonal movements. Connectivity between important areas is critical for species making these movements. Due to the abundance of wilderness and inventoried roadless areas on the Forest, most connectivity corridors for wildlife have not been impacted by management activities. Plan components focus on providing vegetation in appropriate patterns and connectivity to facilitate wildlife movement across the landscape. Other components provide direction for managing infrastructure, forest roads47 in particular, in ways that do not impede wildlife movement.

Habitat connectivity desired conditions

Vegetation patterns vary spatially and temporally across landscapes. Patterns of vegetation provide an inherent degree of connectivity, facilitating animal movement between habitats.

Forest roads do not impede big game and riparian and aquatic species movement during seasonal use. Infrastructure is designed and located to facilitate movement of big game, riparian, and aquatic species. Some secure habitat occurs in big game migration corridors to facilitate big game movement.

NOTE: Chapter 2 lists the objectives for each of the desired conditions, including habitat connectivity, which describes how the Forest Service intends to move toward the desired conditions described in chapter 1. The text for habitat connectivity is on page 88.

Management approach

Program emphasis for improving elk migration corridors should focus on watersheds with low elk security habitat (less than 30 percent). Highway projects bisecting big game crossing routes are coordinated with the Wyoming Department of Transportation to reduce or mitigate animal/vehicle collisions and facilitate connectivity between seasonal habitats. Vegetation activities are generally designed to maintain habitat mosaics within the natural range of variability. Program planning utilizes Wyoming Game and Fish Department mapping of elk and bighorn sheep migration corridors. Maintaining connectivity corridors in riparian habitat focuses on fish, frogs, and toads, as well as other riparian species. Highway projects in riparian areas are coordinated with the Wyoming Department of Transportation to mitigate connectivity issues.

Though the desired condition for habitat connectivity in streams calls for limited barriers, barriers may be created or maintained to block the spread of invasive or non-native species. Additionally, natural barriers may be removed to provide additional habitat for native species.

1986 Forest Plan direction that is retained

Connectivity objectives for lynx habitat are outlined in appendix D Northern Rockies lynx management objectives ALL 01, HU 06.

Note: Chapter 5 is the Plan’s standards and guidelines, which includes standards, guidelines, and references to other applicable guidance. There are no standards or guidelines developed by the Forest to protect terrestrial connectivity and only one guideline for providing for aquatic connectivity.

Habitat connectivity

Guideline 19: New, replacement, and reconstructed stream crossing sites (culverts, bridges, and other stream crossings) should be designed to provide and maintain passage for fish, other aquatic species, and/or riparian associated terrestrial species. Constructed barriers may be maintained in instances where native species benefit from species isolation.

[111] Guideline supplements Forest Service Handbook 2509.25 Region 2 Watershed Conservation Practices Handbook Management Measure (3)

Part three—Design criteria 5.4 Species diversity Chapter 5 Standards and guidelines Shoshone National Forest Proposed Land Management Plan

Page 125

1986 Forest Plan direction that is retained

Connectivity standards and guidelines for lynx habitat are outlined in appendix D Northern Rockies lynx management standards ALL S1, LINK S1 and guideline ALL G1.

Other guidance

Forest Service Handbook 2509.25 Region 2 Watershed Conservation Practices Handbook Management Measure (4).

Environmental Assessment

Bridger-Teton National Forest

Land and Resource Management Plan Amendment: Pronghorn Migration Corridor

SUMMARY

The Bridger-Teton National Forest proposes to amend its 1990 Land and Resource Management Plan (Forest Plan) to allow continued successful migration of the pronghorn (Antilocarpa americana) that summer in Jackson Hole and winter in the Green River basin in Wyoming. The Forest Plan Amendment would designate a Pronghorn Migration Corridor and create a standard requiring that projects, activities and infrastructure authorized by the Forest Service in the corridor be designed, timed and/or located to allow continued successful migration. The migration corridor to which this amendment would apply extends from the Forest boundary near the Green River Lakes Road north of Pinedale in Sublette County, Wyoming to the Forest boundary with Grand Teton National Park northeast of Kelly in Teton County, Wyoming. It is within the Pinedale and Jackson Ranger Districts of the Bridger-Teton National Forest.

Because the proposal would not result in significant changes to multiple-use goals and objectives for long-term land and resource management, the proposed amendment is considered to be "non-significant" according to the planning regulations at 36 CFR 217. Therefore, the amendment can be authorized in a Decision Notice after completion of this Environmental Assessment (EA). In this EA, the Forest Service evaluates the Proposed Action and the "No Action" alternative of not amending the Forest Plan.

Based on this EA, the responsible official will decide whether or not to amend the Forest Plan as described. The Responsible Official is the Forest Supervisor of the Bridger-Teton National Forest Kniffy Hamilton.

Decision Notice & Finding of No Significant Impact

Pronghorn Migration Corridor Forest Plan Amendment

USDA Forest Service
Bridger-Teton National Forest
Wyoming

Decision and Reasons for the Decision

Background

The pronghorn (Antilocarpa americana) that summer in Jackson Hole migrate annually between there and wintering areas in the Green River basin. Documented round trip migration distances from 175 to 330 miles make this the longest known terrestrial animal migration in the 48 contiguous states. Typically, the pronghorn migrate through the corridor in April or May and again in October or November. These pronghorn are a part of the impressive panorama of free-ranging native Rocky Mountain mammals in northwest Wyoming. This landscape and its wildlife draw tourists from around the world and support a robust regional economy.

A significant portion of the full migration route of these pronghorn is within the Bridger-Teton National Forest. The Forest portion extends from the Forest boundary near the Green River Lakes Road north of Pinedale in Sublette County, Wyoming to the Forest boundary with Grand Teton National Park northeast of Kelly in Teton County, Wyoming. It includes approximately 47,000 acres within the Pinedale and Jackson Ranger Districts of the Bridger-Teton National Forest.

Managing this migration corridor to facilitate continued successful movement of pronghorn will help ensure protection of this herd and its migration. The purpose of this amendment to the Bridger-Teton National Forest Land and Resource Management Plan (Forest Plan) is to ensure that projects, activities, and facilities authorized by the Forest Service on National Forest System lands within the corridor allow for continued successful pronghorn migration.

It should be noted that the Forest Service by itself cannot guarantee continued successful migration of this herd over the entire migration route. There are numerous factors beyond Forest Service control such as activities on lands under other jurisdictions within the migration route.

Decision

Based upon my review of the Environmental Assessment (EA), I hereby amend the Bridger-Teton National Forest Land and Resource Management Plan by 1) designating a Pronghorn Migration Corridor as shown on the attached map; and 2) adding the following standard, "All projects, activities, and infrastructure authorized in the designated Pronghorn Migration Corridor will be designed, timed and/or located to allow continued successful migration of the pronghorn that summer in Jackson Hole and winter in the Green River basin." This amendment does not remove any current Forest Plan direction for the area encompassed by the corridor; it simply designates the corridor and adds the above standard. This amendment makes no decisions about the compatibility of specific uses with the pronghorn migration, but requires that all uses be found to allow continued migration before they are authorized.

Activities currently authorized by the Forest Service within this migration corridor, including livestock grazing operations, coexist with the currently successful pronghorn migrations, so changes to current activities and infrastructure are not required by this amendment.
Before future activities can be authorized, a determination must be made that the activity will allow continued successful migration.

It is important to note that, while the full length of the pronghorn migration route includes lands under various jurisdictions, this Forest Plan amendment applies only to National Forest System lands within that larger corridor. Furthermore, the amendment does not constrain activities on private land within the Forest boundary.

Reasons for the Decision

I have decided to create the Forest Plan amendment because it meets the purpose and need of ensuring that Forest Service authorized activities and infrastructure allow continued successful pronghorn migration in the corridor. Furthermore, I find that there are no unacceptable impacts from the amendment. As noted above, activities currently authorized by the Forest Service within the corridor coexist with successful migration, so changes to current activities will not be required by this amendment.

Targhee Forest Plan – 1997

Goals – Grizzly Bear Habitat
(2) Allow for unhindered movement of bears (continuity with Yellowstone National Park and adjacent bear management units)

White River National Forest Plan Revision 2002
Record of Decision

COMPONENT 3: ESTABLISHMENT OF MANAGEMENT AREA DIRECTION

Management Area 5.5 – Forested Landscape Linkages: I am placing an emphasis on the importance of landscape linkages. Alternative K places the highest acreage in corridor designation of any alternative. The creation of habitat gaps heightens the risk that suitable habitats will become isolated from each other. Barriers to the movement of species from one suitable habitat patch to another reduce the connectivity of these habitats. When suitable vegetation types and cover conditions are present between patches, species can move between them. Corridors will provide areas for landscape-scale movement, migration, and dispersal of forest carnivores and other wide-ranging wildlife species; safe travel connections between large blocks of forested landscapes across the Forest; and security from intensive recreational and other human disturbances. This is an important step in providing for the maintenance of biodiversity across the forest. This prescription includes many of the aspects of two different management areas included in the Proposed Revised Forest Plan, Corridors Connecting Core Areas (3.55) and Forest Carnivores (5.45).

REGIONAL PLANS

Interior Columbia Basin Ecosystem Management Plan
ICBEMP: Interim Management Direction
Establishing Riparian, Ecosystem and Wildlife Standards for Timber Sales
Regional Forester's Eastside Forest Plan Amendment #2

Alternative 2, as adopted
The interim wildlife standard only altered portions of current Forest Plans. All additional Forest Plan wildlife standards and guidelines not altered in this direction still apply.

d. Scenario A

If either one or both of the late and old structural (LOS) stages falls BELOW HRV in a particular biophysical environment within a watershed, then there should be NO NET LOSS OF LOS from that biophysical environment. DO NOT allow timber sale harvest activities to occur within LOS stages that are BELOW HRV.

(3) Maintain connectivity and reduce fragmentation of LOS stands by adhering to the following standards:

INTENT STATEMENT: While data is still being collected, it is the best understanding of wildlife science, today, that wildlife species associated with late and old structural conditions, especially those sensitive to 'edge', rely on the connectivity of these habitats to allow free movement and interaction of adults and dispersal of young. Connectivity corridors do not necessarily meet the same description of 'suitable' habitat for breeding, but allow free movement between suitable breeding habitats. Until a full conservation assessment is completed that describes in more detail the movement patterns and needs of various species and communities of species in eastside ecosystems, it is important to insure that blocks of habitat maintain a high degree of connectivity between them, and that blocks of habitat do not become fragmented in the short term.

a) Maintain or enhance the current level of connectivity between LOS stands and between all Forest Plan designated 'old growth/MR' habitats by maintaining stands between them that serve the purpose of connection as described below:
   1) Network pattern – LOS stands and MR/Old Growth habitats need to be connected with each other inside the watershed as well as to like stands in adjacent watersheds in a contiguous network pattern by at least 2 different directions.
   2) Connectivity Corridor Stand Description – stands in which medium diameter or larger trees are common and canopy closures are within the top one third of site potential. Stand widths should be at least 400 ft. wide at their narrowest point. The only exception to stand width is when it is impossible to meet 400 ft with current vegetative structure AND these 'narrower stands' are the only connections available; (use them as last resorts). In the case of lodgepole pine, consider medium to large trees as appropriate diameters to this stand type.

      If stands meeting this description are not available in order to provide at least 2 different connections for a particular LOS stand or MR/Old Growth habitat, leave the next best stands for connections. Again, each LOS and MR/Old Growth habitat must be connected at least 2 different ways.

3) Length of Connection Corridors – The length of corridors between LOS stands and MR habitats depends on the distance between such stands. Length of corridors should be as short as possible.
4) Harvesting within connectivity corridors is permitted if all the criteria in (2) above can be met, and if some amount of understory (if any occurs) is left in patches or scattered to assist in supporting stand density and cover. Some understory removal, stocking control, or salvage may be possible activities, depending on the site.

b) To reduce fragmentation of LOS stands, or at least not increase it from current levels, stands that do not currently meet LOS that are located within, or surrounded by, blocks of LOS stands should not be considered for even-aged regeneration, or group selection at this time. Non-regeneration or single tree selection (UEAM) activities in these areas should only proceed if the prescription moves the stand towards LOS conditions as soon as possible.

e. Scenario B

Within a particular biophysical environment within a watershed, if the single, existing late and old structural (LOS) stage is WITHIN OR ABOVE HRV, OR if both types of LOS stages occur and BOTH are WITHIN OR ABOVE HRV, then timber harvest can occur within these stages as long as LOS conditions do not fall below HRV. Enhance LOS structural conditions and attributes as possible, consistent with other multiple use objectives.

The intent of the following direction is to maintain options by impacting large and/or continuous stands of LOS as little as possible, while meeting other multiple use objectives.
(2) Maintain connectivity as directed in Scenario A, (3)

Southern Rockies Lynx Management Direction
Record of Decision

INTRODUCTION

Risks to Lynx and Lynx Habitat

The LCAS identified risk factors affecting movement (pp. 2-17 to 2-19) as highways and associated development and private land development.

> Within lynx home ranges, highways and associated high-intensity uses and developments may constrain habitat use and impede daily movements. At a broader scale, lynx are known to disperse and make exploratory movements across long distances and varied habitat and terrain. Maintaining connectivity within and between lynx subpopulations is an important consideration to maintain long-term persistence. However, the Forest Service has limited authority over highways and no authority to manage activities on private land. This decision provides guidelines applicable to maintaining connectivity within the limits of the Forest Service's jurisdiction.

RECREATION MANAGEMENT

Developed Recreation

There are 25 existing alpine ski areas in the Southern Rockies Lynx Amendment area, encompassing 82,704 permitted acres. Most ski areas were constructed well before the lynx was listed…

Under Alternative F-modified, the management direction would only apply to the development of new ski areas and to expansions of existing ski areas and would not affect existing ski area facilities or operations, with minor exceptions. Since the U.S. Fish and Wildlife Service concluded in their 2003 Remand Notice that there is no evidence showing that recreational activities exert a population-level impact on lynx, Alternative F-modified applies guidelines, rather than standards. To assure that lynx habitat connectivity is maintained, Alternative F-modified includes standards ALL S1 and LINK S1.

The management direction in Alternative F-modified will minimize the potential impacts of ski areas and other developed recreation sites on lynx habitat. Existing facilities and operations would not be affected. New developments and expansions would need to be designed in accordance with the management direction, which in most cases would have only minor effects.

LINKAGE AREAS

Highways

Highways impact lynx by fragmenting habitat and impeding their movement. With human population growth, highways tend to increase in size and traffic density. As traffic lanes, volumes, speeds and rights-of-way increase, the effects on lynx are increased.

The LCAS recommended one objective, two standards, and a guideline directly or indirectly related to highways and connectivity. These are reflected in Alternative B, Objective ALL O1, Standards ALL S1 and LINK S1 and Guidelines ALL G1 and LINK G1. Objective ALL O1 and Standard ALL S1 are intended to maintain connectivity. Standard LINK S1 provides a process for identifying wildlife crossings across highways. Guideline LINK G1 encourages retaining in public ownership National Forest System lands located within linkage areas.

In comments on the Draft EIS, some people said more should be done than just identifying highway crossings. Others questioned whether wildlife will even use highway crossing structures.

The U.S. Fish and Wildlife Service identified connectivity as an important consideration in the Southern Rockies (USDA Fish and Wildlife Service 2000b and 2003). The selected alternative will provide management direction for those aspects within the authority of the Forest Service that will contribute to the conservation of lynx. Only minor effects to the existing road system, resource management programs and the traveling public would be anticipated as a result of the management direction under Alternative F-modified.
The Colorado Department of Transportation (DOT) and Wyoming DOT coordinate with the Forest Service to identify areas where efforts could be made to reduce lynx mortality and to improve highway permeability to lynx movement. There will be some additional time and costs associated with evaluating and implementing methods to avoid or reduce effects of highways on lynx.

Habitat Connectivity
Maintaining habitat connectivity is particularly important in the Southern Rockies Amendment area, which is separated from lynx habitat to the north in Wyoming and distant from populations of lynx in the Northern Rockies and Canada. Objective ALL O1 and standard ALL S1 assure that all management projects in lynx habitat will consider the need to maintain habitat connectivity within and between LAUs and in linkage areas.

ENDANGERED SPECIES ACT

Preliminary recovery objective 2: Ensure that sufficient habitat is available to accommodate the long-term persistence of immigration and emigration between each core area and adjacent populations in Canada or secondary areas in the United States.

The U.S. Fish and Wildlife Service concluded the selected alternative contributes to this recover objective in part, although some concerns remain regarding connectivity within the Southern Rockies and between the Northern Rockies and Southern Rockies.

ALL MANAGEMENT PRACTICES AND ACTIVITIES (ALL). The following objectives, standards and guidelines apply to all management projects in lynx habitat in lynx analysis units (LAUs) in occupied habitat and in linkage areas, subject to valid existing rights. They do not apply to wildfire suppression or to wildland fire use.

Objective ALL O1
Maintain or restore lynx habitat connectivity in and between LAUs and in linkage areas.

Standard ALL S1
New or expanded permanent developments and vegetation management projects must maintain habitat connectivity in an LAU and/or linkage area.

Guideline ALL G1
Methods to avoid or reduce effects on lynx should be used when constructing or reconstructing highways or forest highways across federal land. Methods could include fencing, underpasses or over passes.

Standard LAU S1
Changes in LAU boundaries shall be based on site-specific habitat information and after review by the Forest Service Regional Office.

VEGETATION MANAGEMENT ACTIVITIES AND PRACTICES (VEG). The following objectives, standards, and guidelines apply to vegetation management projects in lynx habitat within lynx analysis units (LAUs) in occupied habitat…

Standard VEG S5
The Standard: Pre-commercial thinning practices and similar activities intended to reduce seedling/sapling density are subject to the following limitations from the stand initiation structural stage until the stands no longer provide winter snowshoe hare habitat.
5. In addition to the above exceptions…pre-commercial thinning may occur provided that:
   c) Projects are designed to maintain lynx habitat connectivity and provide snowshoe hare habitat over the long term

Note: This standard is intended to provide snowshoe hare habitat while permitting some thinning, to explore methods to sustain snowshoe hare habitat over time, reduce hazardous fuels, improve forest health and increase timber production. Project design must ensure any pre-commercial thinning provides an appropriate amount and distribution of snowshoe hare habitat with each LAU over time and maintains lynx habitat connectivity within and between LAUs. Project design should focus on creating irregular shapes for the thinning units, creating mosaics of thinned and un-thinned areas and using variable density thinning, etc.

HUMAN RESOURCE PROJECTS (HU): The following objectives and guidelines apply to human use projects such as special uses (other than grazing), recreation management, roads, highways and mineral and energy development in lynx habitat and lynx analysis units (LAUs) in occupied habitat, subject to valid existing rights. They do not apply to vegetation management projects or grazing projects directly. They do not apply to linkage areas.

<u>Objective HU O2</u>
Manage recreational activities to maintain lynx habitat and connectivity.

<u>Objective HU O4</u>
Provide for lynx habitat needs and connectivity when developing new or expanding existing developed recreation sites or ski areas.

<u>Objective HU O6</u>
Reduce adverse highway effects on lynx by working cooperatively with other agencies to provide for lynx movement and habitat connectivity and to reduce the potential for lynx mortality.

<u>Guideline HU G6</u>
Methods to avoid or reduce effects to lynx habitat connectivity should be used when upgrading unpaved roads to maintenance levels 4 or 5 where the result would be increased traffic speeds and volumes or contribute to development or increases in human activity.

<u>Guideline HU G7</u>
New permanent roads should not be built on ridge-tops and saddles or in areas identified as important for lynx habitat connectivity. New permanent roads and trails should be situated away from forested stringers.

GLOSSARY
Linkage Area – A linkage area provides landscape connectivity between blocks of lynx habitat. Linkage areas occur both within and between geographic areas, where blocks of lynx habitat are separated by intervening areas of non-lynx habitat such as basins, valleys or agricultural lands, or where lynx habitat naturally narrows between blocks.

<u>OTHER DOCUMENTS</u>

Forest Service Strategic Framework for Responding To Climate Change

The Forest Service Mission is to: Sustain the health, diversity and productivity of the Nation's forests and grasslands to meet the needs of present and future generations.
The Nation's forests and grasslands provide clean water, scenic beauty, biodiversity, outdoor recreation, natural resource-based jobs, forest products, renewable energy and carbon

sequestration. Climate change is one of the greatest challenges to sustainable management of forests and grasslands and to human well-being that we have ever faced, because rates of change will likely exceed many ecosystems' capabilities to naturally adapt. Without fully integrating consideration of climate change impacts into planning and actions, the Forest Service can no longer fulfill its mission.

The Forest Service has a unique opportunity and responsibility to sustain forests and grasslands in the United States and internationally. This responsibility includes: 1) stewardship of 193 million acres of national forests and grasslands, 2) partnerships with States, Tribes and private landowners for assisting communities and owners of 430 million acres of private and Tribal forests, and with other federal agencies, 3) international cooperation, 4) research and development to provide science and management tools. These responsibilities make it imperative that we understand and be able to respond to the effects of climate change on the Nation's forest and grassland resources.

While some ecosystems may be able to adapt rapidly enough to maintain viability and productivity in the face of changing climate, the impacts of climate change on most terrestrial ecosystems are expected to occur at a rate that will exceed the capacity of many plant and animal species and communities to migrate or adapt. Ecosystem processes, water availability, species assemblages and the structure of plant and animal communities and their interactions will change. Some of these changes will enhance ecosystem productivity and carbon storage…Under a changing climate, however, many ecosystems will experience widespread mortality, increased fire and insect activity and other disturbances, changes in water regimes and species losses, with associated loss of productivity and resilience and accelerated carbon loss. Disturbance events can also provide opportunities for recovery actions that will facilitate adaptation and enhance resiliency and ecosystem health in a changing climate. Management to maintain vegetation within the historic range of variability will increasingly not be an option in many areas. Strategies based on historical or current conditions will need to be replaced with approaches that support adaptation to the changing conditions of the future.

Strategies to address climate change must encompass two components:

- Facilitated adaptation, which refers to actions to adjust to and reduce the negative impacts of climate change on ecological, economic and social systems; and
- Mitigation, which refers to actions to reduce emissions and enhance sinks of greenhouse gases, so as to decrease inputs to climate warming in the short term and reduce the effects of climate change in the long run.

In the face of current changes and future projections, critical work is needed to help ecosystems adapt to the changes that will occur in our lifetimes and pursue mitigation opportunities that can help ensure sustainable ecosystems for future generations.

Facilitated Adaptation**:** Approaches to facilitating adaptation will need to be regional and site-specific, and they will fall into two major categories. Anticipatory actions intended to prevent serious disruptions due to changing climate may include thinning of forests to increase tolerance to drought and resistance to wildfire or insects, genetic conservation of species, assisted migration of species to suitable habitat, development of wildlife corridors to facilitate migration, or construction of new water storage facilities. Opportunistic actions that take advantage of man-made or natural disturbance events to facilitate adaptation to future climate may include planting of different species or genotypes from those that occurred on a site before disturbance or active conversion of vegetation structure to make it more resilient to changing climate.

Actions that minimize disruptions in the ability of ecosystems to provide ecosystem services and that facilitate adaptation to changing climate must be central priorities for the Forest Service because many of these services may be lost or significantly altered if the ecosystems are left to adapt on their own. Ecosystem health and resilience, productivity, biological diversity, and carbon storage are likely to decrease over large areas without direct intervention and management. Mitigation activities can only provide significant benefits if ecosystems are adapted to their new environments.

Key Terms:

Adaptation –

- *Natural Adaptation* – reactive responses by natural systems to the effects of a changing climate. In some cases, individuals, species, communities or ecosystems may adapt (migrate, shift, modify behavior, etc.); in other cases these entities may perish or cease to exist.
- *Facilitated Adaptation* – initiatives and measure to reduce the vulnerability of natural and human systems against actual or expected climate change effects includes both anticipatory and opportunistic actions.

Ecosystem Services – are commonly defined as the benefits people obtain from ecosystems. They include basic services like the provision of food, fresh water, wood and fiber, and medicine; environmental services like carbon sequestration, erosion control, biodiversity, wildlife habitat, and pollination; cultural services like recreation, ecotourism, and educational and spiritual values; and supporting services like nutrient cycling, soil formation and primary productivity.

Mitigation – actions to reduce emissions and enhance sinks of greenhouse gases, so as to reduce the impacts and effects of climate change.

1) Adaptation to the effects of climate change is essential if we are to sustain forests and grasslands to provide ecosystem services and continue to mitigate greenhouse gases.
2) Management for adaptation will not be possible or needed everywhere; priorities will need to be set to determine the most beneficial outcomes.
3) Improved risk analysis and decision support tools will be critical to facilitate new policies and management approaches in the face of uncertainty.
4) Continual monitoring and incorporation of new science into planning, policies, and decision processes are essential to adaptation and mitigation in a changing climate.

<u>Principles Related to People</u>

1) Alliances and collaboration will be essential to achieving science-based, integrated approaches for adaptation and mitigation.
2) Institutional and public support and encouragement for implementing innovative approaches is essential.
3) Strategies, policies, and actions for addressing climate change will be integrated across all Deputy areas at all levels of the Forest Service.

<u>Goals Focused on Managing the Land</u>

1) **SCIENCE** – **advance our understanding** of the environmental, economic and social implications of **climate change** and related adaptation and mitigation activities on forests and grasslands.

To successfully manage forests and grasslands in a changing environment, the Forest Service needs to translate relevant science into land management applications using improved, coordinated and enhanced monitoring systems, predictive models, decision support tools, and databases. These tools will aid resource managers by monitoring trends and predicting future changes. These tools are also critical to understanding the role of the United States forests and grasslands in international agreements created to mobilize global action to address climate change. Managers and policy makers will be better able to evaluate the effects of management actions, consider alternatives and make decisions in an uncertain, changing environment. Research is also needed to develop improved, cost-effective methods for biomass utilization, bioenergy, fossil fuel substitutes, soil carbon enhancement, storage in wood products and greenhouse gas accounting.

Also needed are unified, multi-scale monitoring systems sufficient for:

- Evaluating national and regional trends;
- Assessing the effectiveness of management activities designed to mitigate climate change and adapt to its effects;
- Assessing progress in working across landscapes and ownerships; and
- Understanding the interactions with environmental, social and economic conditions.

The integration of science, monitoring, and management will aid land managers – federal, State, Tribal and private – and citizens in making decisions and taking actions affecting the Nation's forests and grasslands.

2) **ADAPTATION – Enhance the capacity of forests and grasslands to adapt** to the environmental stresses of climate change and maintain ecosystem services.

The primary focus of efforts on National Forest System lands will be to facilitate the adaptation of ecosystems to the effects of climate change. Many activities currently underway to restore forests and grassland health and reduce the risk of severe wildfires or pest outbreaks (such as thinning overstocked stands, thinning to alter species composition, fuels reduction, and prescribed fire) also serve to restore ecological health and resilience in the face of future stressors. More extensive application of such measures is vital for adaptation of forests and grasslands, and will need to be part of future planning and management actions to address climate change and its impacts. Lack of markets for the by-products of treatment activities and institutional barriers are significant constraints on implementing adaptation-related projects on National Forest System lands. The Woody Biomass Utilization Strategy identifies goals to address the lack of markets and institutional barriers for marketing the by-products of treatment activities.

The Forest Service has authorities and the ability to assist private landowners and communities to voluntarily implement adaptation techniques on their lands, and to work collaboratively with other federal agencies and international partners. Science-based and easily accessible information and tools are essential.

4) **POLICY – Integrate climate change**, as appropriate, **into Forest Service policies, program guidance and communications** and put in place effective mechanisms to coordinate across and within Deputy Areas.

The Chief has made climate change a top issue for the Forest Service because of its significant impacts to forests and grasslands, and to society. The agency has begun considering climate change in policies, program guidance and communications. In particular, several actions constitute important first steps in grappling with the issues of addressing climate change in forest plans, NEPA analysis, and budget guidance. As required by the 2008 National Forest System Land Management Planning Rule, the National Environmental Management System will include a land management component, which could be defined to address adaptation and mitigation on National Forest System lands.

The uncertainties of outcomes in a changing climate will require the Forest Service to be flexible and adaptable. Addressing climate change will depend on reducing institutional barriers and increasing adaptive learning through experimentation. Monitoring and evaluation will assist managers in dealing with uncertainties and the risks of options, decisions and actions. The Forest Service will need to build consideration of climate change into virtually all aspects of agency operations including consideration of life cycle analysis of activities.

There are a variety of national strategies in place or under development that could complement and reinforce a truly cohesive approach to climate change. These include strategies on integrated vegetation management, biomass, open space, ecological restoration, water, research and development and others.

Collaboration and integration structures are essential to effectively coordinate across Deputy Areas. Some Regions and Research Stations have begun to identify governance actions to improve integration. These types of activities should be encouraged and reinforced. Coordination that integrates across regions and stations will assure that efforts are complementary and not redundant. Unless more effective integration and coordination mechanisms are put into place, this strategic framework has little chance of meaningful implementation.

7) **ALLIANCES – Establish, enhance and retain strong alliances and partnerships** with federal agencies, State and local governments, Tribes, private landowners, non-governmental organizations, and international partners to provide sustainable forests and grasslands for present and future generations.

APPENDIX 1 - CLIMATE CHANGE STRATEGY GOALS & RECOMMENDATIONS		
Number	**Goal**	**Recommendation**
1	**Science - Advance our understanding** of the environmental, economic and social implications of **climate change** and related adaptation and mitigation activities on forests and grasslands.	1.1 **Develop and implement internal mechanisms** to assure a systematic, interactive dialogue between researchers, public and private land and resource managers, and other users to promoted effective alignment of climate change science delivery efforts. (Links to Recommendation 4.1)
		1.2 **Review and adjust priorities** for the most critical focus areas for Forest Service research, development and application activities, including: (1) key **knowledge gaps** in the economic, social and environmental effects of climate change; (2) **implications of land use and land cover change** feedbacks to climate change; and (3) **effects of potential adaptation and mitigation actions** related to forest and grassland ecosystems and products.
		1.3 **Effectively move science into application,** including synthesis of current research and monitoring information, incorporating science into decision support tools, disseminating new knowledge to managers, and integrating tools into common data and analysis structures. Among other things, decision support tools should focus on: (1) **predicting the ecological effects of climate change** at national, regional and local scales; (2) **predicting the effects of management activities** on the ability of forest and grassland communities and their component species to adapt to climate change and provide ecosystem services; (3) **assisting public and private land managers in prioritizing activities** to maximize effectiveness of adaptation strategies in the face of limited resources; and (3) **evaluating the feasibility and impacts of mitigation actions** that involve forests and grasslands and their products.

		1.5 In collaboration with partners and stakeholders, **carry out integrated regional and sub-regional landscape-scale assessments** of the multiple implications of climate change to improve adaptation, mitigation, and conservation activities on forest and grassland ecosystems and the values, outputs and ecosystem services they provide.
		1.6 **Develop improved life cycle analysis of bio-products from forests and grasslands. Promote** development of **methods**, operational processes and decision support tools **to enhance the capacity of these bio-products to offset fossil fuel emissions and to sequester carbon.**
2	**ADAPTATION - Enhance the capacity of forests and grasslands to adapt** to the environmental stresses of climate change and maintain ecosystem services.	2.1 **Set priorities for where, when and how to employ adaptation activities and implement actions** that will: (1) **facilitate adaptation** to the long-term effects of climate change by fostering resilient, productive and functional ecosystems and (2) **prioritize** types and distribution of **management activities** for the greatest benefits to ecosystems and society.
		2.2 **Work with partners,** including other federal agencies, international partners, State and local governments, Tribes, private landowners, managers, consultants, non-governmental organizations, and other stakeholders **to be most effective in** supporting their **efforts to adapt** lands, ecosystems and species to climate change.
		2.3 **Assess how land management activities** (e.g. fire suppression, fuels treatment, post-fire rehabilitation, timber harvest, forest health and invasive species management, ecological restoration and watershed management) **contribute toward adaptation objectives** and how they can be modified to better facilitate adaptation to climate change at various spatial scales.
		2.4 **Ensure that effects of climate change adaptation activities are monitored** (using the monitoring system established under Recommendation 1.4) **and that new knowledge is documented,** reported and used effectively to modify future management actions.

3	**MITIGATION - Promote the management of forests and grasslands to reduce the buildup of greenhouse gases,** while sustaining the multiple benefits and services of these ecosystems.	3.1 **Participate in the development of protocols for carbon accounting** at the international, national, regional and state levels that fully incorporate the potential for forests, forest products and grassland ecosystems and products to mitigate the build-up of greenhouse gases. Develop a consistent approach to guide that participation. Develop a national-level central 'clearinghouse' for information and Forest Service positions on carbon protocols to provide consistency across efforts.
		3.3 **Identify opportunities across all ownerships** for aforestation, reforestation, and forest management to reduce greenhouse gas emissions and increase sequestration domestically and globally.
		3.4 **Work internationally and with States and other partners to identify opportunities to reduce the rate of conversion of forests and grassland ecosystems** to other uses, and in cooperation with partners, facilitate participation by landowners in programs, including market incentives to retain forest cover.
4	**POLICY - Integrate climate change into all Forest Service policies, program guidance, and communications** and put in place effective mechanisms to coordinate across and within Deputy Areas.	4.1 **Create a rapid national analysis of the implications of climate change** for the Nation's forests and grasslands and our capacity to respond to them, including economic and social costs and benefits to the agency and society.
		4.2 **Implement** the appropriate **mechanisms** and institutional structures **to promote effective collaboration between Deputy Areas of Research, National Forest System and State & Private Forestry** to assure that relevant and helpful research and science is being conducted and distributed.
		4.3 **Address climate change as a part of agency plans and direction** to the field, including: (1) program budgeting, (2) forest planning and NEPA, and (3) strategic plans at various levels (Forest Service Strategic Plan, Ecological Restoration Plan, Cohesive Fuels Management Strategy, Water Strategy, Open Space Conservation Strategy and others).

		4.4 **Evaluate and remove the institutional barriers**, policies and constraints that exist to implementing effective management activities to address climate change.
		4.5 **Implement approaches and incentives to encourage managers to make responsible decisions in the face of uncertainty.**
		4.7 **Promote innovation by incorporating the results of Environmental Management System's** scientifically-designed monitoring into decision-making.
6	**EDUCATION - Advance awareness and understanding** regarding principles and methods for sustaining forests and grasslands, and sustainable resource consumption in a changing climate.	6.1 Work with scientists, land and community managers, educators and communicators to **translate climate change science into** accurate, audience-appropriate and **easily accessible tools and information.**

Kootenai National Forest Wildlife Approach Areas

Introduction

Maintaining wildlife population connectivity through identification of corridors/linkage zones has been examined by a variety of experts and managers…

In general terms, corridors/linkage zones are areas where animals can find food, shelter, and security in order to move across the landscape. They are areas where there are lower densities of human site developments and lower risk to wildlife. Direction associated with NF lands related to corridors and linkage zones are found in a number of areas including: the grizzly bear recovery plan and the Northern Rockies lynx management direction. Corridors/linkage zones were considered at the broader forest-wide scale and included in the draft final plan desired conditions and guidelines. As some point these corridors/linkage zones cross what are termed “fracture lines” (e.g. valley bottoms with highways, railways) where animal movement may be hindered and mortality risk may be elevated. These areas are termed “approach areas”. Providing a safe way for wildlife to approach, cross, and then leave a fracture line is the focus of this paper.

Providing safe and secure areas of wildlife movement across the Kootenai national Forest is one management component needed to assure continued species diversity. The focus area for management is the National Forest System (NFS) lands adjacent to major motorized vehicle routes (highways and railways). These routes have been called “fracture lines” b/c of the increased mortality risk to wildlife as they attempt to move across these features and the potential for fragment habitat and separate or isolate portions of a species population. NSF lands that lie adjacent to these linear features may provide a way for wildlife to approach and leave safely before and after crossing one of these fracture lines. The identification and delineation of these areas, termed "approach areas" and the subsequent management of NFS lands within those areas were based on direction developed by the IGBC Public Lands Wildlife Linkage Taskforce (2004) headed by the regional office. Delineations of approach areas also identifies private lands where land exchange, conservation easement or direct acquisition may be appropriate to improve management options for one or more wildlife species.

Management Considerations

In order to connect large land areas and populations of highly mobile species, planning an effective linkage zone includes public lands, private lands, and issues relating to transportation corridors.

Corridors/Linkage areas/Approach Areas Desired Condition

Corridors/linkage areas and associated approach areas provide for wildlife movement (e.g. migration/dispersal) and genetic interactions. Corridors/linkage areas and associated approach areas provide secure habitat conditions for wildlife movement (for species such as Canada lynx, grizzly bear and wolverine) between large blocks of habitat and/or seasonal habitats o a localized and landscape scale, especially across valley bottoms and other 'fracture zones'. These areas provide cover and often connect key habitat components for those species that use that particular area. NFS lands contribute to linkages between landscapes, unless such landscape isolation is determined to be beneficial.

The Forest cooperates with MT and ID State departments of transportation and private landowners to allow movement of wildlife across valley bottoms between large blocks of habitats on NF lands while considering public safety (reduce automobile/wildlife associated accidents).

Current Forest Plan Guidelines

1. The construction of new permanent roads, opening currently restricted roads to long term motorized use (more than 2 years), motorized trails, and site developments that reduce security and tend to make wildlife avoid use of these areas should not occur I established approach areas. When necessary to construct a new permanent road through established approach areas, motorized use of that road should be restricted.
2. Vegetation management activities in established approach areas should maintain or improve habitat conditions, such as visual cover, for continued and future use of the area.

Proposed Forest Plan Guidelines

1. Avoid activities that reduce security or tend to make wildlife avoid use of corridors/linkage zones and approach areas such as construction of new permanent roads, motorized trails, or site developments; and opening currently restricted roads and trails to motorized use within those areas.
2. Maintain appropriate amounts and distribution of natural foods and hiding cover in corridors/linkage zones and approach areas to meet the subsistence and movement needs of target wildlife species.
3. Manage dispersed recreation use to maintain suitability of approach areas for identified target species
4. Manage human, pet and livestock foods, garbage and other potential wildlife attractants to minimize the risk of conflicts between people and wildlife in approach areas
5. Pursue mitigating, moving and/or reclaiming developments and disturbed sites that conflict with the objective of providing wildlife linkage.

Attachment B: Recommended management direction to maintain wildlife linkage on public lands along highways (from IGBC Public Lands Linkage Taskforce Report 2004)

Recommended Management Direction	**Objective**
1. Maintain appropriate amounts and distribution of natural foods and hiding cover in linkage zones to meet the subsistence and movement needs of target wildlife species.	Maintain food/cover/movement
2. Avoid constructing new recreation facilities or expanding existing facilities (e.g. campgrounds, visitor centers, lodges, etc.) within linkage zones.	Maintain security/avoid mortality risk/avoid habitat loss
3. Avoid other (non-recreational) new site developments or expansions that are not compatible with subsistence and movement needs of target species in linkage zones (e.g. special use developments, gravel pits, etc.).	Maintain security/avoid mortality risk/avoid habitat loss
4. Pursue mitigating, moving and/or reclaiming developments and disturbed sites that conflict with the objective of providing wildlife linkage.	Maintain security/avoid mortality risk/restore lost habitat
5. Manage dispersed recreation use to maintain suitability of approach areas for identified target species. Avoid issuing new permits or additional use days for commercial recreation activities (e.g. outfitter and guide permits) that may conflict with wildlife linkage objectives.	Maintain security/avoid mortality risk and displacement
6. Manage roads and trails in linkage zones to facilitate target species movement and limit mortality risk, displacement and disturbance.	Avoid mortality risk, displacement and disturbance
7. Manage livestock grazing to maintain wildlife forage and hiding cover and to minimize disturbance, displacement and mortality of target wildlife species.	Maintain food/cover/avoid mortality risk
8. Work with adjacent landowners, planners, and other interested parties to improve linkage opportunities across multiple jurisdictions (e.g. cooperative agreements, land consolidations, exchanges, acquisitions, easements, etc.).	Enhance linkage opportunities
9. Manage human, pet and livestock foods, garbage and other potential wildlife attractants to minimize the risk of conflicts between people and wildlife.	Provide for human safety/avoid wildlife mortality risk

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Zoological Special Interest Areas:
Tongass National Forest – Pack Creek

Terrestrial Mammal Management Indicator Species (MIS)

Bears at Middle Creek:
…The majority of bear bedding currently occurs on the north side of the creek and on the gravel bars at the apex of the alluvial fan. Bear trails are concentrated near the stream, but there are also important corridors linking this drainage with others to the north and south…

…Human use of the shelter and the estuary meadows has likely influenced long-term patterns of use by bears (e.g. the conspicuous lack of bedding in the large tree forest at the base of the west-side alluvial fan). Most human use of this southern estuary in Windfall Harbor is on the western side of the creek. Commercial guides agreed in 2000 to confine their visits to the western side to prevent displacing bears from habitats further up the creek. SEAWEAD has offered two suggestions for future management of visitor use in this area:

1. To emphasize protection of bear access to habitat, discourage human use of the Windfall Harbor estuary and focus a limited amount of use at the existing shelter. This location would provide a long-distance viewing opportunity that would not significantly affect bears in the estuary and along the anadromous stream. The beach on the west side of Windfall Harbor is likely an important travel corridor for bears that travel to and from drainages to the north. Strict emphasis on protection of bear resources argues for limited use of this shoreline, including the shelter, such that the area would be free of human occupation as much as possible.
2. A compromise between bear and human use of the Windfall Harbor estuary may be achieved if guided and non-guided use is restricted to the beach area at the base of the west-side alluvial fan. This area offers a broad view of the meadow and creek without placing observers in the immediate vicinity of the concentrated trail and bedding areas near shore. The viewing site should be approached from the shelter. All food should be left in bear-proof containers in the shelter to reduce the possibility of food conditioning. Travel time to and from the viewing site should be minimized to reduce disturbance to the west-side bear travel corridor. Because the close proximity to important grazing resources, human behavior at the viewing site should be controlled to reduce offensive scents, loud noises, and abrupt movements. Duration of site occupancy might also be restricted. Disturbance of some bears will likely occur under this scenario because of overlapping use on the west-side travel corridor and the occurrence of high-value grazing habitats in close proximity to the viewing site.

As noted above, the western shore of Windfall Harbor between Pack Creek and Windfall Creek is used by bears as a travel corridor between high value habitats at the estuaries, while the eastern shore has no anadromous estuarine habitat and far less evidence of use by bears…

POTENTIAL FISH AND WILDLIFE PROJECTS

Introduction

…Sensitive wildlife habitats that could be impacted by visitor use would include avian nest sites, amphibian breeding ponds, seal and sea lion haul outs and important bear fishing sites and travel corridors. Human caused impacts to these sites can best be mitigated by restricting or discouraging visitor use of such sites. The specific locations of some of the more sensitive sites at risk from visitor presence (i.e. amphibian breeding ponds) should not be made common public knowledge in order to protect the site.

Avoid Impacting Sensitive Wildlife Habitats

These would include avian nest sites, amphibian breeding ponds, seal and sea lion haul outs, and important bear fishing sites and travel corridors…Devise a permanent strategy whereby impacts to the resource and recreational opportunities may be mitigated.

DEPARTMENT OF INTERIOR

BUREAU OF LAND MANAGEMENT

Dillon Resource Management Plan and EIS

Desired Future Condition
- (bullet 3) provide suitable habitat and condition to allow wildlife, species movement between large blocks of habitat, and seasonal and special habitats on a localized and landscape scale.

Alternative A
Under current management, specific wildlife travel corridors or linkage corridors between major habitat areas would not be delineated, and potential impacts would be considered on a case by case basis during project and activity planning.

Alternative B
…wildlife migration/dispersal corridors that provide connectivity for special status species such as lynx, grizzly bear, and wolf (as well as wildlife in general) would be managed to reduce conflicts between listed species and land use authorizations and activities.

Management actions would include:
- Evaluate projects and authorizations proposed on public lands in this area that may increase habitat fragmentation, create physical barriers to movement or potentially increase mortality.
- Food storage strategies…
- Amend grazing permits..

These actions would apply to all public lands that contain relatively intact habitat and migration corridors between units of the BDNF.

Alternative C
…wildlife migration/dispersal corridors would be delineated as described under Alternative B, but additional management actions would apply. Management actions to reduce potential risks to grizzly bear, wolf and lynx would include:
- Coordinate with others to identify critical barriers and potential passage locations…
- Evaluate projects and authorizations proposed on public lands in this area that may limit the effectiveness of the corridor by increasing habitat fragmentation, creating physical barriers, or potentially increasing mortality

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Pinedale Resource Management Plan
Record of Decision

2.3.16 Wildlife and Fish Habitat Management
Management Goals
Maintain or enhance aquatic and wildlife habitat.

Maintain functioning big game habitats and migration corridors that allow free movement and use of habitats.

2.3.17 Special Designations and Management Areas
Management Goals
Trapper's Point ACEC Management Goal. Preserve the viability of the big game migration bottleneck, cultural and historic resources, and important livestock trailing use.

DEPARTMENT OF INTERIOR

Department of the Interior Task Force on Climate Change
Report of the Subcommittee on Land and Water Management

ADAPTATION ISSUES AND OPTIONS

COMMON THEMES AND DOI-WIDE OPTIONS
Theme 2: Land, Resource and Species Management Plans Need to be Revised to Reflect Climate Change Effects.
Nearly all of the working groups of the Subcommittee on Land and Water Management identified a need to revise management plans to reflect effects of predicted climate conditions…

Theme 3: Definitions for Key DOI Agency Terms, such as "Natural" and "Unimpaired."…
Option 3: Define Key DOI Agency Terms in the Context of a Changing Climate…

Theme 6: Encouraging and Supporting Partnerships for Adapting to Climate Change…
Option 6: Develop an Interior Climate Adaptation Partners Program. Develop a DOI Adaptation Partners (ICAP) Program that provides guidance and possible financial incentives for developing cross-jurisdictional, public/private partnerships that contribute to the conservation of species, natural communities and lands and waters placed at risk by changing climate conditions…

…A financial incentives fund could increase the ability of individual management units to work with private partners who need compensation to take lands out of agricultural production, delay timber harvest, or take other actions in order to maintain a corridor or protected area…

SPECIES MIGRATION AND HABITAT CHANGE
Statement of the Issue
Climate change causes species and natural communities to shift in latitude and/or elevation (primarily northward or upward) across the landscape, perhaps away from DOI-managed lands.

Description of Issue
Plants and animals only reproduce, grow and survive within specific ranges of climate and environmental conditions. When conditions change beyond their tolerance, both plant and animal species may respond by shifting range boundaries or changing the density of individuals within their ranges. Predicted climate changes will make the current ranges inhospitable for many resident species on DOI lands. Following suitable habitat conditions, these species will generally attempt to migrate northward or upward.

This 'species migration' is not the short-term seasonal migration that waterfowl perform each year, but long-term shifting of entire species or local communities to new home ranges. These natural communities will not be replaced suddenly. Individual species will migrate to new areas or die off, placing stress on other species in the community that depend on them for food or habitat. Species losses will eventually cascade through many natural communities and landscapes. Other species will invade empty niches left behind, bringing with them changes to the historical landscape and the ecological services and benefits to which people are accustomed.

A wide variety of natural and man-made barriers can prohibit the natural migration of plants and animals to suitable new locations. Highways, urban areas, rivers, agricultural lands, pipelines, dams, unseasonably low river flows, habitat fragmentation, and lack of connectivity between water sources are just a few obstacles to migration. Even highly mobile species may face serious obstacles to successful migration if their food and habitat requirements cannot cross barriers or do not exist in new areas.

Migratory waterfowl, Neotropical birds, anadromous fish (those that migrate from saltwater to freshwater to spawn) and some insects such as Monarch butterflies offer unique challenges. These species travel great distances during their life cycle, generally from wintering to breeding habitats. Loss of any portion of essential habitat along their migration routes may cause serious populations declines. For example, much of the Prairie Pothole wetlands in the upper Midwest is predicted to dry due to climate change. This drying would eliminate critical breeding grounds for ducks and geese along the central flyway.

Anadromous fish are of particular concern to DOI because they provide significant ecological, economic, and cultural values to native peoples, rural Alaskans, and American society as a whole. Many salmon species are already suffering serious declines due to past and present human-induced habitat modifications and other stresses that are not yet well understood. Climate changes are expected to cause additional stresses, possibly pushing some populations to the brink of collapse. Actions could be taken to increase our understanding of fish responses to changing climate conditions and to reduce other stressors to fish populations.

Statement of Options

Option 1: Assess Vulnerabilities: Species Migration. Conduct a screening level vulnerability assessment of ecosystem shifts in relation to DOI lands.

Option 2: Encourage Regional Inventory and Monitoring Partnerships. Develop regional partnerships to build on existing biodiversity monitoring programs to inform regional-scale decisions for species on DOI lands.

Option 3: Identify and Highlight Species Migration Case Studies. Use selected case studies to educate and inform resource managers on successful species migration and relocation projects.

Option 4: Develop Predictive Models for Species Response. Develop planning models to predict species response.

Option 5: Promote Regional Partnerships for Species Migration and Relocation. Promote regional partnerships to enhance the success of species migration and relocation in response to climate change. This option is more fully described under DOI-Wide Option 6, “Develop an Interior Climate Adaptation Partners (ICAP) Program”.

Analysis of Options

Option 1: Assess Vulnerabilities: Species Migration.

DOI could conduct a vulnerability assessment of ecosystem shifts in relation to DOI lands. The first phase of the assessment could begin by using regional-scale models of climate change predictions and ecosystem responses to create a series of regional maps that overlay expected ecosystem shifts onto DOI lands. These initial maps could then be used to focus national DOI resources on climate change species migration hot-spots. The initial assessment would be regional aimed at completing all regions within a short timeframe.

A second phase of the vulnerability assessment would focus on the species migration hotspots identified in the initial assessment. At this scale, the assessment would focus on identifying individual species and their specific habitats that are expected to either migrate away from protection of DOI lands or be locally extirpated due to climate change. These species will need specific intervention either to protect species health, or to ensure continuance of the services (ecosystem, economic, or cultural) they provide. The cost of a second-level vulnerability assessment would be medium and the timeframe would be medium to long, depending on the availability of resources and the findings of the initial regional assessments. There would be ample opportunity for partnerships with other agencies and with existing partnerships as data are developed and compared.

Option 2: Encourage Regional Inventory and Monitoring Partnerships.
DOI could develop regional partnerships to build on existing biodiversity monitoring programs. For example, these could build upon existing partnerships between DOI and sister Federal agencies, such as the EPA and USDA and other partnerships such as the National Biological Information Infrastructure and NatureServe

As discussed in DOI-Wide Option 1, adaptive management provides a framework for decision making in the face of uncertainty about human and ecological responses to climate change. This framework includes an iterative decision-making process that involves an initial assessment of conditions, a decision, and monitoring for results. As information is received through the monitoring process, understanding and management decisions are updated by what is learned. Therefore, inventory and monitoring information is necessary for both the initial assessment and for the iterative management decisions inherent in adaptive management.

Few DOI land management units have complete biological inventories of species. Additionally, DOI has no cohesive, systematic program for monitoring change over time in the distribution of species and communities. Inventories will be critical to assessing climate change impacts and to developing management responses to those impacts. During the time that DOI conducts the initial regional-scale vulnerability assessments mentioned in Option 1, managers of DOI lands can begin evaluating existing gaps. Our lands do not exist in a vacuum. Rather, they exist in a matrix with other Federal, State, private, non-profit and corporate neighbors. DOI resource managers can begin developing partnerships at various organizational levels for filling ecological data gaps and for monitoring ecological trends that would help guide our adaptive management strategies into the future.

At the national level, DOI could explore strategic partnerships with one or more well-established national programs to identify current biological resources and assess changes in response to climate change. Joining in one or several of these programs would provide a more complete picture of the biological resources on and adjacent to DOI lands allowing DOI land managers to see their resources and make management decisions in the context of the larger landscape…

Managers at regional and local scales could develop other partnerships to deal with more local issues and to begin developing local and regional strategies for meeting the challenges climate change poses to their resources. These would complement the activities of the national programs previously discussed. By enabling DOI to monitor for changes using the same data and parameters as these other organizations, collaboration on monitoring would promote adaptation partnerships. The direct cost to DOI would likely be in the low-to-medium range and the savings could be substantial as compared to setting up completely new and independent DOI monitoring programs.

<u>Option 3</u>: Identify and Highlight Species Migration Case Studies.
Selected case studies could be used to educate and inform resource managers on successful species migration and relocation projects…

<u>Option 4</u>: Develop Predictive Models for Species Response.

In an uncertain climate future, models will be important tools for predicting how plants and animals are expected to respond to climate changes and for adapting and revising management plans accordingly. These models would allow managers to analyze scenarios that incorporate local and regional temperature, rainfall, and stream flow, as well as selected management actions and to predict responses of plant and animal communities…

<u>Option 5</u>: Promote Regional Partnerships for Species Migration and Relocation.
DOI could promote regional partnerships to enhance the success of species migration and relocation in response to climate change…In particular, DOIs success in both its Healthy Lands Initiative and its Cooperative Conservation Initiative could serve as examples.

TERRESTRIAL CARBON SEQUESTRATION
<u>Statement of Opportunity</u>
…DOI is poised to play a key role in reducing the amount of CO_2 in our atmosphere through terrestrial carbon sequestration. There is an opportunity to reduce DOIs carbon footprint through specific mitigation actions, such as minimizing or offsetting residual carbon emissions through a comprehensive carbon sequestration program…

<u>Analysis of Options</u>
<u>Option 3</u>: Create Habitat Restoration Partnerships
DOI could use its statutory authorities, existing policies and regulations, programs and expertise to work with private landowners and CO_2 emitters to restore significant habitat while helping to offset CO_2 emissions. An important component of the option is to understand where to best establish habitat linkages. A plant and wildlife habitat gap analysis could be used to strategically determine where important plant and wildlife habitat linkages (i.e. wildlife and ecosystem corridors) are needed across the landscape. The results would guide private lands programs and broaden the impact of a comprehensive carbon program to restore native wildlife habitat.

DOI's land base provides anchors of biodiversity that could serve as a foundation for our conservation efforts. Linking these lands together as corridors will require public/private partnerships aimed at cooperatively working with private landowners. Strategic habitat conservation through a well conceived terrestrial carbon sequestration program may accomplish a number of public policy goals, including offsetting CO_2 emissions and conserving nationally important natural resources…

DOI could establish collaborative efforts with the USDA Forest Service Farm Service, USDA Agency and Natural Resources Conservation Service and with non-governmental organizations to look for ways to provide incentives to private landowners as part of a broad terrestrial carbon sequestration program. Options include using existing wetlands, grasslands and conservation reserve programs.

U.S. FISH AND WILDLIFE SERVICE

U.S. Fish and Wildlife Service Climate Change Strategic Plan for the 21st Century

OUR VISION
As a leading conservation organization, we see ourselves:

- Depending on our 95 million acre National Wildlife Refuge System to play a critical role in ensuring habitat connectivity and conserving key landscapes and populations of fish and wildlife;

STRATEGIC GOALS AND OBJECTIVES
Adaptation
Goal 2: We will plan and deliver landscape conservation that supports climate change adaptations by fish, wildlife and plan populations of ecological and societal significance.

While our long-term response to climate change will be determined over the next 5 years as we work collaboratively in developing the National Fish and Wildlife Adaptation Strategy there will be many near-term actions we can take to begin the process of managing fish and wildlife adaptation to climate change. Near-term conservation delivery will apply vulnerability assessments and focus on…(2) reducing habitat fragmentation and building connectivity by means such as habitat corridors…(7) addressing key ecological processes…

> Objective 2.2 – Promote Habitat Connectivity
> Climate change will interact with non climate stressors such as land-use change, fire, and habitat fragmentation from urban, suburban and agricultural development. Protecting contiguous and un-fragmented habitat and enhancing connectivity between protected areas using linkages and corridors will facilitate the movement of fish, wildlife and plan species in response to habitat protection and landscape scale habitat linkages and corridors. By joining the habitat protection and management capacities of the Service (e.g. national Wildlife Refuge System, Partners for Fish and Wildlife Program and North American Wetlands Conservation Act) with those of partners, we will help build this connectivity within and between landscapes.

Goal 5 – We will build capacity to understand, apply and share terrestrial carbon sequestration science and work with partners to sequester atmospheric GHGs while conserving fish and wildlife habitat at landscape scales.

Objective 5.5 – Facilitate International Carbon Sequestration
One of our most important roles in carbon sequestration may well be to facilitate carbon sequestration activities internationally…We will work through our Wildlife Without Borders and Multinational Species Programs to provide funding and technical assistance to increase carbon sequestration, restore habitat and increase connectivity.

U.S. Fish and Wildlife Service Draft 5-Year Action Plan for Responding to Climate Change

ADAPTATION

Goal 2 - We will plan and deliver landscape conservation that supports climate change adaptations by fish, wildlife and plan populations of ecological and societal significance.

Objective 2.1 – Take Conservation Action for Climate Vulnerable Species

FY 2011-13

- The Science Advisor will ensure that the results of the vulnerability assessments are spatially integrated with recommendations for landscape-scale habitat connectivity in order to provide a landscape-level overview of opportunities for climate-vulnerable species to migrate and colonize new habitats.

Objective 2.2 – Promote Habitat Connectivity
Climate change will interact with non-climate stressors such as land-use change, fire and habitat fragmentation from urban, suburban and agricultural development. Protecting contiguous and un-fragmented habitat and enhancing connectivity between protected areas using linkages and corridors will facilitate the movement of fish, wildlife and plant species in response to climate change. Through conservation design, we will work with partners to identify needed habitat protection and landscape-scale habitat linkages and corridors. By joining the habitat protection and management capacities of the Service (e.g. national Wildlife Refuge System, Partners for Fish and Wildlife Program and North American Wetlands Conservation Act) with those of partners, we will help build this connectivity within and between landscapes.

FY 2009

- The ANWRS, AMB and AFHC will work with the RDs to demonstrate how Service programs can promote habitat connectivity to achieve population objectives. AFHC will provide a progress summary and final report including proposed funding redirections.

FY 2010-11

- RDs, working through LCCs, will ensure that climate change is addressed in existing on-the-ground projects to promote habitat connectivity among protected areas to achieve objectives through habitat acquisition or restoration. The projects should characterize the carbon sequestration potential of habitat that is conserved or restored.

Goal 5 – We will build capacity to understand, apply and share terrestrial carbon sequestration science and work with partners to sequester atmospheric GHGs while conserving fish and wildlife habitat at landscape scales.

Objective 5.5 – Facilitate International Carbon Sequestration
One of our most important roles in carbon sequestration may well be to facilitate carbon sequestration activities internationally…We will work through our Wildlife Without Borders and Multinational Species Programs to provide funding and technical assistance to increase carbon sequestration, restore habitat and increase connectivity.

FEDERAL INTERDEPARTMENTAL

Interagency Grizzly Bear Committee
Memo Re: Support for the concept of linkage zones

...Habitat fragmentation is one of the issues complicating the conservation of grizzly bears and many other species of wildlife. Habitat fragmentation is the process of separating populations of animals and their habitats into smaller and smaller units. Small, fragmented populations of any species are less likely to survive. The main factor causing habitat fragmentation is human development, especially when development occurs in a linear fashion. Development in mountain valleys and transportation systems such as highways and railroads are common problems for wildlife. If we do not maintain the opportunities for linkage of wildlife populations across these areas of human development, we will have difficulty securing the future of wildlife species such as the grizzly.

To address the issue of habitat fragmentation, the IGBC supports the identification of those areas within and between the major grizzly bear ecosystems where wildlife can live or move between existing large blocks of relatively secure habitat. These areas are called linkage zones. Linkage zones occur primarily between large blocks of public lands. Cooperation and coordination between public land managers, fish and game agencies, private landowners, and state and federal transportation agencies is required to maintain linkage zones that work for wildlife. The IGBC supports this cooperation and coordination.

Especially important in this effort is the cooperation and support of state and federal highway departments to work with wildlife agencies to enhance crossing possibilities for wildlife within linkage zones. A critical part of this effort is support of research and monitoring to identify the best sites for crossing enhancement structures, and the design and placement of such structures at such sites when the opportunity arises through highway improvement and redesign. We urge highway departments to cooperate in this effort.

Another key factor in linkage zone implementation is close and careful cooperation with private landowners to allow them to participate in linkage zone implementation if they choose to do so. The IGBC supports a careful approach that involves private landowners, local governments and all stakeholders in linkage zone activities.

In summary, the IGBC believes linkage zone identification and the maintenance of existing linkage opportunities for wildlife between the large blocks of public lands in the range of the grizzly bear are fundamental to healthy wildlife. Wildlife habitat conservation and the eventual recovery of listed species such as grizzly bears will require connections between populations. Maintaining linkage opportunities will benefit all wildlife species and will help assure healthy populations of the wildlife species we all value.

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Bark beetle effects on a seven-century chronosequence of Engelmann spruce and subalpine fir in Colorado, USA

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ABSTRACT

Many important trends in forest development across landscapes and centuries are difficult to measure directly, and a space-for-time substitution in a chronosequence may provide useful insight at these scales. The value of chronosequences for forest ecology and management depends on a number of sources of variation, including geographic differences in site productivity, differences in climate over long periods, and the presence or absence of rapid events such as fire, windthrow, and insect outbreaks. Confidence in the value of a chronosequence may be increased if later resampling shows that each site followed the predominant trajectory expected from the chronosequence pattern. We resampled a 700-year chronosequence of Engelmann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*) three decades after the initial sampling. The original chronosequence suggested long-term stasis in both biomass and production after about 200 years of stand developments in the absence of major fire, beetle outbreaks, and windstorms. Three decades later, a spruce beetle (*Dendroctonus rufipennis*) outbreak had reduced spruce biomass by 68% and total stand biomass by 44% across the chronosequence (to an average of 7.8 kg m^{-2}). There remained no trend in total stem biomass with stand age, averaging 13.9 kg m^{-2} of stemwood across all ages. Stem production averaged 0.15 kg m^{-2} yr^{-1} between 1984 and 2013, higher than the 0.09 kg m^{-2} yr^{-1} estimated in 1984. Over the three decades, stand biomass shifted from about 2/3 spruce to 2/3 fir. Stands may be selected for chronosequences based on an absence of rapid events that substantially change stand structure, but this may limit the ability of a chronosequence to represent real long-term patterns across landscapes.

1. Introduction

Forests of spruce (*Picea*) and fir (*Abies*) cover vast portions of the northern hemisphere. A variety of ecological factors appear to sustain the codominance of trees in these forests, including shared abilities to survive for decades and centuries in highly shaded locations, and differential susceptibility to pests (e.g. Peet, 1981; Veblen, 1986a,b; Seymour, 1992; Aplet et al., 1988; Nishimura et al., 2010). Forests dominated by Engelmann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*) dominate many landscapes from central British Columbia and Alberta in Canada southward to Arizona and New Mexico in the United States (Alexander, 1984). Long-term changes in composition, production, and biomass develop at century time scales in these forests, posing major challenges for testing ideas about the changes in these forests over time.

A chronosequence approach has the potential to identify trends expected through time based on patterns among stands of different ages across landscapes, assuming that a predominant trajectory should be followed within individual stands (Walker et al., 2010). The critical assumption of a chronosequence is that variations in ecological factors across space and time are relatively small compared to the predominant trajectory over time. Differences in site factors may confound any time-related pattern, as would any trends in climate across centuries. The occurrence and legacies of rapid change events such as fires, insect outbreaks, and windstorms could add variance that further limits the utility of chronosequences. These three sources of variation (ecological site factors, climate, and rapid events) may even challenge the idea that a clear predominant trajectory should be expected for forest development across landscapes and centuries.

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Aplet et al. (1988, 1989) used a chronosequence approach to determine the likely changes in forest composition, stemwood biomass, and stemwood production across seven centuries of change in forests of Engelmann spruce and subalpine fir in northern Colorado. They chose nine stands representing different ages since stand-initiating disturbance (likely severe fire) and used tree ages, diameters, growth, recruitment, and coarse woody debris to examine stand development in response to chronic factors (competition, minor windthrow, endemic bark beetle activity, root diseases) in the absence of major mortality events. Later, Rebertus et al. (1992) presented a general course of forest development (citing Whipple and Dix, 1979; Peet, 1981; Veblen, 1986a, and Aplet et al., 1988) that, in the absence of broad-scale disturbance, follows a continuum that can be divided into four broad phases: colonization, spruce exclusion, spruce reinitiation, and second-generation spruce–fir "old growth." Colonization follows a major disturbance and consists of recruitment of both spruce and fir under conditions of available resources. As the canopy closes, conditions become inhospitable for spruce regeneration, and the stand enters the spruce exclusion period, while fir continues to recruit. Eventually, mortality of large trees, either through competition-induced mortality or as fir reaches the end of its lifespan, opens the canopy and spruce recruitment reinitiates. With time, this spruce recruitment joins the fir as an all-aged forest or forms a second cohort in a long-period cycle of recruitment and exclusion. Various authors (Rebertus et al., 1992; Aplet et al., 1988; Peet, 1981) noted that existence of second-generation forest is unlikely to persist for long, if it is indeed ever reached, as virtually all stand age class structures show evidence of stand initiating disturbance events, and stands containing trees >500 years old are rare.

Chronosequence studies depend on the assumption that the only significant difference between sites is time, and Aplet et al. (1989) cautioned that "[i]n spite of restricting slope, aspect, parent material and climate in this study, other unknown plot-to-plot variation (e.g. initial density and species composition, site quality, stand history) may exist." Rebertus et al. (1992) noted that various authors concluded that differences in site quality among stands affect developmental dynamics such that a variety of stand structures may result: actual trajectories may vary substantially in a population of stands around any central tendency. On especially dry and wet sites, slow recruitment may prevent the onset of a spruce exclusion phase, and Aplet et al. (1989) suggested that, on relatively high-quality sites, overstory mortality early in stand development keeps the canopy open and allows for continuous spruce recruitment. Other authors have reported similar differences in stand trajectory depending on site quality (Whipple and Dix, 1979; Peet, 1981).

Despite apparent differences among sites, Aplet et al. (1989) reported generally stable stand-level biomass from 125 through 700 years of stand development. Stemwood production peaked early in stand development, and then dropped by about half from age 250 years onward. The decline in stemwood production mirrored the decline in stand leaf area. Beyond 175 years since stand establishment, spruce comprised about two-thirds of stem biomass and half of stem production. The distribution of sizes and ages of fir was relatively constant after 200 years, though the distributions for spruce continued to change as the initial even-aged cohort progressed through time. Total biomass and production of stems showed no trend after 200 years.

Did the lack of trend result from long-term patterns of consistent production and mortality (leading to no change in biomass), or did the lack of apparent trend reflect confounding variations of ecological site factors, time, and rapid events that obscured the trends actually followed through the history of each stand? Aplet et al. (1988) marked individual trees so these questions could be tested by resampling in future decades. We report on the changes in stem biomass and production over the next three decades, addressing two questions:

1. Did each stand follow the trend in stemwood biomass and production over time that was suggested by the original chronosequence?
2. Would a remeasurement show the same pattern as the original chronosequence?

Two additional questions were developed once it was clear that rapid mortality from spruce beetles (*Dendroctonus rufipennis*; Colorado State Forest Service, 2013) had substantially altered the stands:

3. Did mortality differ with tree size?
4. Did mortality differ with stand age?

2. Methods

The methods of the original sampling of the chronosequence were described in detail by Aplet et al. (1988, 1989). The stands are located in the upper watersheds of the Cache La Poudre River and Laramie River in Larimer County, Colorado, USA, between 3000 m and 3200 m, most on north-facing aspects (Fig. 1; see Table 1 of Aplet et al. (1989) for details on location and site, where Stand 1 in the present study corresponds to Stand 2 in the 1989 paper, etc.). Spruce and fir comprised >97% of overstory trees and biomass, with minor amounts of lodgepole pine (*Pinus contorta*). Tress of all sizes were cored as close to the base as possible, and stand ages were determined from the oldest, largest trees, all of which displayed ring patters indicative of growth in the open. No stands showed evidence of trees that survived the stand-initiating disturbance.

All of the original plots from 1984 were resampled in 2013 except for the youngest stand (about 125 years old in 1984), which had not been permanently marked for resampling. Each stand was sampled with three plots, either 0.05 ha plots (the youngest resampled stand) or 0.10 ha plots (all others). The resampling entailed locating each original tree and measuring current diameter at breast height (1.4 m), and tallying tree condition (live, dead standing, or dead fallen), but we made no attempt to assess cause of death. A few trees were missing tags, but they were identified reliably based on species, size and location within plots. The diameter cut-off for measurement in 1984 was 5 cm, and new recruits that passed this threshold were tallied by diameter and species (Table 1).

The biomass of stems was estimated using allometric equations for volume, based on diameter and height, and wood density (see methods in Aplet et al., 1989). The diameters were measured directly, and heights were estimated based on locally derived linear relationships between diameter and height in 1984 (measured with a clinometer: spruce, $n = 390$ $r^2 = 0.86$; fir, $n = 307$ $r^2 = 0.88$). Stemwood volume was calculated with equations from Myers and Edminster (1972), and volume was converted to mass based on typical densities (368 kg m^{-3} for spruce, and 433 kg m^{-3} for fir, Wenger, 1984). The values for fir and spruce are given separately, and the total stand values are slightly higher than the sum of the two species owing to inclusion of minor contributions of lodgepole pine.

Stemwood in the first sampling was determined from 10-year growth increments on cores. For the resampling, productivity was calculated from gross stem increments between 1984 and 2013. The production estimates for the 1984 sampling would be somewhat low, as the growth of any tree that occurred between 1975 and 1984 was included only if the tree was alive in 1984. The production estimates for the 2013 sampling did include any

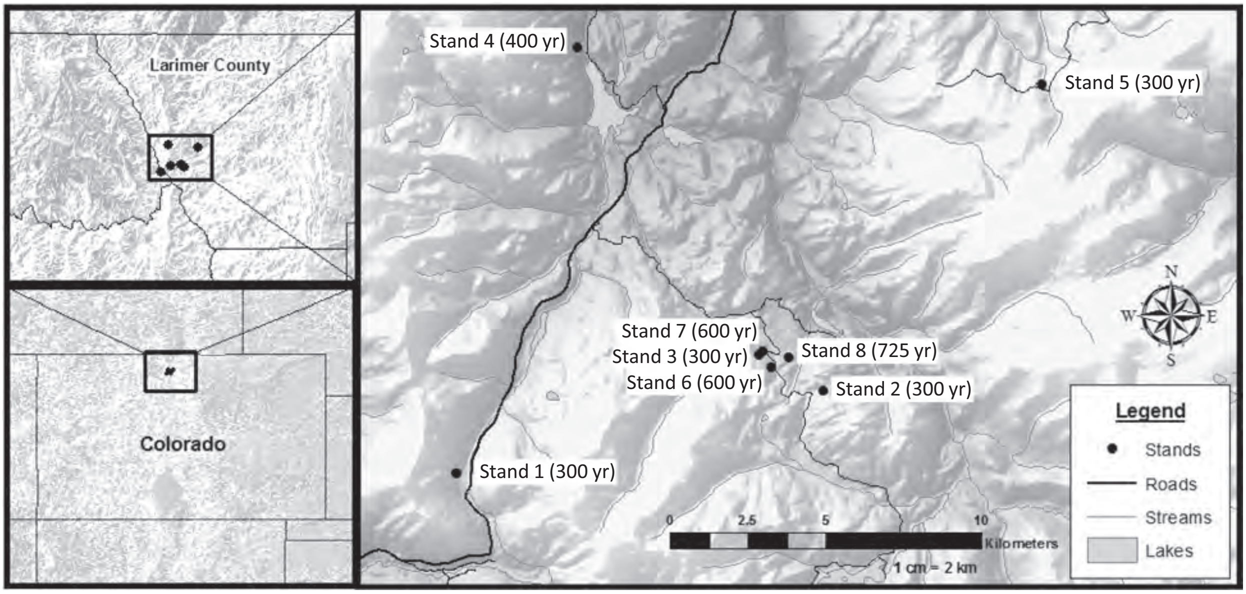

Fig. 1. Location of sampled stands. Stand 1 is near Cameron Pass on Colorado Highway 14. Chambers Lake appears near Stand 4 for reference.

Table 1
Stand descriptions.

Stand number	Stand age (1984)	Density (trees ha^{-1})				Quadratic mean diameter (cm)				Basal area (m^2 ha^{-1})			
		1984		2013		1984		2013		1984		2013	
		Fir	Spruce	Fir	Spruce	Fir	Spruce	Fir	Spruce	Fir	Spruce	Fir	Spruce
1	170	1307	1740	513	873	14.9	16.1	14.6	15.1	22	36	9	16
2	270	1007	1480	1187	813	15.0	22.0	14.4	18.7	18	56	19	22
3	270	1313	1080	1587	560	16.1	26.8	15.3	22.1	27	62	29	21
4	370	1947	1080	1867	500	14.5	30.6	14.8	20.5	32	65	32	17
5	370	1480	1113	1560	513	12.9	26.0	14.3	22.4	25	60	25	20
6	570	1860	953	1767	247	15.3	28.5	15.5	23.3	34	56	33	11
7	570	2660	827	2663	653	15.4	25.3	16.9	20.8	50	42	59	22
8	695	1693	987	1660	360	15.1	29.1	15.2	23.4	31	67	30	15

increment on trees that were alive in 1984 but dead in 2013, with exception of trees that had died and fallen (preventing precise remeasurement of diameter). Measurements made on dead trees also introduce errors associated with shrinkage relative to live trees. We realize that these two methods produce results that are not easily compared, but we were limited by circumstances and compare the two figures with caution.

The values for each stand were averaged from the three plots in each stand (the experimental units of the study). Biomass mortality was calculated as the annual average of the mass of trees that died between 1984 and 2013, with mass determined on the diameter measured in 2013. Net production was the difference between gross stem production and stem mortality. Basal area of dead trees was reported in 1988, but no estimates of dead trees basal area made in the 1984 sampling.

Long-term trends, and relationships between tree size and mortality, were examined with linear and curvilinear routines in CurveExpert Professional 2.2©. The changes over 29 years within stands were tested with paired *t*-tests (in Excel) for stand means of 1984 and 2013. We chose a significance level of $P < 0.05$ to protect against Type I errors.

3. Results

Spruce beetle mortality resulted in substantial reductions in spruce in the larger size classes in each stand, but fir diameter class structure changed very little from 1984 to 2013 (Fig. 2). Table 2 shows changes in live-tree density from 1984 to 2013 for small- (5–15 cm), medium- (25–35 cm), and large-diameter (>50 cm) trees. Spruce experienced reductions in density of medium- and large-diameter trees from 25 to 100 percent in all stands. Overall, only one third of spruce over 30 cm in 1984 survived to 2013. Small-diameter spruces suffered considerably less mortality except in the youngest stand, where mortality exceeded fifty percent, and in Stand 7, in which small-diameter spruce actually increased slightly in density. Fir, in contrast, showed considerably more variability, displaying reductions in density of small- and medium-diameter trees greater than fifty percent in the youngest stand but less change in the density of small-diameter trees and highly variable change in medium-diameter trees across the remaining centuries of the chronosequence. Aggregating all trees across the chronosequence, fewer than half of the fir trees >30 cm diameter in 1984 survived to 2013, indicating that the relative stability or increase in density of medium-sized fir was due to growth into that class over the thirty years. Overall, mortality showed no trend with stand age for either fir or spruce, but large trees of both species had much greater mortality rates than smaller trees (Fig. 3). No trees of either species >60 cm survived.

Total stem biomass showed no trend with stand age in either 1984 or 2013, but high mortality of spruce led to lower biomass for all stand ages in 2013 (Figs. 4 and 5). Fir biomass remained unchanged over the three decades, but spruce biomass declined

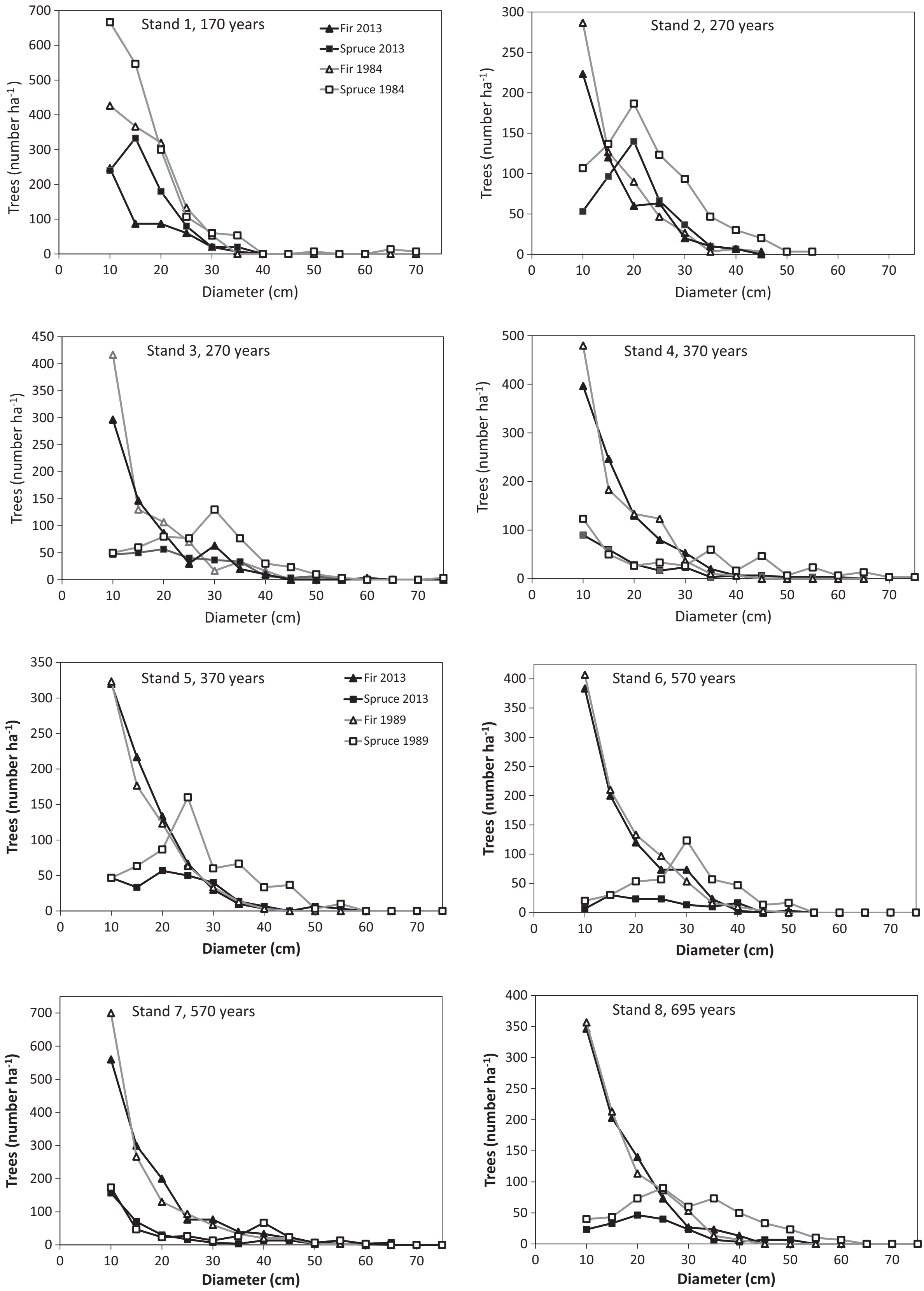

Fig. 2. Diameter distributions for stands (ages for 1984 sampling date).

by over two-thirds. The gross stem production was increased from the 1975–1984 period to the 1984–2013 period (though differences in measurement methods make direct comparisons difficult), but about two-thirds of the spruce stem production occurred on trees that did not survive to 2013. Fir mortality matched fir production, leading to zero net increment for fir. Spruce mortality

Table 2
Change in density for three size classes of trees.

Stand number	Stand age (1984)	Fir			Spruce		
		5–15 cm	25–35 cm	>50 cm	5–15 cm	25–35 cm	>50 cm
Percent change 1984–2013							
1	170	−58	−50		−53	−65	−100
2	270	20	0		−38	−67	−100
3	270	23	−40	0	−12	−66	−60
4	370	−3	57		−13	−69	−82
5	370	7	−14		−27	−58	−25
6	570	−5	38		−27	−87	−100
7	570	−11	25	50	3	−75	−29
8	695	−4	−25		−32	−78	−83

was very large, averaging 0.27 kg m^{-2} yr^{-1} across the 29 years, leading to an average net increment of −0.21 kg m^{-2}.

The original chronosequence showed no evidence of major mortality events, and the pattern showed that gross stem production should equal mortality, with no net gain or loss in stem biomass with stand age. Both the 1984 and 2013 chronosequences showed the same pattern of zero slope for biomass and production, but the onset of high mortality from the spruce beetle outbreak dropped biomass by about half by 2013. The biomass chronosequence in 2013 fell outside the 95% confidence interval of the 1984 chronosequence period (Fig. 6).

4. Discussion

Aplet et al. (1988, 1989) predicted that the stable age- and size-class distributions reached by subalpine fir by the end of the third century of stand development should persist, and those predictions appear to have held up, as the fir diameter class structure remains relatively unchanged after 30 years for stands over 300 years old (Fig. 2). In stand 1, which Aplet et al., considered in the highly competitive "spruce exclusion stage," characterized by intense self-thinning, the high rates of mortality observed in small-diameter trees of both species relative to other stands is consistent with the low rates of radial growth in small-diameter trees observed in Stand 1 in 1984 (Aplet et al., 1988). The fact that diameter class distributions have not changed while more than half of the firs >30 cm have died suggests that the fir population is dynamic but stable in structure, as predicted in 1984.

Spruce, in contrast, did not meet expectations because the rapid changes from the beetle outbreak interrupted the chronic patterns of change. Aplet et al., predicted that, in the absence of disturbance, the stand initiating cohort, evident in both age-class (Aplet et al., 1988) and diameter-class (Aplet et al., 1989) distributions would continue to move through the population until spruce reached the end of its natural lifespan. Instead, the spruce beetle outbreak largely eliminated that cohort from all developing stands, regardless of stand age. The effect was to accelerate the transition from a unimodal diameter class distribution in most stands to a flatter or "inverse-j" shaped distribution predicted of old-growth forest. The exception was Stand 1, which in 1984 still had not developed

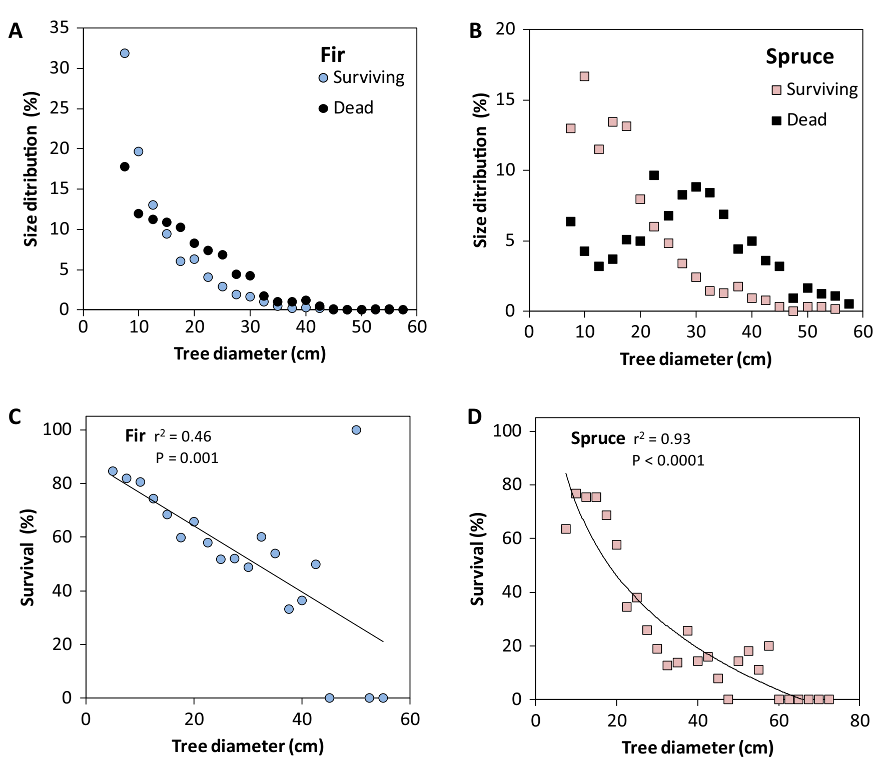

Fig. 3. Across all stands in the chronosequence, the density of trees <25 cm dbh was greater than larger trees for both species (A and B). Survival from 1984 to 2013 declines strongly with tree dbh (C and D).

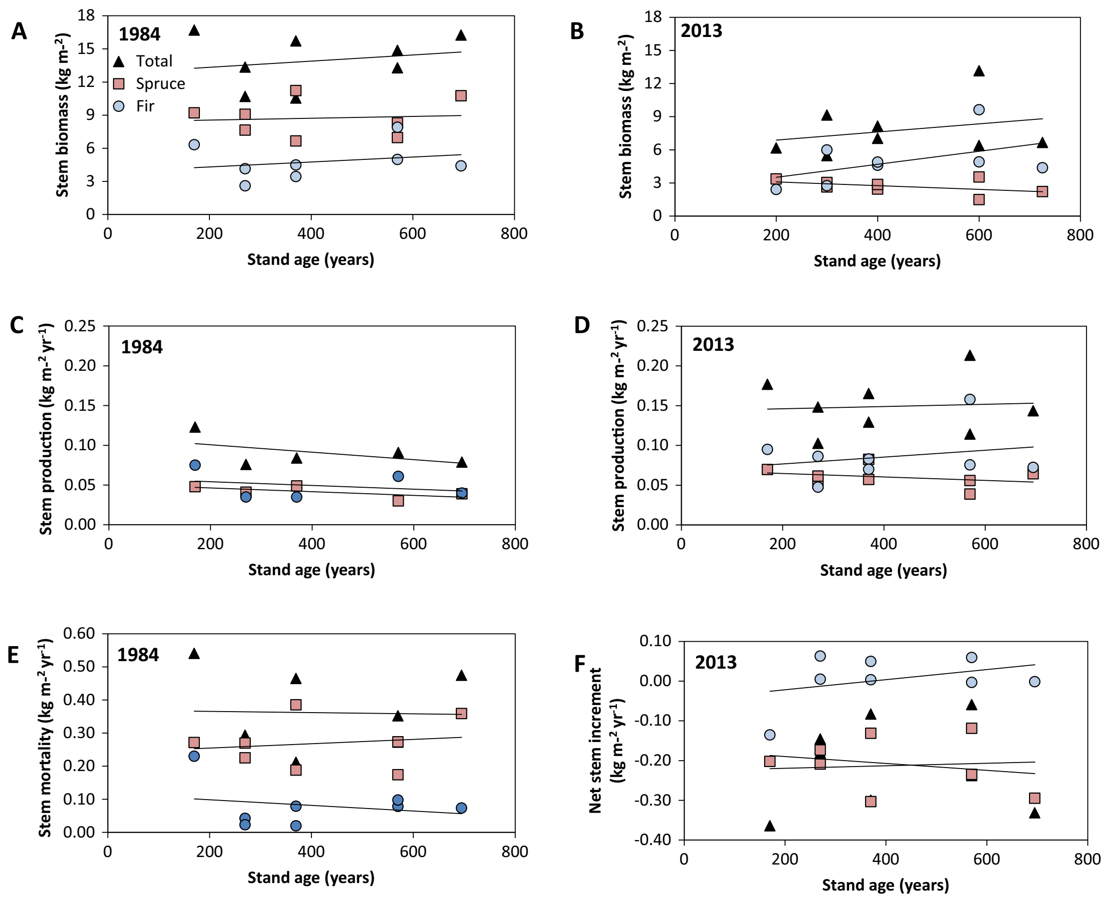

Fig. 4. The effect of stand age was not significant for stem biomass, production, mortality, or net increment for either species in the original sampling (left), or in the second sampling (right). Note that the number of observations of stemwood production in 1984 (c) is limited to five, due to the loss of production data from three stands.

a unimodal diameter class structure. It was still in the process of self-thinning of slow-growing, small-diameter trees, a process evident in the relatively high reduction in small-diameter trees observed here (Table 2). While we did not confirm the cause of death for each tree, it is likely that spruce beetle hastened the mortality that was already ongoing in this stand. In contrast, Stand 7, which exhibited high growth rates in a newly "reinitiated" cohort of small-diameter spruce in 1984 (Aplet et al., 1988), exhibited an increase in the number of small-diameter spruce by 2013, suggesting that vigorous, second generation spruce may be more resistant to bark beetles than suppressed or larger trees of the original cohort. Hart et al. (2015) similarly found smaller spruce that survived or recruited following a 1940s spruce beetle outbreak on the nearby White River Plateau were relatively unaffected by the recent regional outbreak that affected our stands.

The chronosequence examined here exhibited some behavior consistent with predictions and some behavior that was unexpected. Spatial variation across the landscape may have led to variation in patterns of stand development. Aplet (1987) observed that stands with a higher site index (Stands 3, 4, and 7) had a very brief or non-existent "spruce exclusion" phase, due to higher rates of mortality on higher sites (Franklin et al., 1987), while Stands 6 and 8 seem barely to have entered the "spruce reinitiation" stage, despite their great age. Temporal variation may have two features, with factors related to stand age combining with time-related factors such as the weather patterns for specific sets of years. Buechling and Baker (2004) and Sibold and Veblen (2006) observed a link between fire occurrence and drought in forests in this area, indicating that the environment for establishment and growth likely vary over time. Higuera et al. (2014) detected a long-term drying trend over the past 1500 years, and Smith et al. (2015) found a recent increase in drought-associated mortality in nearby forests, even in the absence of lethal beetle activity, indicating that climate may not be as stable as previously assumed in these stands. For a variety of reasons, stands may exhibit a range of behaviors within a chronosequence, including skipping a stage entirely. Still, there are insights to be gained by examining stands of different ages that exhibit consistent behaviors across stand ages, despite some differences among sites or plot histories.

Of the three factors that can confound a chronosequence – ecological site factors, climate, and rapid events – the rapid change wrought by spruce beetles had a large impact on the three-decade changes in these forests. We do not know how unusual this event was across this landscape or over a period of centuries. Hart et al. (2014) identified four periods of broad-scale spruce beetle outbreak across northwestern Colorado over the past 300 years, but most sites exhibited only one or two outbreaks, and only one, on the opposite side of Rocky Mountain National Park from our study site, exhibited as many as four. In our chronosequence, few of the very old spruce survived in 2013, which might suggest that this outbreak has been unprecedented over several centuries. However, the original selection of stands was based on the occurrence of very old trees, and it is possible that historical beetle outbreaks could have killed the majority of old spruce across the landscape, leaving only patches of old spruce to be sampled in 1984. Baker and Veblen (1990) concluded, based on observations of historical photos, that spruce beetle epidemics were widespread and common historically and "may have played a role comparable

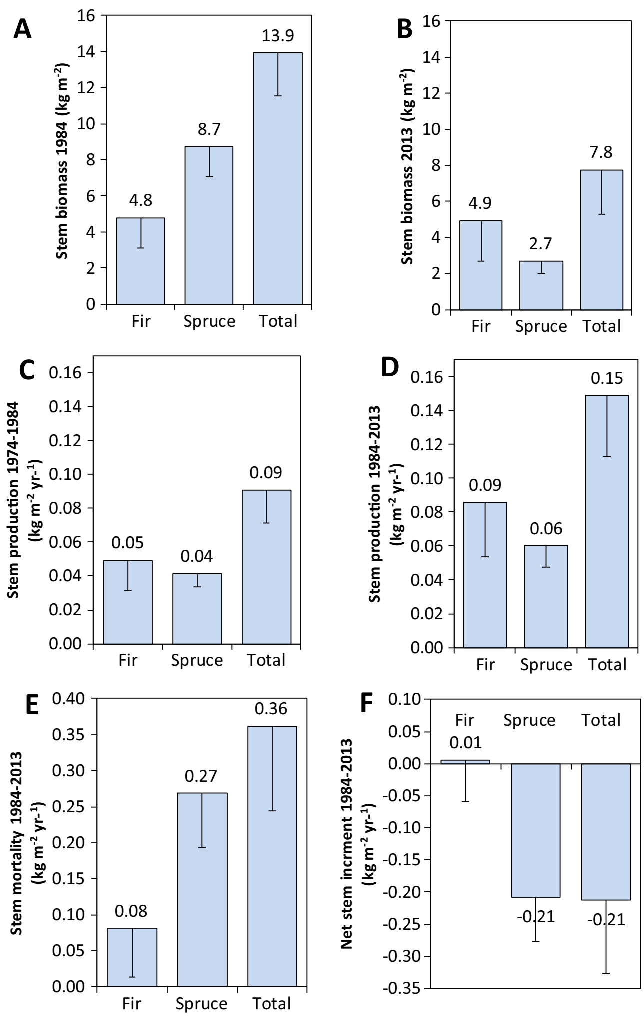

Fig. 5. Grand averages (bars are standard deviations) of all stand ages, showing the spruce biomass declined (A and B) despite sustained production (C and D), as a result of high mortality. Fir production equaled mortality, giving no change in biomass (and near-zero net increment, F).

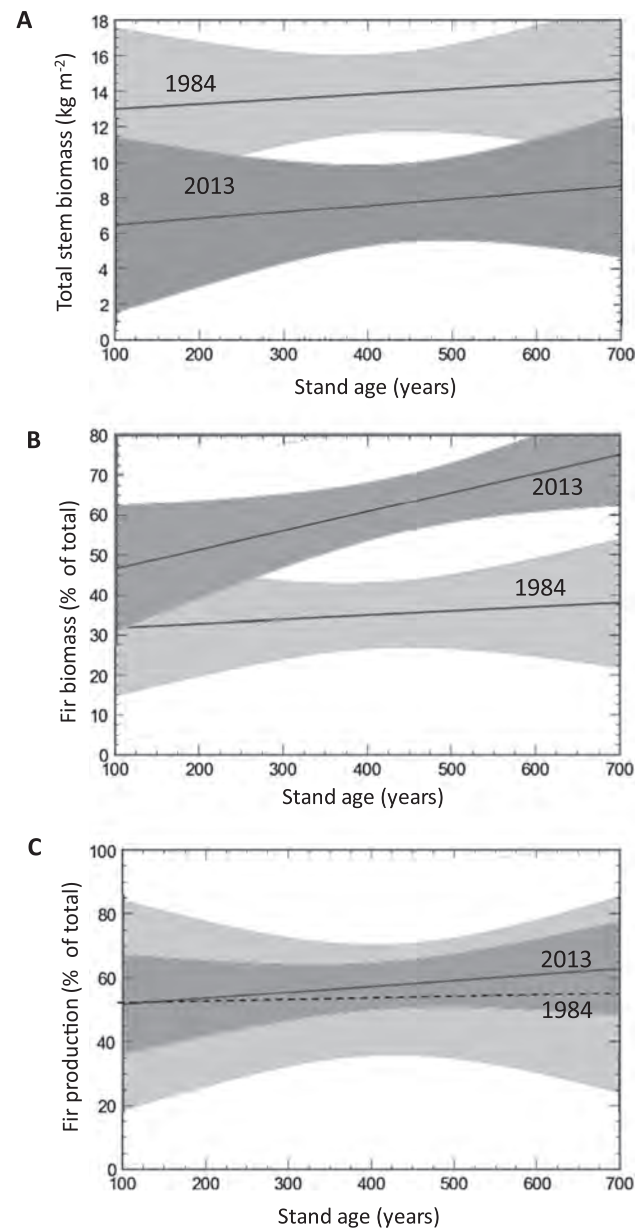

Fig. 6. Total stem biomass did not relate significantly to stand age in either period (A), but the trends for each period fell outside the 95% confidence interval of the other period. The trends in the fir composition of stem biomass also differed between periods (B), and the age-trend was significant ($r^2 = 0.57$, $P = 0.03$) for 2013. The contribution of fir to stand stem production showed no trend with age, and did not differ between periods (C).

to that of fire." They found that stands across western Colorado differed in the intensity of past effects, leaving open the possibility that much of the study landscape experienced beetle epidemics historically, and the chronosequence was anomalous. However, none of the stands exhibited evidence of abrupt growth increases in extracted cores, suggesting that widespread beetle-caused mortality had not occurred in this landscape for several centuries prior to the recent outbreak. Veblen et al. (1991) examined two stands in our study area and similarly found "no evidence of an epidemic."

Perhaps the most notable change in three decades was the shift in biomass composition, moving from two-thirds spruce in 1984 to two-thirds fir in 2013. This shift is not surprising, given the magnitude of the beetle outbreak, and closely mirrors the results reported by Veblen et al. (1991) for spruce–fir landscapes that experienced a spruce beetle outbreak in the 1940s. The shift underscores the influence of rapid-change events that are typically not included in chronosequence studies. The development of spruce forests across landscapes and regions, decades and centuries, may be heavily influenced by rapid changes of fire, windstorms, and insect outbreaks (Rebertus et al., 1992; Veblen et al., 1994; DeRose and Long, 2007; O'Connor et al., 2015). In the forests studied here, rates of stem production were on the order of 0.1 kg C m^{-2} yr^{-1} and even higher in the first century (Aplet et al., 1989), so standing live biomass in these forests equals only about a century of production, indicating that mortality is high throughout stand development. Competition-driven mortality (self-thinning) may account for substantial amounts of mortality, but the low accumulation of stem mass over multiple centuries likely indicates substantial mortality from wind and insects over the centuries. The dramatic loss of live stem mass between 1984 and 2013 might take less than a century to be replaced at current rates of stem growth, if further mortality events are not substantial.

How well would the 1984 chronosequence represent patterns across a greater range of topography? Binkley et al. (2003) sampled 18 spruce/fir stands in Rocky Mountain National Park, between 10 and 40 km south from the chronosequence plots (and about 80 m higher). The 18 forests were chosen to span the full range of aspects and topography, including upper and lower slopes and concave and convex slopes. The stem biomass of these 250- to 450-year-old stands averaged 19.9 kg m^{-2}, about 40% more than the average of the 1984 chronosequence plots. Fir biomass dominated the 18 stands (55% of stand total), in contrast to spruce dominance on the 1984 chronosequence (65% of stand total). Stem production was also about 60% greater for the 18 sites spread across the topographic gradients, although fir averaged a consistent 56% of total stem production in both cases.

Across these gradients of space and time, what trends might be expected to be strong? Spruce and fir clearly tend to persist with neither species fully displacing the other. Veblen (1986b) suggested that coexistence is maintained in equilibrium in stand over 250 years by differences in fecundity and mortality rates between the two species. In contrast, Aplet et al. (1988) posited a non-equilibrium model rooted in the interplay of population dynamics traceable to the stand-initiating disturbance. Even if differences in fecundity, tolerance, and lifespans did produce a tendency toward competitive exclusion, varying factors are likely to introduce enough randomness that near-complete dominance by either species is unlikely. As Lertzman et al. (1998) found in trying to classify fire regimes, heterogeneity in the timing, intensity, and spatial distribution of disturbances challenges our ability to describe a "typical" course of forest stand development in this type.

When appropriately applied, chronosequences offer invaluable insights into the nature of vegetation change, but they have their limitations (Walker et al., 2010). Some studies have assessed the validity of chronosequence designs by resampling the same ecosystems at later dates, finding that minor and even major trends from the chronosequences were not supported by remeasurements. Johnson and Miyanishi (2008) examined classic chronosequences on sand dunes, across landscapes as ponds fill in and become forests, forest development following deglaciation, and forest establishment on abandoned agricultural fields. They concluded that each of these textbook examples presented incorrect predictions of ecosystem change, and that the chronosequence approach was invalid. Hollingsworth et al. (2010) resampled permanent plots in the classic riverside-terrace age sequence along the Tanana River, and found that spruce stands of different ages seem to follow independent trajectories rather than a common trend, and nitrogen-fixing alder (*Alnus tenuifolia*) increased across the whole age sequence rather than remaining restricted to the young stages as expected from the original chronosequence. Some of the deviations from the chronosequence trends were driven by herbivory (beavers, snowshoe hares) and weather (drought and snow damage to trees).

More recently, though Harmon and Pabst (2015) tested the classic chronosequence of developing Douglas-fir forest in the Pacific Northwest by following three 0.4 ha plots over one hundred years of development and found that population and community behavior were consistent with predictions from chronosequences but that, similar to our study, ecosystem-level trends were not. Rather than the convex flattening of biomass accumulation predicted by theory, they found biomass to increase linearly, a phenomenon they attributed to high heterogeneity of mortality at the plot scale. They conclude, "[T]esting hypotheses developed from chronosequences and other reconstructions with long-running temporal observations ... is a necessary step."

In the ecology and management of forests, the value of chronosequences and resampling of individual stands depends in large part on whether forests actually follow a common trajectory over time. Rates of current processes such as stand growth may be strongly related to stand age, and clear patterns may show up in both chronosequences and in re-sampling of individual stands (e.g., Ryan et al., 1997). Forest features that reflect the long-term legacies of contingent events, such as localized windstorms or population irruptions of seedling–browsing herbivores, may not show predictable trends over time (e.g., Bernadzki et al., 1998). Walker et al. (2010) conclude that chronosequences are best suited for the study of vegetation development where trajectories are convergent, have low species diversity, and are infrequently disturbed, all of which appear to apply here. Johnson and Miyanishi (2008) exhort all authors employing chronosequences to provide "strong tests" of critical assumptions. The consistency of pattern in age and size distributions (Aplet et al., 1988, 1989) and the lack of evidence of past disturbances in the tree ring record for this area (Veblen et al., 1991) indicate that disturbances of the scope and scale of the recent beetle outbreak were rare historically. The species diversity of this developmental sequence (spruce and fir only) is obviously low, and the convergence toward an all-age, old-growth structure suggest that this chronosequence meets Walker et al.'s criteria and Johnson and Miyanishi's test.

The fundamental influence of historical legacies challenges expectations that forests should show stable patterns over large areas and spans of time. If forests should not be expected to follow a single predominant trajectory, then both chronosequence and resampling approaches should not be expected to produce clear patterns across stand ages. At the scale of multiple stands and landscapes, the central tendencies from chronosequences and resampling of individual stands may have too much variation to be very useful. Clearly, the chronosequence studied here no longer meets the requirement of a common disturbance history, as the spruce beetle epidemic affected each stand at a different point in its development. Still, there is value in continuing to study these plots, as they may reveal the effect of a regional epidemic on stands at different stages of development. The future development of a post-beetle stand may depend on its pre-beetle structure, and some stands appear to have fared better than others despite heavy mortality across stands. Alternatively, the apparent conversion of all stands to an all-age, old-growth structure may confer future resilience to disturbances, like bark beetle outbreaks, that preferentially target one size or age class over another.

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Forest Roads: A Synthesis of Scientific Information

Editors

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Forest Roads: A Synthesis of Scientific Information

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Abstract

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Effects of roads in forested ecosystems span direct physical and ecological ones (such as geomorphic and hydrologic effects), indirect and landscape level ones (such as effects on aquatic habitat, terrestrial vertebrates, and biodiversity conservation), and socioeconomic ones (such as passive-use value, economic effects on development and range management). Road effects take place in the contexts of environmental settings, their history, and the state of engineering practices, and must be evaluated in those contexts for best management approaches.

Keywords: Roads, roadless areas, forest ecosystems, geomorphology, hydrology, habitat fragmentation, biodiversity, nonmarket values, heritage values, economic development, grazing, mineral resources, fire.

Summary

Roads are a vital component of civilization. They provide access for people to study, enjoy, and commune with forested wildlands and to extract an array of resources from natural and modified ecosystems. Roads have well-documented, short- and long-term effects on the environment that have become highly controversial, because of the value society now places on unroaded wildlands and because of wilderness conflicts with resource extraction.

The approach taken in this report is to identify known and hypothesized road-related issues and to summarize the scientific information available about them. The report identifies links among processes and effects that suggest both potential compatible uses and potential problems and risks. Generalizations are made where appropriate, but roads issues and road science usually cannot be effectively separated from the specific ecologic, economic, social, and public lands management contexts in which roads exist or are proposed.

General Consideration of Road Networks at Intermediate and Large Scales

Across a forest or river basin, the access needs, economic dependencies, landscape sensitivities, downstream beneficial uses of water, and so on can be reasonably well defined, but these relations tend to differ greatly from place to place. An effective synthesis of road issues draws local experts together to thoroughly evaluate road and access benefits, problems and risks, and to inform managers about what roads may be needed, for how long, for what purposes, and at what benefits and costs to the agency and society.

Road effects and uses may be somewhat arbitrarily divided into beneficial and detrimental. The largest group of beneficial variables relates to access. We identified access-related benefits as harvest of timber and special forest products, grazing, mining, recreation, fire control, land management, research and monitoring, access to private inholdings, restoration, local community critical needs, subsistence, and the cultural value of the roads themselves. Nonaccess-related benefits include edge habitat, fire breaks, absence of economic alternatives for land management, and jobs associated with building and maintaining the roads.

Undesirable consequences include adverse effects on hydrology and geomorphic features (such as debris slides and sedimentation), habitat fragmentation, predation, road kill, invasion by exotic species, dispersal of pathogens, degraded water quality and chemical contamination, degraded aquatic habitat, use conflicts, destructive human actions (for example, trash dumping, illegal hunting, fires), lost solitude, depressed local economies, loss of soil productivity, and decline in biodiversity.

For each variable, we sought expert assistance from scientists actively engaged in research related to roads and asked them for information, with emphases on results and conciseness rather than exhaustive descriptions, in the following categories: issues relevant to their topic; science findings; an assessment of reliability, confidence, and limitations inherent in the data; the degree to which the information could be generalized to larger geographic scales than in the original research; secondary links for each topic to other topics; and the ability of the existing knowledge to address the issues raised.

Road development histories crucial to understanding their effects—All roads were not created equal and do not behave the same. Road networks differ greatly in how they developed through time and how they were laid out over terrain; they carry this history into their present performance. The geographic patterns of roads in forest landscapes differ substantially from place to place, with commensurate differences in environmental effects. For example, ridgetop, midslope, and valley floor roads all behave

differently, based on the topography they cross, the degree and type of interaction with stream networks, their stability in and response to storms, and their effects on wildfire, wildlife, and vegetation. Distinguishing among the effects of building, maintaining, using, decommissioning, or abandoning roads is crucial because each of these actions affects the environment in many ways.

Knowledge of the state of road systems in national forests is inadequate—We currently lack sufficient information to develop a comprehensive history of the building and maintaining of national forest roads or their current condition. The inventories of the roads differ widely, in both content and status, and frequently lack sufficient information to define benefits, problems, and risks.

Roads create interfaces and ecotones—Roads are long, which creates large amounts of interface within the landscapes traversed. The strength of the interactions at these interfaces differs with time and space; it is controlled by the contrast between adjacent resource patches or ecological units. These interfaces may regulate the flow of energy and materials between adjacent systems. Such sites are sensitive. They have relatively high biodiversity, affect critical habitat for rare and endangered species, and serve as refuges and source areas for pests and predators.

Road management involves important tradeoffs—Almost all roads present benefits, problems, and risks, though these effects differ greatly in degree. Roads provide motorized access, which creates a broad spectrum of options for management but forecloses other options, such as nonmotorized recreation or wildlife refugia. Even a well-designed road system inevitably creates a set of changes to the local landscape, and some values are lost as others are gained; for example, road density and fish populations correlate negatively over a large area in the interior Columbia basin. The basin's environmental assessment shows that subbasins with the highest forest-integrity index were largely unroaded, and subbasins with the lowest integrity had relatively high proportions of moderate or greater road density. In general, greater short- and long-term watershed and ecological risks are associated with building roads into unroaded areas than with upgrading, maintaining, closing, or obliterating existing roads.

Confounding variables are difficult to separate from road-related ones—Changes in the habitat of terrestrial vertebrates, frequency of road kill, and transmission of forest diseases result from road use, not from the presence of the road itself. Separating effects of roads from other landscape and ecological modifications that result from changes in land use that roads enable is often impossible.

Direct Physical and Ecological Effects

Geomorphic effects of roads range from chronic and long-term contributions of fine sediment into streams to catastrophic mass failures of road cuts and fills during large storms. Roads may alter channel morphology directly or may modify channel flow and extend the drainage network into previously unchanneled portions of the hillslope. The magnitude of road-related geomorphic effects differs with climate, geology, road age, construction practices, and storm history. Improvements in designing, constructing, and maintaining roads can reduce road-related erosion at the scale of individual road segments, but few studies have evaluated long-term and watershed-scale changes to sediment yields when roads are abandoned or obliterated.

Roads have three primary effects on **hydrologic processes**: (1) they intercept rainfall directly on the road surface and road cutbanks and affect subsurface water moving down the hillslope; (2) they concentrate flow, either on the surface or in an adjacent ditch or channel; and (3) they divert or reroute water from paths it otherwise would take were the road not present. Problems of road drainage and transport of water and debris—especially during floods—are primary reasons roads fail, often with major structural, ecologic, economic, or other social consequences. The effect of roads on peak streamflow depends strongly on the size of the watershed; for example, capture and rerouting of water can remove water from one small stream while causing major channel adjustments in another stream receiving the additional water. In large watersheds, roads constitute a small proportion of the land surface and have relatively insignificant effects on peak flow. Roads do not seem to change annual water yields, and no studies have evaluated their effect on low flows.

Forest roads can significantly affect **site productivity** by removing and displacing topsoil, altering soil properties, changing microclimate, and accelerating erosion. The direct effect of roads on soil productivity is estimated to range from 1 to 30 percent of the landscape area in managed forest lands. Losses of productivity associated with road-caused accelerated erosion are site specific and highly variable in extent.

Natural populations of animal species are affected by **habitat fragmentation** caused by the presence of roads and by avoidance of areas near roads by some species and attractiveness to those areas by other species. Fragmented populations can produce increased demographic fluctuation, inbreeding, loss of genetic variability, and local extinctions. Roads fragment habitat by changing landscape structure, dissecting vegetation patches, increasing the amount of edge, decreasing interior area, and increasing the uniformity of patch characteristics. Road-avoidance behavior is characteristic of large mammals such as elk, bighorn sheep, grizzly bear, caribou, and wolf. Some studies have shown that the existence of a few large areas of low road density, even in a landscape of high average road density, may be the best indicator of suitable habitat for large vertebrates.

On the other hand, roads and their adjacent environment qualify as a distinct **habitat** and result in changes at the species, population, and landscape scales. Some species are associated with edges, including those that use roads as corridors to find food. Roads facilitate **biological invasion** in that disturbed roadside habitats are invaded by exotic (non-native) plant and animal species dispersed by wind, water, vehicles, and other human activities. Roads may be the first points of entry for exotic species into a new landscape, and the road can serve as a corridor for plants and animals moving farther into the landscape. Invasion by exotic species may have significant biological and ecological effects if those species are able to displace natives or disrupt the structure and function of an ecosystem.

Indirect and Landscape-Scale Effects

The effects of roads on **aquatic habitat** are believed to be widespread, although direct, quantitative cause-effect links are difficult to document. At the landscape scale, correlative evidence suggests that roads are likely to influence the frequency, timing, and magnitude of disturbance to aquatic habitat. Increased fine-sediment composition in stream gravel—a common consequence of road-derived sediments entering streams—has been linked to decreased fry emergence, decreased juvenile densities, loss of winter carrying capacity, and increased predation of fishes and can reduce benthic organism populations and algal production. Roads can act as barriers to migration, lead to water temperature changes, and alter streamflow regimes. Improper culvert placement where roads and streams cross can limit or eliminate fish passage. Roads greatly increase the frequency of landslides, debris flow, and other mass movement. At the landscape scale, increasing road densities and their attendant effects are correlated with declines in the status of some non-anadromous salmonid species.

Roads can cause a wide variety of effects to **terrestrial wildlife**. Some species, such as gray wolf and grizzly bear, are adversely affected by repeated encounters with people. Roads can increase harassment, poaching, collisions with vehicles, and displacement of terrestrial vertebrates, which affect many large mammals such as caribou, bighorn sheep, mountain goat, pronghorn antelope, grizzly bear, and gray wolf. It is estimated that 1 million vertebrates are killed annually on roads in the United States. Direct mortality of large mammals on forest roads is usually low, except for those with a home range straddling a road. Forest roads pose a greater hazard to slow-moving migratory amphibians than to mammals. Nearly all species of reptiles seek roads for cooling and heating. Vehicles kill many of them, making well-used roads a population sink.

Chemicals applied to and adjacent to roads can enter streams by various pathways. The effect on **water quality** depends on how much chemical is applied, the proximity of the road to a stream, and the weather and runoff events that move chemicals and ediments. Dust produced by vehicles moving on unpaved roads reduces **visibility** and generates airborne particulates that can pose health hazards, such as in areas with soils containing asbestiform minerals.

Direct Socioeconomic Effects

A variety of products harvested from forests are being transformed into medicinals, botanicals, decoratives, natural foods, and other products, called **nontimber** or **special forest products.** The harvest of these products usually depends on road access. The Forest Service is required by law to permit access to **private inholdings** but can require the owners to comply with standards that apply to building roads on or through national forest land.

Economic pressures affect roads and road use, and roads have multiple economic consequences. Both **benefits and costs** are associated with building, maintaining, and using forest roads. The **economic** effects relate to forest access and user-communities, including loggers, silviculturists, fuels managers, and recreationists. The network of roads on national forest lands has both positive and negative effects on most Forest Service land management programs. Reducing road densities could result in increased timber-harvesting costs, for example. Roads have replaced stock drives for transporting sheep and cattle to and from mountain grazing allotments. Road-related issues associated with **energy and mineral resources** are access rights, property rights, and benefits and detrimental effects. **Public recreational users** of national forests depend on roads for access. Altering the road networks will affect such uses differently across the landscape.

Indirect Socioeconomic Effects

The increasing density of roads in and adjacent to many forest, shrub, and rangeland areas is an important factor in the changing patterns of disturbance by **fire** on the landscape. Roads provide access that increases the scale and efficiency of **fire suppression**, and roads create linear firebreaks that affect fire spread. The benefits roads provide for fire prevention and fire management carries an associated cost: increased access has increased the role of human-caused ignitions. And road networks have resulted in changes in fuel patterns and fire regimes at the broad scale.

Roads also affect many less measurable attributes of the national forests, including **passive-use values:** those values that people hold for things they may not expect to use themselves but that they believe should exist for future generations. For example, building roads in roadless areas may reduce passive-use value significantly; decommissioning of roads may increase such value. But decommissioning of roads also is likely to reduce active-use values. Roads themselves sometimes have **heritage value** because of historical or cultural significance.

The aim of this synthesis is to focus on the scientific information about the benefits, uses, and physical and biological effects of forest roads. Because all aspects of roads in forests have become of great interest to the American public, research is underway in many domains. This document represents the information available as of the date of publication.

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Contents

Introduction and Objectives

Roads have become vital components of the human use of forested systems. Without roads, development of the economic activity critical to the quality of modern life would have been difficult, and roads remain central to many forest uses today. Roads provide access for people to study, enjoy, contemplate, or extract resources from natural and modified ecosystems. Building and maintaining roads is controversial, however, because of the kinds of uses they enable, concerns about their short- and long-term effects on the environment, and the value that society now places on unroaded wilderness (Cole and Landres 1996, Williams 1998).

Decisions about roads—locating, building, maintaining, and decommissioning them—are complex because of the many tradeoffs required. The statement by Chomitz and Gray (1996) that “rural roads promote economic development, but they also facilitate deforestation” exemplifies recent experiences. And a tradeoff exists between access by roads for recreation and resource extraction with the potential effects of that access on biodiversity. Roads have been evaluated from physical, biological, and socioeconomic points of view, often under only one perspective in isolation from the others. Such an approach is useful for identifying issues, but it can lead to conflict and poorly informed policy choices because it may unnecessarily play one set of values against another. For example, a road justified only by economic criteria at the expense of ecological ones—or vice versa—is likely to be questioned by advocates of the missing criteria. A unified approach to analyze building, maintaining, or decommissioning roads is needed to allocate resources wisely. This report represents our attempt to summarize the known desirable properties of roads and their known effects on the landscape, based on the scientific information currently available.

The approach taken was to enumerate the known or hypothesized issues and then provide a summary of the scientific information available about those issues. We provide a synthesis that attempts to reveal where links between processes and effects suggest both potential compatible uses and potential problems and risks.

We find that roads cannot be separated from the ecologic, economic, social, or public land management context in which they exist or are proposed. A virtually limitless variety of context factors renders any single, generalized synthesis to be of limited applicability and value. An effective synthesis of all the interactions of roads, the environment, and people can best be attempted by looking at road systems in actual places where the myriad effects of roads are not hypothesized or generalized. For example, across a national forest or river basin, the array of access needs, economic dependencies, landscape sensitivities, downstream beneficial uses of waters, and so on can be reasonably well defined and will tend to differ greatly from any other place. A synthesis of the effects of roads in a specific context can be attempted by drawing local experts together to thoroughly evaluate road and access benefits, problems, and risks, to inform managers about what roads may be needed, for how long, for what purposes, and at what costs to the agency and society.

The Forest Service recently published a document *Roads Analysis: Informing Decisions About Managing the National Forest Transportation System* (USDA FS 1999), which can be considered a specific application of watershed analysis or a cumulative effects analysis, wherein the principal objective is to focus on road effects. For example, roads analysis and watershed analysis have common steps that include:

- Setting up the analysis
- Describing the situation

- Identifying issues
- Assessing benefits, problems, and risks
- Describing opportunities and setting priorities
- Reporting results and conclusions

Similar approaches to watershed analysis or cumulative effects analysis are being adopted widely by federal (for example, Regional Ecosystem Office [REO] 1995), state (for example, Washington Forest Practices Board 1995), and private (for example, NCASI 1992) agencies and organizations. The exact steps and organization of the analysis are somewhat modified by each application, but the conceptual framework is similar. The focus of each analysis can change, depending on the principal reason for doing it (such as timber production, wildlife, or ecosystem integrity); for example, an analysis focused on timber production in a watershed or region would look at effects on and of road development, water quality, wildlife, recreation, and economics. Exactly the same set of issues would emerge if the focus were on water quality, wildlife, or recreation. The perspective and conclusions might be different, but the issues and approach would be the same.

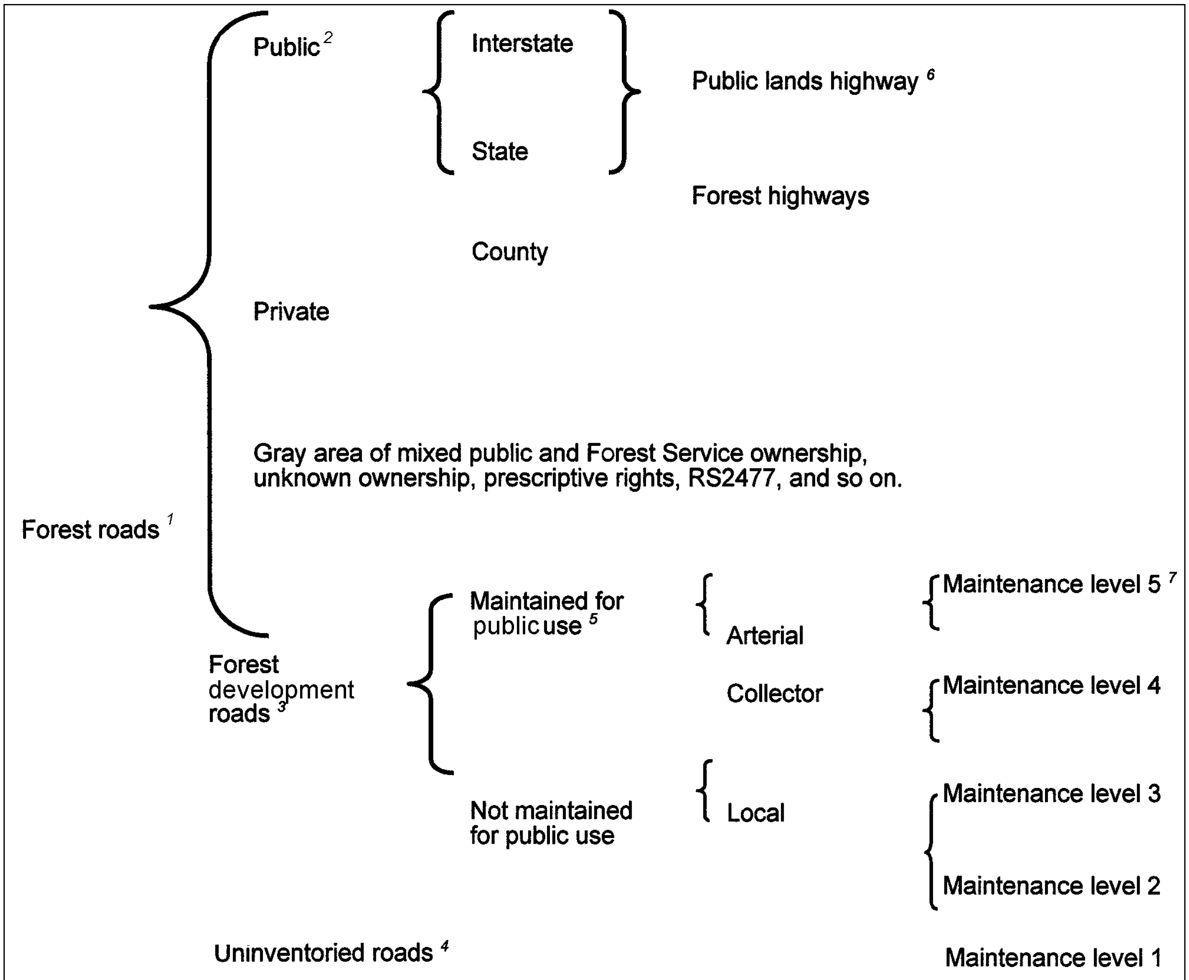
The roads analysis (USDA FS 1999) is intended to be an integrated, ecological, social, and economic approach to transportation planning. It uses a multiscale approach to ensure that the identified issues are examined in context, and it is based on science. Analysts are expected to locate, correctly interpret, and use relevant existing scientific literature in the analysis, disclose any assumptions made during the analysis, and reveal the limitations of the information on which the analysis is based. The analysis methods and the report are to be subjected to critical technical review.

This science synthesis complements the roads analysis by summarizing some of the available scientific information on how roads affect an array of ecological, social, and economic resources. The approach used in this document is mostly reductionist; it is not intended to be a comprehensive encyclopedia of all available knowledge about road effects; but this information, together with the extensive list of questions posed in the roads analysis, should assist interdisciplinary teams in understanding and applying the best available science appropriately to existing and potential road systems in specific geographic contexts, across the national forest system. Commonly used definitions for Forest Service roads are listed in figure 1.

General Considerations of Roads Networks

In this section, we consider what the body of scientific work on roads allows us to understand about how roads function in the landscape. This paper details specific positive and negative consequences of roads; here, we attempt to distill this information into key observations relevant to road policy considerations. The work is a synthesis of a large body of information from many sources. Inevitably, the synthesis creates potential for interpretations beyond the more generally accepted facts about roads contained in the rest of the document. Nevertheless, we believe they represent a reasonable set of principles consistent with the best scientific knowledge.

Road effects and uses may—somewhat artificially—be divided into beneficial and deleterious effects. In the former category, most variables relate to access, with a second group of beneficial uses not related to access. We identified the following access-related benefits or needs: timber acquisition, grazing, mining, recreation, fire control, land management, research and monitoring, access to private inholdings, restoration, community critical needs, subsistence, and the cultural value of the roads themselves. Non-access-related benefits or needs included edge habitat, fire breaks,



[1] **Forest roads:** Roads wholly or partially within, or next to, and serving a national forest and needed to protect, administer, and use the national forest and to use and develop its resources.
[2] **Public roads:** Roads under the jurisdiction of, and maintained by, a public authority that are open to public travel
[3] **Forest development roads:** Forest roads under the jurisdiction of the Forest Service.
[4] **Uninventoried roads:** Short-term roads associated with fire suppression; oil, gas, or mineral exploration or development; or timber harvest not intended for forest-development transportation and not necessary for resource management. Regulations require revegetation within 10 years.
[5] **Maintained for public use:** Forest development roads open to unrestricted use by the general public in standard passenger cars, including those roads closed seasonally or for emergencies.
[6] **Public lands highways, forest highways:** A coordinated Federal Lands Highway Program includes forest highways, public lands highways, park roads, parkways, and Native American reservation roads under the jurisdiction of and maintenance by a public road authority other than the Forest Service and open to public travel.
[7] **Maintenance level 5:** Roads that provide a high degree of user comfort and convenience. Normally double lane, paved facilities, or aggregate surface with dust abatement; the highest standard of maintenance.
Maintenance level 4: Roads that provide moderate user comfort and convenience at moderate speeds. Most are double lane, and aggregate surfaced. Some may be single lane. Some may be dust abated.
Maintenance level 3: Roads open and maintained for travel by a prudent driver in a standard passenger car. User comfort and convenience are not considered priorities. Typically low speed, single lane with turnouts and native or aggregate surfacing.
Maintenance level 2: Roads open for use by high-clearance vehicles. Passenger car traffic is discouraged. Traffic is minor administrative, permitted, or dispersed recreation. Nontraffic-generated maintenance is minimal.
Maintenance level 1: These roads are closed. Some intermittent use may be authorized. When closed, they must have barricades, berms, gates, or other closure devices. Closures must exceed 1 year. When open, a road may be maintained at any other level. When closed to vehicular traffic, they may be suitable and used for nonmotorized uses, with custodial maintenance.

Figure 1—Legal basis and definitions for roads in the national forests.

the absence of economic alternatives for land management, some positive effects on water quality, and the jobs associated with building and maintaining these systems. This analysis uncovered factors that could lessen negative effects of roads by better integrating engineering approaches with knowledge of road effects.

Negative consequences include effects on hydrology, geomorphic features such as debris slides, sedimentation, habitat fragmentation, predation, road kill, invasion by exotic species, dispersal of pathogens, water quality such as chemical contamination, aquatic habitat, use conflicts, human actions (for example, trash dumping, illegal hunting, fires), the cost of lost solitude, local economies, soil productivity, communities, and biodiversity.

For each variable, we sought expert assistance from scientists actively engaged in research related to roads and asked for information in the following categories, with emphases on results and conciseness rather than exhaustive descriptions: issues relevant to the topic variable; science findings; an assessment of the reliability, confidence, and limitations inherent in the data; the degree to which the information could be generalized to larger geographic scales than those of the original research; the secondary links from this topic to other topics; and the ability of the existing knowledge to address the issues raised.

We note that the limitations of science set the bounds for subsequent interpretations, we offer a synthesis of the available scientific information, and we consider how these science-based observations might be used in developing future road policy.

Despite the shortcomings described, we believe that the available science on road effects can provide considerable guidance in evaluating benefits and costs associated with roads. Our interpretation of the scientific literature leads to the following observations.

Roads differ greatly—All roads are not created equal and do not behave the same. Road networks differ greatly in development through time and layout over terrain, and they carry this history into present performance. In many parts of the National Forest System, the major roads were built in the 1950s and 1960s, with secondary and tertiary feeder roads following as the road networks expanded into watersheds. In other areas, logging roads developed from previous road systems used for mining in the Rocky Mountain and Southwestern states or agriculture in the southern Appalachians, Ozarks, and New England. Thus, changes in road standards through time (for example, width, construction methods, position in the landscape) have affected different parts of road networks. Consequently, each road network commonly contains a collection of old and new types and standards of roads designed for various purposes that cross terrain of differing sensitivities. This mosaic of road segments has implications for how roads will be managed in the future (Gullison and Hardner 1993).

The geographic patterns of roads in forest landscapes differ substantially from place to place, with commensurate differences in environmental effects. In the glaciated terrain of southeastern Alaska, for example, main roads were built on the broad, major valley floors, and the high-value timber that grew on lower hillslopes was brought downhill to them. In forests along the west side of the Sierra Nevada in California, on the other hand, major roads were built along broad ridges, with secondary roads leading down into headwater areas. The main roads into western Oregon forests entered watersheds along narrow stream bottoms and then climbed the adjacent steep, unstable hillslopes to access timber extending from ridge to valley floor. These configurations, combined with local geology and climate, resulted in very different effects of roads on watershed, wildlife, vegetation, recreation, and disturbance processes.

Even in the same region, road effects differ by landscape position. Ridgetop, midslope, and valley floor roads all produce different effects, based on the topography they cross, the degree and type of interaction with stream networks, the stability and response to storms, and the effects on fire, wildlife, and vegetation.

Different phases of road development have different effects on the landscape. Distinguishing among the effects of building, maintaining, usage, decommissioning, or abandoning of roads is crucial because they usually affect the environment in several ways.

Road development history crucial to understanding effects—The effects of roads differ over time. Some effects are immediately apparent (such as loss of solitude or creation of edge), but others may require an external event, such as a large storm, to become visible (such as road-related erosion or mass movement). Still other effects may be subtle, such as increased susceptibility to invasion by exotics, pathogens noticed only when they become widespread in the landscape, or increased road use as recreation styles and motor vehicles change.

With time, roads often adjust to the ecosystems they are embedded in. Some segments blend with the landscape and reach a new ecological and hydrological balance, or better, a metastable state. Such a state will be different for a road transecting old-growth forest than for a road in an otherwise highly disturbed landscape. A critical issue in the decommissioning of a road is whether disrupting the new environmental balance created by the presence and aging of the road is desirable. As other segments of the road age, however, some features (such as culverts and disrupted subsurface drainage paths) become increasingly unstable; the probability of failure increases with road age. Sometimes, decommissioning a road can have significant environmental effects because the road has become part of the evolving landscape.

Decommissioning also can avert significant future environmental effects of the road. One last precaution in generalizing about the environmental effects of roads is to determine the age and condition of the road and evaluate the degree of landscape adjustment to the road and vice versa. Roads produce long-term legacies on the landscape. Many roads built by the Roman Empire centuries ago have disappeared from the landscape, but their legacies remain in the sediment layers of Italian lakes (Hutchinson 1973) and in strips of unique vegetation growing on limestone soils (derived from the limestone slabs used to build the road) in landscapes of acid podzolic soils (Detwyler 1971). In Lago di Montesori, Italy, the building and use of Via Cassia resulted in a pulse of eutrophication that lasted 2,000 years before it abated when the road was abandoned (Hutchinson 1973). Strips of fern populations in the Caribbean National and Luquillo Experimental Forests in Puerto Rico, serve as indicators of the skid trails abandoned more than six decades ago in these wet forests (Garcia-Montiel and Scatena 1994). These legacies are useful in historical reconstruction of landscapes because they help to explain the relevance of yesterday’s activities to today’s landscapes (Burel and Baudry 1990). In the process, more is learned about ecosystem resilience and how ecosystems continuously adjust to change.

We do not currently have sufficient information to develop a comprehensive picture of the construction or maintenance history or the current condition of the roads comprising our national forest road networks. Although much information on roads exists at a variety of scales (district, forest, region), and some national forests have invested in inventorying and developing road databases, no common framework or database exists for accessing road development information. For environmental consequences, little information exists on old, abandoned roads that still pose risks of failure. Other data important to defining effects, such as the location and configurations of road-stream crossings, are not available for most places. Without such a database, developing a comprehensive picture of where the road system currently stands, what parts of it need work, and where restoration activities should be focused will be difficult and analyses may be limited at best.

Knowledge of the State of Road Systems in National Forests Is Inadequate

Road inventories for the national forests are highly variable, frequently incomplete or inaccurate, and lack information needed to define benefits, problems, and risks. For most national forests, the inventory contains very limited, transportation-related data, such as road maintenance level and surface type. These data, though useful for some purposes, may be wholly inadequate to address such considerations as sedimentation hazards, migration barriers, landslide potential, road-stream connectivity, or other important aspects of the environmental effects of roads. Other useful data may exist in various forms, but because they are not systematically collected or maintained, they are nearly impossible to access for analysis. Without suitable data, some important aspects of the analysis of roads cannot proceed.

Roads create interfaces and ecotones—Because roads have great length, the interface surface between roads and the ecosystems of the landscape traversed is maximized. Naiman and Décamps (1997) recognized that the strength of the interactions at these interfaces differs with time and space, and it is controlled by the contrast between adjacent resource patches or ecological units. They compare these interfaces to semipermeable membranes regulating the flow of energy and materials between adjacent systems. They note that interfaces “have resources, control energy and material flux, are potentially sensitive sites for interactions between biological populations and their controlling variables, have relatively high biodiversity, maintain critical habitat for rare and endangered species, and are refuge and source area for pests and predators.” The road interface may be split into two zones (roadside and ecotone) to highlight the difference between vegetation along the roadside and vegetation in the zone at the interface of the road. That interface can be sharp or gradual and form an ecotone that differs from both the roadside and the adjacent natural ecosystem.

The width of the surface of a road differs from the width of its ecological influence (Auerbach and others 1997; Forman, in press; Forman and others 1997; Larsen and Parks 1997; Reck and Kaule 1993). For example, a road may be 30 feet wide, but it may influence an additional 80 feet of adjacent land because of disturbance during construction and the buffer zone for the pavement, making the road effectively 110 feet wide. That same road has an ecological influence that can extend an additional unknown distance from storm water runoff—influence over the home range of wildlife, geomorphic alterations upstream and downstream, distance its noise and dust carry, and views it provides.

Road management usually involves important tradeoffs—Almost all roads present benefits, problems, and risks, though these effects differ greatly in degree. Roads permit motorized access, which creates a broad spectrum of options for management but forecloses other options, such as wilderness, nonmotorized recreation, or some types of wildlife refugia. Even a well-designed and well-built road system inevitably creates a set of changes to the local landscape, and some values are lost as others are gained.

Tradeoffs accompany specific decisions about roads, such as construction method. Full-bench road construction, for example, may decrease the risk of fill slope failure, but it also may increase the potential for groundwater interception with attendant water quality risks.

In public wildlands management, road systems are the largest human investment and the feature most damaging to the environment. Thus the choices about what roads are needed, for what purposes, for how long, and at what cost—to public ecological resources as well as financial—are critical decisions in managing public lands.

Recent Efforts at Describing Roads in the Landscape May Be Helpful

Roads can be thought of as ecosystems—Synthesis of the effects of roads on terrestrial ecosystems may be facilitated by viewing roads as "techno-ecosystems," as recently described by Lugo and Gucinski (2000). Roads occupy ecological space (Hall and others 1992), have structure, support a specialized biota, exchange matter and energy with other ecosystems, and experience temporal change. Road "ecosystems" are built and maintained by people (techno-ecosystems; Haber 1990) and are characterized by open fluxes of energy and matter and a predominance of respiration over photosynthesis; that is, they are heterotrophic and highly subsidized systems. To appreciate that features associated with roads function as an ecosystem and interact with the surrounding forests requires thinking about the flow of materials, energy, and organisms along road corridors, vegetation zonation, the interaction with the human economy and human activity, and the external forces that converge on the road corridor (Donovan and others 1997; Forman 1995a, 1995b). (See fig. 2).

Roads connect and disconnect—Roads are corridors that can connect contrasting ecosystem types. Because roads provide a somewhat homogeneous condition through the length of the corridor, they provide opportunity for organisms and materials to move along the corridor, thereby increasing the connectivity (Merriam 1984) among those ecosystems interfacing with the road.

The degree of connectivity between roads and streams (that is, the number of stream crossings and areas where roads and streams are near enough to strongly interact) is recognized as a good general indicator of the interactions between the two and of potential effects roads can exert (Wemple 1994). Where both stream and road densities are high, the incidence of connections between roads and streams can be expected to also be high, resulting in more common and pronounced effects of roads on streams than in areas where road-stream connections are less common and dense. (fig. 3).

The economic benefits of roads could be seen as a function of connecting commodities, such as timber, minerals, recreational opportunities, and so on, with potential users.

Roads also can function to disconnect important features of ecosystems. Many roads built next to streams isolate or disconnect streams from their flood plains, with adverse effects to stream dynamics and associated aquatic biota. Roads can block the movement of some animals, such as wolves crossing wide roads or fish being blocked from their upstream movement by perched culverts.

Road density and fish populations correlate across a large area in the interior Columbia basin—One of the few examples of landscape-scale analysis of road influences has been the interior Columbia River basin environmental assessment (Quigley and others 1997). The evaluation of road density and forest and range integrity in that study may serve to illustrate landscape-scale interaction of roads with their surroundings. Forest and range indices of integrity were developed that showed sub-basins having the highest forest-integrity index were largely unroaded and comprised cold forest "potential vegetation groups," or a mixture of moist and cold forest groups. Of the five indicator variables used, the proportion of a subbasin composed of wilderness or roadless areas seemed most closely associated with subbasins having high integrity indices; 81 percent of the subbasins classified as having the highest integrity had

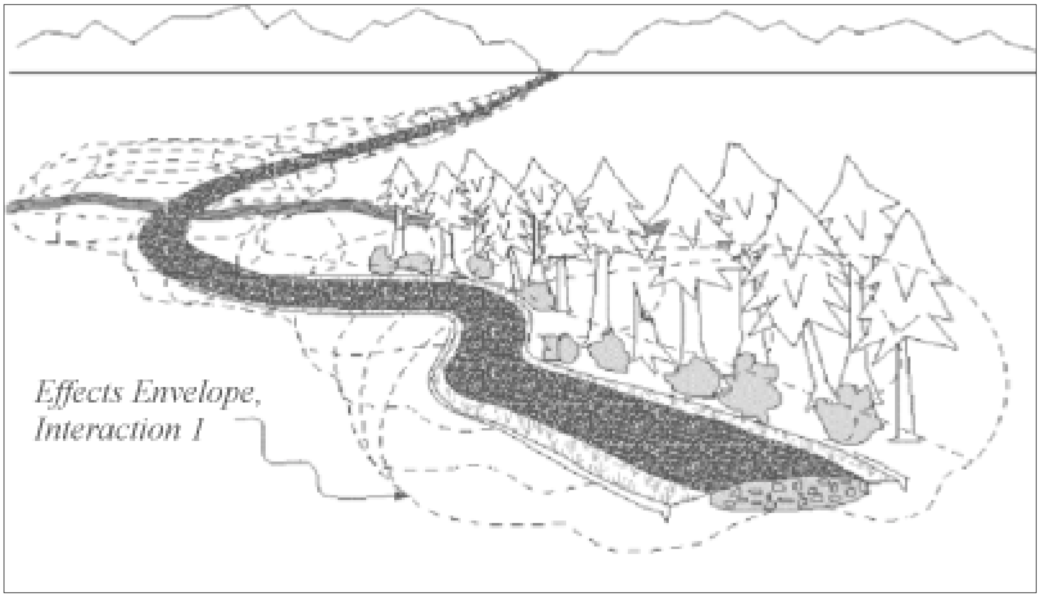


Figure 2—The volume of geographical space occupied by a road, whereby the distance of the road effect is used to define its width and height. The volume changes given the ecological conditions in the area the road traverses (from Lugo and Gucinski 2000).

relatively large proportions of wilderness and roadless areas (>50 percent). Conversely, of subbasins with the lowest integrity, 89 percent had low proportions of roadless and wilderness areas, 83 percent had relatively high proportions of at least moderate road density (0.27 miles/square mile). None of the seven subbasins having high rangeland integrity had areas of moderate or high road densities. The correlation of basin or sub-basin integrity is not total, thereby suggesting that other variables and mechanisms are complex and nonuniform (but see text below for additional caveats).

Recreation surveys suggested the three most highly ranked uses of land administered by the Forest Service and Bureau of Land Management in the interior Columbia basin today are timber, fishing, and hunting. Projected major uses by 2045 will be a shift to motor viewing and day and trail use, even though this area has 70 percent of the un-roaded areas of >200,000 acres remaining in the conterminous 48 states.

Strong fish populations were more frequently found in areas with low rather than high road densities. Supplemental analyses “clearly shows that increasing road densities and their attendant effects are associated with declines in the status of four non-anadromous salmonid species.... They are less likely to use highly roaded areas for spawning and rearing, and, where found, are less likely to be at strong populations levels” (Lee and others 1997).

These findings are a “consistent and unmistakable pattern based on empirical analysis of 3,327 combinations of known species status and sub-watershed conditions, limited primarily to forested lands administered by BLM/FS” (Lee and others 1997). Although unroaded areas are significantly more likely than roaded areas to support strong populations, strong populations are not excluded from roaded watersheds. Possible reasons for this coexistence are that the inherent productivity of some areas allows fish populations to persist despite disturbances linked to roads; real or detectable effects on fish populations may lag behind the initial physical effects in watersheds where roads have been added in the last several years; and the scale of the subwatershed (18,000 acres

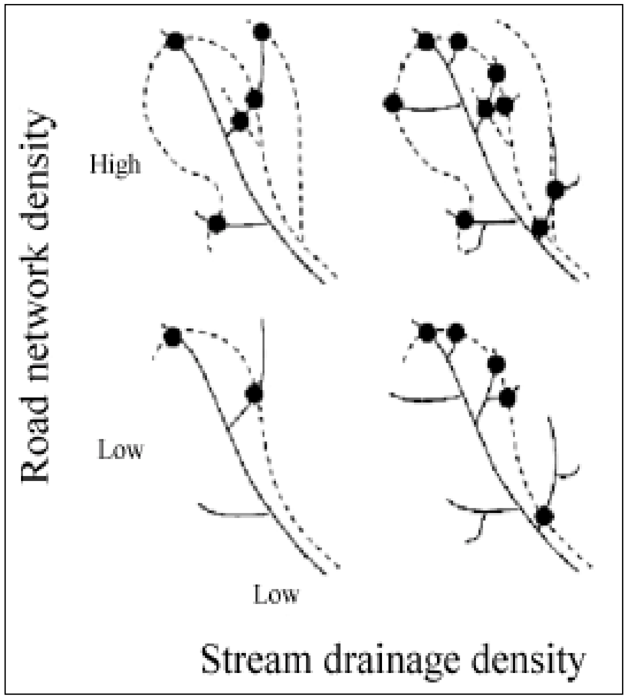


Figure 3—The incidence of road-stream connections, such as stream crossings (the black dots) is related to the density of both roads and streams in the landscape (Swanson and others 2000).

on average) at which strong populations are identified may mask a potential disconnect between the real locations of fish strongholds and roads (identified at resolution of 0.38 square mile). In general, greater short- or long-term watershed and ecological risks are associated with entering an unroaded area than with proceeding continuously with management activities in roaded areas to upgrade, maintain drainage, or close or obliterate existing roads.

Limitations of science—The existing science about roads goes far in establishing what and where problems are likely to arise. More than half a century of research and experience supports designing, building, and maintaining forest roads. Most of the major engineering problems associated with roads have been solved, and a wealth of information exists on many of the physical effects of roads, particularly on hydrologic and geomorphic watershed processes. Information on the biologic effects of roads is improving. Getting this knowledge into practice is more an economic, social, and political issue than a technical one. Less well understood but increasingly studied are the ways that the social and cultural settings of roads influence the benefits, problems, and risks that roads present.

Despite this extensive base of literature and understanding, a striking conclusion from our assessment of the current state of scientific understanding of roads is that virtually no attempt has been made to integrate this information into a comprehensive picture of how roads function in the landscape—physically, biologically, and socially. Despite the ubiquity of roads, no “science of roads” exists. Instead, many disciplines offer their perspectives: engineers study road design and performance, hydrologists evaluate effects of roads on water and sediment, ecologists consider effects on vegetation and wildlife, and transportation planners focus on road layout in relation to other forest resources and uses. Few efforts have been directed toward viewing the gamut of road benefits and effects systematically and simultaneously, or to developing general methods for evaluating risks posed by roads in individual watersheds. Further, the inventory and evaluation of roads is usually limited by ownership: The Forest Service focuses on roads in national forests and generally ignores roads within adjacent ownerships; states evaluate state highways; and the U.S. Department of Transportation evaluates federal highways.

We expect that implementing systematic analyses of road systems in national forests (as part of forest planning and other project planning; USDA 1999) will soon produce abundant examples of intermediate- and large-scale analyses. We hope that those analyses will look beyond ownership to produce a comprehensive evaluation of roads as a system. We have noted that the science information on the benefits of roads is not well developed. The form of scientific approaches for measuring benefits is largely based on economic analyses, which tend to focus on monetary cost differentials produced by the presence or absence of roads. Even in that arena, the data are not rigorously developed. Approaches from the social sciences are based on measurements of public perceptions and public desires, but the total data set does not comprise a highly developed scientific base.

Past studies (with the single, large exception of the interior Columbia River basin environmental assessment) have shed little light on the effects of roads across the whole landscape. Deciphering road effects at large spatial scales is difficult because past studies either focused on the performance of individual road segments, or else road effects were confounded by other simultaneous treatments. Most engineering studies, for example, look at the performance of specific road types (such as arterial, collector), features (road surfaces, cutslopes), or engineered structures (culverts) without examining how the road network functions in relation to adjacent hillslopes and an intersecting stream network. Where roads have been looked at in a watershed context, as in small watershed experiments, effects of roads often have not been distinguished from those of other treatments, such as logging or site preparation, that typically accompany roads. Treatments only of roads are rare and may continue for just a few years before other treatments are applied.

Despite the size of the forest road network, road effects have been examined in only a few places. Much of what we know about forest roads comes from studies in the Appalachians, Pacific Northwest, and Rocky Mountains—areas with known road problems. Given the wide variability in road history, age, construction methods, and use patterns in relation to topography, climate, and social setting, the narrow geographical scope of these studies limits their extrapolation to other regions or their usefulness in addressing more subtle effects.

Research has not typically considered an array of major effects and their interactions. We found only one study (either by way of case study or conceptual framework) addressing the broad range of major road effects. A recent report from the Transportation Research Board that addresses effects of motor vehicles—and by extension, roads—on climate and ecology focuses on the effects of vehicle emissions; only eight pages are devoted to a discussion of the effects of vehicle infrastructure (that is, roads), and the discussion of conserving biodiversity is limited to selected variables. Another recent paper focuses almost exclusively on the ecological damage posed by roads with scant attention to their potential benefits (Forman and others 1997). We know of no studies that provide a systematic way of evaluating risks and benefits from building, using, and removing roads. Such studies are needed to assess tradeoffs among the exceptionally diverse roles of roads in forest landscapes.

Recommendations

This overview of scientific information leads us to conclude that the emerging science of the effects of roads as networks in the landscape requires considerable new research. Because of the high degree of variability of roads from place to place and region to region, a framework for evaluating benefits, problems, risks, and tradeoffs among them would provide a powerful decisionmaking tool. We believe such a framework is now in place (USDA FS 1999). Conducting these analyses is well within the grasp of capable specialists, planners, and managers who can bring their expertise to the problem of reducing risks from past, current, or planned roads and targeting future road-restoration activities. The science pieces are already developed to analyze and integrate road systems and their effects.

Valid and useful analyses of road systems cannot proceed in the face of outdated, incomplete inventories lacking data needed to address important questions. Accurate and current road inventories that include information relevant to environmental effects analyses are needed.

Long-term and ongoing science initiatives would yield valuable information on how the effects of roads develop and change over time. Areas of research should include the effects of progressive road development and how road effects diminish or increase through time, even under constant road configuration. Some observations suggest, for example, that roads systems increasingly connect surface water flow paths to streams over decades, via gullies and landslides in steep terrain. Effects of road restoration practices also need to be evaluated in long-term studies, because both effects and practices are likely to evolve over time. Research on social and cultural perspectives on road use and presence is a key area for future work.

Organization of Sections That Follow

Several possible models might be used to organize a discussion of the ecological and physical effects of roads in forested landscapes. The most logical organization might start from the smallest scale of measurable effects and proceed to the landscape scales. At present, however, our knowledge is too imperfect and too fragmented to fully appreciate and integrate landscape-scale effects. Thus, we have used an approach that goes from the most direct effects to the secondary and indirect effects of forest roads. To a large degree, this model implies we will proceed from understanding effects of road segments to understanding effects of a road network.

We list physical effects first, stressing geomorphic and hydrologic processes, followed by effects on site productivity. Then we move to effects of habitat fragmentation, biological invasion, and other habitat changes that roads introduce. The direct effects—especially the physical ones, such as increased sedimentation and increased risk of slides and debris flows—are much affected by road design and placement on the landscape. Thus, when consequences of roads are aggregated at the landscape scale, the proportion of old roads to new ones that incorporate improved engineering design must be taken into account.

Indirect physical, biological, and landscape-scale effects, sometimes known only from empirical relations, constitute the next set, and include aquatic habitat effects both observed in instream consequences and broad-scale potential effects. Changes in the habitat of terrestrial vertebrates, road kill, and transmission of forest diseases by road traffic are even more complicated, in that they introduce effects not from the road itself, but from road use. Such effects clearly can be stopped by closing a road, but they also can be reduced or altered by changing patterns of road use, allowing for a range of options different from the options roads introduce just by their presence. Lastly, conserving biodiversity is such a broad and unexplained topic that we can sketch only a few of its aspects; we cannot state unequivocally what specific roles roads have in the interplay of populations, modified habitat, the new techno-ecosystem, road kill, and the complex ecological results when alien species modify forest landscapes. We also cannot separate the effects of roads from land-use changes on adjacent lands made accessible by roads; all modify species composition and survival of their populations.

We have addressed socioeconomic effects of roads in forest systems in a manner that follows the pattern introduced in the discussion of physical and biological effects: namely, we examine direct effects first, followed by a discussion of indirect effects or effects at a larger, landscape scale.

Some studies have separated road effects from land-use effects, including timber harvest on adjacent lands; other studies have not. Thus, this synthesis may have allowed these effects to be combined. Although we have made every effort to remove these confounding factors, the reader must carefully evaluate the data presented and consider to what degree we have succeeded.

The following sections are summary discussions of the interaction of roads with adjacent landscape components. They also briefly summarize the available information about the effects of roads on the environment and deliberately have been kept short with references provided for further study.

Direct Physical and Ecological Effects

Geomorphic Effects, Including Sedimentation and Landslides

Issues—More than 50 years of research and many case examples place the effects of forest roads on geomorphic processes squarely at the heart of the debate prompting reexamination of existing and future road networks on public lands. Geomorphic effects of forest roads range from chronic and long-term contributions of fine sediment into streams to catastrophic effects associated with mass failures of road fill material during large storms. The interactions of roads and land surfaces are often complex; for example, on one part of the hillslope, roads may trigger mass failures, and roads downslope from them may trap material derived from these failures. Roads and road building may alter channel morphology directly or may modify channel flow paths and extend the drainage network into previously unchannelized portions of the hillslope. Economic effects of road failures during storms has been discussed; less clearly understood are the cumulative or downstream consequences of road-related changes to geomorphic processes. Major issues motivating concern about road-related erosion include potential degradation of aquatic habitat and water quality and risks to public safety and structures downstream.

Findings—Roads affect geomorphic processes by four primary mechanisms: accelerating erosion from the road surface and prism itself by both mass and surface erosion processes; directly affecting channel structure and geometry; altering surface flow paths, leading to diversion or extension of channels onto previously unchannelized portions of the landscape; and causing interactions among water, sediment, and woody debris at engineered road-stream crossings. These mechanisms involve different physical processes, have various effects on erosion rates, and are not uniformly distributed either within or among landscapes. In steep forest lands prone to landsliding, the greatest effect of roads on erosion rates is from increased rates of mass soil movement after road building. Mass soil movements affected by roads include shallow (three to several feet deep) debris slides, deep-seated (depths of tens of yards) slumps and earth flows, and debris flows (rapid channelized and fluidized movements of water, sediment, and wood). Of these, effects of roads on debris slides and flows have been the most extensively studied, typically by landslide inventories using some combination of sequential aerial photography and ground verification. Accelerated erosion rates from roads because of debris slides range from 30 to 300 times the forest rate, but differ with terrain in the Pacific Northwest, based on a unit area in forest lands ranging from the U.S. Pacific Northwest to New Zealand (Sidle and others 1985). After the 1964 flood in the Pacific Northwest, Swanson and Dyrness (1975) documented increased rates of landslide frequency up to 30 times the rates in unmanaged forested areas. Similar inventories have been conducted elsewhere in the Western United States including Idaho (Megahan and others 1978), Washington (Reid 1981), and northern California, each documenting increased rates of landsliding in road areas relative to unmanaged forested areas. The magnitude of road-related mass erosion differs with climate, geology, road age, construction practices, and storm history. Several studies in the Eastern United States show that landslides are driven more by storm magnitude and geology than by land use. A threshold of 5 inches of rain per day (Eschner and Patric 1982) and metasedimentary geology are associated with large debris slides in the Appalachians. Road drainage can cause small slides in road fills; nevertheless, some major landslides originate in undisturbed forest land (Neary and Swift 1987, Neary and others 1986).

Road-related mass failure results from various causes. Typical causes include improper placement and construction of road fills and stream crossings; inadequate culvert sizes for water, sediment, and wood during floods; poor road siting; modification of surface or subsurface drainage by the road surface or prism; and diversion of water into unstable parts of the landscape (Burroughs and others 1976, Clayton 1983, Furniss and others 1991, Hammond and others 1988, Larsen and Parks 1997, Larsen and Simon 1993). Effects of roads on deep-seated mass movements have been much less extensively studied, although cases are documented of road building apparently accelerating earth-flow movement. This can occur by destabilizing the toe area or diverting water onto the earth-flow complex (Hicks 1982). Little is documented about the potential for increased mass failures from roads resulting from decay of buried organic material that has been incorporated into road fills or landings during road building. Anecdotal evidence is abundant that failures occur predictably after decay of the organic material.

Although mass erosion rates from roads typically are one to several orders of magnitude higher than from other land uses based on unit area, roads usually occupy a relatively small fraction of the landscape, so their combined effect on erosion may be more comparable to other activities, such as logging. Studies by Swanson and others (1981) in the Oregon Coast Range, for example, showed that although unit-area erosion from roads was 30 times greater than the increase from clearcutting alone, road-related landslide erosion accounted for just three times as much accelerated slide erosion in the watershed when the area in roads and clearcuts was taken into account. Road and clearcut erosion were nearly equal in a study in the west side of the Cascade Range in Oregon (Swanson and Dyrness 1975). In the Klamath Mountains of southwest Oregon, erosion rates on roads and landings were 100 times those on undisturbed areas, but erosion on harvested areas was 7 times that of undisturbed areas (Amaranthus and others 1985).

A related point is that only a few sites can be responsible for a large percentage of the total erosion. For example, major erosional features occupied only 0.6 percent of the length of roads studied by Rice and Lewis (1986).

Although road location, design, construction, and engineering practices have improved markedly in the past three decades, few studies have systematically and quantitatively evaluated whether these newer practices result in lower mass erosion rates (McCashion and Rice 1983). Retrospective analysis of road-related landslides in the Oregon Coast Range suggests some reduction in slide frequencies because of improved road siting and building (Sessions and others 1987). No large storms occurred during the study period, however, so these practices remain largely untested. Currently, several studies are ongoing to evaluate road-related mass movements and the influence of road design after several large floods in 1996 in the Pacific Northwest and 1997 in California. These studies are likely to substantially improve understanding of whether “best management practices” are effective in reducing mass erosion from roads, and which specific practices influence mass failure response.

Surface erosion from road surfaces, cut banks, and ditches represents a significant and, in some landscapes, the dominant source of road-related sediment input to streams. Increased sediment delivery to streams after road building has been well documented in the research literature for the Pacific Northwest and Idaho (Bilby and others 1989, Donald and others 1996, Megahan and Kidd 1972, Reid and Dunne 1984, Rothacher 1971, Sullivan and Duncan 1981) and in the Eastern United States (Kochenderfer and others 1997; Swift 1985, 1988). Rates of sediment delivery from unpaved roads are highest in the first years after building (Megahan and Kidd 1972) and are closely correlated to traffic volume on unpaved roads (Reid and Dunne 1984, Sullivan and Duncan 1981). Surface-erosion problems are worst in highly erodible terrain, particularly landscapes underlain by granite or highly fractured rocks (Megahan 1974b, Megahan and Ketcheson 1996). In the Eastern United States, poorly designed and managed forest access and county roads are major sources for higher sediment input rates to streams (Hansen 1971, Patric 1976, Van Lear and others 1995). Roads were identified as the major source of sediment in the Chattooga River basin, where 80 percent of the road sources are unpaved, multipurpose roads (forest and county) paralleling or crossing tributary streams (Van Lear and others 1995). The largest sediment losses were during road building and before exposed soils were protected by revegetation, surfacing, or erosion control materials (Swift 1985, 1988; Thompson and others 1996; Vowell 1985). Soil loss from skid roads in West Virginia ranged from 40 tons/acre during logging, to 4 tons/acre the first year after logging, to 0.1 ton/acre 1 year after logging was completed (Hornbeck and Reinhart 1964). Raw ditch lines and roadbeds are continuing sources of sediment (Miller and others 1985), usually because of lack of maintenance, inadequate maintenance for the amount of road use, excessive ditch line disturbance, or poorly timed maintenance relative to storm patterns (Swift 1984, 1988).

Extensive research has demonstrated that improved design, building, and maintenance of roads can reduce road-related surface erosion at the scale of individual road segments. Key factors are road location, particularly layout relative to stream systems (Swift 1988, USDA FS 1999), road drainage (Haupt 1959), surfacing (Burroughs and King 1989, Kochenderfer and Helvey 1987, Swift 1984), and cut slope and fill slope treatments (Burroughs and King 1989, Swift 1988). Many studies show that surfacing materials and vegetation measures can be used to reduce the yield of fine sediment from road surfaces (Beschta 1978, Burroughs and others 1984, Kochenderfer and Helvey 1987, Swift 1984).

Few studies have evaluated long-term and watershed-scale changes to sediment yields as roads are abandoned, obliterated, or restored. Personnel at Redwood National Park are undisputed experts in road restoration at a watershed scale; they have developed, tested, and applied road-restoration techniques at a scale virtually unprecedented throughout the world (Ziemer 1997). Since Redwood National Park was expanded in 1978, 134 miles of the 300 miles of road within park boundaries have been restored or obliterated. This work has removed about 1,300,000 cubic yards of material from stream crossings, landings, and unstable road benches. The volume of material is about equal to the long-term average annual sediment discharge near the mouth of Redwood Creek (Ringgold, n.d.). To evaluate the success of removing this volume of material, the delivery mechanism, timing, and proportion of the removed material that actually would have found its way to the channel without the restoration activity, the quantity of new material introduced by erosion caused by the restoration work itself, and the relative proportion of the treated areas compared to untreated areas at comparable risk in the basin must be known. Such evaluations are uncommon.

Roads interact directly with stream channels in several ways, depending on orientation to streams (parallel, orthogonal) and landscape position (valley bottom, midslope, ridge). The geomorphic consequences of these interactions, particularly during storms, are potentially significant for erosion rates, direct and off-site effects on channel morphology, and drainage network structure, but they are complex and often poorly understood. Encroachment of forest roads along the mainstem channel or flood plain may be the most direct effect of roads on channel morphology in many watersheds. Poorly designed channel crossings of roads and culverts designed to pass flow also may affect the morphology of small tributary streams, as well as limit or eliminate fish passage. Indirect effects of roads on channel morphology include the contributions of sediment and altered streamflow that can alter channel width, depth, local gradients, and habitat features (pools, riffles) for aquatic organisms (Harr and Nichols 1993).

Roads in midslope and ridgetop positions may affect the drainage network by initiating new channels or extending the existing drainage network. By concentrating runoff along an impervious surface, roads may decrease the critical source area required to initiate headwater streams (Montgomery 1994). In addition, concentrated road runoff channeled to roadside ditches may extend the channel network by eroding gullies or intermittent channels on hillslopes and by linking road segments to small tributary streams (Weaver and others 1995, Wemple and others 1996a). These effects of roads on the channel network have implications for slope stability, sedimentation, and streamflow regimes.

An emerging focus of the postflood studies in the Pacific Northwest is the importance of designing roads to accommodate disturbances (see “Hydrologic Effects” below), particularly in the area of road-stream crossings, which are implicated in most documented road failures (Furniss and others 1997). Another facet of this research is recognizing that roads can serve both as sources (by initiating landslides) and sinks (by trapping debris flows) of sediment during large events (Wemple and others 1996a).

Reliability of findings—These findings represent a broad synthesis of more than 50 years of research on geomorphic effects of roads in a wide range of physiographic and land-use settings. Although they are generally well supported by field, small watershed, and plot studies, specific effects of roads are strongly influenced by local factors, including road building techniques, soil and geology, precipitation and runoff regimes, and topography. As with hydrologic studies, evaluating effects of roads on geomorphic processes is further limited by the short timeframes (one to several years) during which such effects typically are monitored. Few studies have placed road effects in a broad landscape or watershed setting.

Generalizability—Most studies of roads have been conducted in only a few landscapes (the Pacific Northwest, Rocky Mountains, Appalachians, interior highlands, and Piedmont), so the ability to generalize to other terrains is limited. Statements about effects of roads on mass erosion are limited to those landscapes affected by such processes. A large part of the United States, including the Central States, Piedmont, and the coastal plain in the East, do not experience mass erosion processes in the forest. For the most part, only historical road-building practices (pre-1990) have been rigorously evaluated, either by scientists or by the landscape itself through large floods. Little is known, however, about geomorphic effects of old mining and arterial roads (older than 50 years).

Secondary links—The geomorphic and hydrologic effects of roads are closely related. Restoration strategies to reduce either geomorphic or hydrologic effects are likely to be quite different, however, which underscores the need to clearly identify objectives for restoration. For example, practices to reduce road network extension of surface flow paths by draining water back into the subsurface could have the unintended consequence of destabilizing fill slopes. Both the mass erosion and fine-sediment delivery issues are closely linked to concerns about aquatic habitat.

Conclusions—As with the hydrologic issues, evaluating geomorphic effects of roads needs to be addressed at several scales: individual road segments, intermediate-sized watersheds, and the entire road network in the river basin (which may include private lands and roads and roads built for a broad range of purposes, not just forest operations). Key directions for future research work are to systematically evaluate the relation between improved road practices and mass-erosion rates, particularly in light of mid-1990s floods in the Pacific Northwest and California; develop a conceptual and analytical framework for evaluating how roads in different landscape positions (valley bottom, midslope, ridgetop) interact with streams; develop empirical data on the amount of drainage-network extension and drainage-density increases resulting from roads in different geomorphic settings; and place geomorphic effects of roads in broader landscape contexts by using sediment budget and disturbance budget approaches.

Hydrologic Effects

Issues—The interaction between forest roads and water lies at the heart of several key issues surrounding the effects of roads on the environment. At the scale of individual road segments, designing and building roads to drain or channel water away from the road surface is one of the main problems facing road engineers, and it reflects the substantial effects that roads can have on hillslope hydrology. Road drainage problems and water and debris passage problems—especially during floods—are primary reasons for road failure, often with major structural, ecologic, economic, or social consequences. For example, of the $178 million spent on flood recovery on Forest Service lands in the Pacific Northwest Region after the 1996 floods, more than 70 percent was to fix road damage; most of the damage resulted from water drainage problems that, in turn, triggered mass movements (Cronenwelt, n.d.). At a broader scale, roads can influence the size and timing of streamflows from watersheds, with possible consequences for downstream channels and aquatic ecosystems. For these reasons, many road restoration projects are explicitly or implicitly focused on the ways roads influence the routing of water, with consequences for erosional processes.

Findings—Roads have three primary effects on water: they intercept rainfall directly on the road surface and road cutbanks and intercept subsurface water moving down the hillslope; they concentrate flow, either on the surface or in an adjacent ditch or channel; and they divert or reroute water from flow paths that it would take were the road not present. Most hydrologic and geomorphic consequences of roads result from one or more of these processes. By intercepting surface and subsurface flow, for example, and concentrating it through diversion to ditches, gullies, and channels, road systems effectively increase the density of streams in the landscape. This changes the amount of time required for water to enter a stream channel, which alters the timing of peak flows and hydrographic shape (King and Tennyson 1984, Wemple and others 1996a). Similarly, concentration and diversion of flow into headwater areas can cause incision of previously unchanneled portions of the landscape and initiate slides in colluvial hollows (Mongomery 1994). Diversion of streamflow at road-stream crossings is a key factor contributing to road failure and erosional consequences during large floods (Furniss and others 1998, Weaver and others 1995).

Hydrologically, different parts of the road system behave differently. All roads are not created equal and do not perform the same during storms, and the same road segment may behave differently during storms of different magnitudes. Recent, detailed examination of hydrographs at stream crossings with culverts shows that during the same storm, some road segments contribute substantially more flow to channels than others, primarily owing to differences in the amount of subsurface water intercepted at the cut bank (Bowling and Lettenmeier 1997, Wemple and others 1996b). As storms become larger or soil becomes wetter, more of the road system contributes water directly to streams. Slope position has a profound effect on the magnitude of hydrologic change caused by roads. Discharge from hill slopes, height of cut bank, density of stream crossings, soil properties, and response to storms all differ with slope position.

Although hydrologic effects of roads have been studied for more than 50 years, systematic studies with long-term measurement of the full range of potential interactions between water and roads are few. Most studies have emphasized geotechnical issues, including road design, culvert size and placement, and erosion control from road surfaces (see Reid and others 1997, for bibliography; Swift 1988). Of those studies that have attempted to look at the hydrologic behavior of roads, most have been part of small (typically 0.3 to 2 square miles) watershed experiments, where roads were a component of the experimental treatment, which often included other silvicultural practices. Key studies and locales of this type include those by Rothacher (1965, 1970, 1971, 1973), Harr and McCorison (1979), Harr and others (1975), Jones and Grant (1996), and Thomas and Megahan (1998) in western Oregon; Ziemer (1981, 1998) and Wright and others (1990) in northern California; King and Tennyson (1984) in central Idaho; Reinhart and others (1963), Hewlett and Helvey (1970), Swank and others (1982, 1988) in the southern Appalachians, Helvey and Kochenderfer (1988) in the central Appalachians; and Hornbeck (1973) and Hornbeck and others (1997) in the northern Appalachians. Very few studies have focused on the hydrologic behavior of roads alone; in the Pacific Northwest and Rocky Mountains, maximum measurement periods during which roads were the only treatment range from 1 to 4 years (Wemple 1994). Most studies have been conducted as “black box” experiments comparing streamflow hydrographs before and after road building, with little ability to identify key processes. Exceptions include the work of Megahan (1972), Keppeler and others (1994), and Wemple (1994) on subsurface flow interception and Luce and Cundy (1994) and Ziegler and Giambelluca (1997) on road-surface runoff. Few studies have focused on road effects, on hydrology in arid or tropical areas, or on areas dominated by snow hydrology, permafrost, and wetlands.

Even fewer published studies have explicitly considered how road networks affect the routing of water through a basin. We therefore have little basis to evaluate the hydrologic functioning of the road system at the scale of an entire watershed or landscape. Few published studies to date have identified how roads in different landscape positions might influence the movement of water through a basin. Montgomery (1994) looked at the effect of ridgetop roads on channel initiation, and Wemple (1994) documented the magnitude of drainage network enlargement caused by roads in different slope positions.

Based on studies of small watersheds, the effect of roads on peak flows is detectable but relatively modest for most storms; insufficient and contradictory data do not permit evaluation of how roads perform hydrologically during the largest floods. Roads do not appear to affect annual water yields, and no studies have evaluated their effects on low flows. In some studies, roads produced no detectable change in flow timing or magnitude (Rothacher 1965, Wright and others 1990, Ziemer 1981), but in other basins, average time to storm peak advanced and average peak magnitude increased after road building for at least some storm sizes (Harr and others 1975, Jones and Grant 1996, Thomas and Megahan 1998). In a study in Idaho, peak stormflow magnitude increased in one basin and decreased in another after road building, an effect the authors attribute to subsurface flow interception by roads and desynchronization of delivery of water to the basin outlet (King and Tennyson 1984). A whole-tree logging operation in New Hampshire that resulted in 12 percent of the area in roads (Hornbeck and others 1997) showed a maximum average increase of growing-season peak flows of 63 percent in the second year after harvest. This increase disappeared as the forest regenerated, and only 2 of the 24 peak flows in the 6th through the 12th growing seasons showed statistically significant increases. Dormant-season peak flows generally decreased because cutting changed snowmelt regimes. Helvey and Kochenderfer (1988) concluded that typical logging operations in the central Appalachians do not increase flows sufficiently to require larger culverts to accommodate them. Forest harvesting without roads in the southern Appalachians increased stormflow volumes by 11 percent and peak flow rates by 7 percent (Hewlett and Helvey 1970, Swank and others 1988). Harvesting an adjacent watershed with 4 percent of the area in roads increased stormflows by 17 percent and peak flows by 33 percent. Four years later, peak flows dropped to a 10-percent increase after 40 percent of the road system was closed and returned to forest (Douglass and Swank 1975, 1976). Collectively, these studies suggest that the effect of roads on basin streamflow is generally smaller than the effect of forest cutting, primarily because the area occupied by roads is much less than that occupied by harvest operations. Generally, hydrologic recovery after road building takes much longer than after forest harvest because roads modify physical hydrologic pathways, but harvesting principally affects evapotranspiration processes. The hydrologic effect of roads depends on several factors, including the location of roads on hillslopes, characteristics of the soil profile, subsurface water flow and ground-water interception, design of drainage structures (ditches, culverts) that affect the routing of flow through the watershed, and proportion of the watershed occupied by roads.

Most road problems during floods result from improper or inadequate engineering and design, particularly at road-stream crossings but also where roads cross headwater swales or other areas of convergent groundwater. Road redesign that anticipates and accommodates movement of water, sediment, and debris during infrequent, but major storms should substantially reduce road failures and minimize erosional consequences when failures occur. Recent studies after large floods in the Pacific Northwest highlight the importance of water diversion by roads and road-related structures (that is, plugged culverts, ditches) in contributing to road-related failures (Donald and others 1996, Furniss and others 1997). A typical failure resulted from culverts sized only to accommodate the flow of water, but not the additional wood and sediment typically transported during major floods. The culverts became obstructed and diverted water onto the road surface, into neighboring drainages unable to adjust to the increase in peak flow from the contributing basin, or onto unchanneled hillslopes. “Cascading failures” were common, where diversion or concentration of flow led to a series of other events, ultimately resulting in loss of the road or initiation of landslides and debris flows. Analysis of the probability of large floods and how they relate to the design life of roads indicates that most road crossings are likely to have one or more large floods during their lifetimes. Consequently, designing roads with large storms in mind is prudent and well within the reach of current engineering practices (Douglass 1977; Furniss and others 1991, 1997; Helvey and Kochenderfer 1988). The potential for stream diversion on wildland roads indicates that the environmental consequence of road failure during large storms is an option to consider.

Although the ability to measure or predict the hydrologic consequence of building or modifying a specific road network might be limited, general principles and models can be provided that, if followed, may decrease the negative hydrologic effects of roads. These principles will be useful during upgrading or decommissioning of roads to meet various objectives. A partial list of principles includes:

- Locate roads to minimize effects; conduct careful geologic examination of all proposed road locations.
- Design roads to minimize interception, concentration, and diversion potential, including measures to reintroduce intercepted water back into slow (subsurface) pathways by using outsloping and drainage structures rather than attempting to concentrate and move water directly to channels.
- Evaluate and eliminate diversion potential at stream crossings.
- Design road-stream crossings to pass all likely watershed products, including woody debris, sediment, and fish—not just water.
- Consider landscape location, hillslope sensitivity, and orientation of roads when designing, redesigning, or removing roads.
- Design with failure in mind. Anticipate and explicitly acknowledge the risk from existing roads and from building any new roads, including the probability of road failure and the damage to local and downstream resources that would result. Decisions about the acceptable probability and especially consequences of failures should be informed through explicit risk assessments. The many tradeoffs among road building techniques to meet various objectives must be acknowledged. For example, full bench road construction may result in lower risk of fill slope failure, but it also may increase the potential for groundwater interception; outsloping of the road tread may reduce runoff concentration on the road surface but also increase driving hazard during icy or slippery conditions.

Reliability of findings and generalizability—Hydrologic effects of roads are strongly influenced by landscape condition, road design and construction, and storm history. Generalizability of paired-watershed studies is limited by the short timeframes (one to several years) during which road effects alone are typically monitored. In addition, most road studies have been done in only a few landscapes where road problems are common (the Pacific Northwest, Rocky Mountains, and Appalachians), thereby limiting the ability to generalize to other terrain. The general principles represent reasonable interpretations of the available scientific knowledge, however. Some landscapes may be much more sensitive than others to certain key processes, such as interception of subsurface flow and drainage network extension resulting from gullying. For this reason, the specific range of hydrologic effects likely to be encountered needs to be evaluated by both regional and landscape scales.

Secondary links—The hydrologic effects of roads are strongly linked to their sediment and geomorphic effects. Other links can be found with wildlife (for example, road-created wetlands) and invasion by exotics (for example, microclimate related to water availability above and below the road prism), but these links have received little scientific attention.

Conclusions—Future efforts to redesign, restore, or remove road systems because of hydrologic concerns should have clear objectives: What hydrologic processes are considered problems? Where do they occur? What can be done about them? What degree of hydrologic alteration is considered acceptable? This type of evaluation of roads is best accomplished in the context of a watershed analysis (USDA FS 1999). Key areas for future research are to develop analytical models that allow managers to display the predicted hydrologic consequences of alternative road-network designs (these types of models are still in their infancy but should be more widely available in the next 2 to 3 years), expand process-based studies of how roads affect specific hydrologic mechanisms (for example, subsurface flow interception or channel network extension) in different geomorphic settings evaluate at the landscape scale the extent of links between the road and stream networks in different landscapes, and relate type and size of road failures to specific design practices and landscape position.

Site Productivity

Issue—The presence of roads commits a soil resource, and where roads occupy formerly productive land, they affect site productivity.

Findings—Forest roads can have significant effects on site productivity by removing and displacing topsoil, altering soil properties, changing microclimate, and accelerating erosion. The direct effects of taking land out of production by removing trees and displacing soil, or removing soil during building and maintaining roads, has been estimated to range from 1 to 30 percent of the landscape area in managed forest lands (Megahan 1988a). In the Western United States, tractor and ground-cable systems average about 10 percent of the area affected by roads to support harvest operations, and skyline and helicopter operations average 2 percent (Megahan 1988b). Studies in Eastern U.S. forests have consistently found that 4 to 5 percent of the total forested area is taken out of forest production by building roads during logging operations, although more than 50 percent of this area may be reforested within 8 years, but at reduced growth rates and productivity. Total road length required to support logging operations depends on the harvest and silvicultural systems and topographic configuration, but the area disturbed may be surprisingly consistent (Douglass and Swift 1977, Robinson and Fisher 1982, Swank and others 1982, Swift 1988).

Measurable declines in tree growth are common where soil is excavated to build the road prism. Evidence of off-site effects of roads on productivity is conflicting, though road-associated mass erosion may scour soil from steep slopes. Road building changes soil physical properties including depth, density, infiltration capacity, water holding capacity, and gas exchange rate, nutrient concentrations, and microclimate. Fertile topsoils, often containing most of the organic matter and plant nutrient capital of a site, frequently are buried under road fills or sidecast and may be rendered inaccessible to plant roots. Trees can grow on any portion of a closed road, but they can grow only on cut and fill slopes on open roads. Sites are harshest and soils poor or nonexistent on road cuts and the cut portion of road treads. Tree height and diameter growth is reduced on these portions of the road (Smith and Wass 1979, 1980, 1985). Growth is sometimes enhanced on or below fill portions of roads because of reduced competition and greater soil depth. Pfister (1969) documents a 30-percent increase in height growth of western white pine (*Pinus monticola* Dougl. ex D. Don) adjacent to outsloped roads. Megahan (1988a) suggests that this increase is due to enhanced soil moisture below outsloping roads. Smith and Wass (1980) document significant declines of 23 percent in height growth for lodgepole pine (*Pinus contorta* Dougl. ex Loud.) and 20 percent for Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) below insloped roads, which they attribute to loss of available water through redirected drainage flow. Improper fill placement and drainage can cause upslope groundwater to rise, and the changed soil moisture kills trees (Boelter and Close 1974, Stoeckeler 1965), although not commonly. Loss of nutrient capital is inevitable with soil disturbance from road building (Swanson and others 1989), but isolating this effect from other site changes has proved difficult. An indirect indication of nutrient loss is the marked growth response of plants on road fills after fertilizer is applied. Fertilizer applied to a granitic road fill in Idaho increased growth of vegetation by 32 to 116 percent (Megahan 1974a), but such increases are not documented after fertilizer is applied on undisturbed soils. Both surface and mass-erosion rates increase after road building, and often roads accelerate erosion on the slope below. Downslope damage generally is associated with mass erosion when a landslide originates from a road and causes scour on lower slopes or gullies related to concentrated road drainage (Megahan 1988a). This problem is widespread on steep slopes of the Pacific States and in the northern Rocky Mountains (Burroughs 1985, Swanson and others 1981), although Megahan (1988b) estimates that productivity is reduced on about 0.3 percent of forested land at a broad scale. These effects may range from decades (Ice 1985) to more than 85 years (Smith and others 1986). Road treads are highly compacted compared to natural soils, but compaction is not a productivity issue so long as roads are open and the running surface is bare. Road decommissioning must take compaction into account in restoring productivity, and various “ripping” treatments are routinely applied to decompact road surfaces.

Reliability, confidence, and generalizability—Direct effects of roads—including lost productivity because of the area occupied by roads themselves, and diminished productivity on cut slopes and road treads on closed roads—are well documented and general in geographic extent. Losses of productivity associated with road-caused, accelerated erosion are site specific and variable in extent, but they are commonly reported for all steep-slope landscapes. Rates of reforestation along road fills are high in the Pacific Northwest and Eastern United States and slower in the inland West and Southwest. Road-caused nutrient imbalances or declines often are confounded by other effects (notably soil moisture losses) in Western States.

Conclusions—A substantial amount of information is available on productivity in road fills and cut slopes and strong anecdotal, but obvious, evidence of lack of productivity on road treads. Information on effects of roads on adjacent site productivity is limited, and variable results confound attempts to generalize and accurately predict effects.

Secondary links—Applying salt to roads is discussed in “Water Quality” and its effects on plant damage are discussed in “Forest Diseases,” both below. Erosional processes and rates are discussed extensively in “Geomorphologic Effects,” above. Loss of site productivity represents a long-term economic loss, and quantifying such losses is confounded by the difficulty in establishing or even estimating the degree of soil productivity changes associated with roads.

Habitat Fragmentation

Issues—Natural populations of animal species are reduced by habitat loss caused by road building and by the animals' avoidance of areas near roads. Populations can be fragmented into smaller subpopulations, thereby causing increased demographic fluctuation, inbreeding, loss of genetic variability, and local population extinctions.

Findings—Habitat loss has broader effects than just the conversion of a small area of land to road surface. Roads fragment by changing landscape structure and by directly and indirectly affecting species. Habitat effects of roads on the landscape include dissecting vegetation patches, increasing the edge-affected area and decreasing interior area, and increasing the uniformity of patch characteristics, such as shape and size (Reed and others 1996). Whenever forest roads are built, changes in habitat and modified animal behavior will lead to changes in wildlife populations (Lyon 1983). Road-avoidance behavior is characteristic of large mammals such as elk (*Cervus canadensis*), bighorn sheep (*Ovis canadensis*), grizzly (*Ursus arctos horribilis*), caribou (*Rangifer tarandus*), and wolf (*Canis lupus*). Avoidance distances of 300 to 600 feet are common for these species (Lyon 1985). Road usage by people and their vehicles has a significant role in determining road avoidance by animals. In a telemetry study of movement by black bear (*Ursus americanus*), bears almost never crossed interstate highways, and they crossed roads with little traffic more frequently than those with high traffic volumes (Brody and Pelton 1989). Bobcats (*Lynx rufus*) crossed paved roads in Wisconsin forests less than expected, possibly to minimize interactions with vehicles and people (Lovallo and Anderson 1996). A few studies have related genetic changes in populations simply to the presence of roads (Forman and others 1997), but the distribution of roads in the environment also must be considered. Road density is a useful index of the effect of roads on wildlife populations (Forman and others 1997). Wolves in Wisconsin are limited to places with pack-area mean road densities of 0.7 mile/square mile or less (Mladenoff and others 1995). Some studies have shown that a few large areas of low road density, even in a landscape of high average road density, may be the best indicator of suitable habitat for large vertebrates (Rudis 1995).

Reliability, confidence, and limitations—The evidence is strong that forest roads displace some large mammals and certain birds such as spotted owls (*Strix occidentalis)* and marbled murrelets (*Brachyramphus marmoratus*) and that displaced animals may suffer habitat loss as a result. Effects of roads on small mammals and songbirds are generally described as less severe, with changes expressed as modifications of habitat that cannot readily be classified as detrimental or beneficial. This interpretation is also probably true for amphibians and reptiles.

Generalizability—For large mammals, general principles have been explained, above, that can be applied to project decisions.

Secondary links—Habitat fragmentation is linked to other habitat-related topics and also links with access-related topics, particularly timber, where the density and distribution of roads is a key technical and economic question.

Conclusions—Specific issues related to wildlife can be addressed directly. Integration with other technical, economic, and social issues (such as timber availability and recreational access) have to be dealt with by management.

Habitat

Issues—Road building introduces new edge habitat in the forest. The continuity of the road system also creates a corridor by which edge-dwelling species of birds and animals can penetrate the previously closed environment of continuous forest cover. Species diversity can increase, and increased habitat for edge-dwelling species can be created.

Findings—Roads and their adjacent environment qualify as a distinct habitat and have various species, population, and landscape-scale effects (Baker and Knight 2000, Dawson 1991, van der Zande and others 1980). Some research has attempted to describe habitat modifications caused specifically by roads, but most of this work is species and site specific (Lyon 1983). Surveys of songbirds in two national forests of northern Minnesota found 24 species of birds more abundant along roads than away from them (Hanowski and Niemi 1995). Close to half these species were associated with edges, including birds like crows (*Corvus brachyrhynchos)* and blue jays (*Cyanocitta cristata)* that use roads as corridors to find food. Turkey hens (*Megapodiidae*) in North Carolina nested near closed and gated logging roads and used them extensively in all stages of brood development (Davis 1992). One study showed that habitat in the road-side right-of-way supports a greater diversity of small mammals than do adjacent habitats (Adams and Geis 1983), but this finding may not apply to forest roads with only narrow cuts and fills on either side. The similarity between forest roads and transmission-line rights-of-way may be important in assessing the contribution of roads to habitat. Studies have shown that wide transmission-line corridors support grassland bird communities of species not found in the forest, and narrow corridors produce the least change from forest bird communities (Anderson and others 1977). The same study notes that increasing edge diversity of birds, for instance, may negatively affect abundance of interior species (see “Biological Invasions,” below).

Reliability, confidence, and limitations—Limited species and site-specific data exist describing the immigration of particular species into habitat created by roads. Detailed information on specific habitat characteristics affected by the building and presence of roads is lacking. The relation of microclimate, vegetation distribution, and water supply to the road network needs to be described.

Generalizability—In general, road building fragments habitat and creates habitat edge, thereby modifying the habitat in favor of species that use edges. Edge-dwelling species generally are not threatened, however, because the human-dominated environment has provided ample habitat for them. Any habitat modifications attributed to the road may be insignificant compared to the effects of the activity, such as timber harvest, for which the road was built.

Secondary links—Links exist to other habitat-related topics and also to biological invasions.

Conclusions—Science information about the underlying principles related to this issue is incomplete. Further study is needed before anything more than site- and species-specific analyses can be undertaken.

Biological Invasions

Issues—A widely cited generalization about biological invasion is that it is promoted by disturbance. Building roads and subsequently maintaining them (including ditch clearing, road grading, and vegetation clearing) in the interior of a forest represents disturbances that create and maintain new edge habitat. These roadside habitats can be invaded by an array of exotic (non-native) plant species, which may be dispersed by “natural” agents such as wind and water as well as by vehicles and other agents related to human activity. Roads may be the first point of entry for exotic species into a new landscape, and the road can serve as a corridor along which plants move farther into the landscape (Greenberg and others 1997, Lonsdale and Lane 1994). Some exotic plants may then be able to move away from the roadside into adjacent patches of suitable habitat. Invasion by exotic plants may have significant biological and ecological effects if the species are able to disrupt the structure or function of an ecosystem. Invasion also may be of concern to land managers, if the exotic species disrupt management goals and present costly eradication problems.

Findings—Although few habitats are immune to at least some invasion by exotic plants, predicting which species will become pests usually is difficult. Assessing the scale of a biological invasion problem is complicated by the lag between when an exotic is introduced and when it begins to expand its distribution and population size in a new area. Cowbirds *(Molothrus ater)*, for example, can be introduced into forested environments by roads and subsequently affect populations of Neotropical migratory birds through nest parasitism. The spread of pathogens where roads act as vectors is described in "Forest Diseases," below. Few environmentally benign approaches to exotic plant control or eradication have been tested.

Reliability, confidence, and limitations—Field studies of exotic plants tend to focus on a particular geographic region, and observed patterns of road-supported invasion may not apply to other regions. In general, however, observations suggest that biological invasion is often a negative effect of extending roads into forest interiors. Such effects should be considered in the design and execution of road network extensions.

Generalizability—Observations in different settings suggest that the exotic species that successfully invade and the scale of invasion problems differ regionally. Some exotic species can become significant pests, and others remain fairly benign.

Secondary links—Consequences of biological invasions link to habitat quality issues (including changes in plant community structure and function), other edge effects, and effects on sensitive or threatened species.

Conclusions—Information to assess the degree of risk relies on case studies; the risks may be slight or significant. A less than ideal science base exists for identifying which exotic species pose the greatest threat and what preventive or remedial measures are appropriate. Retrospective studies may help identify directions. One study showed that abandoned roads had fewer exotics (both in number of species and frequency of individuals) than did roads that were in use.

Indirect and Landscape-Scale Effects

Aquatic Habitat

Issues—The effects of roads on aquatic habitat are believed to be widespread and profound, and evidence is documented through empirical associations and direct mechanistic effects, although the mechanistic effects become fuzzy when direct, quantitative, cause-effect links are sought. Several studies correlate road density or indices of roads to fish density or measures of fish diversity. Mechanisms include effects of fine sediment, changes in streamflow, changes in water temperature caused by loss of shade cover or conversion of groundwater to surface water, migration barriers, vectors of disease, exotic fishes, changes in channel configuration from encroachment, and increased fishing pressure. A growing body of work indicates that the complexity of habitat and the predictability of disturbance influences species diversity. At the landscape scale, correlative evidence suggests that roads are likely to influence the frequency, timing, and magnitude of disturbance, which are likely to influence community structure.

Findings—Increased fine-sediment composition in stream gravel has been linked to decreased fry emergence, decreased juvenile densities, loss of winter carrying capacity, and increased predation of fishes. Increased fine sediment can reduce benthic organism populations and algal production. Increased sediment production associated with roads is discussed in detail in “Geomorphic Effects,” above. Survival of incubating salmonids from embryos to emergent fry has been negatively related to the proportion of fine sediment in spawning gravels (Chapman 1988, Everest and others 1987, Scrivener and Brownlee 1989, Weaver and Fraley 1993, Young and others 1991). Increased fine sediment in stream gravel can reduce intragravel water exchange, thereby reducing oxygen concentrations, increasing metabolic waste concentrations, and restricting movements of alevins (Bjornn and Reiser 1991, Coble 1961, Cordone and Kelley 1960). Survival of embryos relates positively to dissolved oxygen and apparent velocity of intragravel water, and positively to gravel permeability and gravel size (Chapman 1988, Everest and others 1987). Consequently, juvenile salmonid densities decline as fine sediment concentrations increase in rearing areas (Alexander and Hansen 1986, Bjornn and others 1977, Chapman and McLeod 1987, Everest and others 1987, Shepard and others 1984). Increases in fine sediment also can reduce winter carrying capacity of streams by loss of concealment cover (Bjornn and others 1977, Chapman and McLeod 1987, Thurow 1997) and by increasing the likelihood of predation (Chapman and McLeod 1987). Pools function as resting habitats for migrating adults, rearing habitats for juveniles (Bjornn and Reiser 1991), and refugia from natural disturbances (Sedell and others 1990). Pools that lose volume from sediment (Jackson and Beschta 1984, Lisle 1982) support fewer fish (Bjornn and others 1977), and fish that reside in them may suffer higher mortality (Alexander and Hansen 1986). Similarly, populations of tailed frogs can be severely reduced or eliminated by increased sedimentation (Corn and Bury 1989, Welsh 1990), presumably because of their dependence on unembedded interstitial areas in the stream substrate where they hide and overwinter (Brown 1990, Daugherty and Sheldon 1982). Increased sediment reduces populations of benthic organisms by reducing interstitial spaces and flow used by many species and by reducing algal production, the primary food source of many invertebrates (Chutter 1969, Hynes 1970).

The effects of roads are not limited to those associated with increases in fine-sediment delivery to streams; they can include barriers to migration, water temperature changes, and alterations to streamflow regimes. Improper culvert placement at road-stream crossings can reduce or eliminate fish passage (Belford and Gould 1989), and road crossings are a common migration barrier to fish (Clancy and Reichmuth 1990, Evans and Johnston 1980, Furniss and others 1991). In a large river basin in Washington, 13 percent of the historical coho habitat was lost as a result of improper culvert barriers (Beechie and others 1994). Roads built adjacent to stream channels pose additional effects. Changes in temperature and light regime from removing the riparian canopy can have both positive and negative effects on fish populations. Sometimes increased food availability can mitigate negative effects of increased summer water temperatures (Bisson and others 1988). Beschta and others (1987) and Hicks and others (1991) document negative effects, including elevation of stream temperatures beyond the range of preferred rearing, inhibition of upstream migrations, increased disease susceptibility, reduced metabolic efficiency, and shifts in species assemblages. Streamflow stability and predictability (size, timing, duration, and frequency) also strongly influence salmonid densities by influencing reproductive success and overwintering survival (McFadden 1969). For example, high flows after spawning can wash out eggs or displace fry, thereby increasing mortality (Latta 1962, Mortensen 1977, Shetter 1961).

The effect of roads on peak flows is relatively modest (see “Hydrologic Effects,” above), and the issues of changing stability and predictability because of roads may be of little importance to aquatic habitat suitability.

Road-stream crossings have effects on stream invertebrates. Hawkins and others (in press) found that the aquatic invertebrate species assemblages (observed versus expected, based on reference sites) were related to the number of stream crossings above a site. Total taxa richness of aquatic insect larvae (mayflies, Ephmeroptera; stoneflies, Plecoptera; and caddisflies, Trichoptera) were negatively related to the number of stream crossings. Another study (Newbold and others 1980) found significant differences between macroinvertebrate assemblages above and below road-stream crossings.

Several studies at broad scales document aquatic habitat or fish density changes associated with road density or indices of road density. Eaglin and Hubert (1993) show a positive correlation with numbers of culverts and stream crossings and amount of fine sediment in stream channels, and a negative correlation with fish density and numbers of culverts in the Medicine Bow National Forest. Macroinvertebrate diversity negatively correlates with an index of road density (McGurk and Fong 1995). Increasing road densities are associated with decreased likelihood of spawning and rearing of non-anadromous salmonids in the upper Columbia River basin, and populations are negatively correlated with road density (Lee and others 1997).

Reliability, confidence, and limitations—Research evidence of increased erosion and sediment delivery to streams resulting from roads is strong. Subsequent habitat changes from such processes as pool filling and cobble embeddedness are well documented, but these effects depend heavily on channel geometry, flow regimes, and so on. Thus, they range widely in time and space. Measured changes in stream temperature after canopy removal are strong but biological response is highly variable, and existing literature speculates on possible mechanisms. Empirical evidence relating road density to habitat and population response at landscape scales is fairly new. The study by Lee and others (1997) has a large database and is analytically sound, but it demonstrates a statistically valid population response only for non-anadromous salmonids. Because roads are not distributed randomly on the landscape, these studies can be confounded by other landscape variables that may control biological response. This issue is addressed by Lee and others (1997).

Generalizability—Broad-scale patterns in the distribution of roads and fish suggests that the effects of roads are common and widespread across a range of environments and conditions (Bettinger and others 1998, Lee and others 1997). Changes in aquatic habitat resulting from increased erosion and sediment delivery are highly controlled by lithology and slope, however. Road-derived sediment in granitic terrain typically results in an increase in the proportion of fine bedload. In fine-textured parent materials, suspended load may increase but not change pool filling and cobble embeddedness. Changed timing and size of peak and low flows resulting from roads have different implications for storm-generated and snowmelt-dominated hydrologic regimes, and they result in different biological effects for oversummer and overwinter egg survival. The effect of cover removal on elevated stream temperature depends on the rate of vegetation recovery and appears to be brief in the Eastern United States (Swift 1983).

Secondary links—Responses by aquatic habitat depend on geomorphic and sediment changes associated with roads. Road-associated changes in nutrients and hazardous chemical spills are also linked but are issues addressed elsewhere in this report.

Conclusions—Road effects on aquatic habitat and population response are well documented and overwhelmingly negative, but results differ among sites. Measures of the cumulative effects of roads that are closely related to mechanism (for example, the length of roads connected by direct surface-flow paths to streams or the miles of potential habitat blocked by culverts) would be more likely to produce stronger relations between roads and aquatic habitat elements than would road density.

Landscape-Scale Effects on Fish

Issues—The decline of anadromous fish in many parts of the country, especially the salmonids in the West, has led to much research on the diverse causes. Among those, the relation of roads to intensity of land use and adverse effects on aquatic habitats has been discussed in several recent studies and publications (Meehan 1991, Naiman and others 1992, Spence and others 1996). The discussion centers on three themes: the correlation of road density to fish habitat and fish populations is not strong; the legacy of past road building is so vast and budgets for maintaining roads so low that the problems will be with us for a long time; and road building practices have improved in the last decade to the point where we need not worry about the effects of roads on aquatic systems. The scientific assessment for the interior Columbia basin provided an opportunity to examine these issues at a broad, landscape scale in this ecoregion.

Findings—Roads contribute more sediment to streams than does any other land management activity (Gibbons and Salo 1973, Meehan 1991), but most land management activities, such as mining, timber harvest, grazing, recreation, and water diversions, depend on roads. Most of the sediment from timber harvest activities is related to roads and road building (Chamberlain and others 1991, Dunne and Leopold 1978, Furniss and others 1991, MacDonald and Ritland 1989, Megahan and others 1978) and the associated increases in erosion rates (Beschta 1978, Gardner 1979, Meehan 1991, Rhodes and others 1994, Reid 1993, Reid and Dunne 1984, Swanson and Dyrness 1975, Swanston and Swanson 1976). Serious degradation of fish habitat can result from poorly planned, designed, located, built, or maintained roads (Furniss and others 1991, MacDonald and others 1991, Rhodes and others 1994). Roads also can affect water quality through applied road chemicals and toxic spills (Furniss and others 1991, Rhodes and others 1994), and the likelihood of toxic spills reaching streams has increased with the many roads paralleling them.

Roads directly affect natural sediment and hydrologic regimes by altering streamflow, sediment loading, sediment transport and deposition, channel morphology, channel stability, substrate composition, stream temperatures, water quality, and riparian conditions in a watershed. For example, interruption of hillslope drainage patterns alters the timing and magnitude of peak flows and changes base stream discharge (Furniss and others 1991, Harr and others 1975) and subsurface flows (Furniss and others 1991, Megahan 1972). Road-related mass soil movements can continue for decades after roads have been built (Furniss and others 1991). Such habitat alterations can adversely affect all life stages of fish, including migration, spawning, incubation, emergence, and rearing (Furniss and others 1991, Henjum and others 1994, MacDonald and others 1991, Rhodes and others 1994).

Poor road location, concentration of surface and subsurface water by cross-slope roads, inadequate road maintenance, undersized culverts, and sidecast materials all can lead to road-related mass movements (Lyons and Beschta 1983, Swanston 1971, Swanston and Swanson 1976, Wolfe 1982). Sediment production from logging roads in the Idaho batholith was 770 times higher than in undisturbed areas; about 71 percent of the increased sediment production was due to mass erosion (Megahan and Kidd (1972), leaving 29 percent due to surface erosion.

In granitic land types, sedimentation is directly proportional to the road distance (Jensen and Finn 1966). For instance, 91 percent (66,000 cubic yards) of the annual sediment production by land-use activities (72,200 cubic yards) in the South Fork of the Salmon River (Idaho) is attributed to roads and skid trails (Arnold and Lundeen 1968). King (1993) determined that roads in the Idaho batholith increase surface erosion by 220 times the natural rates per unit area. Roaded and logged watersheds in the South Fork of the Salmon River drainage also have significantly higher channel-bed substrate-embeddedness ratings than do undeveloped watersheds (Burns 1984).

Roads greatly increase the frequency of landslides, debris flow, and other mass movements (Dunne and Leopold 1978, Furniss and others 1991, Megahan and others (1992). Mass movement along the west side of the Cascade Range in Oregon was 30 to 300 times greater in roaded than in unroaded watersheds (Sidle and others 1985). Megahan and others (1992) found that 88 percent of landslides in Idaho are associated with roads. Roads were the primary factor in accelerated mass movement activity in the Zena Creek drainage (Idaho batholith) after the 1964-65 winter storms (Gonsior and Gardner 1971). Of 89 landslides examined along the South Fork of the Salmon River, 77 percent originated on road hillslopes (Jensen and Cole 1965). Cederholm and others (1981) found increases (above natural rates) in the percentage of fine sediment in fish spawning habitat when road density exceeded 2.5 percent of the Clearwater River watershed in Washington. Increased stream-channel sedimentation in Oregon and Washington watersheds east of the Cascade Range also is associated with road density (Anderson and others 1992).

Road-stream crossings can be a major source of sediment to streams and result from channel fill around culverts and subsequent road-crossing failures (Furniss and others 1991). Plugged culverts and fill-slope failures are frequent and often lead to catastrophic increases in stream channel sediment, especially on abandoned or unmaintained roads (Weaver and others 1995). Unnatural channel widths, slope, and streambed form are found upstream and downstream from stream crossings (Heede 1980), and these alterations in channel morphology may persist for long periods. Channelized stream sections resulting from riprapping roads adjacent to stream channels are directly affected by sediment from side casting, snow removal, and road grading; such activities can trigger fill-slope erosion and failures. Because improper culverts can reduce or eliminate fish passage (Belford and Gould 1989), road crossings are a common migration barrier for fish (Clancy and Reichmuth 1990, Evans and Johnston 1980, Furniss and others 1991).

Key aspects of aquatic habitat are pools and instream wood (positive attributes) and fine sediment (negative attribute). From an analysis of stream-inventory data for the Columbia River basin (Lee and others 1997), pools declined with increasing road density and were highest in wilderness areas. Relations between wood and surface fines were less clear. In Oregon and Washington, where wood frequency was measured, it was higher for Forest Service lands managed as wilderness or in areas with moderate use; it was significantly related to road density in the northern Cascades, southern Cascades, Columbia Plateau, northern glaciated mountains, and Blue Mountains but not in the Upper Klamath. Only the Lower Clark Fork and central Idaho mountains had sufficient data to model the relation of wood frequency to surface fines. In these latter two areas, the relation with road density was not significant, although the highest mean values of five sediments were associated with the highest road-density class.

Analysis of fish distribution and status data for seven species of anadromous and resident salmonids in the Columbia basin showed that the frequency of strong populations generally declined with increasing road densities. Additional analyses of road effects focused on four non-anadromous species, because effects of roads and other land uses on anadromous species may be masked by migrational and ocean-related factors (for example, dam passage, predation, harvest). Three species showed significant road effects when either occupied spawning and rearing areas were distinguished from unoccupied areas or strong status was differentiated from depressed status. The analysis suggested a decreasing likelihood of occupancy, or a decreasing likelihood of strong status if occupied, with increasing road density. No other variables except ground-slope showed the consistent patterns across all species shown by the road-density measures.

The investigation of the influence of roads on population status clearly showed an increasing absence and a decreasing proportion of strong populations with increasing road density for several subgroups of fish. Additional evidence suggested that the lowest mean road-density values (number of road miles per unit area) are always associated with strong population status.

This trend is apparent for Yellowstone cutthroat trout *(Onchorynchus clarki bouvieri)*, even though it was the only subgroup that did not show a significant road effect in a logistic regression analysis. The lack of statistical significance in the face of apparent trends, however, points to complex interactions among the explanatory variables that are not adequately addressed in the relatively simple logistic model. Consistent, significant effects for other species may be further testament to the presence and pervasiveness of the effects. Strong relations between roads and the distribution and status of these species were detected despite the potential confounding effects of other variables (such as harvest, non-native introductions, and other habitat factors).

These results show that increasing road densities and their attendant effects are associated with declines in the status of four non-anadromous salmonid species. These species are less likely to use highly roaded areas for spawning and rearing and, if found, are less likely to have strong populations. This consistent pattern is based on empirical analysis of 3,327 combinations of known species' status and subwatershed conditions, limited primarily to forested lands administered by the Forest Service and the Bureau of Land Management. The relation would not be expected to be as strong on the con-forested, lower gradient lands administered by the bureau. Of the four species examined, the redband trout is the only one supported by the low-gradient lands. Only in forested, high-elevation areas could redband trout status be clearly associated with road-density changes.

Most aquatic conservation strategies acknowledge the need to identify the best habitats and most robust populations to use as focal points from which populations can expand, adjacent habitat can be usefully rehabilitated, or the last refugia of a species can be conserved in unroaded areas where biophysical processes are still operating without effects from many human disturbances. These refugia also provide necessary experimental controls for evaluating the effects of land management activities in other areas. The ecological importance of unroaded areas has been highlighted in the Columbia basin assessment as well as other reports (FEMAT 1993, Henjum and others 1994).

The overlap of unroaded areas—both within and outside designated wilderness areas—with stronghold watersheds for fish and with important conservation watershed efforts in the Columbia basin also was examined. Designated wilderness and unroaded areas are important anchors for strongholds throughout the basin. Unroaded areas occupy 41 percent of the area with known and predicted strongholds in the east-side environmental impact statement area. One-third of this area is outside designated wilderness. Of the known and predicted strongholds in the upper Columbia basin area, 68 percent are unroaded, of which 37 percent are outside wilderness.

Aquatic integrity in the Columbia basin was analyzed in relation to road densities and integrity ratings for other resources (forest, range, hydrology). Forest clusters with the highest integrity ratings were associated with low road densities; low integrity ratings corresponded with moderate or higher road densities. For example, the range cluster with the highest aquatic and composite integrity also had mostly low road densities. But the relations between road densities and integrity ratings for other range clusters were more variable.

The legacy of road building in the Pacific Northwest is enormous. The FEMAT report (1993) notes that federally managed forest lands in the range of the northern spotted owl contain about 180 000 kilometers (111,600 miles) of roads. A major portion of this road system may constitute a potential threat to riparian and aquatic habitats through sedimentation. An estimated 250,000 stream crossings (about 1.3 per kilometer [2.3 per mile]) are associated with these roads, and a significant number of culverts are thought to be unable to withstand storms with a recurrence interval greater than 25 years (FEMAT 1993), a hypothesis tested and affirmed by the February 1996 flood. Analysis suggests more than 205 000 kilometers (127,000 miles) of roads are on Forest Service and Bureau of Land Management lands in the Columbia River basin. Many stream crossings exist, with high densities of crossings in steep, highly dissected terrain and low densities in drier and flatter terrains. Many of the culverts or stream crossings are expected to perform poorly in flood events with recurrence intervals of more than 25 years, similar to their west-side counterparts identified in the FEMAT report. Even with adequate culvert size, lack of maintenance of a road network of this size could lead to significant road-drainage problems and accompanying effects on aquatic habitat.

Budgetary constraints on land management agencies may lead to lack of maintenance, resulting in progressive degradation of road-drainage structures and functions, increased erosion rates, and the likelihood of increased erosion (Furniss and others 1991). Problems are greatest with older roads in sensitive terrain and roads functionally abandoned but not adequately configured for long-term drainage. Applying erosion prevention and control treatments to high-risk roads can drastically reduce risks for future habitat damage and can be both effective and cost-effective. In watersheds that contain high-quality habitat and have only limited road networks, large amounts of habitat can be secured with small expenditures to apply storm proofing and decommissioning activities to roads (Harr and Nichols 1993).

For federal forests with moderate to high road densities, the job of maintaining roads may be expensive because many road networks have not been inventoried to determine their influence on riparian or aquatic resource goals and objectives. Substantial increases in sedimentation are unavoidable even when the most cautious road-building methods are used (McCashion and Rice 1983, Megahan 1980). Improving road-building and logging methods, however, can reduce erosion rates and sediment delivery to streams. The amount of sedimentation or hydrologic alteration from roads that aquatic species can tolerate before a negative response appears is not well known, though general effects of sediments on fishes are known. Sediment exceeding natural background loads can fill pools, silt spawning gravels, decrease channel stability, modify channel morphology, and reduce survival of emerging salmon fry (Burton and others 1993, Everest and others 1987, MacDonald and others 1991, Meehan 1991, Rhodes and others 1994).

Rice (1992) documents an 80-percent reduction in mass erosion from forest roads and about a 40-percent reduction in mass erosion from logged areas in northern California that resulted from improvements in forest practices beginning in the mid-1970s. Megahan and others (1992) used the BOISED sediment-yield production model to evaluate the effects of historical and alternative land management in an Idaho watershed (South Fork Salmon River). They report that current management practices, properly implemented, could reduce sediment yield by about 45 to 90 percent when compared with yields caused by the historical land use in their study watershed. If the improved road design currently practiced by the Boise National Forest is used, however, total accelerated sediment yields are still 51 percent more than natural ones. These improved road designs plus maximum erosion mitigation lead to 24-percent increases over natural yields in unroaded areas. Helicopter logging results in 3-percent increases over natural yields, and wildfire increases sediment yield about 12 percent over natural loads (Megahan and others 1992).

Megahan and others (1995) evaluated the effects of helicopter logging and prescribed burning on south-facing slopes of headwater drainages in the Idaho batholith by using paired watersheds monitored from 1966 to 1986. Average annual sediment yields show a statistically significant increase of 97 percent persisting for the 10 years of posttreatment study after logging and burning. Accelerated surface erosion primarily result from the prescribed burning, not the helicopter logging, because burning results in most of the bare-soil exposure and in connecting the affected area to streams. Surface erosion rates in the logged and burned areas are about 66 times greater than those on undisturbed slopes. The conclusion is that current best management practices can reduce sediment yields compared with historical practices. But the risk of increased sedimentation from forest management continues, particularly with such activities as road building, timber harvest, and prescribed burning.

Temporary roads may have fewer adverse effects than do permanent roads, depending on the extent to which they are decommissioned. As indicated by the analyses for the Columbia basin, distinguishing the direct effects of roads from the cumulative effects of other activities associated with roads is sometimes difficult. Thus, temporary roads may reduce the direct effects of roads, but effects of activities for which the temporary roads were built still will affect the environment.

Reliability, confidence, and limitations—The relations among roads, aquatic species and their habitats, and other variables analyzed for the Columbia basin were developed from predicted road density data developed from actual subsampled road data and a rule-based model. The method used in developing road density classes is not a substitute for actually mapping roads, but the rule-based model approach provides a tool for predicting road densities across a large landscape, when existing road data are incomplete or out of date. Also, the rule-based model assures that the method used in developing road densities is consistent throughout the Columbia basin. The final road density model had inherent uncertainties because of incomplete data layers, limitations of the sampling design, and the limitations of a rule-based model.

A few road types could not be predicted by using this rule-based approach, despite its general utility. For instance, Yellowstone National Park was assigned a road density class of *none* because no unique rule-based model combinations existed for predicting the park's road system. Roads inside the park are based on human recreational interests, which were not accounted for in the model.

Generalizability—Because the Columbia basin assessment was designed specifically as a broad-scale analysis, the relation of roads and aquatic species and their habitats can be applied at the large-landscape scale. Those relations may not be the same for federally managed lands outside the Pacific Northwest, particularly the Columbia basin, although aquatic habitat loss and alterations, which include effects of roads, are associated with the decline of many fish species throughout North America (Miller and others 1989). Those general relations also may differ at finer scales because of specific biophysical characteristics, such as geology and soils, and use of actual rather than predicted road densities.

The declines in population status of non-anadromous salmonids in the Columbia basin should be viewed as indicating the types of responses that may be experienced by other native aquatic species in similar habitats. The species most like the non-anadromous salmonids in distribution or habitat requirements would be expected to show the most similar responses. This group would include the anadromous species—such as steelhead, stream chinook salmon, and Pacific lamprey—that broadly overlap in range with the non-anadromous salmonids and use many of the same habitats for significant portions of their life. No logical reasons exist to expect anadromous fishes to be immune to the effects of habitat change evident in the non-anadromous species. The ranges of other species—including sculpins, dace, and some suckers—also overlap considerably, and these species may follow similar trends in population abundance and distribution.

Although unroaded areas are significantly more likely to support strong populations, strong populations are not excluded from roaded watersheds. Several possible reasons for this coexistence have been suggested: The inherent productivity of some areas allows fish populations to persist despite disturbances linked to roads; real or detectable effects on fish populations may lag behind the initial physical effects in watersheds where roads have been built in the last several years; and the scale of the subwatershed (19,800 acres on average) at which strong populations are identified may mask a potential disconnect between the real locations of strongholds and roads (which are identified at 1-square-kilometer [0.39-square-mile] pixels). This issue of scale can be resolved with a midscale or subwatershed analysis. The fact that strong salmonid populations can coexist in many roaded areas provides opportunities to determine the reasons, which may be instructive for both watershed restoration and future road building. Given current information, the assumption that because roads and strong fish populations coexist in some watersheds, they will in others is not prudent, however. In general, greater short- or long-term watershed and ecological risks are associated with entering an unroaded area than with proceeding cautiously with management activities in roaded areas to close and obliterate existing roads. The data strongly suggest a closer examination of the stronghold subwatersheds and their roaded condition.

Secondary links—The effects associated with roads reach beyond their direct contribution to disruption of hydrologic function and increased sediment delivery to streams. Roads provide access, and the activities that accompany access magnify the negative effects on aquatic systems beyond those caused solely by the roads themselves. Activities associated with roads include fishing, recreation, timber harvest, livestock grazing, and agriculture. Roads also provide avenues for stocking non-native fishes.

Unfortunately, inadequate broad-scale information on many of these attendant effects for the Columbia basin prevents identification of their component contributions. Similarly detailed analyses are needed to address the relations between roads and fish at a landscape scale in other ecoregions.

Conclusions—The range of specific case studies for broad-scale assessment of road relations in the Columbia basin provides a substantial base of information on which to evaluate the direct effects of roads and the cumulative effects of activities associated with roads on aquatic habitats and species in the Northwest.

Terrestrial Vertebrates

Issue—Effects of roads on vertebrate populations act along three lines: direct effects, such as habitat loss and fragmentation; road use effects, such as traffic causing vertebrate avoidance or road kill; and additional facilitation effects, such as overhunting or overtrapping, which can increase with road access.

Findings—In recent research in the interior Columbia River basin, Wisdom and others (2000) identify more than 65 species of terrestrial vertebrates negatively affected by many factors associated with roads. Specific factors include habitat loss and fragmentation, negative edge effects, reduced densities of snags and logs, overhunting, overtrapping, poaching, collection, disturbance, collisions, movement barriers, displacement or avoidance, and chronic, negative interactions with people. These factors and their effects on vertebrates in relation to roads are summarized from Wisdom and others (2000) as follows:

Road construction converts large areas of habitat to nonhabitat (Forman 2000, Hann and others 1997, Reed and others 1996); the resulting motorized traffic facilitates the spread of exotic plants and animals, further reducing quality of habitat for native flora and fauna (Bennett 1991, Hann and others 1997). Roads also create habitat edge (Mader 1984, Reed and others 1996); increased edge changes habitat in favor of species that use edges, and to the detriment of species that avoid edges or experience increased mortality near or along edges (Marcot and others 1994).

Species dependent on large trees, snags, or logs, particularly cavity-using birds and mammals, are vulnerable to increased harvest of these structures along roads (Hann and others 1997). Motorized access facilitates firewood cutting, as well as commercial harvest, of these structures.

Several large mammals are vulnerable to poaching, such as caribou, pronghorn antelope, mountain goat, bighorn sheep, wolf, and grizzly bear (Autenrieth 1978, Bruns, 1977, Chadwick 1973, Dood and others 1986, Greer 1985, Gullison and Hardner 1993, Horejsi 1989, Knight and others 1988, Lloyd and Fleck 1977, Luce and Cundy 1994, Mattson 1990, McLellan 1990, McLellan and Shackleton 1988, Mech 1970, Scott and Servheen 1985, Singer 1978, Thiel 1993, Van Ballenberghe and others 1975, Yoakum 1978). Roads facilitate this poaching (Cole and others 1997).

Gray wolf and grizzly bear experience chronic, negative interactions with humans, and roads are a key facilitator of such interactions (Mace and others 1996, Mattson and others 1992, Thiel 1985). Repeated, negative interactions of these two species with humans increases mortality of both species and often causes high-quality habitats near roads to function as population sinks (Mattson and others 1996a, 1996b; Mech 1973).

Carnivorous mammals such as marten (*Martes americana*), fisher (*M. pennanti*), lynx (*Lynx canadensis*), and wolverine (*Gulo luscus*) are vulnerable to overtrapping (Bailey and others 1986, Banci 1994, Coulter 1966, Fortin and Cantin 1994, Hodgman and others 1994, Hornocker and Hash 1981, Jones 1991, Parker and others 1983, Thompson 1994, Witmer and others 1998), and overtrapping can be facilitated by road access (Bailey and others 1986, Hodgman and others 1994, Terra-Berns and others 1997, Witmer and others 1998). Movement and dispersal of some of these species also is believed to be inhibited by high rates of traffic on highways (Ruediger 1996), but this has not been validated. Carnivorous mammals such as lynx also are vulnerable to increased mortality from highway encounters with motorized vehicles (as summarized by Terra-Berns and others 1997).

Reptiles seek roads for thermal cooling and heating, and in doing so, these species experience significant, chronic mortality from motorized vehicles (Vestjens 1973). Highways and other roads with moderate to high rates of motorized traffic may function as population sinks for many species of reptiles, resulting in reduced population size and increased isolation of populations (Bennett 1991). In Australia, for example, 5 million reptiles and frogs are estimated to be killed annually by motorized vehicles on roads (Ehmann and Cogger 1985, as cited by Bennett 1991). Roads also facilitate human access into habitats for collecting and killing reptiles.

Many species are sensitive to harassment or human presence, which often are facilitated by road access; potential reductions in productivity, increases in energy expenditures, or displace-ments in population distribution or habitat use can occur (Bennett 1991, Mader 1984). Exam-ples of such road-associated effects are human disturbance of leks (sage grouse [*Centrocercus urophasianus*] and sharp-tailed grouse [*Tympanuchus phasianellus*]), nests (ferruginous hawk [*Buteo regalis*]), and dens (kit fox [*Vulpes macrotis*]). Another example is elk avoidance of large areas near roads open to traffic (Lyon 1983, Rowland and others 2000), with elk avoidance increasing with increasing rate of traffic (Wisdom and others 2000, Johnson and others 2000).

Bats are vulnerable to disturbance and displacement caused by human activities in caves, mines, and on rock faces (Hill and Smith 1984, Nagorsen and Brigham 1993). Cave or mine exploration and rock climbing are examples of recreation that could reduce population fitness of bats that roost in these sites (Nagorsen and Brigham 1993, Tuttle 1988). Such activities may be facilitated by human developments and road access (Hill and Smith 1984).

Ground squirrels often are targets of recreational shooting (plinking), which is facilitated by human developments and road access (Ingles 1965). Many species of ground squirrels are local endemics; these small, isolated populations may be especially vulnerable to recreational shooting and potentially severe reductions or local extirpations of populations.

Roads often restrict the movements of small mammals (Mader 1984, Merriam and others 1988, Swihart and Slade 1984), and consequently can function as barriers to population dispersal and movement by some species (Oxley and Fenton 1974).

Many granivorous birds are attracted to grains and seeds along roadsides and as a result have high mortality from collisions with vehicles (Vestjens 1973). And pine siskens (*Carduelis pinus*) and white-winged crossbills (*Loxia leucoptera),* for example, are attracted to road salt, which can result in mortality from vehicle collisions (Ehrlich and others 1988).

Terrestrial vertebrates inhabiting areas near roads accumulate lead and other toxins that originate from motorized vehicles, with potentially lethal but largely undocumented effects (Bennett 1991).

In summary, no terrestrial vertebrate taxa seem immune to the myriad of road-associ-ated factors that can degrade habitat or increase mortality. These multifaceted effects have strong management implications for landscapes characterized by moderate to high densities of roads. In such landscapes, habitats are likely underused by many species that are negatively affected by road-associated factors. Moderate or high densities of roads sometimes index areas that function as population sinks that otherwise would function as source environments were road density low or zero.

Reliability, confidence, and limitations—General effects of roads and road-associated factors on a wide variety of vertebrate taxa are well documented from a broad range of studies conducted in North America, Europe, and other areas (Bennett 1991, Forman and Alexander 1998, Mader 1984, Trombulak and Frissell 2000, Vestjens 1973). Reliability of such effects at large, landscape scales, and for many taxa, is compelling and unequivocal. Reliability of site-specific, small-scale effects, with focus on single species, is less certain. For many species at local scales, the array of factors that could affect habitats or populations have been neither well studied nor documented. Despite such limitations, current knowledge of broad-scale effects on a variety of taxa is highly certain and provides an overarching paradigm from which likely or presumed effects on single species at local scales can be inferred. The many factors associated with roads suggests that mitigating such effects succeeds best at large scales, when focused on multiple species, and when based on a combination of aggressive road obliteration and protection of roadless areas (Trombulak and Frissell 2000).

Generalizability—Although the summary of road-associated effects on vertebrates described here is taken from research conducted in the interior Columbia River basin (Wisdom and others 2000), results likely apply to several species occupying a diversity of forest and rangeland environments in North America. At least four reasons account for this presumed high generalizability: the road and road-associated effects described by Wisdom and others (2000) were synthesized from research conducted across the world; the synthesis focused on multiple species encompassing diverse taxa and environmental requirements; the synthesis addressed an extreme range of environmental conditions on federal lands administered by the Forest Service, the Bureau of Land Management, and state, private, and tribal landowners; and the synthesis focused on large-scale, overarching effects common to many species and conditions.

Secondary links—Many road-associated effects on terrestrial vertebrates are intimately linked to managing human activities related to road access. Accordingly, mitigation of road-use effects requires effective control of human access to roads related to managing livestock, timber, recreation, hunting, trapping, and mineral development.

Conclusions—Comprehensive mitigation of the full array of road-associated effects on terrestrial vertebrates of conservation concern poses one of the most serious of land management challenges. Balancing such mitigation with socioeconomic desires will be controversial and contentious. Comprehensive efforts to mitigate road-associated effects on terrestrial vertebrates is well suited to testing as a large-scale management experiment developed and implemented jointly by managers, researchers, and the public.

Road Kill

Issues—Large numbers of animals are killed annually on roads. In selected situations, such as for some amphibians with highly restricted home ranges, populations of rare animals may be reduced to dangerous sizes by road kills.

Findings—An estimated 1 million vertebrates a day are killed on roads in the United States (Lalo 1987). Studies show that the number of collisions between animals and vehicles is directly related to the position of the nearest resting and feeding sites (Carbaugh and others 1975). Because most forest roads are not designed for high-speed travel, and the speed of the traffic is directly related to the rate of mortality, direct mortality on forest roads is not usually an important consideration for large mammals

(Lyon 1985). An exception is forest carnivores, which are especially vulnerable to road mortality because they have large home ranges that often include road crossings (Baker and Knight 2000). Forest roads pose a greater hazard to small, slowly moving, migratory animals, such as amphibians, making them highly vulnerable as they cross even narrow forest roads (Langton 1989). Nearly all species of reptiles use roads for cooling and heating, so many of them are killed by vehicles. Highways and other roads with moderate- to high-speed traffic function as population sinks for many species of reptiles, resulting in reduced and increasingly isolated populations (Wisdom and others 2000). Predators and scavengers are killed while they feed on road-killed wildlife, as are other species attracted to roads because of salts or vegetation, or because roads facilitate winter travel (Baker and Knight 2000). Although countless animals are killed on roads every year, documented road-kill rates are significant in reducing populations of only a few rare species in North America, and these kills generally are on high-speed highways (Forman and others 1997).

Reliability, confidence, and limitations—A large body of data documents annual road kill, and wildlife science can describe the factors that put wildlife at risk, but little research has focused on how to mitigate the effects on wildlife populations.

Generalizability—Most road-kill questions will be related to individual species and geographic sites, but general principles such as the frequency of travel between known resting and feeding areas for individual species can be used in project decisions.

Secondary links—Road-kill issues link to habitat fragmentation, predation, and access issues.

Conclusions—The issues can be addressed based on site and species. Difficulty will arise in integrating road kill with the social and economic issues related to mitigation.

Forest Diseases

Issues—In general, the existence of roads seems to have little effect on forest tree diseases, but there are some examples where building or using roads caused significant local effects. Nearly always, the negative effects can be ameliorated through simple modifications in how they are built and used. The one benefit of roads, as it pertains to tree diseases, is to provide access for silvicultural activities that protect resources, such as the ability to inoculate decay fungi into trees to create wildlife habitat (Bull and others 1997). One negative effect includes the movement of people on the roads, which allows the pests to be introduced. Road building also may set the stage for an insect attack that further stresses the trees and then a disease outbreak that kills them (Boyce 1961).

Findings—A significant forest disease problem associated with roads is Port-Orford-cedar root disease. This disease of Port-Orford-cedar (*Chamaecyparis lawsoniana* (A. Murr.) Parl.) is a root disease caused by the fungus *Phytophthora lateralis*. Spores of the fungus are carried in water or contaminated soil to uninfected areas. Roads of any sort in the very limited geographic range of the primary host provide a way to move soil—along with the fungus—from infected to uninfected areas. Spread of the fungus can be checked by careful planning to reduce entry to uninfected areas, road closures, partial road closures during wet weather, attention to road surfaces and drainage of possibly contaminated water to streams, wash stations to remove soil from vehicles before entry to uninfected areas, and sanitation strips to remove host plants from near roadsides (Kliejunas 1994, Roth and others 1987, Zobel and others 1985). Building and maintaining roads may exacerbate root diseases. Wounded trees and conifer stumps created and not removed during road building provide infection courts for annosus root disease; the disease may then spread through root contacts to kill a patch of trees

(Otrosina and Scharpf 1989). Trees damaged or stressed by road building—through direct wounding of stems and roots, covering of roots with side castings, or compacting of soil over roots—become susceptible to various tree diseases. Armillaria root disease is benign in deciduous stands where only injured trees are attacked but more serious in conifer stands where pockets of disease are initiated (Shaw and Kile 1991). Oak decline is associated with poor sites, older stands, and road building or other disturbance (Wargo and others 1983). Black stain root disease (*Leptographium wagneri*) attacks stressed conifers associated with disturbance, especially compaction caused by road building; in pinyon pine (*Pinus monophylla*), it is associated with roads and campsites (Hansen 1978, Hansen and others 1988, Hessburg and others 1995). Droopy aspen disease is associated with road building and compaction, but the pathogen identity is unknown (Jacobi and others 1990, Livingston and others 1979). Sap streak disease in sugar maple is associated with compaction from roads and from direct injury to trees (Houston 1993).

Road building can be planned to help reduce the spread of some forest tree diseases: mistletoe is spread by the forcible ejection of the mistletoe seeds. In young plantations or pole-sized stands, roads can subdivide an area to prevent mistletoe seeds from reaching a healthy stand (Hawksworth and Wiens 1996). In Texas, roads could be planned to separate a portion of a stand with oak wilt from healthy trees. The act of building the road (if extensive enough) severs root connections and prevents tree-to-tree movement of the pathogen (Appel and others 1995, Rexrode and Brown 1983). In other areas, new or established roads may have the unintended effect of breaking the continuity of host roots and thus halting the spread of laminated root rot (*Phellinus weirii*) and other root diseases (Hadfield 1986, Thies and Sturrock 1995).

Roads indirectly contribute to disease spread by giving people access to remote forests and ways to transport material long distances. New pockets of both oak wilt and beech bark disease (Houston and O'Brien 1983) may have resulted from moving firewood from the forest to a homesite (Appel and others 1995, Rexrode and Brown 1983). Pitch canker (*Fusarium subglutinans*) was recently reported on Monterey pine (*Pinus radiata*) in California; previously, it had been found on little-leaf and slash pines in the South. A single introduction is thought to be responsible; 117 vegetative compatibility groups are found in Florida but only 5 in California, and 70 percent of the isolations in California are from a single group, likely carried on a tree transported as an ornamental (Correll and others 1992, Storer and others 1995). Campers who use roads to get to remote sites in Colorado and other states have caused significant mortality by carving on aspen and birch, which provides pathways for various fungi that cause cankers and quickly kill the trees. Many trees are unintentionally damaged, for example, when campers hang a gas lantern on a branch too close to the trunk of a tree, thereby causing heat damage.

One abiotic disease has caused significant damage. In the Lake Tahoe basin in California, trees were killed by salt put on the roads to reduce ice. This problem also has appeared in some areas of the Midwest and east coast (Kliejunas and others 1989, Scharpf 1993, Scharpf and Srago 1974). Needle and rust diseases spread long distances by spores and do not appear to be influenced by roads or road building.

Reliability, confidence, and limitations—Field studies tend to focus on a single disease or an insect-disease complex; many of these centers are associated with or influenced by compaction or tree damage associated with roads.

Generalizability—Problems, where they exist, appear to be specific to the pathogen, host, and site.

Conclusions—In general, land managers appear to have the information and technology needed to handle most road, road building, and disease interactions. Additional science-based information is needed to understand and manage the interactions between compaction and black stain root disease and between compaction and droopy aspen disease.

Predation

Issues—The introduction of roads into the closed forest environment creates corridors by which predators can enter and affect native populations.

Findings—Forest roads create corridors by which predators, especially people, can enter the forest environment and affect wildlife populations. Nest depredation of songbirds may increase by predators attracted to edges. Evidence for edge effects, however, is highly variable (Paton 1994). Although evidence has been found for local edge effects in cowbird parasitism and nest depredation, their effects on bird populations is not documented. Geographic location and large-scale patterns in the amount of forest and nonforest habitats may be more important in determining the reproductive success of forest songbirds (Donovan and others 1997, Robinson and others 1995). Forest carnivores apparently travel on roads in winter when snow is deep, and thus the road system alters and enhances their ability to move (Paquet and Callaghan 1996). Wolves and grizzly bears are two key species that have chronic, negative interactions with people, and roads are a key facilitator. Repeated, negative interactions of these two species with people increase mortality of both species and often cause high-quality habitats near roads to be population sinks (Wisdom and others 2000). High road densities are associated with a variety of negative human effects on several wildlife species (Brocke and others 1988). People directly affect snakes by collecting, harassing, and killing them (Wisdom and others 2000). Increases in illegal hunting pressure, facilitated by roads, also negatively affect populations. Moose, wolves, caribou, pronghorn antelop, mountain goat, and bighorn sheep are particularly vulnerable to this kind of predation (Lyon 1985, Wisdom and others 2000).

Reliability, confidence, and limitations—Limited data exist on the effects of introducing natural predators as a result of road building. The evidence is strong that human predation, either legally in game management programs or illegally, is greatly facilitated by roads and can significantly affect populations of animals.

Generalizability—General principles related to human effects on wildlife populations are understood by wildlife managers and can be applied to species and site-specific management.

Secondary links—Predation links to other habitat-related topics, such as fragmentation and road kill, and also to people-related topics such as recreation.

Conclusions—Species-specific issues related to predation facilitated by roads can be addressed for specific sites. Predation related to illegal hunting facilitated by improved access can be addressed by legal measures, or, where legal remedies are ineffective, by closing or decommissioning roads where wildlife values are high.

Biodiversity and Conservation

Issues—Previous issues in this section may be synthesized by the concept of biodiversity. Biodiversity is, in simplest terms, the variety of life and its processes (Keystone Center 1991). Recent syntheses (Heywood and Watson 1995) emphasize the reciprocal relation between biodiversity—conceived as genetic and species diversity—and ecosystem function. The many species comprising the biodiversity of an area play roles essential to ecosystem function and are the source of variation that enables an ecosystem to adapt to change. The healthy, functioning ecosystem, in turn, supports the many species living within it. Appreciating this reciprocity means that biodiversity can be taken as a natural measure of the ecosystem as a whole and thus can integrate the many concerns listed.

Some species may play more important roles than others in the normal functioning of an ecosystem. For example, keystone species may define the major structural elements of an ecosystem, as Douglas-fir does for forests in the Pacific Northwest, or they may—by virtue of their position in a complex trophic structure—act to maintain the diversity as keystone predators do for herbivores. On the other hand, the many species that do not appear to serve an important role in an ecosystem constitute a reservoir of potential adaptation to change. Because an ecosystem cannot predict change, the diversity of species acts as a hedge against it.

Biodiversity is vital to long-term ecosystem function, and human activities that decrease biodiversity can impair it. Our working hypothesis, then, is that measures of biodiversity provide the best integrative assessment of the effects of roads on ecosystems.

Findings—Roads can have major adverse effects on biodiversity, many of which are already described (Forman and Collinge 1996). A recent review by Forman and Hersperger (1996) usefully distinguishes these aspects of the road-biodiversity interaction:

- Road density: As road density increases, thresholds may be passed that cause some species to go locally extinct. The probability of extinction depends, in part, on body size, with larger animals requiring larger residual populations to prevent their extinction.
- Road-effect zone: The effects of roads can extend over some distance from their centers, such that their "effective widths" can be many times their actual widths.

Reliability, confidence, and limitation—The confidence in the general negative relation between roads and biodiversity is high. The current primary limitation, however, is on the utility of measures of biodiversity for assessing road effects. First, both the status of keystone and other important species must be assessed, which seems fairly straightforward. But, second, the status of the pool of all the other species that form the basis for adaptation to change must be assessed, and how to do this assessment is much less clear.

Landscape ecology as well as fragmentation and viability analysis contain relevant scientific uncertainties. Two critical uncertainties must be resolved to understand how roads affect fragmentation and population viability. First, in the mechanistic analysis of the effects of roads and roadlike entities, such as power lines, on landscape fragmentation and species viability, the question of the "effective width" of roads is open. Kiester and Slatkin (1974) predict that, for species using conspecific cuing for movement strategies and habitat selection (likely most vertebrates), a spatially localized source of mortality in an area of otherwise suitable habitat can act as an active sink, drawing individuals in as residents die, making it likely that the new individuals will die as well. Consider a road traversing the habitat of a territorial or conspecific-cuing species. Those individuals whose home range overlaps a road have some probability of being hit each time they venture across it. Eventually they are killed, and their neighbors, in the process of constantly testing the boundaries of their home ranges, move into the vacated area next to the road and themselves run the risk of road mortality. The question is, How far from a road does this probability of mortality spread? Second, at the landscape scale, the relation between patterns of dispersal of individual species and measurements of fragmentation must be clarified. Current information (Schumaker 1996) indicates that most of the commonly used measures of fragmentation do not predict habitat connectivity for individual endangered species; rather, a model of fragmentation must be derived from species-specific dispersal characteristics. This kind of analysis is now available for only a few species.

Generalizability—Exactly how roads affect biodiversity in any particular place is a matter of the devil being in the details. The results given here would generally apply to any area.

Secondary links—Appreciation of biodiversity itself is an important part of the passive-use value of biodiversity. In particular, the aesthetic appreciation of biodiversity through an understanding of how biodiversity is sublime (rather than just beautiful) is now leading to a new link between biodiversity and passive-use value (Kiester 1997).

Conclusions—Forman and Hersperger (1996) conclude " ...that a quantum leap in focus on the ecological effects of roads is warranted, and that the foundations are in place for effective research, planning, public education, and action."

Water Quality

Issues—Roads provide access to and increase the opportunity for applying a variety of chemicals in national forests. Some applications target the roads, such as with road surface treatment; other chemicals are intended for adjacent ecosystems to control pests and fertilize vegetation. Materials also are added to roads by traffic, such as asbestos from brake linings, oil leakage, and accidental spills. Some portion of applied and spilled chemicals eventually reaches streams by drift, runoff, leaching, or adsorption on soil particles. Roads also increase the nutrient delivery to streams by removing vegetation, rerouting water flow paths, and increasing sediment delivery. And roads increase the likelihood of toxic spills associated with accidents along streamside corridors.

Findings—Chemicals applied on and adjacent to roads can enter streams by various pathways. The likelihood of water-quality deterioration from ground applications is a function of how much chemical is applied, the proximity of the road to a stream, and the rainfall, snowmelt, and wind events that drive chemical and sediment movement. The risk is a function of the likelihood of water-quality deterioration and exposure of organisms, including people, and how susceptible the organisms are to the pollutant or pollutants. (A large proportion of Forest Service roads are low standard and few if any chemicals are applied, so the risk of chemical contamination for most Forest Service roads is relatively low.) Chemicals are applied directly to roads and adjacent rights-of-way for various purposes, including dust abatement, stabilizing the road surface, deicing, fertilizing to stimulate plant growth on road cuts and fills, and controlling weeds and the invasion of nonweedy plants onto the roadway (Furniss and others 1991, Norris and others 1991, Rhodes and others 1994). Applied chemicals can enter streams directly when they are applied, but little is known about the effects of these chemicals on stream biota (Furniss and others 1991). Norris and others (1991) provide a comprehensive review of the types and amounts of fertilizers, pesticides, and fire retardants applied to forests in the United States, although little information is given to distinguish road-related from aerial applications. They report that most herbicides are applied by ground-based equipment, presumably using roads for access; that ground-based applications in or near aquatic zones can result in chemicals entering streams by drift or direct application; and that these problems are more serious when the chemicals are applied from the air. Movement of sediment containing adsorbed chemicals is possible, and the risk increases with increasing persistence (Norris and others 1991). The amount of input by this pathway is thought to be small, however; it is a more likely pathway for entry of salts applied for de-icing and of fertilizers applied to road fills.

Increased nutrient supply to streams from roads is proportional to the area disturbed and maintained free of vegetation and the amount of sediment delivered. Increased nutrients rarely have detrimental effects on stream water quality, but they may modify the composition of aquatic biota (Hawkins and others, in press). Few studies examining watershed responses to logging separate the effect of road building from those of the broader disturbance associated with removing timber. In one such study, Swank (1988) monitored stream chemical composition during the pretreatment, road building, logging, and posttreatment phases in a cable-logged watershed in the southern Appalachian Mountains. No stream chemical response was found to result from the road-building phase of the watershed treatment. Nutrient movement to streams often increases significantly after timber harvest operations (Frederiksen and others 1973, Hornbeck and others 1973, Likens and others 1970, Pierce and others 1972, Swank and Waide 1988). The primary intent of these studies was to assess onsite nutrient losses, with changes in water quality a secondary concern. All cited studies report increases in nitrogen cation and phosphorus concentrations in streams after treatment. In general, nutrient loss to streams is roughly proportional to how much vegetation was removed. For example, three studies at Hubbard Brook in New Hampshire compared three treatments: clearcutting with a herbicide treatment to suppress vegetation regrowth (Likens and others 1970), clearcutting without suppressing regrowth (Pierce and others 1972), and strip cutting of one-third of the forest (Hornbeck and others 1973); the three studies found nitrogen concentrations in streams reduced, most by the first treatment, less by the second, and least by the third. These findings suggest that residual or reestablished vegetation immobilizes released nutrients, thus diminishing the disturbance effect. Although roads might not respond in the same way because of drainage rerouting, we expect that nutrient mobility is proportional to the area maintained in a disturbed, non-revegetated state.

Hazardous chemical spills from vehicle accidents can pose a direct, acute threat of contamination to streams. The risk of hazardous chemical spills resulting from vehicle accidents adjacent to waterways is recognized and documented by the National Forest System and by state transportation departments (IDT 1996). Risk-analysis models of accident-related chemical spills are available, but they are designed for paved roads in nonmountainous terrain. Models take into account risk to human health, traffic frequency, vehicle type, and proximity to water. Possible contaminants include any substance being transported, such as fuel, pesticides, chemicals used in mining, fertilizers, and fire retardants.

Reliability, confidence, and limitations—Both anecdotal and scientific bases for linking increased access provided by roads to increased use of a wide variety of introduced chemicals are strong. Potential delivery to streams is mainly anecdotal, and few models are available for predicting delivery. Evidence for increased nutrient delivery to streams from disturbance by roads is strong, but it is confounded by other management activities such as logging.

Generalizability—The use of chemicals that are potential contaminants is well known and often described. The likelihood of routinely or accidentally spilled chemicals is related to type and frequency of traffic, but determining probabilities of spills accurately is difficult or impossible, especially for accidents. The likelihood of contaminants reaching a stream differs widely from site to site; it is most strongly controlled by stream proximity and road drainage features. Soluble and persistent elements and compounds adsorbed on sediment particles have increased probability of contaminating waterways.

Secondary links—Roads have strong links to aquatic health and biological response. A large body of literature exists on bioassays, but little information is available on transport, toxicity, and persistence of potential contaminants in natural systems. Terrestrial effects of chemicals, such as damage to vegetation by road salt, are not addressed here.

Conclusions—Most of the information is anecdotal or requires extrapolation from other studies (nutrient issues). The degree to which aquatic organisms are affected by applied and routinely spilled chemicals is poorly known or not understood in most places. Better information on effects is needed to make decisions about chemical application, road drainage control, and road location. Better models of chemical spill risks on forested roads are needed.

Air Quality

Issues—Dust emitted into the atmosphere by vehicles moving on unpaved roads contributes to reducing visibility and to suspending airborne particulates that can pose health hazards. Issues revolve around the contribution of national forest roads to regional and urban air pollution and what effects maintaining, paving, and shutting down roads on national forests have on this problem. Roads built into or surfaced with serpentinitic rock may contain asbestos-type minerals that could pose a hazard to people exposed to dust from the road surface.

Findings—Scientific literature on this topic is scarce. A study of degraded visibility and its causes in 16 national parks and wilderness areas on the Colorado Plateau, by the Grand Canyon Visibility Transport Commission (available online at http://www.nmia.com/gcvtc/), found that dust from unpaved roads could be a contributing factor. Soils in the Southwest are often very fine textured, and once dust is made airborne by vehicles, it can remain suspended for a long time and be transported long distances by the wind. The commission recommended that the Environmental Protection Agency (EPA) require further study and mitigation of these effects.

The amount of dust emitted into the atmosphere is estimated by a formula that considers the number and speed of vehicles traveling on a road in a given period, the relative humidity, and the composition of the road surface. This model was developed and reviewed by the Department of Transportation and the EPA. Related information about calculations for paved roads can be found at http://www.epa.gov/ttn/chief/ap42/ch13/related/c13s02-1.html.

Dust emissions also raise issues of human health. Where national forests are close to urban areas, dust from national forest roads can contribute to the burden of airborne particulate matter from a wide variety of sources including transportation and industrial activities. The fine fraction of airborne particles with diameters less than 2 microns have been found to contribute to human health problems and increased mortality, especially in young children, old people, and people with lung problems such as asthma and emphysema. Particles of this size and smaller cannot be effectively cleared by human lungs and therefore accumulate. How much road dust from forest roads contributes to the fine particulates in urban atmospheres is not currently known for most cities because the EPA is just beginning wide-spread monitoring of fine particulates, and reliable results will take at least 3 years to gather.

Unpaved roads built into or surfaced with serpentine materials can generate dust containing asbestos or asbestiform minerals. Although few such roads exist, methods have been developed to determine the extent of ambient asbestos coming from them.

During commercial use of unsurfaced roads, watering or other dust-abatement treatment (such as the addition of lignin sulfonate or calcium chloride) is often required by the Forest Service or other road manager to reduce dust emissions and conserve the fine fraction of the road surface. Such treatments do not accompany noncommercial uses, however, and they include most of the traffic for such roads.

The EPA has proposed a regional haze rule calling for more regions to do the kind of analysis done by the Grand Canyon Commission. Such analyses are likely to find similar emissions from unpaved roads and similar visibility problems elsewhere. EPA's recent tightening of the National Ambient Air Quality Standard on the effects of fine particles on human health are likely to require similar analyses of particle emissions, especially as they affect urban air quality. Analyzing the entire transportation system, including national forest roads, would be a logical approach to finding the most efficient means of controlling air pollution. Under emissions-trading scenarios, treatments, like paving or closure to reduce emissions of particles from national forest roads might qualify for highway funds, as cost-effective adjuncts to upgrading major arterials to reduce air pollution.

Reliability, confidence, and limitations—The basic models of dust emission and transport down-wind are generally reliable and widely used by the EPA in regulatory decisions. Much of the basic data to make these calculations for national forest roads have not been collected; thus, most estimates of the emissions are based on very coarse estimates of the conditions that produce dust emissions. Effects of the amount of road maintenance on emissions also are not well understood. The effects of road closures on dust emissions are not easily predicted because they depend on the details of how traffic is rerouted from closed sections and what emissions are created by the rerouted traffic pattern.

Generalizability—Models of emissions are relatively easy to generalize to many parts of the country, if reliable data are collected to use in them.

Secondary links—Reductions in visibility negatively affect recreational values because beauty is one of the major attractions to national forest visitors. Improving national forest roads to reduce dust emissions could be linked to regional transportation plans aimed at reducing air pollution. Such a link might make Forest Service roads eligible for highway funds.

Conclusions—Emissions from national forest roads would need to be included in regional analyses of air emissions. Models to make these analyses are available, but data to represent national forest roads would have to be collected and included in the analysis.

Direct Socioeconomic Effects

Timber Programs

Issues—Road closures are expected to strongly affect Forest Service timber programs. On federal timberlands, the timber program and an extensive road network evolved simultaneously. Many roads were built by purchasers or with purchaser credits from timber sales, but these roads served a variety of users. By the late 1980s, about 25,000 timber sales were recorded per year (of more than $300) supplying 14 percent of the U.S. timber harvest. This harvest supported some 125,000 direct jobs in many communities, mostly in the Western United States. By 1997, the proportion of total U.S. harvest supplied from federal lands had dropped by half because of efforts to protect various habitats for species at risk of extinction.

Along with the evolution of the existing road network went the development of logging systems designed for site conditions, soil-compaction concerns, and costs. Such systems (except for some forwarder systems) are designed to minimize skid distances, both in harvest units and at road-based landings. The most commonly used logging systems (cable yarding or ground-based skidding systems) depend on direct access to a stand. Helicopter and cut-to-length (harvester-forwarder) systems depend on access to nearby stands (usually less than a mile).

Findings—In steep terrain, reducing road densities may require longer cable yarding distances, and because yarding distance is a significant cost factor, especially in thinnings (Hochrein and Kellogg 1988; Kellogg and others 1996a, 1996b) timber harvesting costs likely will increase. In addition, greater reliance could be placed on helicopter logging, which would increase logging costs by as much as 2.5 times. Another result could be more wood left behind in the forest because logs must be bucked to their optimum length to maximize the payload of the helicopter.

In gentler terrain, a reduction in road densities could lead to an increased use of cut-to-length (harvester-forwarder) systems or more reliance on cable yarding. Primary transportation distance (movement of logs from stump to landing) is a variable significantly affecting the productivity of ground-based skidding (Tufts and others 1988) as well as harvester-forwarder systems (Kellogg and Bettinger 1994). Lanford and Stokes (1996) note, however, that at least with similar primary transportation distances in the Southeast, harvester-forwarder systems have comparable costs per unit harvested to traditional ground-based skidder systems, yet with lower environmental effects. If cable yarding replaced some ground-based systems, costs could increase by 1.4 times or more (Kellogg and others 1996b).

Logging cost increases (all else held constant) would reduce the likelihood that proposed sales would sell and lead to reduced harvest. The Forest Service's Washington, DC, office provided an estimate of the extent of these harvest reductions. They estimated that harvests would be reduced by 6 percent in the Northern Region (Montana, northern Idaho, North Dakota, and northwestern South Dakota), 90 percent in the Intermountain Region (southern Idaho, Nevada, Utah, and western Wyoming), and 17 percent in the Pacific Northwest Region (Oregon and Washington). If the issue involves only the use of secondary roads into sale units or just reliance on temporary roads for local sale access, then these effects may be overstated.

More difficult to determine are the long-term effects of focusing future management activities in only the roaded sections of national forests, where one of the primary management tools is stand manipulation through timber-sale contracts. Some management activities, such as prescribed fire, are not road dependent but most of the techniques for stand manipulation require some type of access.

Another issue is how changes in one region relate to changes elsewhere in North America. Reductions in federal timber harvest largely in the West are offset by increases in harvest elsewhere (mostly in Canada and on private timberlands in the South). These offsetting changes are usually sufficient to reduce consumer effects to modest, so that the largest effects are borne by producers (and their employees) in the affected regions.

Reliability, confidence, and limitations—Studies document the effect of skid distances and different logging systems on logging costs (Kellogg and Bettinger 1994, Kellogg and others 1996a, Lanford and Stokes 1996, Tufts and others 1988). Some of these studies were used to support timber appraisal processes. The effect of higher logging costs (because of more expensive logging systems) on stumpage prices has been well documented in the literature (for example, Jackson 1987); stumpage values have to be greater than logging costs for sales to be sold. Increasing logging costs, all else held constant, will result in fewer sales (or more sales being below cost). The effects listed in the findings are uncertain after one to two years because of the ability timber sale planners have to redesign timber sales, including their ability to change harvest unit locations.

Generalizability—The results are generalizable. What does differ are the values for timber throughout the West and the opportunities for less road-dependent logging systems.

Secondary links—The secondary effect of greatest concern is the potential loss of access to stands for forest management activities that remove individual trees. Although much of the current controversy is over final harvest, many other silvicultural practices depend on timber-sale contracts and timber removals to achieve various stand and landscape conditions. Often the forest road network was designed to allow access to multiple stands. Identifying the optimal network in light of potential additions or reductions in roads is difficult (Dean 1997). In addition to considering the loss of access, planners need to consider costs of alternative road building or rebuilding, landslide risks, and expected environmental effects, when they evaluate road management alternatives (Sessions and others 1987). Algorithms to incorporate road management alternatives in forest planning efforts have been described for traditional optimization techniques (Jones and others 1991), as well as heuristic methods (Bettinger and others 1998, Weintraub and others 1995). The effects of road management alternatives on timber programs is a site-specific problem, depending on the road system that exists, the road management alternatives examined, and the condition (age, volume, and so on) of the harvestable timber stands affected by the alternatives. For example, areas of mature forest stands in nonreserved land allocations may be most affected by near-term changes in the road network.

Conclusions—Roads and timber-program issues have been much studied, including attention to the ability to trade off more intensive management on the roaded parts of national forests with the unroaded portions. The ability to address immediate effects (say, for the next fiscal year) is very high, but beyond several years, the ability to predict effects greatly diminishes because no opportunities are available for mitigating the effects of changes in sale location or design. Finally, economic effects tied to changes in timber flows are very real. Roughly 10 direct jobs are generated for each 1 million board feet of harvest from national forests in the West. In addition, payments in lieu of taxes account for significant parts of local government funds in much of the rural West.

From a planning perspective the ability to examine tradeoffs in road system alternatives is moderate. Examinations into the theoretical complexity of road network planning problems have led to the development of planning models designed for integrating road decisions with land management decisions (Bettinger and others 1998; Jones and others 1986, 1991; Nelson and Brodie 1990; Sessions and Sessions 1997; Weintraub and others 1994, 1995; Zuuring and others 1995). These models are particularly useful for measuring tradeoffs among the quantifiable management benefits and costs associated with changes in the road network. Not all issues relevant to a decision can be adequately quantified, however, because the output or response relations are not known or are just being developed. For example, the response variables can be complex and may depend on activities in adjacent stands (see Bettinger and others 1998). In addition to the complex planning model, data development (both geographic information system

[GIS] and associated tabular inventories) is one of the main challenges. The ability to collect and use GIS data as well as the attributes of a road system (and related resources) is evolving and, over time, analyses now based on current data will progressively become more precise and accurate.

Nontimber Forest Products

Issues—A variety of products harvested from the abundant biotic resources of the North Temperate Zone forests are being transformed into medicinals, botanicals, decoratives, natural foods, and a host of other novel and useful products. These renewable, vegetative natural resources harvested for personal or commercial use are called nontimber or special forest products. Consumer forces, changing social climate, and expanding global markets are contributing to the increasing development of these products as viable economic options for sustaining rural communities. Ginseng (*Panax quinquefolius*), goldenseal (*Hydrastis canadensis*), coneflower (*Echinacea angustifolia*), and St. John's wort (*Hypericum perforatum*)—all plants found on national forest lands are major contributors to a multibillion-dollar herbal and botanical industry. Access to these resources has important economic value to those rapidly growing industries. Plants harvested from the wild are "wildcrafted" by harvesters from local communities or contract crews brought in from elsewhere. Particularly for the local harvesters, who operate under the permit system of various public and private land ownerships and who often have low income, access by road to the resource becomes a critical cost factor. In addition, roads create openings important to maintaining diverse species in abundance. How roads will affect the survival and sustainability of nontimber forest products and how access to nontimber forest products will be influenced remain important issues. Both issues are important to the people and communities that already depend on these herbs, shrubs, lichens, fungi, algae, and micro-organisms as part of their economy.

In 1992, the herbal-medicinal market was estimated at just under $1 million and growing at a rate of 13 to15 percent per year (Mater 1997). Traffic USA, a program of the World Wildlife Fund that monitors commercial trade in wild plants and animals, estimates annual retail sales of medicinal plants in the United States in 1997 at $1.6 billion and rising. Of the 25 top-selling herbs in U.S. commerce (Brevoort 1998), more than 50 percent are included in the 1,400 plant species found and traded in the United States. Moss and lichens, harvested extensively from public forest lands and exported to worldwide markets, were valued at more than $14 million in 1995 (Vance and Kirkland 1997). Demand is increasing for huckleberries and mushrooms, important foods harvested for commercial and personal use. In 1995, less than 1 million pounds of the matsutake (*Tricholoma magnivelare*) mushroom were harvested, but in 1997, in one 8-week period, 1.2 million pounds were harvested, which provided the Forest Service with $365,935 in revenue from permit sales (Smith, n.d.). Floral greens are an important mainstay for several markets in the Pacific Northwest. A 1989 study (Schlosser and others 1991) showed that the total value of floral and Christmas greens earned $128.5 million in product sales with about $48 million paid to harvesters, which supported the employment of about 10,000 people and about 675,000 acres in production west of the Cascades. On a single ranger district (Hood Canal Ranger District, Olympic National Forest) from February 1996 through February 1997, 1,500 permits were sold for commercial harvest of greens, bringing in revenue of $63,835. Christmas boughs have continued to increase in demand, and by 1995, harvest in the Pacific Northwest was approaching 20 million pounds per year (Savage 1995).

Findings—Market growth is documented (Mater Engineering 1992, 1993a, 1993b). Collection activities permit information, environmental and other assessments, and maps with roads indicated are part of the written procedures and permitting instructions at forests and districts affected by special forest products. Costs of harvest are recognized as a factor in permit prices, and they influence contract bids in these assessments. Market value is related to cost; increasingly difficult access as plants become scarce may be factored into market value. An assessment in the Southern Region (Alabama, Arkansas, Florida, Georgia, Kentucky, Louisiana, Mississippi, North Carolina, Oklahoma, Puerto Rico, South Carolina, Tennessee, Texas, Virgin Islands, and Virginia) identified dozens of plants and products for which free use and commercial permits are issued. Illegal collection is considered a problem in many areas, and some documentation exists in Oregon with the Bureau of Land Management, Forest Service, and state enforcement personnel. Although not explicitly, roads play a role in illegal taking, as well as in monitoring harvest activities. Other reports and inventories have maps indicating roads that offer access to nontimber forest products and often act as a means of pinpointing the desirable harvesting areas. For example, in the special forest products inventory (Karen Theiss and Associates 1996) created for Trinity County, California, roads were used extensively to describe how to find areas where wildcrafters could harvest a particular species.

Reliability, confidence, and limitations—Much of the documentation that relates to special forest products can be found in forest and environmental assessments and in recent reports and papers published in journals and books (Molina and others 1997, Savage 1995, Thomas and Schumann 1993, Vance 1997). In some of these documents, roads are addressed directly about use and compliance with reciprocal agreements where they are in effect. Historically, special products have been administered as a byproduct of timber contracting and road building. The same benefits accrued by recreational collectors of mushrooms, berries, and so on in those areas also could be enjoyed by commercial harvesters. No formal documentation of these benefits going to commercial harvesters is available. Note that some states (e.g., Oregon) require anyone transporting any such product, including firewood, on public roads to have a legal permit or bill of sale.

Generalizability—Generalizing the need for roads or road decommissions for nontimber forest products is impossible. Some populations of harvestable species will benefit from the disturbance caused by building and maintaining roads, and other populations will be harmed. Although enforcement of illegal harvest might be hampered, so would legal harvest. But market forces adjusting for reduced harvest (product scarcity) is unpredictable, and whether any increased value would be transferred to the harvester is not known.

Secondary links—Habitats and plant community structure of some commercially harvested species are linked to roads. From an assessment of 45 commercial species in Oregon, 30 percent can be found in openings and along roadsides. It also is well known that certain species require undisturbed mature forest and would not benefit from the gaps and disturbance caused by roads. Because of the specific habitat requirements of, for example, wild ginger, pitcher plants, and shade-loving mosses, roads would not directly benefit these plants. Some of these species are listed as sensitive, and ready access threatens their survival. Documentation exists for habitat requirements of almost all commercial plants and fungi. Other habitat concerns are related to maintaining roads. A special forest products inventory created for Trinity County, California, suggests that harvesters stay away from roadsides because some Bureau of Land Management and Forest Service districts routinely spray herbicides and pesticides.

Communities and sustainable economies—Many rural areas need more sustainable and diversified economies, for which they may require assistance. The Forest Service recognized this need and developed economic action programs aimed to help communities strengthen their local economies through a range of forest-based resources, including nontimber forest products.

Conclusions—Information on habitat requirements for many of the commercial species is available, and retrospective studies may show how road closures affect species composition; for example, in the prevalence of native versus exotic species (Parendes and Jones 2000). Developing appropriate policies and implementing them for most special forest product species would benefit from information and models that predict regional and general effects from building or closing roads on the species' harvest and sustainability. Information on the economic effects on various components of the industry—from harvester's overhead to product price—is needed. These questions must be answered to determine how building or decommissioning roads would affect the sustainability of particular commercial species and hence the sustainability of the economies reliant on them.

The effects of roads on the economic, social, and biological factors and their effects outlined above need to be documented. Although roads are generally recognized as major components of recreational and commercial-harvest activities that affect hundreds of species in the national forests, systematic studies that integrate these components, much less any individual component, have not been carried out. Only fragmented information on these biological resources, products, uses, values, and habitat considerations is available. Case studies will provide information on local or regional scales, but a comprehensive model of the relation of roads to special forest products nationally requires a comprehensive special forest products database. In addition, an integrated strategy for special forest products that addresses community and resource sustainability together would benefit from targeted and integrated research-based information.

Grazing and Rangeland Management

Issues—According to the 1995 draft RPA program, about 46.2 million acres of national forest lands are considered suitable for livestock grazing. Producing livestock can be an important part of local economies, and livestock grazing is deeply rooted in the culture of the American West and sanctioned by legislation. Grazing was first authorized on national forest lands by the Organic Administration Act of 1897 and confirmed by many later appropriations acts (USDA FS 1989). The Public Rangelands Improvement Act of 1978 reinforced a national policy that public rangelands were to be “managed...so that they become as productive as feasible for all rangeland values.” The network of roads on national forest lands has both positive and negative effects on rangelands and the administration of the grazing program. Roads have mostly replaced driveways as a means for transporting sheep and cattle to and from mountain allotments. As a result, these driveways have dramatically improved in rangeland health. Until the 1970s, livestock driveways were considered “sacrifice areas” in the range-management discipline (Stoddart and Smith 1955). Thus, national forest roads can promote ecosystem management objectives along alternative transportation corridors, which they replace. Roads can simultaneously lead to ecosystem changes that reverse rangeland management objectives, however, and increase the administration of the range management program. Administratively, national forest roads allow range conservationists to access allotments quickly by using vehicles rather than horses. But the same roads can produce conflicts between users of the national forests, such as between livestock grazing and recreation interests. And roads can reduce permittee operating costs by providing motorized access to allotments.

Findings—Essentially no scientific information exists that analyzes the ecological, administrative, or economic effects of roads on administering the Forest Service range-management program. Preliminary unpublished analyses from the interior Columbia River basin ecosystem management project addressed the road issue from the perspective of ecological responses to the presence or absence of roads. The analyses found correlations between changes in vegetation composition, riparian functioning, and fire regimes and the presence of forest roads. They could not conclude any cause-and-effect relations from these correlations, however. The program also found higher road densities to be associated with diminished ecological integrity, including those based on range criteria.

To assess the importance of national forest roads for administering the grazing program, as well as their economic value to permittees, an ad hoc interdisciplinary team was formed to provide a nominal assessment. The findings below reflect the input of the team:

- Roads in national forests are essential for administering the grazing program, allowing timely access to allotments. Compliance enforcement was mentioned in particular as an activity greatly benefiting from forest roads. The principal reasons cited were that agency downsizing has resulted in high workloads for remaining range conservationists, which does not allow them sufficient time to carry out their duties; guard stations have been closed; Forest Service personnel no longer have the option of spending nights in the field in some places; and many allotment plans incorporate Forest Service roads into their approved grazing system or as driveways to and from the allotment; for example, in the Black Hills, all driveways are along roads.
- Roads can reduce permittee operating costs by providing motorized access to allotments. The team estimated that, if all national forest roads were closed, permittee costs would increase by three to five times. These costs would accrue from increased riding time, cost of horses and riders, and added equipment costs (such as horse trailers). The grazing program derives benefit from only part of the road system, however, and if arterial and collector roads remained open, the expected cost increases would be less, from none to a twofold increase.
- Roads can heighten conflicts among users of national forests, such as cattlemen and recreationalists, although some evidence shows that concerns about road conditions actually can cause some forest visitors to slightly, but measurably, shift their focus of attention from grazing encounters to roads (Mitchell and others 1996).

Reliability, confidence, and limitations—No peer-reviewed studies have assessed the effects of national forest roads, or roads in general, on livestock grazing or ecosystem management. The results from the Columbia River basin program are tentative and show no causal relations. The results of studies examining the influence of roads on forested landscapes must be carefully extended because the results from studies in Eastern forested landscapes may not apply to Western forested landscapes (Miller and others 1996). The results of the interdisciplinary-team assessment are heavily weighted towards the Rocky Mountain Region (Colorado, Kansas, Nebraska, South Dakota, and eastern Wyoming) and thus may not represent a national perspective.

Generalizations—National forest roads are an important part of range-allotment plans. Roads are also important for administering the grazing program on national forest lands. Ecologically, roads may have a negative effect on rangelands; however, the environmental effects of not having roads are unknown. The team concluded that closing some roads would be acceptable from the perspective of managing the grazing program if the process was systematically evaluated first.

Secondary links—Effects of roads on spread of non-indigenous weeds (biological invasions), wildlife-livestock interactions, and recreation-grazing interactions (particularly with four-wheeling interests) are important.

Conclusions—No science-based information was found on how national forest roads affect livestock grazing. Many questions remain, including the cost of closure to permittees, and the effects of road closure on administering range management programs, including the weeds program, and on compliance.

Energy and Mineral Resources

Issues—The road-related issues associated with energy and mineral resources fall into three overlapping categories: access rights, property rights, and benefits and negative effects. The extractive industries want, and have certain legal rights to, access to public lands to explore for energy and mineral deposits. The access may be on existing forest roads or may require building new roads. The Forest Service road system facilitates providing energy and mineral resources extracted from public lands, which can benefit society. The negative environmental effects of roads used in support of nonrenewable resource extraction are covered in the earlier sections of the synthesis. Mineral developments and oil fields in and of themselves can affect the environment negatively, such as by loss of habitat, increased noise, and added particulate emissions in the air and water, but these effects can be attributed only secondarily to roads; that is, without the road, mineral development might not have taken place.

These issues are a consequence of the inherent nature of the resources and their treatment under existing law. The defining characteristic of energy and mineral resources is nonrenewability; energy and mineral resources are finite, so extraction inevitably leads to resource exhaustion. Depleted deposits must be replaced either through domestic exploration and mine or field development or through importation. In many places, national forest lands are underlain by deposits of nonrenewable resources, some of which are privately held, that make demand for access inevitable.

Federal law and Forest Service policy clearly support exploration for and extraction of resources from public lands. Leasable resources (that is, metallic minerals found on acquired lands and all energy resources) are managed under the Mineral Leasing Act of 1920. Locatable minerals, primarily the metallic ones on public domain lands, are managed under the Mining Law of 1872. Saleable minerals (that is, common varieties such as gravel) are managed under the Mineral Materials Act of 1947. These laws predate the National Forest Management Act of 1976 and the Multiple Use Sustained Yield Act of 1960.

Findings—Under the Mining Law of 1872, U.S. citizens and firms have the right to explore for and stake claims to selected minerals on all public domain lands not specifically withdrawn from mineral entry. Claims are valid in perpetuity or can be converted to private property rights (that is, patented) assuming that appropriate legal requirements are fulfilled. The Forest Service cannot unilaterally deny exploration access to national forest public domain lands, although the agency does have the right to withdraw specific areas from further mineral entry. The agency cannot prevent staking of a claim on these lands, and a claim holder is entitled to use the surface for activities attendant to exploring for, developing, and extracting minerals, within the limits set by federal, state, and local environmental laws. The agency cannot block an otherwise legal patent (that is, deny a claim holder the right to convert the claim to private property). The Congress can, and has, placed a moratorium on new patents, but the moratorium could be lifted in the future. In any event, hundreds of thousands of patented and unpatented claims are already held within the administrative boundaries of the national forests.

The Forest Service has considerably more control over the location of exploration and development activities for leasable minerals than it has for locatable minerals. For national forests and grasslands with completed oil and gas leasing EISs, petroleum exploration activities are restricted to areas designated as appropriate in those documents. The regions also are taking an active role in directing access for leasable minerals. For example, the Northern Region is attempting to restrict oil and gas exploration to areas relatively near existing roads. This approach is not without potential for controversy, however. Decommissioning of roads could be perceived as a de facto withdrawal of the adjacent lands from exploration. The circuit courts are split on the question of whether failure to offer lands for lease is tantamount to withdrawal.

The Forest Service is required by law to provide reasonable access to valid existing mineral rights, regardless of their form, whether unpatented claim, lease, or private property, as a patented claim or subsurface mineral right. An unpatented claim is an implied property right that can be held, sold, or inherited, and access is regulated under the Mining Law of 1872. Patented claims are private property, and access is regulated under the Alaska National Interest Land Conservation Act of 1980 (ANILCA). Coal, oil and gas, and mineral leases also offer a limited form of property right. The rights to individual energy and mineral resources may be held by different legal entities, and the mineral rights may be severed from the surface, which is termed a “split estate.” Access to unpatented inholdings, patented claims, leases, and severed mineral rights can be restricted but seldom denied. Access may be by the existing road system or require new roads. The Forest Service is neither required by law nor expected by industry to build or maintain energy and mineral access roads. Roads built for other reasons (for example, in support of recreation development) might be paid for by the Forest Service but also be used by a mining or energy firm. The firm is always required to maintain the road or to pay for road maintenance called for by their activities; they frequently pay through a reimbursement arrangement with the agency.

The Forest Service can affect the location and design of roads built on national forest lands to support energy and mineral activities. In addition, the agency can sometimes place stipulations on access by limiting road use to certain months, permitting aerial access only, or precluding surface occupancy. Constraints that are unduly expensive to fulfill or so restrictive as to make an otherwise economic mineral deposit uneconomic, however, might well be perceived as denying reasonable access. Temporary roads often are built to facilitate energy and mineral exploration activities. Building plans are subject to review and approval by the agency. If no discovery is made, the exploration firm obliterates the road. Otherwise, the road could be upgraded to permanent status, depending on the circumstances and legal authority. Public use of the road might sometimes be limited because road condition acceptable to the mineral industry might be neither acceptable to, nor safe for, the general public. In addition, other means of access, particularly for exploration, do not require roads, including access by helicopter, foot, horseback, and all-terrain vehicles.

The energy and minerals industries use the existing road system in exploration, development, extraction, and reclamation activities. Only a small portion of the entire road system is affected in any given year, but assuming use of most roads over the long term would be reasonable. Designating a subset of the existing road system as having no future benefit to the industry is not feasible because geographic targets for exploration and development change in response to technological advances and market fluctuations. Limiting mineral exploration access to areas where minerals have already been or are being extracted could preclude future discoveries. Road closures or decommissionings are controversial. Firms wanting to rebuild obliterated roads could face long delays because of the lengthy approval process now in place for building new roads. Such delays could disrupt multiyear exploration and development plans and financing.

The energy and mineral resources produced from national forest lands are essential to the manufacturing, farming, building, and power-generating industries, with a value of $4.3 billion in 1995. Forest Service production represents only a small part of the total value of U.S. production, however. For example, the value of copper produced on national forest lands represents only 1 percent of total U.S. copper. Sometimes, production from national forest lands is a significant percentage of domestic production; national forests produced 80 percent of domestic lead in 1995. Significant amounts of coal and molybdenum also are produced from national forest lands. These contributions to the domestic economy are made possible by use of the forest road system.

Reliability, confidence, limitations, and generalizability—Some case law on energy and mineral access and property rights can be applied more broadly than to the specific litigation reported in it. And for certain situations, existing case law, statutes, and regulations clearly demonstrate the right to reasonable access for existing mineral rights. In numerous other situations, however, the right to access for energy and mineral exploration and development is less clear-cut. Unresolved access issues are associated with both ANILCA and Section 8 of the Lode Law of 1866 (R.S. 2477), which granted right of way across unreserved public domain lands. Considerable debate continues on the degree to which this right has been modified by subsequent legislation.

Secondary links—Roads built to provide access for energy and mineral exploration and development often are heavily used for other purposes. Secondary links can be found to recreation, species endangerment, biological invasions, and many other areas. The effects from energy- and minerals-related roads and road usage are comparable to those of other roads in the Forest Service system built to the same specifications and carrying the same types and amount of traffic. Unpaved Forest Service roads frequently are topped with a layer of aggregate or crushed stone, and the material often has been extracted from Forest Service lands. Thus, the extent of the road system also has implications for the volume of aggregates extracted; fewer miles of road built and maintained implies fewer tons of aggregate and crushed stone extracted.

Conclusions—The legal issues surrounding energy and mineral road access and usage will require the input of the Office of General Council: Pamela Piech (202/720/2515) is an expert on the Mining Law of 1872; James Snow (202/720/6055) is an expert on RS2477 and ANILCA. Little or no research has been published on the secondary links associated with energy and mineral road usage. One key area for future nonlegal research is to determine the landscape-scale effects of energy and mineral development; for example, extensive oil-field road networks may lead to habitat fragmentation.

Another need is to determine exactly which roads are currently being used for access to explore, develop, extract, and reclaim. Quantifying the effects on road condition of nonrenewable resource activities by number and size of vehicles is also important, and another management need is to identify the roads leading to or adjacent to valid existing mineral rights.

Resource-Based Outdoor Recreation

Issues—Almost all the different types of public recreational uses of national forests depend in one way or another on roads for access. Whether, when, and where various recreational uses occur depend on the availability of access to, and the extent and location of, the road system. Altering this system is likely to have widespread and differing effects across different types of uses. In considering the future of roads on national forests, the general question is, "What are the direct, indirect, and secondary effects on recreation from possible changes in national forest road systems?" More specifically, "What are the direct effects of changing the class, spatial density, ecological distribution, maintenance, and total mileage of national forest roads on the density, placement (ecologically and socially), mix, economic value, experience quality, and amount of recreation uses?" As well, "What are the indirect effects on access to views of natural scenery and on the quality of scenic resources, and what are the secondary effects on the economic and social viability of communities in the area and the condition of the forest ecosystem?" Answers to these and many other questions are needed as input when national forest road policies are considered and in seeking to optimize net benefits across multiple roads.

Findings and hypotheses—The relations between roads and recreation on national forests is highly complex and includes many direct, indirect, and secondary links that are not well understood. Research findings specifically addressing these links are limited and uneven across the questions we have posed. Indirect evidence and related research provide the following insights and hypotheses:

- Roads provide corridors of access to a variety of national forest sites, settings, and viewing opportunities for widely diverse users. Almost all recreation use in national forests depends to some degree on road access. Sightseeing, driving outdoors for pleasure, and developed camping are examples of activities that directly use roads as a part of the recreation experience. Backpacking, white-water boating, and birdwatching are examples of activities usually away from roads, but the user still must access areas of interest by using them. Altering road systems can disrupt long-established access and use patterns and, at least in the short run, result in not meeting visitors' expectations. Less road mileage or maintenance, or both, can lead to uneven shifts in recreational opportunities across different user, socioeconomic, and ethnic groups who depend differently on roads for access.

- Roads provide staging access to remote areas and wilderness, but the presence of roads can at the same time reduce opportunities for solitude and perceptions of wildness. The amount, placement, and class of roads are positively correlated with the amount and concentration of recreational uses. But visible roads, greater numbers of users, and sounds from motor vehicles can interrupt solitude and perceptions of wildness for wilderness and other backcountry users.

- As demand for forest recreational opportunities continues to grow locally, regionally, and nationally, even a stable amount and condition of forest roads likely will result in increased congestion, lowered satisfaction, and user conflicts. Outdoor recreation trends show recent strong growth in participation across a wide spectrum of activities and segments of the American public (Cordell and Bergstrom 1991). Projections show this growth is likely to continue well into the future for all nature-based activities except hunting (Bowker and others 1999). At the same time, access to private lands is continuing to decrease and be limited to lessees and friends of the owners (Cordell and others 1999). Public lands are likely to be the destinations of choice for increasing numbers of people looking for high-quality outdoor recreational experiences in natural settings. Several national parks already have limited motorized access to bus tours or other public transportation as one way to address increased congestion from private cars. Continued growth in demand without increases in road systems or limits to use of private cars likely will lead to lowered satisfaction and more conflicts at the more popular national forests (Tarrant and others 1999). Changes in satisfaction likely will differ significantly by setting (for example, as distinguished in the recreation opportunity spectrum [Tarrant and others 1999]). Direct recreational access, the character of and access to scenic views, and provision of increasingly sophisticated visitor services (including rescue and medical services) will depend on the character of the road system in place.

Reliability, confidence, and limitations—Data on national forest use and the relations of roads to that use are unreliable, but a national project is underway to develop an improved use-monitoring system. Data from the customer project provide insights into user perceptions of experience quality related to national forest attributes, including roads (Tarrant and others 1999). Social group differences between users of roaded, near road, and backcountry settings are available for the U.S. population in general, and to some degree for national forest users. Science-based methods are available for examining in more depth the relations between roads, recreational use, visitor satisfaction, and economic values and effects. Little research exists to guide management for optimizing recreational benefits from roads and globally optimizing multiple benefits across the broad range of national forest road uses.

Secondary links—Even though increased use (on the same or fewer miles of forest roads) or changes in the mix of recreational uses, or both, may increase aggregate visitor spending (and thus general economic effect), the distribution of economic effects among economic sectors and regions is likely to be altered. The biophysical effects of recreational use on forest ecosystem conditions are confined mostly to near-road zones, the site of most use. The biophysical condition of affected sites tends to stabilize after each successive increment of recreation use, although the resulting condition may be unacceptable to managers, users, or both. Specific links between recreational use and conditions of ecological components and links between recreational use and other resource uses are not well known.

Conclusions—Quantitative and qualitative methods, research underpinning the recreation opportunity spectrum, and a wealth of related published and unpublished literature dealing with economic values (Bergstrom and Loomis 1999); secondary economic effects (Archer 1996, Bergstrom and others 1990); visitor perceptions and behavior (Tarrant and others 1999; Williams and Patterson, in press), resource and social capacity (Shelby and Heberlein 1986); conflicts, consumption, and future projections of road-based recreation (Cordell and Bergstrom 1991, Bowker and others 1999, Cordell and others 1999), and social justice assessment are available. For the most part, however, existing databases and literature have only indirectly addressed the hypotheses described above that deal specifically with the relations between roads and recreation (for example, Knight and Gutzwiller 1995). Substantial research is needed to better understand direct and indirect relations between road-system characteristics, recreational use, and ecosystem conditions, including issues such as the introduction of exotics, soil erosion, habitat fragmentation, forest-product harvesting, wildlife disturbance, riparian vegetation, and fire.

Indirect Socioeconomic Effects

Fire

Issues—The increasing density of road networks in and adjacent to many forest, shrub, and rangeland areas has been an important factor in changing patterns of disturbance by fire on the landscape. Roads provide access that has increased the scale and efficiency of fire suppression, and roads have created linear firebreaks that affect fire spread. These factors can be useful in both fire suppression and prescribed fire operations. In addition, road access has undoubtedly contributed to increased frequency of human-caused ignitions in some areas.

Findings—That improved road access leads to increased efficiency and effectiveness of fire-suppression activities is a long-held tenet of fire fighting. Much of the effectiveness of past fire-suppression policies probably can be attributed to increased access for ground crews and equipment, particularly under weather and fuel conditions where fire behavior is not severe. Under the severe conditions associated with intense, rapidly spreading fires, the value of forest roads for access or as fuelbreaks is likely to be minimal. Although little has been published in the science literature to quantify these effects, a study in southern California concluded that the road network had been a key factor in determining what suppression strategies were used, both in firefighter access and because roads were widely used for backfiring and burning-out operations (Salazar and Gonzalez-Caban 1987). Early studies of fuelbreak effectiveness in southern California came to similar conclusions (Green 1977). Daily costs of fire-fighting activities unfortunately are of little value in answering the question of how much road access increases efficiency, because fire-fighting agencies tend to put money and resources into fighting fires with access, which confounds the results. In spite of this, strong anecdotal evidence supports this effect.

An important issue in the Western United States is building new roads to allow harvest and prescribed fire to reduce fuel accumulations in ecosystems where past management (principally fire suppression and harvest) have increased the risk of large, severe wildfires (Lehmkuhl and others 1994). The principal concern here is the tradeoff between reducing the effects of wildfire and increasing the risks of road effects on aquatic habitat. In the Columbia basin, scientists concluded that “it is not fully known which causes greater risk to aquatic systems, roads to reduce fire risk, or realizing the full potential risk of fire,” and that more research is needed (Quigley and others 1997). Some potential considerations in setting priorities for forest health treatments have been suggested in an adaptive management framework for addressing this concern (Rieman and Clayton 1997). We currently have few data on how these processes might be affected by road networks, although a study after the 1987 Stanislaus fires in California suggests that cross-slope road networks reduced sediment delivery to debris basins (Chou and others 1994).

The benefits that roads provide for fire prevention and fire management carry an associated cost. For purposes of simplicity, we will highlight them here in place of a second fire section under the “undesirable or negative effects.” Indirect effects of increased access have increased the role of human-caused ignitions, particularly in areas of expanding urban and rural development into wildland interfaces (Hann and others 1997). The high rate of human-caused fires in the Blue Mountains of eastern Oregon is associated with high recreational use in areas with high road densities (Hann and others 1997). The importance of human-caused ignitions as an issue may depend on what resources are considered of concern. For example, in the Southwest, numbers of ignitions go up with access, but numbers of ignitions are not limiting to maintaining fire regimes, but fuel loadings and climatic conditions are (Swetnam and Baisan 1996). Numbers of ignitions are important determinants of fire risk, however, in areas such as wildland-urban interfaces for which maintaining historical fire-regime patterns is not the overriding issue. In addition, numbers of ignitions are important determinants of fire risk in some wildland-urban interfaces where fire intensities are often higher (such as chaparral), and active suppression of ignitions by people may be critical to maintaining historical fire patterns (Conard and Weise 1998).

Road networks have resulted in changes in fuel patterns and fire regimes at the broad scale. If we accept that road networks have been important in effectively suppressing fire and that they alter fire patterns on the landscape, then road systems are, in some sense, linked to changes in fuel patterns and fire regimes. Before fire-suppression activity in the Western United States, fuels were maintained at relatively low amounts in dry forest types, with high fuel loads restricted to small, isolated patches (Agee 1993). As access increased, areas burned by wildfire declined, at least through the 1960s. As a result of suppression supported by access (in part), fuel accumulations increased and areas with moderate to high fuel loadings became larger and more contiguous. This pattern of change has been documented for the entire upper Columbia River basin, where scientists assert that fire suppression has generally been more effective in roaded areas, which has resulted in roaded areas in the upper basin departing further from unaltered biophysical templates (as measured by dominant species, structures, and patterns) than have the unroaded areas (Hann and others 1997). Roads (along with other human disturbances such as clearcutting) contribute to new disturbance patterns at the landscape scale, both by increasing efficiency of fire fighting and providing barriers to fire-spread that are different from natural barriers (Swanson and others 1990). Increased emphasis on removing roads in certain environmentally sensitive areas will reduce access for fire suppression and prescribed fires, potentially leading to increased fuel accumulation and fire hazard in some areas.

Reliability, confidence, and limitations—Logic and anecdotal evidence for the contention that road access increases effectiveness and efficiency of fire suppression efforts are strong, but quantifying this issue in terms of cost savings or size and severity of fires is not well documented. The scientific support for the contention that roads serve as firebreaks is strong, but how important this effect is in controlling the pattern of fire on the landscape is not clear; the ecological implications of this pattern change also are not clear. The secondary effect of roads providing access for timber harvest that has resulted in changing mosaics of fire is strong; the ecological consequences, while strong, are highly variable. Long-term effects on changing fire regimes in the Western United States are well documented. Increased access probably leads to increased human-caused ignitions, but the implications of this increase differ from area to area. Increased ignitions at urban-wildland interfaces are likely to be a problem, but it may be unimportant in affecting fire regimes in less-developed landscapes in the West. Building roads to provide access to reduce fuel in fire-suppressed forests is likely to enhance this activity, but it may carry added risks to aquatic environments over the risk of fire alone.

Generalizability—Most of the concerns addressed here apply primarily to the Western United States. In much of the East, road networks are well developed and relatively stable because of terrain and vegetation differences. Wildfire interactions are likely to be similar to those described for the West, but the effects are likely to be significantly less. In the Southeast, where use of prescribed fire is widespread, roads are frequently used as firebreaks. Much of this activity is on private lands, however, and a high proportion of the road network is state and county highways rather than Forest Service roads.

Secondary links—Fire issues are linked to issues of forest (ecosystem) health and aquatic habitat.

Conclusions—In general, the importance of roads for providing access and firebreaks is well established, although literature on cost-to-benefit ratios is lacking; most evidence is anecdotal. The issue of road access to lessen fire risk and improve forest health in unroaded areas is heating up, and little published research is available to fall back on for resolving the debate.

Forest Research, Inventory, and Monitoring

Issues—Among the benefits that roads provide is access for research, timber and non-timber forest inventories, and monitoring. Although the economic scale of these tasks may be low compared to some other activities, the knowledge derived may be key for managing other access-related uses, in addition to the more general objectives sought. Hence, understanding the relation of roads to inventory and monitoring activities is not a trivial issue.

Findings—Although finding sufficient data for a complete and wide-ranging analysis is difficult, the role roads play in inventory and monitoring access (that is, the cost per plot) can serve as a surrogate for the larger problem. Plot-survey contracts are based on four categories in which the proximity to roads plays a significant part. For example, costs run about $600 per plot when roads allow access to within 0.25 mile of the plot sites. In the same region, cost rises to $1,300 per plot in roadless areas open only to foot access. In the Pacific Northwest, the nearly 650 wilderness plots, of a total of 11,360 in all terrain, had survey costs only about 23 percent greater ($1,460 per wilderness; $1,174 per nonwilderness plot). The data did not permit comparing the cost difference of road-accessed plots in the Pacific Northwest Region over the montane sites in the Pacific Southwest Region, however. More extreme conditions are encountered in Alaska, where roadless areas are vast, yet helicopter access is permitted. The average cost per plot for roadless areas in the Alaska interior has averaged $4,000 per plot for 170 plots. Obtaining good data for comparing areas covered by these approaches is generally difficult because photo-interpretation based on aerial photo coverage is used to supplement ground-survey efforts.

Reliability, confidence, and limitations—Problems of access to survey plots for research, inventory, and monitoring will clearly raise costs of operations. The exact differences can be quantified by taking terrain differences, size of roadless areas, and means of permitted entry into account. For this study, we used only a few data points from limited regions to understand the extent of this issue. More comprehensive analyses are possible with existing data, given the resources to do them. The data are sufficiently robust to suggest that the cost elements relating to access constitute a factor in research, inventory, and monitoring. Whether the magnitude of the contribution of such uses constitutes a significant economic component when compared to, say, recreation is not clear, however.

Generalizability—The data examined for this order-of-magnitude approach were taken from limited observations originating in the Pacific Southwest, Pacific Northwest, and Alaska, with Alaska representing extreme conditions. Corroboration for the observed higher cost resulting from the absence of road access was attained qualitatively for the Eastern Region of the Forest Service.

Secondary links—Access issues have similar aspects whether extraction (such as timber, mining, and grazing), recreation, inholdings, or related activities are considered. The links do suggest that coordination of overlapping uses be a variable examined when road density and road-network planning are considered.

Private Inholdings

Science-based sources of information have not been found on the relations between roads and private inholdings. The following propositions are therefore offered as hypotheses based on judgment, not scientific findings. These propositions do not necessarily apply to inholdings dedicated to mineral and energy exploration or extraction, which are covered in "Energy and Mineral Resources," above.

- The Forest Service is required by law to permit access to private inholdings.
- The Forest Service can require private inholding owners or lessees to comply with official regulations and standards that apply to building roads on or through national forest land. The regulations and standards are documented in writing as official policy, but they are subject to interpretation and application in specific cases by agency line officers.
- The Chief (of the Forest Service) may consult appropriate national forest policy offices and line officers about the sources of scientific documentation used in practice and official regulations, standards, and procedures applicable to roads on or through national forest lands that provide access to private property.
- In general, the scientific documentation of ecological and human effects of roads on or through national forest land provided elsewhere in this synthesis applies to roads that provide access to private inholdings.
- No scientific basis exists for stating propositions about whether the Forest Service subsidizes access to private inholdings or the effect, if any, of Forest Service roads on the market, use, and passive-use values of private inholdings.
- The Chief needs inventory information about the type, number, acreage, location, use, value, and so on of private inholdings on national forest land and the extent to which private inholdings use national forest roads for access. At present, no systematic inventory procedure or documentation can provide comprehensive and valid information of that type.

Nonmarket and Passive-Use Value

Issues—A comprehensive understanding of the economic effects of roads in the national forests must include both effects that can be measured in dollars (market effects) and those with no direct dollar values (nonmarket effects). The influence and importance of market values to land management decisions is obvious, and measuring and comparing effects of management decisions that affect market values are relatively simple. For example, the cost of building and maintaining a road into a forest can be readily compared to the income generated from harvesting the timber accessed by that road. Also important, but far more difficult to measure and compare, are the things people care about for which no market exists, such as access for hunting, bird watching, and wilderness experience.

Natural resource economists have invested much effort over the last several decades to develop and test methods for estimating nonmarket values. The methods can produce useful information, but they are costly and their validity has not yet been demonstrated sufficiently to satisfy many economists (Arrow and others 1993, Cambridge Economics 1992, Mitchell and Carson 1989, Portney 1994).

Economists generally classify nonmarket values as either active or passive. The term “active-use value” applies to goods and services used in some activity like recreational fishing, skiing, or camping. The term “passive-use value” includes two categories (Peterson and Sorg 1987, Randall 1992): things people appreciate without actually using them or even intending to use them (like a distant wilderness or an endangered plant or animal) are called “existence values”; and things people want to remain available for others (such as their descendants) to use and appreciate are called “bequest values.”

Environmental economists often define and measure these nonmarket values in monetary terms, but monetary valuation is often not possible, cost-effective, or appropriate. All nonmarket consequences of national forest roads and of any changes to these roads must be considered in road management and policy decisions. For example, passive-use values are likely to strongly affect decisions about preserving areas without roads or about removing existing roads to create roadless areas. Thus, the nonmarket consequences need to be identified in some way—either in monetary terms or by some other means.

Under regulations of the Comprehensive Environmental Response, Compensation and Liability Act of 1980 (CERCLA), as amended, 42 U.S.C. 9651 (c), a United States Court of Appeals for the District of Columbia ruled in 1989 that passive-use values “...reflect utility derived by humans from a resource and thus, prima facie, ought to be included in a damage assessment.” Thus, if Forest Service roads significantly alter passive-use value, whether positively or negatively, such value needs to be considered in road policy and management decisions. Failure to include these nonmarket values in an economic evaluation, when such values are judged to be important, presents the manager with biased information that could lead to inefficient and unfair allocation of resources.

Significant questions: Under what conditions do people assign passive-use value to national forest landscapes or their attributes? Forest Service officers responsible for road policy and management need to know the forest landscape conditions to which people assign passive-use or other nonmarket values, how such values differ among individuals and groups of people, the strength or significance of the value assigned, how changes in the landscape affect the nonmarket values, and how such values trade off with other forest-related values assigned by affected people.

Do Forest Service roads, road policies, or road management actions strongly affect passive use and other nonmarket values? If so, how and why? A related question is whether the effects of roads on nonmarket values affect people differently and differ by landscape. For example, if the supply of landscape that provides passive-use value is sufficiently large in a given region, small increments of road building or decommissioning may not affect people very much. Many small encroachments could produce severe cumulative effects, however.

Findings—People do assign passive-use value to natural resources, especially roadless areas and natural areas with unique characteristics. And the passive-use value often exceeds the active-use value served (or potentially served) by road access (Bengston and Fan 1997; Brown 1993; Driver and others 1987, 1996; Payne and others 1992; Walsh and others 1984, 1990).

Building roads in roadless areas may reduce passive-use value significantly; decommissioning roads may increase such value. Building roads into roadless areas may serve values that require such access, however, and decommissioning roads may obstruct values and uses that require access. Decisionmakers need to consider all these tradeoffs. Individuals and affected groups often disagree aggressively about the passive-use value of specific roaded and roadless areas and the effects of building or decommissioning those roads (Bengston and Fan 1997). Thus an equity (or distribution) question must be considered: Whose desires should the Forest Service fulfill when stakeholders' values conflict? What criteria should be used to decide among them? What approaches can be taken to resolve the conflict?

The effects of roads on passive-use value differ by location and circumstance. Differences in the quality and uniqueness of landscapes modify passive-use-value effects from building or decommissioning roads. The relation between supply and demand also will affect the extent and strength of a passive-use value. For example, if many substitutes for a given roadless landscape exist, building a road in that area may have little or no effect on its passive-use value, just as the hunter's killing of a single elk does not reduce the passive-use value of elk because the species is still abundant. Likewise, if an abundance of roads are provided to resources that people want for active use, decommissioning or closing one road will have little effect. People with strong attachments to a special place, use, or road may suffer loss, however, unless they can find and adapt to a substitute.

Validly and reliably measuring changes in passive-use and other nonmarket value is costly and can sometimes exceed the cost of being wrong. Managers of national forest roads must understand such values, however, and the circumstances under which they are significant decision factors, to assure that the values can be included where appropriate. A survey-based method called contingent valuation (contingent valuation generally uses surveys or interviews to determine how much people say they would be willing to pay for some nonmarket good) that asks people to state their willingness to pay for nonmarket values can provide a useful indication of relative magnitude, but applying it to passive-use value of public goods is where the method is most vulnerable to flawed results, criticism, and controversy. Studies must be designed and applied carefully and the results interpreted cautiously. Other methods, such as value juries (Brown and others 1995), focus groups, public hearings, and other forms of public participation also can provide useful information. Quantitative measures should be taken only when the scale of the problem justifies sufficient investment for scientifically rigorous results.

If fully and correctly disclosed, the cost of opportunities foregone by preserving a roadless landscape can serve as the price to be paid for the values served by preservation. Preserving a roadless area may sometimes cause an opportunity cost in the form of alternative uses foregone, such as timber harvest, developed recreation, or fire suppression. If the opportunity cost has been fully disclosed to the decisionmaker, a decision to preserve a roadless landscape is a policy acknowledgment that the value created exceeds that opportunity cost. In a decision about whether to designate an area as roadless, opportunity cost can sometimes serve as the price to be paid for whatever values, including intangibles, are served by the designation. Stakeholders and decisionmakers can then decide—by judgment, negotiation, or analysis—whether the gain is worth the price (Bell 1996; Fight and others 1978, 1979; Randall and others 1979).

Reliability and degree of confidence—The scientific literature supports the general propositions that roadless natural landscapes and unique natural features and resources generate passive-use and other nonmarket values; that such values differ among individuals, groups, and landscape conditions; and that disagreement about nonmarket value fuels conflict. Legal precedent also validates policy concern. The effects of roads on passive-use and other nonmarket values have not yet been studied extensively, and the validity and reliability of methods for measuring the necessary values are still questionable.

Generalizability—No science-based procedures, analytical methods, formulas, tables of values, or handbooks are available for applying the general principles we have outlined to specific decisions or to transfer measured values from one place to another. Each project-scale decision requires original human-dimension inventory and assessment techniques, either by technical measurement or through public involvement. Managers making decisions on whether to build or remove roads in specific places always need to consider the principles and questions defined in the findings section. Roadless areas may have significant passive-use and other nonmarket value, depend-ing on the people affected and the availability of substitutes, but obtaining the required information requires original inventory and assessment for each decision. Expensive procedures may not be appropriate where the scale of the problem does not justify the cost.

Research in progress is exploring nonmarket active-use-value transfer (that is, generalizing by formulas and tables) among different site-specific situations. The results thus far are encouraging but not conclusive, although they may offer useful guidance in some situations (Rosenberger and Loomis 2000). We are not aware of any similar work on passive-use-values.

Secondary links—Passive-use value affects public attitudes toward the Forest Service as well as public willingness to accept and support proposed forest policies and plans. Roads and roadless areas sometimes take on symbolic meaning in the broader context of environmental concerns about such things as biodiversity, pollution, and ecosystem health. Passive-use value associated with symbolic issues triggered by changes in road distribution can be an important cause of conflict and litigation.

Conclusions—Extensive scientific evidence exists on passive-use and other nonmarket values in general and on applying them to unique natural environments, environmental accident damage assessment, and sensitive species. Little scientific evidence is available on the relations among roads, roadless landscapes, and passive-use value, however. Published studies demonstrate that people often do assign significant passive-use value to natural areas, including roadless ones, in specific places (Bishop 1978; Brookshire and others 1986; Carson and others 1999; Cicchetti and Wilde 1992; Ciracy-Wantrup 1968; Crowards 1997; Farmer and Randall 1998; Freeman 1993; Krutilla 1967; Krutilla and Fisher 1975; Loomis and White 1996; Mazzotta and Kline 1995; Morton 1999; Walsh and others 1984, 1990). National forest roads can be an important cause of ecological degradation. Under the right conditions and taken together, those studies also imply that national forest roads can cause a significant loss of passive-use values. The actual effect on passive-use value will be specific to the site and situation, however; the only refereed studies we found that document the specific relation between roads and passive-use value are Brown and others (1996) and Champ and others (1997). Rosenberger and Loomis (2000) compiled a comprehensive tabulation of nonmarket recreational values, including a bibliography of 162 studies.

Additional studies are needed to test hypotheses or estimate parameters that apply to specific decisions. General methodological and theoretical research not specifically focused on forest roads is ongoing in several disciplines, including environmental economics, sociology, psychology, political science, and anthropology. Several approaches are being pursued, including social and psychological surveys, ethnographic studies, methods for effective citizen participation, focus groups, citizen and value groups, and monetary valuation. The needed and ongoing research is long term, however, and must not delay making decisions in the short term, based on the best available current knowledge.

Heritage and Cultural Value of Roads

Issues—In addition to satisfying the American penchant for sightseeing by car and other forms of recreation requiring auto travel, roads and their features themselves sometimes have heritage value because of historic significance or architectural features. Roads also may affect areas considered sacred by American Indians or other religious groups. These issues can affect the legal and political framework for Forest Service road policy and management because important historical, social, and cultural values are often part of developing, maintaining, or decommissioning roads. Forest planning for transportation and for individual roads should incorporate information on heritage and cultural values for both roaded and unroaded areas.

Findings—Roads and associated features are part of the history of the nation. Some features are significant for their association with exploration and settlement, others for accomplishments in engineering, and still others for reasons of local history and culture. Roads and other transportation features figured prominently in the early nonindigenous settlement and development of the nation. Roads that were or are significant in this way include early Spanish roads, such as El Camino Real (the Royal Highway) in California and New Mexico; those that follow the routes of American Indian trails (Davis 1961); military roads such as Cook's trail, which crosses the forests of northern Arizona (Scott 1974); and some early routes established for commerce, such as the Santa Fe Trail, which crosses the Cibola National Forest. Given their historical role, such roads (many still in use) often are eligible for the National Register of Historic Places. Of equal importance, historic roads often have special meaning to people who live near them or have used them. Route 66, for example, which crosses the Kaibab National Forest, is considered historically valuable for its role in establishing regular, all-season east-west automobile transportation to California (Cleeland 1988, 1993).

Features forming part of or associated with a road may be historically or culturally valuable for their own merits (Fraser 1987). Bridges and other features built by the Civilian Conservation Corps often are fine examples of engineering and considered eligible for the National Register of Historic Places (Throop 1979). Many such bridges are on Forest Service roads. Roads also may have heritage value as part of a cultural landscape, such as the landscapes associated with homesteading, ranching, or logging. Even roadside advertising can have local cultural significance, such as the hand-painted message along an abandoned highway in the Cibola National Forest that claims "Curandera cures all." The National Park Service and the U.S. Committee of the International Council on Monuments and Sites recognized the heritage value of transportation corridors in a conference held in 1993 (USDI 1993).

Building, maintaining, and decommissioning roads can affect historical and cultural values. Roads often directly affect historical and archaeological sites. Building, maintaining, or decommissioning roads can damage or destroy archaeological sites (Spoerl 1988) with earthmoving equipment used on buried and surface remains, such as structures and other cultural materials. Roads also affect sites indirectly by increasing erosion or by making sites accessible to vandals. Less tangibly, but no less important, roads often affect areas that American Indians consider sacred, may limit their ability to conduct ceremonies that require privacy, and may even diminish the sacred qualities of such places. Building new roads, or adding to existing ones, can affect sacred areas that may qualify for the National Register of Historic Places as Traditional Cultural Properties (Parker and King 1990). The Cibola National Forest has recently been in litigation initiated by Sandia Pueblo over plans to rebuild a road through Las Huertas Canyon in New Mexico. The pueblo claims that the canyon is eligible to be a Traditional Cultural Property. A larger issue in this case is that the road and the traffic it brings affect use of the area for pueblo ceremonies. In northern California, similar issues surrounded the case of the Gasquet-Orleans Road on the Six Rivers National Forest (Theodoratus and others 1979), which concerned road building and resource extraction in an area that local American Indians considered sacred. The dispute over this road lasted many years, and its repercussions continue to be felt.

Generalizability—The findings are partially generalizable to all national forests but not to all decisions. As with sensitive species, some issues arise where heritage and cultural values are especially significant. Because of legal requirements and the intensity of concern among affected stakeholders, however, assessing cultural and heritage values is essential in every Forest Service decision about building or decommissioning roads.

Secondary links—Inadequate participation in road policy decisions by affected stakeholders concerned with heritage or cultural values can lead to litigation and political conflict. It also can stimulate symbolic opposition to the Forest Service on other fronts that even direct amelioration of the heritage or cultural concerns cannot resolve.

Conclusions—Good information is available on cases encountered by the Forest Service; it is generally after the fact, however, and pertains to actions taken to resolve conflicts caused by failure to consider the issues early and effectively in policy and management decisions. Existing information about heritage and cultural values relating to roads and roadless areas often may not be adequate; ongoing inventories tend to be project-specific rather than part of the general program. Obtaining information about sacred places from some American Indian groups is difficult because Forest Service styles of communication and negotiation often are incompatible with these cultures, and revealing sacred values and identifying sacred places to outsiders may be thought to imperil the values in need of protection.

Documentation—Much of the documentation for the heritage and cultural values of roads resides in administrative documents in the 50 state historic-preservation offices and the Advisory Council on Historic Preservation.

Economic Effects and Development

Issues—Both benefits and costs are associated with building, maintaining, and continued use of Forest Service roads. Likewise, benefits and costs are associated with removing existing roads. The issues revolve around whether the good things outweigh the bad things and what the extent of roads should be in national forests.

Findings—Some economic activity is supported by building and maintaining roads: economic activity also is supported by decommissioning roads. Analyses for the 1995 RPA program suggest that about 33 jobs economy wide (nationally) are supported per $1 million expenditure on building and maintaining roads (Alward and others 2000). A reasonable speculation might be that roughly the same rate of employment would be supported by removing existing roads and restoring the land underlying them. Road building and removal represent one-time stimuli to the economy, but maintaining roads is a recurring stimulus. After a road is removed, the jobs supported by road maintenance cease.

The major effects of roads on local economies, however, would be expected to result from the economic activity those roads support by providing access to the national forest and to communities in or near it. On Forest Service roads, that activity includes logging, silvicultural operations, and recreation, among others. Also supported is economic activity that depends on recreation, such as guides, outfitters, and rafting permittees. The roads also provide access for land management and firefighting operations.

Indirect (and approximate) indications of the amounts of economic activity that might be associated with changes in Forest Service roads can be obtained from several sources. Reports indicate that timber harvest from national forests supports about 16.5 jobs economy wide (in the local area) per million board feet harvested (USDA FS 1996). That estimate is conservative because it is based on summed local-area models. Recreational use of national forests supports a range of 1,000 to 2,000 jobs economy wide (nationally) per million trips, depending on the primary activity, based on analyses done for the 1995 RPA program (Alward and others 2000, Archer 1996).

Use of public lands, in general, follows roads. In Alaska, for example, intensity of use by both hunters and nonconsumptive wildlife users follows road corridors (Miller and McCollum 1997). Further, we hypothesize that more casual users—such as scenery gazers, picnickers, car campers, and day hikers that constitute the bulk of national forest recreationists—probably stay closer to the road than do some hunters and backpackers, the minority of national forest recreationists.

Whenever timber is cut and removed from the forest, roads will be needed; even helicopter logging at some point converts to road use by truck hauling. One issue is the quality of the roads and the length of their lives; that is, whether they are permanent and remain after timber harvesting ceases, or temporary and closed after harvest. Permanent roads are available for other activities over time, primarily recreation and management activities. Temporary roads are available for timber activity and some incidental activity during harvest, but when the roads are closed, benefits accruing from those roads cease. That the cost of maintaining a road over time could sometimes outweigh the cost of removing it at the end of one timber harvest cycle and rebuilding it for the next one is at least conceivable. Environmental effects (and cost) of multiple entries and decommissioning of temporary roads must be balanced against those of a single permanent road. Permanent roads cost more to build and maintain than temporary ones, with increased potential for degrading the ecosystem, but they can result in more benefits over longer periods than temporary roads because of the access they allow.

Roads affect spatial patterns of forest use. Changes in roads change those patterns. Recreational users are particularly attracted to or driven away from particular areas by the availability and ease of access. With decreased access to the national forest, some users might drop out and give up outdoor recreation. Others would shift their use to other areas, some on Forest Service land and others off. The result would be reduced economic activity in the locale where forest access was decreased and increased economic activity in areas where displaced users moved. In general, the effects would be reversed if access were increased. Sometimes, however, increased access could lead to decreased use and result in less local economic activity; for example, where new roads and associated commercial activity degrade a viewshed, which could decrease visits to view autumn foliage.

Another result of spatial shifts in recreational use could be to concentrate use in areas to which displaced users move. Concentrated use may increase environmental effects as well as decrease the quality of people's experiences. Crowding imposes costs on existing users in those areas by diminishing the benefits they received from their recreational use because of the inflow of displaced users from areas affected by decreased road access.

Anything that affects the demand for and benefits received from recreation and other uses of Forest Service land has subsequent economic effects, and it may alter development because land uses drive local economic activity. Forests and local economies will be affected differently, depending on the mix of local activities.

Building or removing Forest Service roads and maintaining existing roads can help mitigate ecosystem degradation associated with roads. Note that the tradeoffs are between the expense of minimizing or eliminating environmental degradation associated with Forest Service roads and access to Forest Service lands with associated economic activity.

Many roads are or have been funded by the timber program. Benefits accrue from use of those roads beyond timber, largely for recreation. This contrast presents a classic problem of joint cost allocation, and the accounting problem of attributing cost should not be used as an excuse for looking only at specific programs or components of the Forest Service mission.

The jobs and other economic activity supported by building and maintaining roads must be balanced against the cost of building and maintaining those roads, including costs resulting from choosing not to maintain selected roads. The question is, do the benefits associated with the roads, both direct and indirect from all sources, justify the cost incurred by society, including costs of increased ecosystem degradation from deferred or inadequate maintenance? Reports like this one can provide information on a wide variety of benefits and costs, but answering the question just posed is a policy decision.

Reliability, confidence, and generalizability—Analyses done for the 1995 RPA program provide a broad picture of national effects that can be expressed as averages and rates per unit of activity. They are not site-specific studies, and they do not estimate the effects on local areas. A few recreation-demand studies based on specific sites and regions provide corroborating evidence of the qualitative results (English 1997, McCollum and Miller 1994, Miller and McCollum 1997). The transportation literature contains some studies on roads and development (Berechman 1994, Broder and others 1992, Rephann 1993, Rietveld 1994), but those studies are mainly about highway systems, and though we expect their conclusions to be qualitatively relevant to the types of roads administered by the Forest Service, some attributes of Forest Service roads are so different that creating a complete picture is impossible. A primary gap in knowledge is understanding the links between policy or management actions and their effects on forest-based activity (both in the amount of activity undertaken by users and in the benefits they receive), especially for recreational and noncommodity uses. Changes in road availability and quality affect whether and how much users access the forest in particular areas. Road availability and quality also affect the quality of users' experiences, and thereby affect the benefit they receive. No access or access on a poorly maintained road, for example, could decrease benefit for some activities but have little or no effect on others. We did not find any activity-specific studies documenting the direction and size of such effects. Those factors are relevant because they drive demand for access to Forest Service land and the local economic activity associated with use of these lands.

Further gaps in knowledge exist on the distributive effects of new or improved and degraded or removed roads on forest use in local areas and on local economic activity. To what extent do the existence or lack of Forest Service roads, and their condition, attract or drive away users pursuing particular activities? The general development literature provides some insights and qualitative expectations for Forest Service roads, but empirical findings on the likely size of the effects are absent.

Conclusions—Empirical estimates are not available to document the size of the economic contribution of recreation-dependent commercial activities like guides, outfitters, and rafting permittees. Also missing are empirical estimates of benefits received from and economic activity supported by specific recreation activities in specific areas. Estimates are often obtained from national studies or site-specific studies in other areas and blindly applied to areas being analyzed.

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Appendix

This section draws from the analysis in the main document, with interpretations relevant to roadless and unroaded areas.

Forest Service Roadless Areas: A Synthesis of Science Information

Managing and maintaining existing forest roads has not kept pace with either the shifting balance of forest users or the increased scientific understanding of the ecological effects of roads. In particular, entry into roadless areas merits consideration of both benefits derived and risk of unacceptable impacts. Thus, managing for roadless area protection consists of positive steps such as providing for habitat conservation areas, watershed protection, critical habitat protection, contingency or passive-use values, and related land stewardship objectives. It also consists of restricting actions that may contribute to deteriorating environmental integrity, such as stand-replacing fires or large-scale insect outbreaks.

Questions affecting roadless areas include:

- Are significant and important social values associated with the existence and protection of wilderness and roadless areas?
- Does a road network in itself pose a risk to the integrity (as defined in the interior Columbia River basin study) of roadless forested ecosystems?
- Do roadless areas make substantial contributions to maintaining biodiversity and desirable habitat characteristics?
- Can roadless areas stay intact without management efforts that are facilitated by roads (for example, fire prevention, disease and pest control)?
- Does creating new roads in roadless areas have overriding benefits that outweigh the potential ecological costs?

Existing and perhaps new science information may be needed to assess some or all of the questions posed. In addition, methods from the social sciences are available to conduct surveys and assessments of public perceptions, values, and beliefs to determine the values that roadless areas hold in the mind of the public. This summary of existing information is an attempt to identify the ecological and biophysical characteristics of large nonroaded blocks of the forest and rangeland ecosystems that would permit conclusions about the value of maintaining such landscape features, and to examine the scientific aspects of a possible rationale for road building in currently roadless areas.

Ecological and biophysical aspects of roadless areas—An approach for providing the scientific basis of ecological and biophysical value is to summarize the known information on roadless areas at the landscape or large basin scale and proceed to smaller spatial scales. Questions that may be asked at the larger scale include the following:

- Is retention of existing roadless areas an important as part of a conservation strategy?
- Does the distribution of roadless systems affect the success of conservation strategies?
- Does the size of individual roadless areas affect the success of conservation strategies?

One of the few examples of landscape-scale analysis of road influences is the interior Columbia River basin environmental assessment. Analysis of fish distribution and status data for seven species of anadromous and resident salmonids in the Columbia basin showed that frequency of strong populations generally declined with increasing road densities. Additional analyses of road effects focused on four non-anadromous species, because effects of roads and other land uses on anadromous species may be masked by migrational and ocean-related factors (for example, dam passage, predation, and harvest). Three species showed significant effects from roads, either when occupied spawning and rearing areas were distinguished from unoccupied areas or when strong status was differentiated from depressed. The analysis suggested a decreasing likelihood of occupancy—or a decreasing likelihood of strong status if occupied—with increasing road density. No other variables except ground slope showed the consistent patterns across all species shown by the road density measures.

The investigation of the influence of roads on population status clearly showed an increasing absence and a decreasing proportion of strong populations with increasing road density for several subgroups. Additional evidence suggests that the lowest mean road density values (number of road miles per unit of area) always are associated with strong population status.

Based on the synthesis reported in the main body of this document, this trend is apparent for Yellowstone cutthroat trout, even though it was the only subgroup not showing a significant road effect in a logistic regression analysis. The lack of statistical significance in the face of apparent trends, however, points to complex interactions among the explanatory variables not adequately addressed in the relatively simple logistic model. Consistent, significant effects for other species may be further testament to the presence and pervasiveness of the effects. Strong relations between roads and the distribution and status of these species were detected despite the potential confounding effects of other variables (such as harvest, non-native introductions, and other habitat factors).

These results show that increasing road densities and their attendant effects are associated with declines in the status of four non-anadromous salmonid species. These species are less likely to use highly roaded areas for spawning and rearing and, if found, are less likely to have strong populations. This consistent pattern is based on empirical analysis of 3,327 combinations of known species' status and subwatershed conditions, which were limited primarily to forested lands administered by the Forest Service and the Bureau of Land Management. We would not expect the relation to be as strong on the nonforested, lower gradient lands administered by BLM. Of the four species examined, the redband trout is the only one supported by the low-gradient lands. Only in forested, high-elevation areas could redband trout status be clearly associated with road density changes.

Most aquatic conservation strategies acknowledge the need to identify the best habitats and most robust populations to use as focal points; from these, populations can expand where adjacent habitat can be usefully rehabilitated or the last refugia of a species can be conserved. These strategies also provide necessary experimental controls for evaluating the effects of land management activities in other areas. The ecological importance of unroaded areas has been highlighted in the Columbia basin assessment and in other reports cited in the main body of this paper.

The overlap of unroaded areas within and outside designated wilderness areas with stronghold watersheds for fish and other important conservation watershed efforts in the Columbia basin also was examined. Designated wilderness and unroaded areas are important anchors for strongholds throughout the basin. Unroaded areas occupy

41 percent of area with known and predicted strongholds in the east-side EIS area. One-third of this area is outside wilderness. Sixty-eight percent of known and predicted strongholds in the upper Columbia basin EIS area are unroaded, of which 37 percent are outside of wilderness.

Aquatic integrity in the Columbia basin was analyzed in relation to road densities and integrity ratings for other resources (forest, range, hydrology). Forest clusters with the highest integrity ratings for aquatic organisms were associated with low road densities; low integrity ratings corresponded with moderate or higher road densities. The range cluster having the highest aquatic and composite integrity also had mostly low road densities. The relations between road densities and integrity ratings for other range clusters were more variable, however (FEMAT 1993, Henjum and others 1994, Lee and others 1997). The correlation of basin or subbasin integrity is not total, suggesting the variables and interesting mechanisms are complex and nonuniform. Such data suggest that criteria be developed to examine the role of roadless areas in conservation strategies and permit assessing the risks taken when roadless blocks that are significant features at the landscape level are further intersected by roads.

- Does the distribution of roadless areas contribute to the ecological integrity of forested ecosystems?
- Does a conservation strategy that includes roadless areas need to be spatially explicit?

The distribution and the desirability of having well-distributed roadless area systems pose interesting scientific challenges. Historical trends significantly influenced the extent and distribution of roadless areas. Logging progressed from easily accessible, low-elevation forests to more difficult, high-elevation terrain; thus the remaining road-less areas tended to be at high elevations. We are unaware of a systematic analysis of this issue. Criteria that include assessing how well some roadless areas represent certain native ecosystems should be considered. This is especially the case at lower elevation sites that historically have seen the greatest harvesting effort and attendant road building. If the goal is to have a system of reserves consisting of representative, relatively undisturbed habitats, then roadless areas and the habitat types within them should be distributed over major ecoregions and be derived logically.

- Do corridors connect the high-quality roadless areas?

Biodiversity is, in simplest terms, the variety of life and its processes (Keystone Center 1991). Recent syntheses (Heywood and Watson 1995) emphasize the reciprocal relation between biodiversity—conceived as genetic and species diversity—and ecosystem function. The many species representing the biodiversity of an area play roles necessary for ecosystem function and, importantly, are the source of the variation enabling an ecosystem to adapt to change. The processes of a healthy, functioning ecosystem in turn support the many species. Appreciating the reciprocity means that biodiversity can be taken as a natural measure of the ecosystem as a whole and thus can integrate the many concerns listed.

Some species may play more important roles than others in the normal functioning of an ecosystem. Keystone species, for example, may define the major structural elements of an ecosystem as Douglas-fir does for forests in the Pacific Northwest, or they may—by virtue of their position in a complex trophic structure—act to maintain the diversity as keystone predators do for herbivores. The many species that do not seem to serve an important role in an ecosystem constitute a reservoir of potential adaptation to change.

Because an ecosystem cannot predict change, the very diversity of species acts as a hedge against it. Thus, biodiversity is important to long-term ecosystem function, and human activities that decrease biodiversity can impair it. Our working hypothesis is, then, that measures of biodiversity provide the best integrative assessment of the effects of roads on ecosystems.

Forest roads create corridors that not only permit invasion of alien, weedy species, but also permit entry of predators, including humans, to the forest environment and affect wildlife populations. Limited studies have shown that roads allow exotic species into areas where they historically have been absent or where appropriate habitat was not available (Parendes, 1997). Clearly, these secondary effects are promoted by the existence of roads but are not due to the roads themselves; however, the increase in human access to remote areas allowed by roads has a far more significant effect on native populations. High road densities are associated with a variety of negative human effects on some wildlife species. Black bear populations are inversely related to road density in the Adirondacks (Wisdom and others 2000). Increases in hunting pressure, particularly illegal hunting, have the potential to impact populations. Moose and caribou are particularly vulnerable to this kind of predation (Scott and Servheen 1985). Such connectivity will be important for endangered species where the gene pool is already limited, such as in the case of the Florida panther (*Puma concolor corgi*), and where gene exchange between populations in adjacent habitat may help species viability (Shrader-Frechette 1995). Connectivity also is important for species having large home ranges, and road avoidance or risk from road related mortality constitutes an additional threat to the populations, or may lead to undesirable, even dangerous animal-human interaction, as may be occurring with mountain lion (*Felis concolor*) populations in southern California.

Whenever forest roads are built, modified habitat and changes in animal behavior will lead to changes in risk to viability and distribution and even local extirpation in wildlife populations. Road avoidance behavior is characteristic of large mammals such as elk, bighorn sheep, grizzly bear, caribou, and wolf. Avoidance distances of 100 to 200 yards are common for these species. Road usage by vehicles and humans has a significant role in determining road avoidance behavior. In a telemetry study of black bear movements, interstate highways were almost never crossed, and roads with low traffic volume were crossed more frequently than roads with higher traffic volumes (Wisdom and others 2000.). It appears that in some cases, male bears may actually be using roads as travel corridors (Young and Beecham 1986, Zager 1980). Wolves in Wisconsin are limited to areas with overall mean road densities of 0.07 miles per square mile. Some studies have shown that the existence of a few large areas of low road density, even in a landscape of high average road density, may be the best indicator of suitable habitat for large vertebrates (Wisdom and others 2000.).

- Are roadless areas important to the conservation of high-quality aquatic and terrestrial habitats?

Again drawing on the Columbia River basin assessment, fish with strong populations occurred more frequently in areas with lower road densities. Supplemental analysis further showed that increasing road densities and their attendant effects were associated with declines in the status of four non-anadromous salmonid species. Fish seem to be less likely to use highly roaded areas for spawning and rearing and, where found, are less likely to have strong populations. Patterns based on empirical analysis of 3,327 combinations of known species status and subwatershed conditions are consistent and unmistakable, though limited primarily to forested lands administered by the Bureau of

Land Management and Forest Service. Although unroaded areas are significantly more likely to support strong populations, strong populations are not excluded from roaded watersheds. Possible reasons for this coexistence are that, in general, increased short- or long-term watershed and ecological risks are associated more with entering an unroaded area than with proceeding continuously with management activities in roaded areas to upgrade, maintain drainage, or close or obliterate existing roads (Lee and others 1997). The empirical evidence is correlational and, when the causes for the above observations are fully established, a more complex picture is likely to emerge.

At a more local scale, hydrologic and geomorphic interactions are a potential consequence of road building and presence that can involve altered flow regimes, increased sedimentation, local failures with local and “downstream” consequences for streams, riparian areas, and vegetation cover. For example, the FEMAT (1993) analysis stats, “Management activities in roadless areas will increase the risk of aquatic and riparian habitat damage and potentially impair the capacity of Key Watersheds to function as intended...[while]...most timber-suitable roadless acreage can be harvested either directly from existing roads or from helicopters.” Further, “if all timber-suitable roadless remains unroaded in Option 9, then the estimated reduction for the total regional probably sale quantity is less than 0.2 percent.” In terms of aquatic effects, the Columbia basin assessment summaries include the following statements: “Roads provide access, and the activities which accompany access magnify the negative effects on aquatic systems beyond those solely due to roads.” Among other findings, the assessment “...subwatersheds supporting strong populations were found on Forest Service administered lands (75 percent) and a substantial number (29 percent) are located within designated Wilderness areas and National Parks.” Thus, the data “...clearly show increasing absence and decreasing proportion of strong [fish] populations with increasing density for some subgroups” (FEMAT 1993). Other studies found that the length of road segments connected to the stream network at stream crossings or gully-debris slide tracks amounted to a 40-percent extension of the stream network length in a Cascade Range watershed (Jones and others, in prep; Wemple 1999).

High-quality terrestrial habitats may be affected by the potential for invasion of exotic plants and animals that can displace or threaten native populations; that is, affect biodiversity, which can be increased by roads. Migrating populations of rare amphibians may be killed during road use; disease and pathogens are spread more rapidly and widely if roads are present (Kiester and Slatkin 1974). The preponderance of the negative findings in many scientific studies also suggests that the potential for ameliorating or minimizing the unwanted effects exists, even if it has not been made a prime objective historically. Lastly, some positive ecological results may follow (though they are proportionately less significant) that roads create edge environments exploited by small mammals, can sustain some desirable species, and provide useful niches. Maintaining an optimum balance is a function of the long-term magnitude of road networks; for the present system, the need for additional niches and habitats is difficult to demonstrate.

A full scientific view of the data on roadless areas cannot stop at the local scale, but must ultimately view the presence of roaded and roadless areas in a landscape context and be able to draw the distinction between a large road network and small roadless areas or large roadless areas and a small road network. Again drawing on the Columbia basin assessment, we note that “while unroaded areas are significantly more likely to support strong populations, strong populations are not excluded from roaded watersheds.... the scale of the subwatershed (8000 ha on average) at which strong populations are identified may mask potential disconnects between the real locations of strongholds and roads. The significance of the impacts and benefits will be affected and must withstand rigorous scientific approaches over a spectrum of possibilities and of scales" (Lee and others 1997).

Social, aesthetic, and economic values of roadless areas—The interaction between roadless areas and people's aesthetic and spiritual beliefs about the landscape probably affects people's perceptions in many different ways. We know that passive or "nonuse" values include "existence" and "bequest" value. Existence value pertains to things, places, or conditions people value simply because they exist, without any intent or expectation of use. Bequest value pertains to a desire people may have to allow others, such as future generations, to receive benefit from a resource (Peterson and Sorg 1987, Randall and others 1979). The issues are as follows:

- People assign significant passive-use value to national forest landscapes or attributes.
- Forest Service road policies or management actions affect passive-use values.

People do assign passive-use (nonuse) value to natural resources, and passive-use value may exceed the active-use value served by road access to the resource. Invasion of roads will reduce some aspects of passive-use value in natural areas. Likewise, obliteration of roads may increase such value. Building roads into roadless areas may, however, serve values that require access, and obliterating roads may obstruct values and uses that require access, so tradeoffs need to be considered. Though not universally shared, a strong value is doubtless attached to the continued existence of wilderness and roadless areas, including those in national forests.

The relation between roadless areas and recreation on national forests is highly complex. Research findings are limited and uneven on the issues of direct, indirect, and secondary effects on recreation of altering the national forest road system. Indirect evidence and related research provide the following insights:

- Roads provide corridors of access to various national forest sites, settings, and visual and aesthetic experiences; in fact, almost all recreation in national forests depends to some degree on road access.
- Roads provide access to remote areas and wilderness but at the same time can reduce opportunities for solitude elsewhere.
- The amount of roading and the amount of recreation use are positively correlated, sometimes leading to heavy concentrations of use, and roads may be the only means of enjoyment for persons with some forms of disability.
- Demand for forest recreational opportunities continues to grow regionally and nationally.
- Placement, scale, class, and setting of roads can greatly affect the quality of scenic views of national forests and access to outstanding vistas.

The three most highly ranked uses of lands administered by the Forest Service and Bureau of Land Management in the basin today are timber, fishing, and hunting. Projected uses by 2045 will be motor viewing and day and trail use; this for an area where 70 percent of the unroaded areas of >200,000 acres occurs in the lower 48 states (Cordell and Bergstrom 1991, Tarrant and others 1999).

- Does a roadless area preclude needed access for public services and resources as well as conservation management?

Roadless areas not already congressionally withdrawn (for example, as a designated wilderness area) total about 34 million acres in national forests. Of these, 9 million acres have been identified as suitable for timber production. Management practices and natural resource use may suggest strong reasons for entry into the 9 million acres (Coghlan and Sowa 1997). Timber harvesting using roadless approaches in these areas would lead to greater reliance on helicopter logging systems, which increase logging costs. The FEMAT study (1993) suggests that in key watersheds, the reduction in timber volume would be about 0.3 percent, and reduction by prohibiting entry into existing roadless areas not congressionally withdrawn in all areas considered by FEMAT (that is, the range of the northern spotted owl) would be 6 percent.

For the interactions of grazing rights, grazing access, and roads, essentially no scientific information exists analyzing the ecological, administrative, or economic effects of roads on administering the Forest Service range management program, and the synthesis in the main report did not uncover data specific to the relation of roadless areas and grazing practices (Peterson and Sorg 1987).

That improved road access leads to increased efficiency and effectiveness of fire suppression activities is a long-held tenet of fire fighting. Much of the effectiveness of past fire suppression policies probably can be attributed to increased access for ground crews and equipment, particularly under weather and fuel situations where fire behavior is not severe. Under the severe conditions associated with intense, rapidly spreading fires, the value of forest roads for access or as fuel breaks is likely to be minimal. However, quantification of these effects in published research in the United States is minimal. But it should be noted that indirect effects of increased access have increased the role of human-caused ignitions, and this is particularly true in areas of expansion of urban and rural development into wildland interfaces.

Roadless areas: conclusions—The scientific literature provides a framework of general principles regarding the nonuse values of present roadless areas and may even be extended to apply to areas where road decommissioning may recreate roadless areas. Such values include areas (1) having significant amounts of interior habitat for many forest species now being observed under the “survey and manage” concept of the Northwest Forest Plan, (2) maintaining connectivity of habitat for species having large home-ranges, (3) valuing the existence of forest “reserves” that permit the continued functioning of representative habitat types in a state of least human disturbance, and (4) becoming aware that forest-stream interactions seem to confer somewhat stronger fish viability in areas of low to no road densities. At present, no science-based analytical models, formulas, tables, or handbooks are available that the manager can use to apply the general principles to specific decisions, though pilot efforts are now underway by the USDA Forest Service to develop such tools. Such tools will provide methods that permit judgments about offsetting benefits and impacts from road building and usage, which suggests that we will have the means at hand to decide on an agreed on mix of roaded vs. roadless areas in national forests.

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Review

Biodiversity management in the face of climate change: A review of 22 years of recommendations

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Climate change creates new challenges for biodiversity conservation. Species ranges and ecological dynamics are already responding to recent climate shifts, and current reserves will not continue to support all species they were designed to protect. These problems are exacerbated by other global changes. Scholarly articles recommending measures to adapt conservation to climate change have proliferated over the last 22 years. We systematically reviewed this literature to explore what potential solutions it has identified and what consensus and direction it provides to cope with climate change. Several consistent recommendations emerge for action at diverse spatial scales, requiring leadership by diverse actors. Broadly, adaptation requires improved regional institutional coordination, expanded spatial and temporal perspective, incorporation of climate change scenarios into all planning and action, and greater effort to address multiple threats and global change drivers simultaneously in ways that are responsive to and inclusive of human communities. However, in the case of many recommendations the how, by whom, and under what conditions they can be implemented is not specified. We synthesize recommendations with respect to three likely conservation pathways: regional planning; site-scale management; and modification of existing conservation plans. We identify major gaps, including the need for (1) more specific, operational examples of adaptation principles that are consistent with unavoidable uncertainty about the future; (2) a practical adaptation planning process to guide selection and integration of recommendations into existing policies and programs; and (3) greater integration of social science into an endeavor that, although dominated by ecology, increasingly recommends extension beyond reserves and into human-occupied landscapes.

Contents

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1. Introduction

Climate change poses major new challenges to biodiversity conservation. As atmospheric CO_2 increases over the next century, it is expected to become the first or second greatest driver of global biodiversity loss (Sala et al., 2000; Thomas et al., 2004). Global average temperatures have increased 0.2 °C per decade since the 1970s, and global average precipitation increased 2% in the last 100 years (IPCC, 2007a). Moreover, climate changes are spatially heterogeneous. Some locations, such as the Arctic, experience much larger changes than global means, while others are exposed to secondary effects like sea level rise (IPCC, 2007a). Climate change may have already resulted in several recent species extinctions (McLaughlin et al., 2002; Pounds et al., 2006). Many species ranges have moved poleward and upward in elevation in the last century (Parmesan and Yohe, 2003; Root et al., 2003) and will almost certainly continue to do so. Local communities are disaggregrating and shifting toward more warm-adapted species (Parmesan, 2005). Phenological changes in populations, such as earlier breeding or peak in biomass, are decoupling species interactions (Walther et al., 2002).

These changes raise concerns about the effectiveness of existing biodiversity protection strategies (Halpin, 1997; Hannah et al., 2002; Peters and Darling, 1985; Scott et al., 2002). Biodiversity conservation relies predominately on fixed systems of protected areas, and the mandated goals of many conservation agencies and institutions are to protect particular species assemblages and ecosystems within these systems (Lemieux and Scott, 2005; Scott et al., 2002). With the magnitude of climate change expected in the current century, many vegetation types and individual species are expected to lose representation in protected areas (Araujo et al., 2004; Burns et al., 2003; Lemieux and Scott, 2005; Scott et al., 2002). Reserves at high latitudes and high elevations, on low-elevation islands and the coast, and those with abrupt landuse boundaries are particularly vulnerable (Sala et al., 2000; Shafer, 1999). Landscapes outside of protected areas are hostile to the survival of many species due to human infrastructure and associated stressors, such as invasive species, hunting, cars, and environmental toxins. Such fragmentation directly limits species migration and gene flow. Projected rates of climate change are also faster than they were in the past – so rapid that in situ genetic adaptation of most populations to new climate conditions is not likely (Jump and Penuelas, 2005), nor is migration likely to be fast enough for many species (Davis and Shaw, 2001). Moreover, even if major global action reduced emissions significantly within the next years or capped them at year 2000 levels, the thermal inertia of the oceans will continue to drive climate change for decades and will require adaptive responses (Meehl et al., 2005; Wigley, 2005). A recent update of atmospheric CO_2 growth rate, which has more than doubled since the 1990s as global economic activity increases and becomes more carbon-intensive, makes clear that significant global emissions reductions are a distant goal at best (Canadell et al., 2007).

How should we modify our biodiversity protection strategies to deal with climate change? Here we focus on adaptation strategies. Adaptation is broadly defined as adjustment in human or natural systems, including structures, processes, and practices (IPCC, 2007b). Scientists have written about adaptation with increasing frequency over the last two decades, but developments in this area have progressed slowly. For years, emissions mitigation has largely been the only game in town, with little governmental or private support for climate change adaptation. For instance, the United States National Park Service (NPS) in collaboration with the Environmental Protection Agency (EPA) has created a ‘Climate Friendly Park’ program. It aims to reduce greenhouse gas emissions, but it does not include measures or incentives to park managers to build and test adaptation strategies to preserve biodiversity under climate change. In many ways, adaptation science has begun to develop only very recently in response to recent widespread acceptance by governments and private citizens of the certainty of climate change.

In this paper we review the growing, published literature specifically addressed at biodiversity management and adaptation in the face of climate change. We consider biodiversity to include all types of organisms at all scales, from genes to ecosystems. The genesis for our review was the 2006 annual meeting of the California Invasive Plant Council, where climate change was identified by both researchers and practitioners as a key issue for action. Discussions throughout the meeting, however, made clear that practitioners felt at a loss for practical steps to take. Managers working at local preserves were particularly uncertain about what, if anything, they could do to prepare for climate change. We use this review in order to highlight what actions and actors scientists have so far identified to address climate change, and to explore how recommendations inform an adaptation planning process at various management scales. Scott and Lemieux (2005) reviewed a similar literature but focused on park management. Here we explore adaptation planning across scales and in both protected and unprotected areas.

2. Methods

We used Web of Science, including Science Citation Index Expanded, Social Science Citation Index, and Arts and Humanities Citation databases from 1975 to March 2007, to search for published journal articles on climate change and biodiversity management. We used the search terms "climate change", "global warming", "climatic change", "climate-change" and "changing climate" in all possible combinations with the search string "management OR biodiversity OR adaptation OR conservation OR restoration OR planning OR reserve design OR strategy OR land-use OR landuse OR landscape OR protected area OR park". Articles that discussed strategies for both biodiversity and related ecosystem services were included, but we excluded articles that only addressed ecosystem services such as management strategies for carbon stocks, human infrastructure, and food security. We also did not attempt to review studies that explore climate impacts on ecosystem components and processes without making explicit recommendations for biodiversity management. This literature is large and has been reviewed elsewhere (Kappelle et al., 1999; McCarty, 2001; Walther et al., 2002). From these searches, we identified and read 281 prospective articles, and from these culled those that provided explicit recommendations for management in the face of climate change. An additional four articles published after March 2007 were included, which were found through personal communication.

To analyze recommendations, we created a database in which we recorded every recommendation for action or information in the exact language used in the paper and answered a series of questions designed to synthesize recommendations and identify biases in the literature to date. We asked:

(1) *In what formal and informal contexts does action need to occur?* To answer this question, we categorized recommendations into broad spheres of activity: (1) policy reform, (2) science and technology effort and advances, (3) changes in conservation sector activity including restoration, or (4) changes in individual and community behavior, such as by farmers, ranchers, and other private landowners.

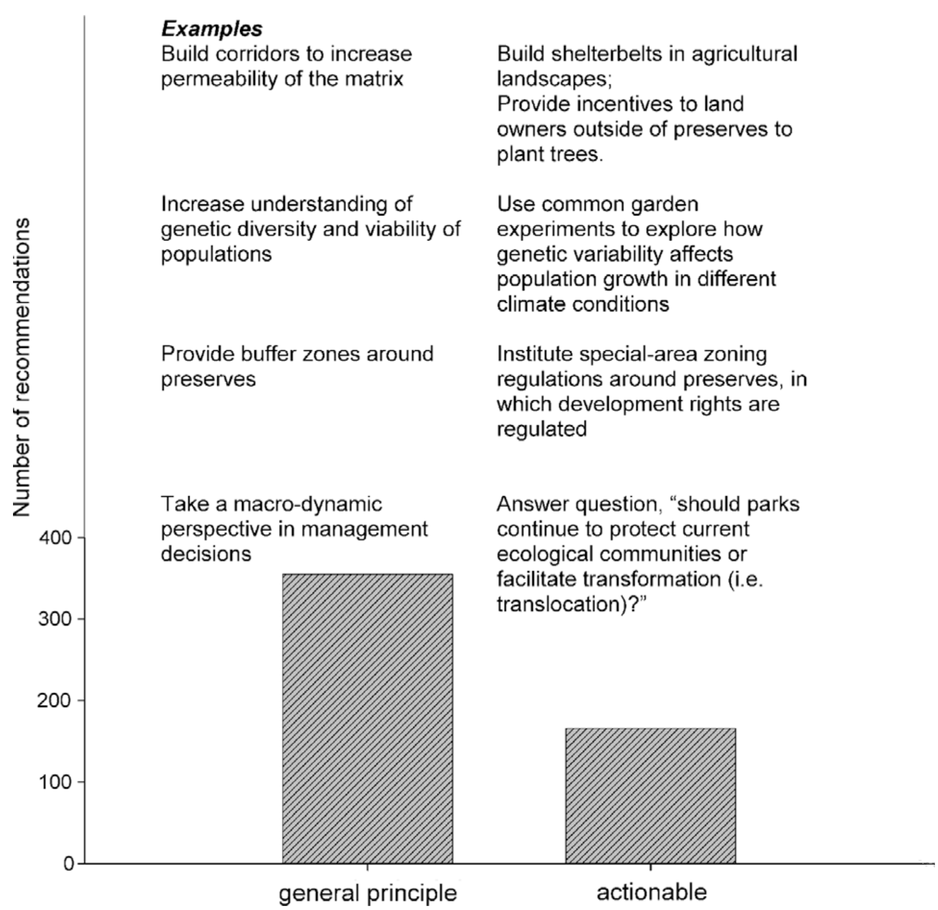

Fig. 1 – Examples and distribution of recommendations classified as "general principle" and "actionable". Most recommendations offer general principles for climate change adaptation but lack specificity needed for implementation.

(2) *What is the basis for the recommendation?* We recorded what information an author used to formulate their recommendation. Categories included empirical experimental data, simulation, literature review, case studies, interviews, or workshops. We also included the term 'ecological reasoning' to encompass studies based predominately on theory and opinion.
(3) *Is the recommendation a general principle or actionable?* A recommendation was considered a "general principle" if it provided a guiding concept, such as "build flexibility", but was generic open-ended and without example of who should act or what one should do (see Fig. 1). "Actionable" was given to those recommendations that identified a very clear who and what and often gave examples, such as, "[in restoration] use a broader range of species than prescribed solely on local basis to build system resilience (Harris et al., 2006)".
(4) *Is the recommendation for biodiversity or for biodiversity and related ecosystem services?*
(5) *Is there a geographic context for the recommendation?*
(6) *Does the article focus on a biome or ecosystem-type?*
(7) *Where in the landscape would the recommendation apply?* We categorized recommendations as applying to reserve (any public or private land-holding dedicated to biodiversity protection and maintenance, synonymous with protected area), or human-use lands (the matrix), or non-specific, meaning the recommendation could be enacted in either reserve or matrix land.
(8) *Does the recommendation describe an information need or a necessary action?* All recommendations for research were categorized as information needs, while the 'action' category included recommendations such as building corridors, reforming policy or buying more land.

To minimize variation in how articles were classified as a function of when they were read (i.e. the 1st paper entered compared to the 100th), records in the database were periodically shuffled by different criteria (i.e. year published or geographic context) and then re-classified. In addition, both authors coded a sub-sample of recommendations. After compiling the database, similar records were grouped into 'recommendation' categories. We tabulated the most common recommendations and ranked them by frequency cited overall.

3. Results and discussion

We recorded 524 recommendations from 113 papers, published in 57 different source journals and three books. Recommendations ranged from calls for specific types of modeling (e.g. inexact-fuzzy multiobjective programming (Huang et al., 1998) to broad shifts in governance structures (Tompkins and Adger, 2004) (Table 1). The number of papers published on this topic has increased dramatically in recent years (Fig. 2). Thirty-three percent of recommendations addressed biodiversity protection in conjunction with related ecosystem services, including forest products, fisheries and hunting, agriculture and grazing, and human health. Recommendations call for research, leadership and reform by a range of actors in several sectors; Emphasis in this set of literature is on science and nature conservation rather than on social or political adaptation measures (Fig. 3), with an emphasis somewhat more focused on reserve land over the matrix (Fig. 4a). Action is weighted more than information needs (Fig. 4b). When information needs were identified, they were overwhelmingly calls for more ecological rather than social scientific data (Fig. 4c). Recommendations are biased toward North America and Europe (Fig. 5a) and forests ecosystems (Fig. 5b).

Recommendations address various stages in an adaptation process, from research needs to methods for impact assessments to large-scale changes in policies by governmental, academic or non-governmental institutions (Table 1). About 70% of recommendations were classified as general principles under our classification scheme rather than specific, actionable strategies or tactics (Fig. 1). For example, seven authors suggest flexibility in management approaches, but only Millar et al. (2007) suggest flexibility and follow with a definition of what that means: willingness to change course, risk-taking including doing nothing, and capacity to reassess conditions frequently. Climate change adaptation work, at least in this literature, is still largely at the "idea" stage – it is based predominately on ecological reasoning rather than specific research, case studies, or empirical data (Fig. 5c), and it is largely nonspecific in the geographic areas or biome types that it targets (Fig. 5a and b). Many articles based on concrete modeling work or empirical studies of species responses to climate change tended either to not elaborate their results to management directives, or to present recommendations in vague terms such as, "restoration should be considered". Alternatively, very specific recommendations were proposed and not generalized for use outside of the target system. There appears to be a need for a happy medium between highly specific recommendations useful only in target areas and highly generalized recommendations that fail to inspire application (Halpin, 1997). This happy medium is likely to emerge rapidly as climate change adaptation science grows.

In the literature reviewed here, few recommendations suggested a process a manager could use to develop an adaptation plan and evaluate its usefulness (but see Hannah et al., 2002). More information on adaptation frameworks are developed in reports by Parks Canada (Welch, 2005), the NCEAS Conservation and Climate Change Working Group 2 (personal communication), and England's Department for Food Environment and Rural Affairs (http://www.defra.gov.uk/wildlife-countryside/resprog/findings/ebs-climate-change.pdf), which were not reviewed here. In practice, planners and managers could apply recommendations in at least three ways. At the broadest scale, long-term planning and policy formulation should tackle adaptation for whole landscapes and regions, with tools like reserve selection, ecosystem management, and landuse zoning schemes. Second, managers of individual reserves might want to know what they can do at their sites, individually or in concert with other sites. Third, rather than initially pursuing an idealized regional, landscape, or site-scale plan, the first practical step for many managers, conservation stakeholders and policymakers is to evaluate and adapt existing conservation plans. In the following

Table 1 – List of recommendations for climate change adaptation strategies for biodiversity management assembled from 112 scholarly articles. 524 records were condensed into 113 recommendation categories and are ranked by frequency of times cited in different articles.

Rank	Recommendation	No. articles	References
1	Increase connectivity (design corridors, remove barriers for dispersal, locate reserves close to each other, reforestation	24	Beatley (1991), Chambers et al. (2005), Collingham and Huntley (2000), Da Fonseca et al. (2005), de Dios et al. (2007), Dixon et al. (1999), Eeley et al. (1999), Franklin et al. (1992), Guo (2000), Halpin (1997), Hulme (2005), Lovejoy (2005), Millar et al. (2007), Morecroft et al. (2002), Noss (2001), Opdam and Wascher (2004), Rogers and McCarty (2000), Schwartz et al. (2001), Scott et al. (2002), Shafer (1999), Welch (2005), Wilby and Perry (2006) and Williams (2000)
2	Integrate climate change into planning exercises (reserve, pest outbreaks, harvest schedules, grazing limits, incentive programs	19	Araujo et al. (2004), Chambers et al. (2005), Christensen et al. (2004), Dale and Rauscher (1994), Donald and Evans (2006), Dyer (1994), Erasmus et al. (2002), Hulme (2005), LeHouerou (1999), McCarty (2001), Millar and Brubaker (2006), Peters and Darling (1985), Rounsevell et al. (2006), Scott and Lemieux (2005), Scott et al. (2002), Soto (2001), Staple and Wall (1999), Suffling and Scott (2002) and Welch (2005)
3	Mitigate other threats, i.e. invasive species, fragmentation, pollution	17	Bush (1999), Chambers et al. (2005), Chornesky et al. (2005), Da Fonseca et al. (2005), de Dios et al. (2007), Dixon et al. (1999), Halpin (1997), Hulme (2005), McCarty (2001), Noss (2001), Opdam and Wascher (2004), Peters and Darling (1985), Rogers and McCarty (2000), Shafer (1999), Soto (2001), Welch (2005) and Williams (2000)
4	Study response of species to climate change physiological, behavioral, demographic	15	Alongi (2002), Chambers et al. (2005), Crozier and Zabel (2006), Dyer (1994), Erasmus et al. (2002), Fukami and Wardle (2005), Gillson and Willis (2004), Honnay et al. (2002), Hulme (2005), Kappelle et al. (1999), McCarty (2001), Mulholland et al. (1997), Noss (2001), Peters and Darling (1985) and Swetnam et al. (1999)
	Practice intensive management to secure populations	15	Bartlein et al. (1997), Buckland et al. (2001), Chambers et al. (2005), Chornesky et al. (2005), Crozier and Zabel (2006), Dixon et al. (1999), Dyer (1994), Franklin et al. (1992), Hulme (2005), Morecroft et al. (2002), Peters and Darling (1985), Soto (2001), Thomas et al. (1999), Williams (2000) and Williams et al. (2005)
	Translocate species	15	Bartlein et al. (1997), Beatley (1991), Chambers et al. (2005), de Dios et al. (2007), Halpin (1997), Harris et al. (2006), Honnay et al. (2002), Hulme (2005), Millar et al. (2007), Morecroft et al. (2002), Pearson and Dawson (2005), Peters and Darling (1985), Rogers and McCarty (2000), Schwartz et al. (2001), Shafer (1999) and Williams et al. (2005)
5	Increase number of reserves	13	Burton et al. (1992), Dixon et al. (1999), Hannah et al. (2007), Hughes et al. 2003, LeHouerou (1999), Lovejoy (2005), Peters and Darling (1985), Pyke and Fischer (2005), Scott and Lemieux (2005) (2007), van Rensburg et al. (2004), Wilby and Perry (2006) and Williams et al. (2005)
6	Address scale problems match modeling, management, and experimental spatial scales for improved predictive capacity	12	Chornesky et al. (2005), Da Fonseca et al. (2005), Dale and Rauscher (1994), Ferrier and Guisan (2006), Guisan and Thuiller (2005), Huang (1997), Hughes et al. (2003), Kueppers et al. (2004), Kueppers et al. (2005), Mulholland et al. (1997), Noss (2001), Root and Schneider (1995) and Root and Schneider (2006)
	Improve inter-agency, regional coordination	12	Bartlein et al. (1997), Cumming and Spiesman (2006), Da Fonseca et al. (2005), Grumbine (1991), Hannah et al. (2002), Lemieux and Scott (2005), Rounsevell et al. (2006), Scott and Lemieux (2005), Soto (2001), Suffling and Scott (2002), Tompkins and Adger (2004) and Welch (2005)
7	Increase and maintain basic monitoring programs	11	Chambers et al. (2005), Cohen (1999), Huang (1997), Rogers and McCarty (2000), Root and Schneider (1995), Schwartz et al. (2001), Shafer (1999), Staple and Wall (1999), Suffling and Scott (2002), Wilby and Perry (2006) and Williams (2000)
	Practice adaptive management	11	Allison et al. (1998), Chambers et al. (2005), Hulme (2005), Lasch et al. (2002), Maciver and Wheaton (2005), Millar et al. (2007), Scott and Lemieux (2005), Staple and Wall (1999), Suffling and Scott (2002), Tompkins and Adger (2004) and Welch (2005)
	Protect large areas, increase reserve size	11	Beatley (1991), Bellwood and Hughes (2001), Burton et al. (1992), Bush (1999), Halpin (1997), Hulme (2005), Morecroft et al. (2002), Peters and Darling (1985), Shafer (1999), Soto (2001) and Watson (2005)

Table 1 – continued

Rank	Recommendation	No. articles	References
8	Create and manage buffer zones around reserves	10	Bush (1999), de Dios et al. (2007), Halpin (1997), Hannah et al. (2002), Hartig et al. (1997), Hughes et al. (2003), Millar et al. (2007), Noss (2001), Shafer (1999) and van Rensburg et al. (2004)
9	Create ecological reserve networks large reserves, connected by small reserves, stepping stones	8	Allison et al. (1998), Collingham and Huntley (2000), de Dios et al. (2007), Gaston et al. (2006), Opdam et al. (2006), Opdam and Wascher (2004), Shafer (1999) and Welch (2005)
	Develop improved modeling and analysis capacity i.e. more effective software, integration with GIS, integrate greater complexity	8	Chornesky et al. (2005), Ferrier and Guisan (2006), Guisan and Thuiller (2005), Guo (2000), Huang et al. 1998, Mulholland et al. (1997), Peters and Darling (1985) and Rounsevell et al. (2006)
	Do integrated study of multiple global change drivers	8	Dale and Rauscher (1994), Desanker and Justice (2001), Donald and Evans (2006), Halpin (1997), Hannah et al. (2002), McCarty (2001), Watson (2005) and Williams (2000)
	Improve techniques for and do more restoration wetlands, rivers, matrix	8	Da Fonseca et al. (2005), de Dios et al. (2007), Dyer (1994), Hartig et al. (1997), Lovejoy (2005), Millar et al. (2007), Mulholland et al. (1997) and Shafer (1999)
	Increase interdisciplinary collaboration	8	Gillson and Willis (2004), Guisan and Thuiller (2005), Hannah et al. (2002), Hulme (2005), Kappelle et al. (1999), Root and Schneider 1995, Soto (2001) and Williams (2000)
	Promote conservation policies that engage local users and promote healthy human communities	8	Chapin et al. (2006), Desanker and Justice (2001), Eeley et al. (1999), Lovejoy (2005), Opdam and Wascher (2004), Ramakrishnan (1998), Tompkins and Adger (2004) and McClanahan et al. (2008)
	Protect full range of bioclimatic variation	8	Bush (1999), Eeley et al. (1999), McCarty (2001), Noss (2001), Pyke et al. (2005), Pyke and Fischer (2005), Shafer (1999) and Thomas et al. (1999)
	Soften landuse practices in the matrix	8	Beatley (1991), Burton et al. (1992), Da Fonseca et al. (2005), Franklin et al. (1992), Hannah et al. (2002), Noss (2001), Williams (2000) and Woodwell (1991)
10	Adopt long-term and regional perspective in planning, modeling, and management	7	Eeley et al. (1999), Ferrier and Guisan (2006), Franklin et al. (1992), Guo (2000), Lovejoy (2005), Millar and Brubaker (2006), Opdam and Wascher (2004), Peters and Darling (1985), Peterson et al. (1997), Scott et al. (2002) and Welch (2005)
	Re-asses conservation goals (i.e. move away from concepts of natural, embrace processes over patterns	7	Franklin et al. (1992), Hulme (2005), Millar et al. (2007), Scott and Lemieux (2005) (2007), Scott et al. (2002) and Suffling and Scott (2002)
	Study species dispersal across landuse boundaries, gene flow, migration rates, historic flux	7	Guo (2000), Halpin (1997), Hughes et al. (2003), Kappelle et al. (1999), Lovejoy (2005), Opdam and Wascher (2004) and Rice and Emery (2003)
	Study species distributions current and historic	7	Da Fonseca et al. (2005), Eeley et al. (1999), Erasmus et al. (2002), Guo (2000), Hannah et al. (2002), Kappelle et al. (1999) and Millar and Brubaker (2006)
11	Broaden genetic and species diversity in restoration and forestry	6	Burton et al. (1992), de Dios et al. (2007), Harris et al. (2006), Maciver and Wheaton (2005), McCarty (2001), Millar et al. (2007), Rice and Emery (2003) and Staple and Wall (1999)
	Develop adaptation strategies now; early adaptation is encouraged	6	Huang et al. (1998), Hulme (2005), Lemieux and Scott (2005), Scott and Lemieux (2005) (2007) and Welch (2005)
	Do not implement CO_2 emission mitigation projects that negatively impact biodiversity	6	Chambers et al. (2005), Klooster and Masera (2000), Koziell and Swingland (2002), Kueppers et al. (2004) and Streck and Scholz (2006), Welch (2005)
	Manage for flexibility, use of portfolio of approaches, maintain options	6	Eeley et al. (1999), Hulme (2005), Kappelle et al. (1999), Lovejoy (2005), Millar et al. (2007) and Welch (2005)
	Validate model results with empirical data	6	Dale and Rauscher (1994), Guisan and Thuiller (2005), Hulme (2005), Malcom et al. (2006), Opdam and Wascher (2004) and Watson (2005)
12	Do regional impact assessments	5	Cohen (1999), Desanker and Justice (2001), Lasch et al. (2002), Lindner et al. (1997) and Suffling and Scott (2002)
	Identify indicator species	5	Chambers et al. (2005), Hulme (2005), Noss (2001), Underwood and Fisher (2006) and Welch (2005)
	Initiate long-term studies of species responses to climate	5	Mulholland et al. (1997), Noss (2001), Opdam and Wascher (2004), Peters and Darling (1985) and Root and Schneider (2006)
	Model species ranges in the future	5	Allison et al. (1998), Da Fonseca et al. (2005), Hannah et al. (2002), Kerr and Packer (1998) and Kriticos et al. (2003)
	Protect refugia current and predicted future	5	Bush (1999), Chambers et al. (2005), Eeley et al. (1999), Noss (2001) and Scott et al. (2002)
	Study adaptive genetic variation	5	Harris et al. (2006), Hughes et al. (2003), Jump and Penuelas (2005), Kappelle et al. (1999) and Rice and Emery (2003)

(*continued on next page*)

Table 1 – continued

Rank	Recommendation	No. articles	References
13	Leadership by those with power senior management, government agencies	4	Scott and Lemieux (2005) (2007), Tompkins and Adger (2004) and Welch (2005)
	Limit CO_2 emissions	4	Hannah et al. (2007), Hannah et al. (2005), Mayer and Rietkerk (2004) and Rogers and McCarty (2000)
	Predict effects of directional climate change on ecosystems, communities, populations	4	Allison et al. (1998), de Dios et al. (2007), Kappelle et al. (1999) and Root and Schneider (2006)
	Preserve genetic diversity in populations	4	Chambers et al. (2005), de Dios et al. (2007) and Lovejoy (2005), Noss (2001)
	Represent each species in more than one reserve	4	Halpin (1997), Millar et al. (2007), Peters and Darling (1985) and Shafer (1999)
14	Create culturally appropriate adaptation/ management options	3	Dixon et al. (1999), Huang (1997), Tompkins and Adger (2004)
	Create education programs for public about landuse practices and effects on and with climate	3	Bush (1999) and Welch (2005), Williams (2000)
	Develop best management practices for climate change scenarios	3	Mulholland et al. (1997), Rogers and McCarty (2000) and de Dios et al. (2007)
	Institute flexible zoning around reserves	3	Halpin (1997), Peters and Darling (1985) and Soto (2001)
	Increase investment in climate related research	3	Lemieux and Scott (2005), Lovejoy (2005) and Peters and Darling (1985)
	Increase communication of knowledge about climate change impacts to policymakers and stakeholders	3	Erasmus et al. (2002), Opdam and Wascher (2004) and Welch (2005)
	Initiate dialogue among stakeholders	3	McKenzie et al. (2004), Rogers and McCarty (2000) and Scott et al. (2002)
	Institute government reform (i.e. adaptive governance)	3	Chapin et al. (2006), Tompkins and Adger (2004) and Williams (2000)
	Locate reserves in areas of high heterogeneity, endemism	3	Halpin (1997), Opdam and Wascher (2004) and Peters and Darling (1985)
	Maintain natural disturbance dynamics of ecosystems	3	Halpin (1997), Noss (2001) and Shafer (1999)
	Practice proactive management of habitat to mitigate warming	3	Halpin (1997), Mulholland et al. (1997) and Wilby and Perry (2006)
	Secure boundaries of existing preserves	3	Hannah et al. (2007), van Rensburg et al. (2004) and Welch (2005)
	Start strategic zoning of landuse to minimize climate related impacts	3	Bush (1999), Solecki and Rosenzweig (2004) and Tompkins and Adger (2004)
	Study and monitor ecotones and gradients	3	Halpin (1997), Lovejoy (2005) and Stohlgren et al. (2000)
	Study effectiveness of corridors	3	Graham 1988, Halpin (1997) and Williams et al. (2005)
	Use predictive models to make decisions on where to situate new reserves	3	Bush (1999), Hannah et al. (2007) and Pearson and Dawson (2005)
15	Anticipate surprises and threshold effects i.e. major extinctions or invasions	2	Bartlein et al. (1997) and Millar et al. (2007)
	Design biological preserves for complex changes in time, not just directional change	2	Bartlein et al. (1997) and Graham (1988)
	Locate reserves at northern boundary of species' ranges	2	Peters and Darling (1985) and Shafer (1999)
	Manage the matrix	2	Eeley et al. (1999) and Lovejoy (2005)
	Practice proactive research on climate change	2	Harris et al. (2006) and Williams (2000)
	Protect many small reserves rather than single large	2	Opdam and Wascher (2004) and Pearson and Dawson (2005)
	Provide education opportunities and summaries of primary literature for management staff to learn and network about climate change	2	Grumbine (1991) and Welch (2005)
	Study and protect metapopulations	2	Crozier and Zabel (2006) and Opdam and Wascher (2004)
	Study processes of change at multiple spatial and temporal scales	2	Dale and Rauscher (1994) and Watson (2005)
	Use GIS to study species distributions and landscape patterns	2	Brown (2006) and Da Fonseca et al. (2005)

Table 1 – continued

Rank	Recommendation	No. articles	References
16	Action plans must be time-bound and measurable	1	Welch (2005)
	Adjust park boundaries to capture anticipated movement of critical habitats	1	Welch (2005)
	Create institutional flexibility	1	Millar et al. (2007)
	Create linear reserves oriented longitudinally	1	Pearson and Dawson (2005)
	Establish cross-national collaboration	1	Desanker and Justice (2001)
	Establish neo-native forests plant species where they were in the past, but are not found currently	1	Millar et al. (2007)
	Experiment with refugia	1	Millar et al. (2007)
	Focus protection on sensitive biomes	1	Scott et al. (2002)
	Focus on annual plants rather than perennials near climate boundaries	1	Buckland et al. (2001)
	Increase wetland protection	1	Hartig et al. (1997)
	Institutional capacity enhancement to address climate change	1	Lemieux and Scott (2005)
	Institute reform to improve support for interdisciplinary, multi-institutional research	1	Root and Schneider (1995)
	Locate reserves so major vegetation transitions are in core	1	Halpin (1997)
	Locate reserves at core of ranges	1	Araujo et al. (2004)
	Manage for landscape asynchrony	1	Millar et al. (2007)
	Manage human-wildlife conflict as change occurs	1	Wilby and Perry (2006)
	Manage populations to reduce temporal fluctuations in population sizes	1	Rice and Emery (2003)
	Develop guidelines for climate sensitive restoration and infrastructure development	1	Welch (2005)
	Need to increase social acceptance of shared resilience goals	1	Tompkins and Adger (2004)
	Promote personal action plans among employees to reduce emissions	1	Welch (2005)
	Protect endangered species ex situ	1	Noss (2001)
	Protect functional groups and keystone species	1	Noss (2001)
	Protect mountains	1	Peterson et al. (1997)
	Protect primary forests	1	Noss (2001)
	Protect urban green space	1	Wilby and Perry (2006)
	Quantify environmental susceptibility versus adaptive capacity to inform conservation planning	1	McClanahan et al. (2008)
	Schedule dam releases to protect stream temperatures	1	Rogers and McCarty (2000)
	Study changes in populations at rear of range rather than only range fronts	1	Willis and Birks (2006)
	Study response of undisturbed areas to climate change	1	Mulholland et al. (1997)
	Study social agency and human decision making	1	Desanker and Justice (2001)
	Study time-series data on species dynamics	1	Erasmus et al. (2002)
	Substitute space for time to study the responses of species to climate change	1	Millar and Brubaker (2006)
	Train more taxonomists	1	Huber and Langor (2004)
	Use caution in predictive modeling because the responses of some species are not well predicted	1	Willis and Birks (2006)
	Use simple decision rules for reserve planning	1	Meir et al. (2004)
	Use social networks for education about climate change	1	Huang (1997)
	Use triage in short-term to prioritize action	1	Millar et al. (2007)

sections, we discuss how recommendations in the literature to date inform these three scales of application.

4. Regional policy and planning

Species historically respond to changing climate with distributional shifts, and many species are expected to lose current habitat representation in the future. In light of this, many recommendations call for greater integration of species protection plans, natural resource management, research and development agendas across wider geographic areas, on longer time-scales, and involving more diverse actors than in current practice. (1) Long-term, regional perspective and (2) improved coordination among scientists, land managers, politicians and conservation organizations at regional scales are among the most frequently cited recommendations to protect biodiversity in the face of climate change (Rank 10 and 6 respectively, see references in Table 1 and for all ranks mentioned hereafter). Increased interdisciplinary collaboration (Rank 9) as well as regional-scale impact assessments are also frequently identified (Rank 12). Recommendations for adaptation to regional policy and planning focus on two comple-

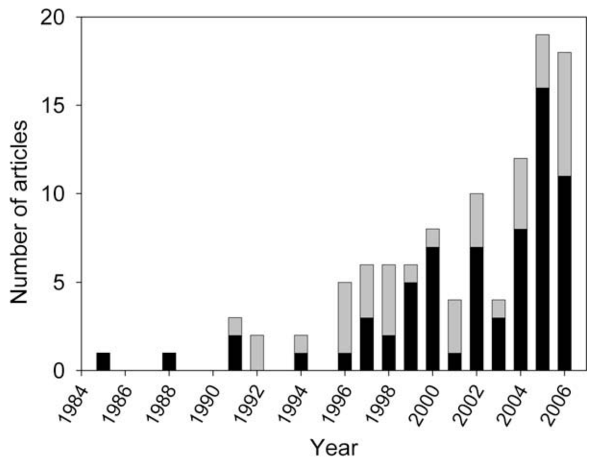

Fig. 2 – Frequency distribution by publication year of papers included in this review, including articles addressing biodiversity only (black) or biodiversity in conjunction with ecosystem services (grey). Records from 2007 were only partially covered in this review and not included.

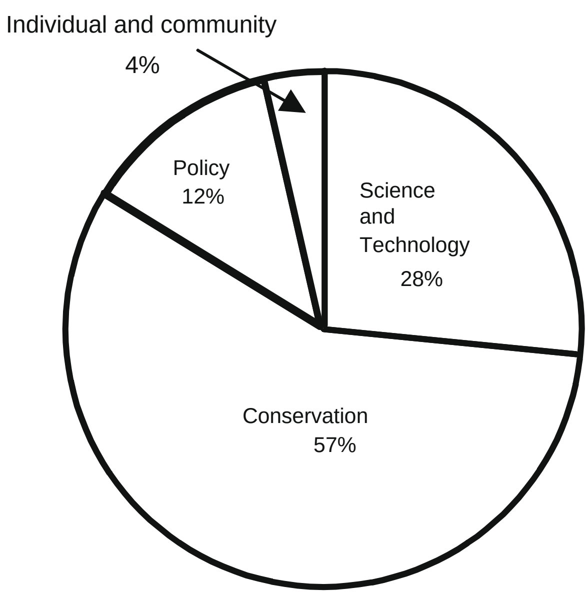

Fig. 3 – Distribution of recommendations calling for climate change adaptation among different activity sectors: conservation (e.g. reserve purchases, management, restoration and regional coordination), science and technology (e.g. research and modeling), policy (e.g. land-zoning, governance structure and institutional capacity), and individuals and communities (e.g. private landowner practices and grassroots action). Recommendations were counted in all applicable sectors.

mentary strategies: reserve planning and improving landscape connectivity. We discuss these issues further in the following two sections.

4.1. *Reserve planning*

Basing reserve acquisition priorities on predictions of future biome, community or individual species distributions under different climate scenarios is one method for climate change adaptation. The guiding principle is that reserves should be accumulated in areas predicted to be hotspots for biodiversity in the future or to provide habitat for species of high conservation value, warranting increased effort to model species distributions in the future (Rank 12). There are, however, several limitations to the accuracy and precision of simulation and analytical models of future species, biome or community distributions, leading some authors to recommend improved modeling capacity as the first step (Rank 9).

Model prediction error results from variation in model types, emissions, landuse and socio-economic scenarios. There are little-understood, but important, interactions between climate change and other global change drivers that could influence where species and habitats occur in the future (Rank 9). Insufficient data on species distributions (Rank 10) the effects of species interactions on distribution (Ferrier and Guisan, 2006; Kappelle et al., 1999), dispersal (Rank 10) and species, community or ecosystem responses to climate change (Rank 4) are also widely expressed concerns and lead authors to advocate for increased research in these areas before models are accepted. For example, bioclimatic envelope modeling uses current species distributions to predict future distributions as a function of climate. For many species such models can be productive, but in cases where species distributions are limited by factors other than climate, this extrapolation will prove misleading. Willis and Birks (2006) discuss the accuracy of bioclimatic models. Species-envelope model runs were conducted for backward predictions of species distributions and compared to paleo-ecological records. Many species distributions were predicted well, but some were largely inaccurate.

Problems of scaling also raise uncertainty (Rank 6), including scaling-down global climate models (GCMs) to fit management scales, or scaling-up empirical observations typically made at small spatial scales to predict larger scale processes (Root and Schneider, 1995). The scales of global climate models (GCM) and management activities simply do not match. Most reserves are smaller than a single grid cell in a GCM. Climate can vary sharply within this scale, and this variation often drives local patterns of species distribution and abundance – particularly in mountainous or coastal areas. Regional climate models, which are only available for small areas of the globe, are a more appropriate choice for management and planning (Dale and Rauscher, 1994; Guisan and Thuiller, 2005; Kueppers et al., 2005; Mulholland et al., 1997), though they remain limited by key uncertainties, assumptions and costs (Root and Schneider, 1995).

Not surprisingly, these inherent limitations of bioclimatic envelope models generate debate about whether and how to apply them to reserve selection. Some strongly advocate including climate change in reserve selection models and locating new reserves with expected changes in climate (Araujo et al., 2004; Bush, 1996; Dyer, 1994; Pearson and Dawson, 2005). Araujo et al. (2004) compare the ability of six existing reserve selection methods to secure European plant species in the context of climate change. They found species loss from protected reserves on the order of 6–11% of taxa for all models, and they conclude that new reserve-selection models specific to climate change are needed. Hannah et al. (2007)

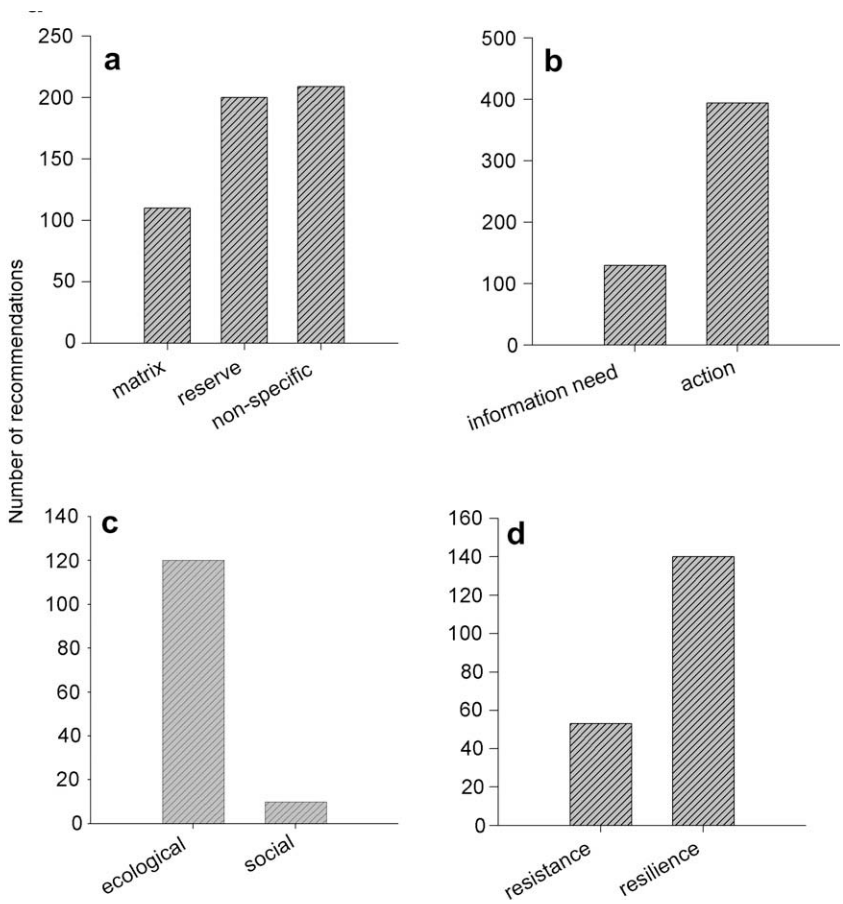

Fig. 4 – Distribution of recommendations among broad categories referring to (a) type of land targeted, (b) information need or action, (c) type of information need, and (d) management goal. Y-axis ranges vary across graphs because not all recommendations fit into every set of categories.

make a compelling case for not waiting to incorporate climate change forecasts into reserve selection models despite uncertainty. They use bioclimatic envelope models to explore the need for additional protected areas to achieve representation for thousands of species in three regions (Mexico, South Africa Cape, and Europe) in current and future climate and find that less land is needed in the long-term if planning models are designed to solve for both current and future conditions simultaneously.

Others argue, however, that given tremendous uncertainty, the priority should be to acquire new reserves in locations that minimize the spatial distances among new and existing reserves so that species can migrate (Allison et al., 1998; Collingham and Huntley, 2000; Halpin, 1997; Opdam and Wascher, 2004; Shafer, 1999). Williams et al. (2005) used a simulation model to estimate that 50% more protected land area in particular locations was needed to create reserve corridors to protect *Proteaceae* in the South African Cape region through 2050. Citing a number of sources of potential error in model results, however, they recommend that as much reserve area as possible be set aside. Such strategies do not require extensive modeling capacity and resources and instead focuse on rapid acquisition of land as it becomes available to create porous landscapes. Other authors reason that to facilitate migration and adaptation potential, reserves should be located with reference to focal species or community distributions, such as in their cores (Araujo et al., 2004; Halpin, 1997) or at their northern boundaries (Peters and Darling, 1985; Shafer, 1999). There seems to be little consensus or data to inform this debate. More research is needed about where in a species' range individuals are most likely to survive, migrate or adapt to rapid environmental change (Willis and Birks, 2006).

Debate also arises around the relative advantages of few large versus several small reserves in the context of climate change. The tension is whether large reserves will be large enough to allow species to track changing climate and remain inside reserve boundaries, and whether small preserves along latitudinal, elevational or other climate gradients will be close enough together for species to move between them. Eleven sources recommend protecting large areas (Beatley 1991; Bellwood and Hughes 2001; Burton et al. 1992; Bush 1996; Halpin 1997; Hulme 2005; Morecroft et al. 2002; Peters and Darling 1985; Shafer 1999; Soto 2001; Watson 2005), while two advocate focusing on many small areas (Opdam and Wascher, 2004; Pearson and Dawson, 2005). Eight suggest a compromise strategy of creating ecological networks of small and large reserves embedded within intermediate land uses (Allison et al., 1998; Collingham and Huntley, 2000; de Dios et al., 2007; Gaston et al., 2006; Opdam et al., 2006; Opdam and Wascher, 2004; Shafer, 1999; Welch, 2005).

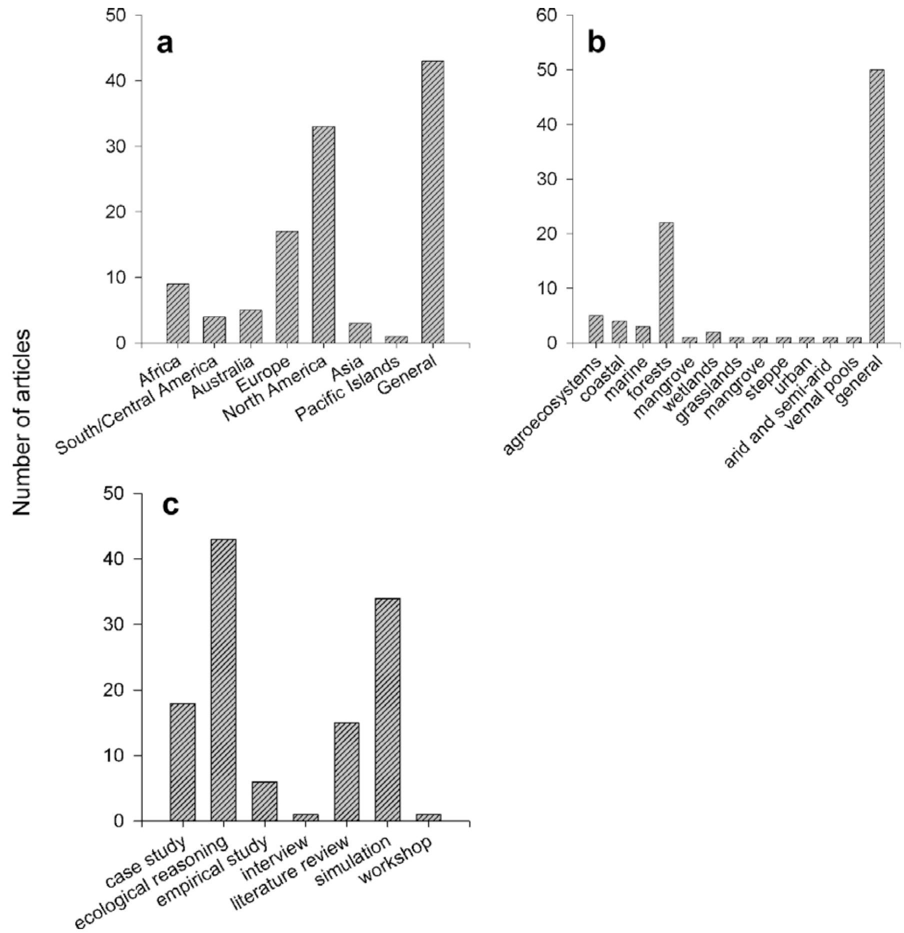

Fig. 5 – The (a) geographic focus, (b) biome focus, and (c) evidence basis for recommendations addressing climate change adaptation strategies for biodiversity management.

What all of the recommendations for reserve selection share is an urge to protect more land rapidly (Rank 5). This push will certainly help buffer biodiversity against climate change as well as other threats. However, climate change is likely to exacerbate existing tensions and tradeoffs between protecting areas and meeting basic human needs. Creating more new reserves might be feasible in some settings but must be guided by targeted, well-informed strategies likely to maximize effectiveness in the face of climate change. In most areas, action in lands outside of reserves must also be a part of climate change strategies for biodiversity conservation (Franklin et al., 1992; Lovejoy, 2005).

4.2. *Landscape connectivity*

To improve landscape connectivity, so that species can move, is the most frequent recommendation for climate change adaptation in the literature reviewed here (Rank 1). Authors recommend some form of corridor creation via the designation of new parks (de Dios et al., 2007; Halpin, 1997; Scott et al., 2002) oriented longitudinally (Eeley et al., 1999; Noss, 2001; Shafer, 1999), or through actions in non-reserve land, such as protecting riparian habitat and railway lines in cities (Wilby and Perry, 2006), or by planting trees and shrubs to create shelterbelts and hedgerows in farmlands (Donald and Evans, 2006; Guo, 2000; Schwartz et al., 2001). There was little guidance in this literature set for corridor implementation beyond common-sense reasoning, however. Illustrative examples of current corridor projects or elaboration of specific ecological or political tactics for corridor creation might help jump-start this process. For example, case studies of the Dutch Ecological Network and other similar national models to plan and link protected areas may be particularly informative at this stage of adaptation planning (Gaston et al., 2006). Further, despite widespread favor for ecological networks, assessment of their effectiveness remains in its infancy. Similarly, the field of corridor ecology, while recognized as integral to conservation practice in fragmented landscapes for years, is still young (see Hilty et al., 2006). Some authors warn of a significant need for more empirical data to support the effectiveness of corridors, optimize their spatial arrangement, and minimize risks of increased transmission of disease or invasive species before the conservation community embraces corridors uniformly as the tool to combat biodiversity loss in the face of global climate change (Graham, 1988; Halpin, 1997; Scott and Lemieux, 2005; Williams et al., 2005).

A second popular recommendation for improving landscape connectivity is to change how we manage the matrix

(Da Fonseca et al., 2005; Eeley et al., 1999; Lovejoy, 2005). Many authors advocate creating buffer zones around reserves (Rank 8) or flexible landuse zoning at reserve boundaries to allow for land swaps in the future as species distributions shift (Rank 14). Others recommend urban planning and zoning to avoid climate-related risks (Rank 14). In general, enlisting people and human communities to 'soften' landuse through sustainable or less damaging practices (e.g. low intensity forestry or alternatives to building sea walls) (Rank 9) and to restore habitat (Rank 9) will facilitate species movement and persistence in the future.

Despite wide acknowledgement, these connectivity strategies were among the most poorly developed recommendations, limited mainly to very general actions (e.g. "build flexibility", "manage the matrix", "modify landuse practices") without identification of kinds of actors that might need to be involved (e.g. reserve managers, policymakers, individuals) or information gaps. Landuse reform likely needs to bring together local governments, urban planners, community groups and conservation organizations and to involve high degrees of coordination across multiple jurisdictions to provide landscape cohesion (Press et al., 1995). Substantial work to flesh out this process, as well as to guide information acquisition, is needed before new forms of management across landuse types can be implemented.

Even with good landscape connectivity, some species will not be able to migrate. For these species – such as dispersal-limited species, those restricted to rare or confined habitat types, or those with life history traits like low reproductive rates – translocations from within their current range to locations suitable in the future are widely advocated (Rank 4). Translocations are a contentious issue because of the challenges associated with moving populations successfully and predicting suitable future habitats, as well as the potential for unintended consequences from introducing new species into existing communities (Lemieux and Scott, 2005; McLachlan et al., 2007). Empirical evidence suggests that animal translocations tend to be unsuccessful and costly (Fischer and Lindenmayer, 2000). Despite these real problems, we did not find discussion of the feasibility of such programs. Climate change adaptation strategies would likely necessitate moving at least some species outside of their current range, an action that has rarely been pursued thus far. To fully evaluate the feasibility of translocations would require stronger understanding of best available methods, potential risks, and policies for regional coordination to avoid situations in which different conservation objectives are put in conflict (McLachlan et al., 2007).

5. Site-scale action

Many land managers feel that there is little they can do about climate change beyond what they are already doing, such as trying to maintain basic ecosystem functioning and mitigate other threats like invasive species and pollution. To a certain extent, recommendations we reviewed validate this perspective. A number of "business as usual" recommendations rank high in their frequency in the literature, e.g. mitigating current threats, such as invasive species and habitat loss (Rank 2), increasing or continuing basic monitoring programs (Rank 7) or managing populations for natural disturbance dynamics (Halpin, 1997; Noss, 2001; Shafer, 1999). Franklin et al. (1992) describe how in forest ecosystems mature trees slow the effects of climate change because they tolerate a wide range of temperatures, while seedling establishment is far more sensitive. Under climate change, removal of long-lived trees will therefore act to intensify and speed-up the rate at which forest ecosystems change compared to intact forests. Restoration and greening efforts function as proactive management to mitigate local-scale warming (Halpin, 1997; Mulholland et al., 1997; Wilby and Perry, 2006). Mulholland et al. (1997) point out that restoration of riparian vegetation, needed to secure wildlife populations and ecosystem services now, will also function to decrease stream temperatures in the future. Wilby and Perry (2006) highlight how green building and landscaping techniques, such as planting green roofs, neighborhood trees, and water structures, will help to counter increasing problems of urban heat-island effects.

Other authors point out that business as usual is probably not enough in many cases. Peters and Darling (1985) suggest that managers consider rescue measures such as adding irrigation or drainage systems to secure sensitive populations. Buckland et al. (2001) anticipate that soil fertility in some grasslands may require manipulation to impede species invasions under warmer conditions. Advice to incorporate a broader range of species and genotypes in restoration and forestry than prescribed based on local provenance was common (Rank 11). This type of strategy would depart significantly from the preference for local genotypes prevailing in restoration and forestry practice to date (Millar and Brubaker, 2006; Millar et al., 2007; Scott and Lemieux, 2007) and warrants increased experimentation to better understand potential costs and benefits (Harris et al., 2006; Rice and Emery, 2003).

5.1. *Resilience versus resistance*

A first step for managers will be to wrestle with the question of whether and when they will attempt to resist biotic change, such as by adding irrigation if precipitation declines, rather than try to build resilience to change, such as by facilitating population adaptive capacity through introduction of a wider range of genotypes. In theory resistant strategies attempt to bolster a system's defenses to rapid environmental change, while resilience strategies attempt to bolster a system's ability to absorb rapid environmental change. More recommendations advocate resilience than resistance strategies (Fig. 4d). However, intensive management actions to protect historical species in their current distributions are widely advocated (Rank 4). The latter align best with a fixed-reserve approach focusing on local species precedence, an approach that will be increasingly costly and challenging to maintain as directional global changes accelerate.

For some species and systems, options other than intervention might not exist. Resistance approaches designed to maintain the status quo are nevertheless risky – they may leave systems vulnerable to total collapse if interventions are not maintained or compromise other system components (Harris et al., 2006; Walker et al., 2002). For example, the removal of invasive species has sometimes resulted in unpredicted and negative impacts to ecosystem structure and function (Zavaleta et al., 2001). Managing for resilience (sensu Holling, 1973) on the other hand explicitly focuses on increasing the flexibility and ability of systems to adapt and self-organize in response to change. To build resilience to climate change into systems, however, may require radical shifts in perspective for many conservation stakeholders and re-evaluation of conservation goals (Rank 10). Land managers might need to view a broader range of ecosystem states as desirable, such as novel or dynamic local assemblages that maintain functioning and trophic complexity but not necessarily species identity (Hulme, 2005), or to re-evaluate operational definitions and guidelines, such as what constitutes an invasive species or when a species can be added to a risk list (Scott and Lemieux, 2005; Scott et al., 2002).

Examples of broad perspective shift are found in the restoration literature. Millar and Brubaker (2006) emphasize the use of paleo-ecological perspectives to guide restoration goals and interventions. They ask that managers and restoration practitioners "make friends with physical and climatic change," arguing for instance that which species are deemed 'natural' or 'invasive' depends on the spatial and temporal resolution of data used to inform perspective. For example, Monterey pines (*Pinus radiata*) are considered native to a small region of California in which they were found at the time of European colonization. The species has since naturalized widely in California from landscaping plantings and is targeted for removal as an unwanted exotic in these regions. Paleo-ecological records of *P. radiata* reveal strong climate-driven dynamics in range, with widespread distribution during favorable periods and retreat during unfavorable periods. Millar and Brubaker (2006) suggest that naturalized populations be restored rather than removed in locations where *P. radiata* thrived when the climate was similar to the present or predicted future. Pearsall (2005) describes an experimental landscape-scale project in North Carolina, USA designed to test a range of restoration options for combating peat-land loss as a result of rising sea level. Options include oyster bed formation, dune formation, native plant establishment, as well as nonnative plant establishment. The experiment is scheduled to run for 25 years with regular evaluation intervals. Bradley and Wilcove (in press) imagine a "transformative restoration" in which the plant species used to repopulate restoration sites are determined by future climate conditions rather than historical presence. For example, based on results of bioclimatic envelope models, areas in the Great Basin ecoregion of the Western US may be restored best with plants introduced from the Mojave Desert, a more arid, neighboring biome. These projects share a broad, long-term and pragmatic perspective on acceptable restoration outcomes, one that may be necessary to tackle climate change.

A key strategy for building the adaptive capacity of systems is to enhance diversity at various scales. Diverse populations tend to be more adaptable, placing a premium on protecting and managing for high genetic diversity (Rank 13). Capturing the full range of bioclimatic variability within preserves and across landscapes and designing high species, structural, and landscape diversity into constructed and managed systems are also recommended (Rank 9). Pockets of outlier vegetation, areas of high endemism, ecotones, and refugia that protected species during climate shifts in the past are anticipated to be important sources for species re-colonization and radiation in the future, as well as provide retreats for migrating or translocated species (Rank 12). Willis and Birks (2006) discuss methods that combine genetic and paleo-ecological evidence to identify sites with distinctive patterns of genetic diversity that resulted from past geological events and refugial isolation.

Resistance and resilience strategies are not mutually exclusive. Very special communities or organisms that are of high conservation value may warrant highly invasive, intense and costly management regimes to maintain them. Regimes for intensive management are likely to be implemented through existing threatened species management frameworks, such as recovery plans. For more widespread populations, communities and ecosystems, which often provide important ecosystem services, a focus on resilience might be most appropriate. At the site-scale, managers need to address a host of practical issues such as the cost and cost-effectiveness of adaptation options, their compatibility with existing regulatory and institutional constraints, and their likely effectiveness in the absence of coordination with adjoining private lands.

6. Adapting existing conservation plans

The existing literature does provide an array of actions for managers to build on and consider incorporating into existing conservation plans. A practical first step to climate change adaptation planning is to evaluate the likely outcomes for biodiversity of continuing current management and conservation directions. Most conservation policies and management plans do not yet explicitly consider climate change (Chambers et al., 2005; Groves et al., 2002; Hannah et al., 2002; Scott and Lemieux, 2007). A consistent theme in the literature is at the very least to immediately appraise current conservation and management practice in the context of climate change (Rank 2) with the goal of developing and adopting specific climate change adaptation policies in the near future (Rank 11). The literature here contained some suggestions for how to do this. A few articles emphasized the use of models to guide evaluation and adaptation of existing practices. For example, Christensen et al. (2004) used a simulation model to investigate a coupled system of plants and grazers in the Inner Mongolia Steppe under different climate scenarios. They determined that grasslands were likely to undergo a state-transition to shrublands if existing grazer densities are maintained, and they advocate reducing grazers in this area as well as in other semi-arid managed grassland systems. Hulme (2005) provided a general overview of how mathematical models can integrate long-term demographic and climate data to set climate change-appropriate harvest or stocking schedules or to forecast pest outbreaks.

Some authors highlight existing efforts that are well-suited to tackle climate change and warrant increased funding and research. Donald and Evans (2006) argue that agri-environment incentives and easement programs in the US and the EU, which are growing due to shifts in farm policies, warrant increased funding priority because of their potential to improve habitat availability and landscape connectivity across managed ecosystems. They discuss how these policies could be modified to tackle climate change directly. Site-specific climate conditions and biotic responses could be mapped on to landscapes and used to prioritize locations for farm diversification. Similar gains could be made by targeting other private landowner biodiversity enrichment programs, like the USDA Forest Legacy Program (http://www.fs.fed.us/spf/coop/programs/loa/flp.shtml) or the National Wildlife Federation's Urban Backyard Wildlife Program (http://www.nwf.org/gardenforwildlife/).

6.1. *Holistic strategies*

Issues that currently challenge conservation practice may need to be addressed before the added stress of climate change complicates them further. Communities of local users are often in conflict with conservation objectives (Chan et al., 2007; Suffling and Scott, 2002). Identifying opportunities for reduced conflict and increased synergy between conservation and local communities will become more important as climate changes. A number of authors warn that conservation policies must create positive economic outcomes for local peoples to buffer them against potentially dramatic shifts in livelihoods that will accompany climate shifts (Rank 9). Adaptation requires community buy-in and participation (Chapin et al., 2006). To this end, conservation policies that foster learning and participation (Ramakrishnan, 1998) and provide options that are culturally and economically appropriate, such as those that honor traditional management systems and do not rely on expensive technologies, are more likely to be embraced and implemented (Rank 14). McClanahan et al. (2008) argue that climate-informed conservation planning necessitates site-specific understanding of environmental susceptibility and societal capacity to cope and adapt. They illustrate this process for five western Indian Ocean countries with respect to coral reef conservation. Locations with high environmental susceptibility and low adaptive capacity will be most difficult to secure effectively in the future, while those with low environmental susceptibility and high adaptive capacity will be easiest. Locations with low environmental susceptibility and low adaptive capacity are good candidates for biodiversity investment, but to be effective these locations also require investments in human infrastructure, livelihood diversification and social capital.

Climate change is acting in concert with multiple other drivers of biodiversity loss including habitat degradation, soil loss, nitrogen enrichment, and acidification. Strong policies must simultaneously address more than one issue (Watson, 2005) or risk exacerbating environmental problems in the process of trying to combat them. Emission reduction programs are a significant push for many governments, organizations and individuals. They warrant an important place in any climate change combat strategy (Rank 13). A number of authors in this review urge, however, that emissions reduction programs and the Clean Development Mechanisms (CDMs) in the Kyoto Protocol be implemented in ways that simultaneously address carbon sequestration, biodiversity conservation and human livelihoods, rather than carbon sequestration in isolation (Rank 11).

Finally, climate change provides a much-needed impetus to evaluate how conservation policies respond to change in general. Climate change is only one of several global environmental trends to which biodiversity and its conservation must respond. Uncertainty in the climate change arena and about the future in general should not limit action to strengthen existing conservation strategies, with a focus on enhancing the ability of ecosystems to absorb and recover from rapid and unpredictable change.

7. A complete strategy

Climate change challenges conservation practice with the need to respond to both rapid directional change and tremendous uncertainty. Climate change adaptation therefore requires implementation of a range of measures, from short- to long-term and from precautionary and robust to more risky or deterministic, but specifically anticipatory (Fig. 6). To cer-

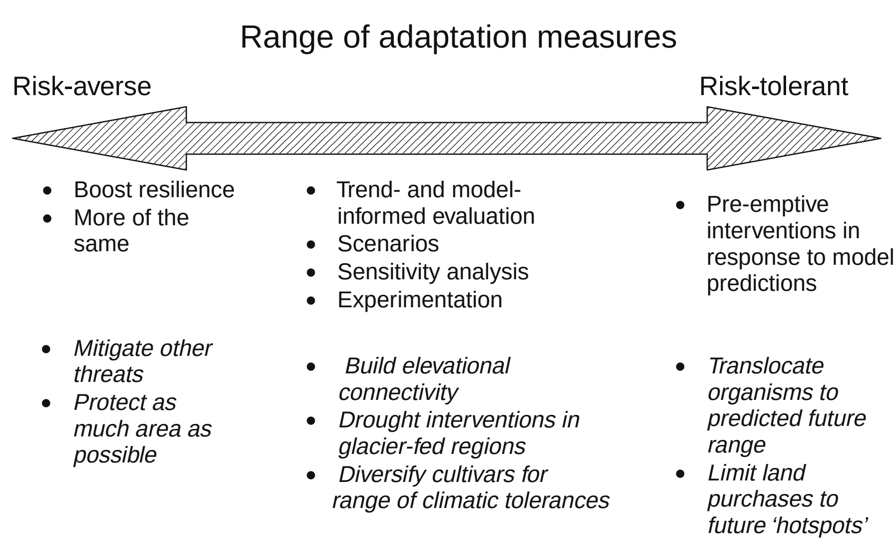

Fig. 6 – Adaptation measures classified along a risk continuum. Under each risk category are examples of general approaches followed by examples of specific adaptation measures. A complete strategy should span a risk continuum.

tain degree, risk tolerance of individual actors will guide strategy selection. Millar et al. (2007) discuss how managers must proactively decide whether to adopt deterministic or indeterministic approaches.

Each type of approach has benefits and drawbacks. Precautionary measures such as restoration, increased monitoring of species distribution, and increased investment in reserve protection do not necessarily require highly certain and precise climate change predictions, but such precautionary steps will help managers respond to current biodiversity threats as well as threats that emerge in the future. Precautionary measures alone, however, will not expand our ability to absorb and respond to rapid directional changes in climate, nor do they capitalize on available predictive information and efforts. In worst-case climate scenarios, over-reliance on bet-hedging measures may spread resources too thin or prove insufficient to help biodiversity weather the rapid changes underway. On the other hand, forecast-interventions bear significant risks if they are too deterministic, not robust to alternative futures or have negative unanticipated consequences (Suffling and Scott, 2002). They could also deliver great rewards and should be weighed with sensitivity analyses and scenarios, tested in pilot programs, and implemented initially at small scales (McLachlan et al., 2007). Scenario building – done in ways that are amenable to local data limitations and useable by policymakers and managers – is particularly apt for exploring the range of magnitudes and direction of possible futures and trends without commitment to specific forecasts (Brown, 2006; Millar et al., 2007).

While the range of recommendations in the literature is great, four consistent, broad themes emerge in this review for conservation stakeholders to apply to climate change planning and adaptation: (1) the need for regional institutional coordination for reserve planning and management and to improve landscape connectivity; (2) the need to broaden spatial and temporal perspective in management activities and practice, and to employ actions that build system resilience; (3) the need to incorporate climate change into all conservation planning and actions, which will require increased research and capacity to forecast future conditions and species responses and to deal effectively with unavoidable uncertainty; and (4) the need to address multiple threats and global change drivers simultaneously and in ways that are responsive to and inclusive of diverse human communities and cultures. Action along each of these fronts will involve difficult tradeoffs, barriers to implementation, and collaboration across diverse actors.

Action will also require an adaptation planning process or series of processes appropriate for various scales and applications. Most of the literature to date fails to distinguish adap-

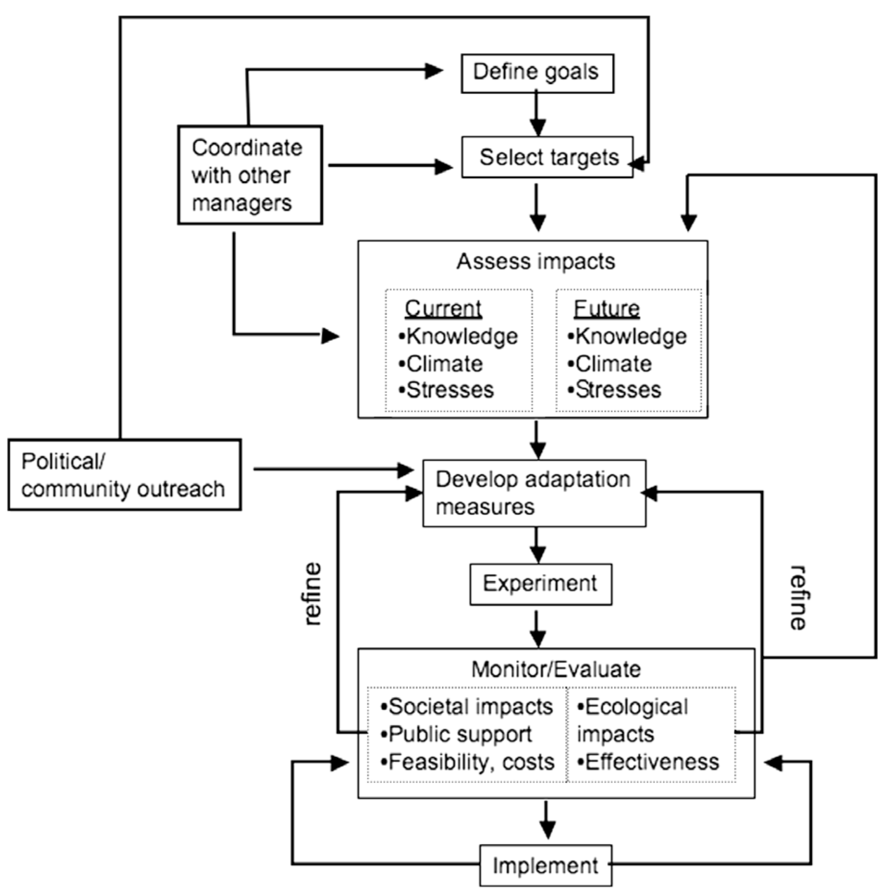

Fig. 7 – Adaptation planning involves at least a few key steps, each complex and requiring collaboration among actors such as land managers, the public, scientists, funders and lawmakers. Recommendations reviewed here address aspects of these steps, but without specifying where they fit in relation to one another.

tation from climate change impact assessment, or adaptation planning from implementation. These are distinct steps in an as-yet largely undefined process that the recommendations we survey could inform. We propose a series of general steps that should be modified, elaborated, and tailored to specific needs (Fig. 7). Key to any adaptation planning process will be to follow the principles of adaptive management (Rank 7), in which later steps inform earlier steps in an iterative and on going process.

8. Conclusions

Widespread calls exist for immediate action to adapt conservation practice to ongoing climate change in order to ensure the persistence of many species and related ecosystem services. However, the majority of recommendations in the published journal literature lack sufficient specificity to direct this action. Over the last 22 years, general recommendations have been reiterated frequently without the elaboration necessary to operationalize them. Greater effort to increase the availability and applicability of climate change adaptation options for conservation—through concrete strategies and case studies illustrating how and where to link research agendas, conservation programs and institutions—is badly needed.

Recommendations to date also largely neglect social science and are overwhelmingly focused on ecological data (Fig. 4c). This bias is alarming given the obvious importance of human behavior and preferences in determining conservation outcomes (Watson, 2005) and the increasingly important role of multi-use public and private lands in conservation practice. A holistic landscape approach to conservation, driven by a vision of humans and other species co-mingling across reserves and developed lands, has gradually gained prominence over the last 20 years. In their seminal paper, Peters and Darling (1985) provided a number of recommendations that continue to be widely advocated (Table 1), but they did not address the roles of conservation and restoration in human-dominated landscapes. These ideas emerge strongly in more recent literature highlighting a need to integrate ecology with other disciplines and approaches that explicitly address the roles of institutions, policy, politics and people in successful conservation strategies.

Finally, few resources or capacity exist to guide an adaptation planning process at any scale (Hannah et al., 2002; Scott and Lemieux, 2007; Welch, 2005). Such a process would place the sea of adaptation ideas and recommendations in framework and provide practitioners with tools, roles and a structure to evaluate what ideas might be useful and feasible for particular situations. Large-scale adaptation efforts that incorporate many of the recommendations found in this review are currently underway, including governmental efforts such as by Parks Canada or DEFRA in England, and by international non-governmental organizations such as The Nature Conservancy and the Wildlife Conservation Society. Well-documented case studies that focus not only on the outcome but also on the development process of adaptation plans are a promising avenue. These efforts can best enhance and encourage more widespread climate change adaptation, particularly at smaller scales, by capturing what they learn and disseminating it widely.

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CONSERVATION ECOLOGY

Report

USDA Forest Service Roadless Areas: Potential Biodiversity Conservation Reserves

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ABSTRACT. In January 2001, approximately 23 x 10^6 ha of land in the U.S. National Forest System were slated to remain roadless and protected from timber extraction under the *Final Roadless Conservation Rule.* We examined the potential contributions of these areas to the conservation of biodiversity. Using GIS, we analyzed the concordance of inventoried roadless areas (IRAs) with ecoregion-scale biological importance and endangered and imperiled species distributions on a scale of 1:24,000. We found that more than 25% of IRAs are located in globally or regionally outstanding ecoregions and that 77% of inventoried roadless areas have the potential to conserve threatened, endangered, or imperiled species. IRAs would increase the conservation reserve network containing these species by 156%. We further illustrate the conservation potential of IRAs by highlighting their contribution to the conservation of the grizzly bear (*Ursos arctos*), a wide-ranging carnivore. The area created by the addition of IRAs to the existing system of conservation reserves shows a strong concordance with grizzly bear recovery zones and habitat range. Based on these findings, we conclude that IRAs belonging to the U.S. Forest Service are one of the most important biotic areas in the nation, and that their status as roadless areas could have lasting and far-reaching effects for biodiversity conservation.

INTRODUCTION

In January 2001, the Clinton administration promulgated its *Roadless Area Conservation Rule,* which states that 237,000 km^2 of inventoried roadless areas (IRAs) within the U.S. National Forest System will remain roadless and protected from timber extraction (USDA Forest Service 2000). These lands represent 31% of the National Forest System or 2.5% of the total U.S. land base (DeVelice and Martin 2001). They would increase the amount of strictly protected land area in the United States in IUCN categories I–III from 4.8 to 8.5%. Beyond these most basic statistics, few studies have analyzed the potential contribution of IRAs to biodiversity conservation (Martin et al. 2000, DeVelice and Martin 2001).

DeVelice and Martin (2001) assessed the extent to which IRAs could contribute to building a representative network of conservation reserves in the United States. Using ecoregions as their unit of analysis (Ricketts et al. 1999), they found that IRAs could potentially expand ecoregional representation, increase the area of reserves at lower elevations, and increase the size of conservation areas to provide refuge for wide-ranging species. However, in their assessment they did not evaluate the contribution of IRAs toward the conservation of biodiversity and populations of specifically threatened, endangered, or imperiled species.

The lands belonging to the USDA Forest Service contain more than 80% of mammal and reptile species and more than 90% of the bird, amphibian, and fish species in the United States, including many that have been extirpated from large portions of their presettlement ranges (USDA Forest Service 1997). According to the NatureServe database, more than 1400 of these species have been designated as threatened and endangered (TE) species under the *Endangered Species Act* (ESA). The *Forest Service Roadless Area Final Environmental Impact Statement* identified approximately 400 TE or proposed species found on USDA Forest Service land and an estimated 220 (55%) that are directly or indirectly associated with IRAs (USDA Forest Service 2000). IRAs provide or influence designated critical habitat for at least 30 of these species (USDA Forest Service 2000).

However, the ESA list is not a complete listing of imperiled species. There are numerous species that are globally rare or threatened with extinction but for

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various reasons do not appear on the ESA TE species list. Many of these species also occur on USDA Forest Service land. To fill this gap, we supplemented the TE species list with species categorized as critically imperiled or imperiled according to NatureServe's central database.

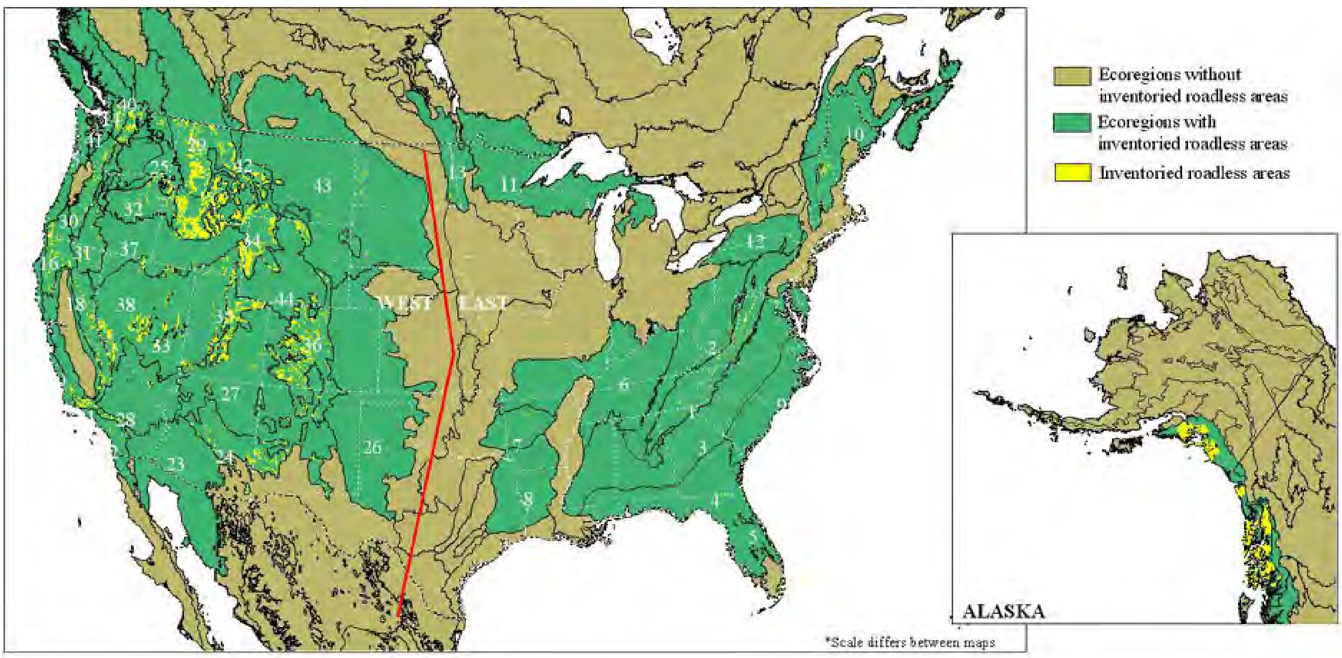
The objective of this paper is to assess three critical questions associated with IRAs:

> Is there a high concordance between IRAs and ecoregions of particular biodiversity values?
>
> Do IRAs overlap with threatened, endangered, or imperiled species?
>
> Is there potential for IRAs to assist in the conservation of wide-ranging species, such as the threatened grizzly bear (*Ursos arctos horribilis*), in the conterminous United States?

METHODS

We obtained the spatial coverages of the inventoried road areas (IRAs) in vector format from the roadless area conservation Web site (Table 1).

Fig. 1. Overlap of USDA Forest Service inventoried roadless areas (IRAs) with ecoregions that contain USDA Forest Service lands. The bold line indicates the separation of IRAs into three geographic regions: east, west, and Alaska.



1. Appalachian-Blue Ridge Forests
2. Appalachian Mixed Mesophytic Forests
3. Southeastern Mixed Forests
4. Southeastern Conifer Forests
5. Florida Sand Pine Scrub
6. Central US Hardwood Forests
7. Ozark Mountain Forests
8. Piney Woods Forests
9. Middle Atlantic Coastal Forests
10. New England-Acadian Forests
11. Western Great Lakes Forests
12. Allegheny Highlands Forests
13. Northern Tall Grasslands
14. British Columbia Mainland Coastal Forests
15. Central Pacific Coastal Forests
16. Klamath-Siskiyou Forests
17. Northern California Coastal Forests
18. Sierra Nevada Forests
19. Madrean Sky Islands Montane Forests
20. California Interior Chaparral & Woodlands
21. California Montane Chaparral & Woodlands
22. California Coastal Sage & Chaparral
23. Sonoran Desert
24. Arizona Mountains Forests
25. Palouse Grasslands
26. Western Short Grasslands
27. Colorado Plateau Shrublands
28. Mojave Desert
29. North Central Rockies Forests
30. Central and Southern Cascades Forests
31. Eastern Cascades Forests
32. Blue Mountains Forests
33. Great Basin Montane Forests
34. South Central Rockies Forests
35. Wasatch and Uinta Montane Forests
36. Colorado Rockies Forests
37. Snake-Columbia Shrub Steppe
38. Great Basin Shrub Steppe
39. Okanogan Forests
40. Cascade Mountains Leeward Forests
41. Puget Lowland Forests
42. Montana Valley and Foothill Grasslands
43. Northwestern Mixed Grasslands
44. Wyoming Basin Shrub Steppe
45. Northern Pacific Coastal Forests
46. Pacific Coastal Mountain Tundra & Ice Fields

Table 1. Data sources. All data web data sources were accessed in February 2001.

Database name	Source
USDA Forest Service roadless area database	http://roadless.fs.fed.us/documents/feis/data/gis/coverages/index.shtml
World Wildlife Fund ecoregions database	Ricketts et al. 1999
NatureServe central databases	NatureServe
Protected areas database	Conservation Biology Institute and World Wildlife Fund
Grizzly bear recovery area boundaries	U.S. Fish and Wildlife Service and University of Montana

Ecoregions

As seen in Fig. 1 and Table 1, we evaluated the potential benefit of IRAs for biodiversity conservation using the ecoregions and biological importance rankings provided in Ricketts et al. (1999). Using ArcView 3.2, we combined the IRAs and ecoregion coverages, both in vector format. To facilitate interpretation, we separated our analysis into three geographic regions, i.e., the eastern United States, the western United States, and Alaska, following the methodology used by DeVelice and Martin (2001).

Ricketts et al. (1999:7) defined an ecoregion as " ... a relatively large area of land or water that contains a geographically distinct assemblage of natural communities." Ecoregions were selected as the units of analysis because they integrate ecological, biological, and geographic considerations into land-use decision making and are being used to establish priorities for large-scale conservation efforts (Omernik 1995*a*,*b*, Ricketts et al. 1999, Groves et al. 2002). Where ecoregions extend into either Canada or Mexico, we included only those portions within U.S. boundaries for all analyses. Although we would have preferred to maintain ecoregional contiguity, the spatial nature of USDA Forest Service lands and the applicability of the *Endangered Species Act* required strict adherence to political boundaries.

Ricketts et al. (1999) classified the biological importance of each ecoregion based on species distribution, i.e., richness and endemism, rare ecological or evolutionary phenomena such as large-scale migrations or extraordinary adaptive radiations, and global rarity of habitat type, e.g., Mediterranean-climate scrub habitats. They used species distribution data for seven taxonomic groups: birds, mammals, butterflies, amphibians, reptiles, land snails, and vascular plants (Ricketts et al. 1999). Each category was divided into four rankings: globally outstanding, high, medium, and low. The rankings for each of the four categories were combined to assign an overall biological ranking to each ecoregion. Ecoregions whose biodiversity features were equaled or surpassed in only a few areas around the world were termed "globally outstanding." To earn this ranking, an ecoregion had to be designated "globally outstanding" for at least one category. The second-highest category, or continentally important ecoregions, were termed "regionally outstanding," followed by "bioregionally outstanding" and "nationally important" (Ricketts et al. 1999). Although our analyses focused on those ecoregions characterized as globally and regionally outstanding, even the lowest category, nationally important, contains important biodiversity in a local context.

Threatened, endangered, and imperiled species

Currently, public land managers are required to monitor populations of threatened and endangered (TE) species and, where appropriate, develop management plans to conserve these populations and their habitat requirements (U.S. Fish and Wildlife Service 1973). Previous studies have analyzed the distribution of TE species based on counties, or boroughs in Alaska, and identified high-concentration areas of TE species and associated habitats (Dobson et al. 1997, Flather et al. 1998, Stein et al. 2000). Despite their valuable findings, these previous studies were limited by the coarse level of spatial resolution and the use of political units of disparate sizes. To avoid similar limitations with our analysis, we use data of a finer resolution to identify levels of concordance between the locations of IRAs and TE species.

The NatureServe central database (Table 1) provided the finer-resolution data for the identification of the locations of TE species. Data for this database are developed by state natural heritage programs and managed by NatureServe. Natural heritage programs have documented and tracked the occurrence of threatened, endangered, and imperiled species for nearly 30 yr (Jenkins 1985, 1988, 1996). The system assigns global conservation status ranks known as "element global ranks" or "G-RANKS" to species and communities that are intended to estimate the extent of their imperilment or vulnerability. Conservation status ranks are assigned based on an assessment of rarity, the extent of recent decline of populations, threats, biological fragility, and other factors (Stein et al 2000). The most imperiled species and communities are ranked G1, and the most stable ones are ranked G5.

The NatureServe central database includes fields for federal ESA listing status and for global conservation status. We selected records of species that are federally listed as threatened or endangered (TE) according to the U.S. Fish and Wildlife Service or the National Marine and Fisheries Service and those that are ranked by NatureServe as critically imperiled (G1) or imperiled (G2). The output file was a vector file of 109,125 occurrences of species with G1 or G2 rankings or federal ESA listings. These occurrences were collated into 7.5-min quadrangles from the U.S. Geological Survey. The largest quadrangles, in the southern part of the United States, are 179 km^2. We used two data products for our analyses. The first contains only TE species (Fig. 2), and the second contains TE, G1, and G2 species (Fig. 3). The spatial resolution of the locational data varied according to the equipment and methodologies that natural heritage programs used in collecting the data. However, the maximum uncertainty for the data set was less than the area of a quadrangle grid cell.

Fig. 2. Threatened and endangered (TE) species distributions by the 7.5-min quadrangles of the U.S. Geological Survey.

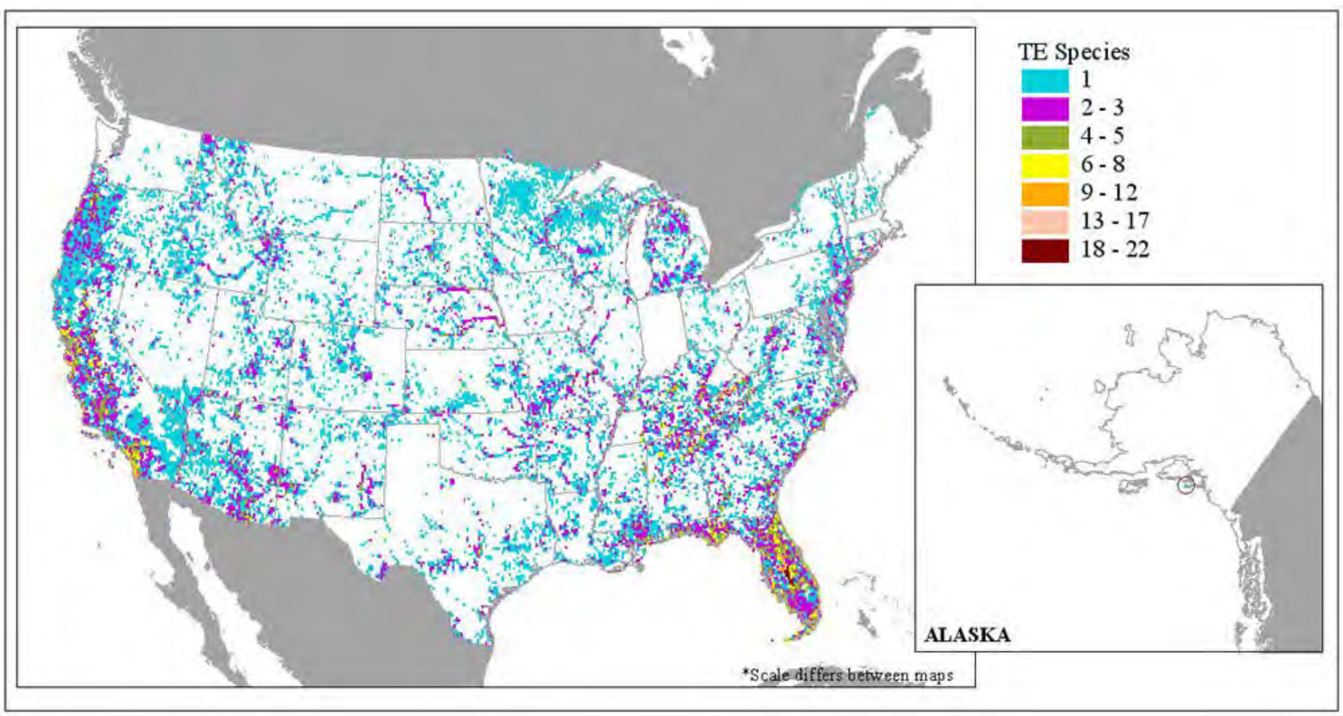
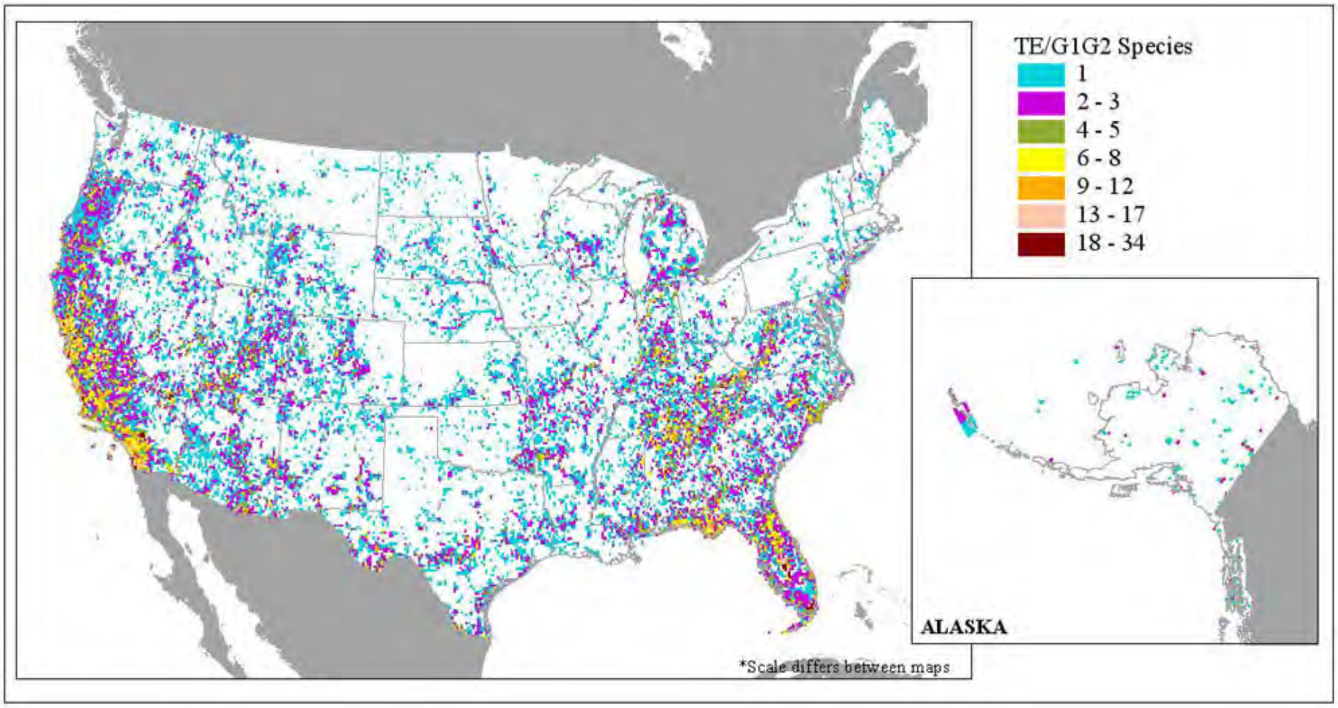


Fig. 3. Threatened and endangered (TE) species and critically imperiled (G1) and imperiled (G2) species distributions by the 7.5-min quadrangles of the U.S. Geological Survey.



The TE, G1, and G2 data sets demonstrate only a moderate degree of overlap. These discrepancies occur partly because the NatureServe system evaluates only biological factors, whereas species are assigned to federal listings for both scientific and political reasons. There are 75,000 occurrences of TE species, but only 27,000 are ranked G1 or G2 by the NatureServe system. Of the 1409 ESA-listed TE species in the NatureServe database, 1109 are ranked G1 or G2. Conversely, there are 5997 species ranked G1 or G2 that are not classified as TE species. Of the 61,000 occurrences of G1 and G2 species recorded in the NatureServe database, more than 33,000 occurrences lack a TE species designation. One of the reasons for the disparity between the high concordance of species but the low concordance of occurrences is the fact that certain species are wide-ranging. For example, the grizzly bear, which is a threatened species but not a G1 or G2 species, is recorded often across its wide range, so that it accounts for far more records than a narrow endemic species that is both TE and listed as G1 or G2.

The NatureServe database contains information gaps (Table 2). However, although the missing data for Idaho, Montana, and Washington are critical for the conservation of individual species, the lack of them served only to make our analysis a more conservative estimate of the potential contributions of IRAs to species conservation. There are no IRAs in Massachusetts and only one in Maine, with a total area of 24 km^2.

We overlaid both the TE species and TE/G1–G2 species databases with the uniquely named IRAs to identify the percentage of IRAs that contain known occurrences of TE or G1–G2 populations. In instances where multiple quadrangles containing species occurred within a single IRA unit, we erred on the conservative side and used only the quadrangle that contained the most species, i.e., we assumed that multiple quadrangles would contain the same species.

We also analyzed the relative increase in conservation reserves that IRAs would confer to TE and TE/G1–G2 species. We overlaid the TE and TE/G1–G2 databases with a conservation area database compiled by the Conservation Biology Institute and World Wildlife Fund (Table 1). This database includes all federal, state, county, and municipal public lands and some private lands. The private lands have not been systematically surveyed and do not include conservation easements. We used only lands that are classified for strict biodiversity conservation, which we define as those designated as categories I–III by the IUCN. Category I is for Strict Nature Reserves/Wilderness Areas, category II covers National Parks, and category III includes National Monuments (The World Conservation Union 1978, The World Conservation Union 1994). Hereafter we refer to the areas that meet these criteria as "conservation reserves." We did not include protected-area categories IV–VI, which allow road building, timber harvesting, and other extractive activities in our analysis. Of 78 x 10^6 ha of National Forest land, 14 x 10^6 ha are designated as National Wilderness Areas, and an additional 2.5 x 10^6 ha are classified as Special Designated Areas that are IUCN category I reserves. The remaining 61.5 x 10^6 ha of National Forest land, which are not classified as conservation reserves, are governed by periodic management plans that may allow or restrict resource uses and extraction.

Table 2. Gaps in data available for this study.

State	Missing data
Idaho	Fish data
Maine	Animal data
Massachusetts	All data
Montana	Canada lynx, bull trout, gray wolf data
Washington	Most animal data

Grizzly bear case study

Finally, because national analyses can obscure important details of individual species, we also analyzed the potential contribution of IRAs to grizzly bears (*Ursos arctos horribilis*), specifically in relation to the regions designated as grizzly bear recovery areas by the U.S. Fish and Wildlife Service (Table 1). We overlaid these grizzly bear recovery zones with the IRAs to assess the concordance of these areas. We chose grizzly bears because they are a federally listed threatened species in the conterminous United States and require large and contiguous habitat areas to survive.

All spatial databases were in vector format and put into a common projection prior to the overlap analysis. All spatial estimates derived from our analyses were obtained by summarizing the area of overlap of the respective GIS databases. One caveat of our methodology is that the combination of multiple GIS layers may lead to the propagation of spatial errors and increased uncertainty (Flather et al. 1998, Heuvelink 1998). This concern is a generalized methodological one. Our errors are no greater or smaller than those of any similar analysis that uses multiple spatial data from multiple sources. The TE species databases, protected areas database, and IRA coverages represent a vast collection of data from many sources. It is likely that errors are associated with each of these layers. However, most of our analyses were conducted at a sufficiently broad scale that we believe the error rate is not large enough to affect our ultimate conclusions.

RESULTS

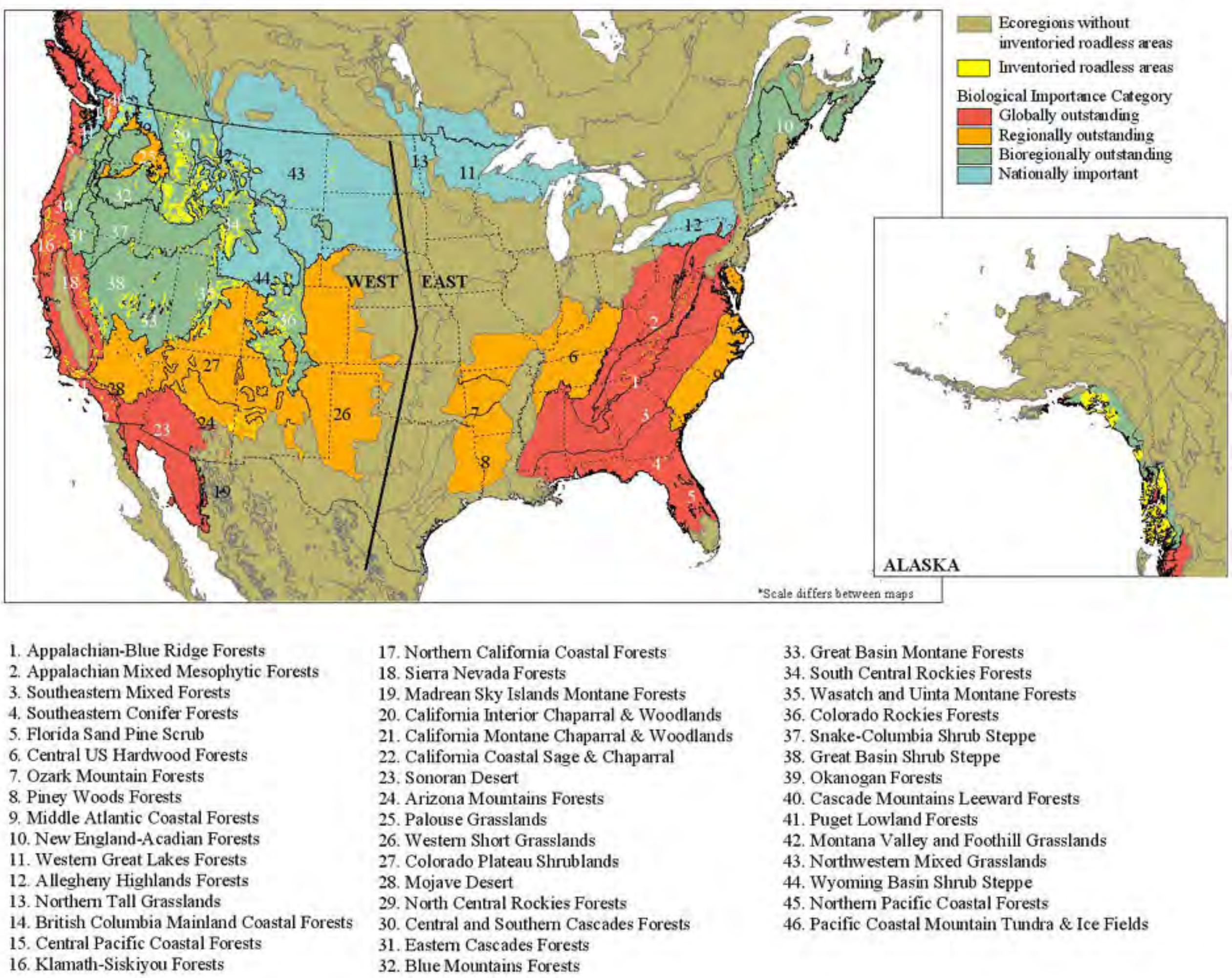
Ecoregions

Across the United States, we found that more than 20% of inventoried roadless areas (IRAs) were located within ecoregions that have been classified as globally outstanding (Table 3, Fig. 4). In the eastern region, approximately 70% of the IRAs are found in globally or regionally outstanding ecoregions (Table 3, Fig. 4). More than 50% of these forests occur in two Appalachian ecoregions, the Appalachian-Blue Ridge forests and the Appalachian mixed mesophytic forests. Both are considered globally outstanding for their diverse endemic species, which range across many taxa (Stephenson et al. 1993, Ricketts et al. 1999). The vast majority of the IRAs in eastern forests are less than 10.1 km^2 in size, and few are adjacent to existing wilderness areas (DeVelice and Martin 2001).

Table 3. Distribution of inventoried roadless areas (IRAs) by category of ecoregion biodiversity as per Ricketts et al. (1999). The percentage is the percentage of IRAs that fall into that particular category.

Biodiversity category	km^2	Percentage
Globally outstanding	50,221	21.2
Regionally outstanding	12,648	5.4
Bioregionally outstanding	164,600	69.5
Nationally important	9268	3.9

Fig. 4. Overlap of USDA Forest Service inventoried roadless areas and ecoregions classified by biological importance (see Ricketts et al. 1999).



In the western region, IRAs are found predominantly in bioregionally outstanding ecoregions, with only 18% in globally or regionally outstanding ecoregions (Table 3, Fig. 4). Although globally and regionally outstanding IRAs are found mainly in the states of California, Oregon, Washington, and Arizona, the intermountain west contains most of the nation's bioregionally and nationally important IRAs. Western IRAs are on average larger than eastern IRAs, and the vast majority are adjacent to existing wilderness areas. If the IRAs were combined with the wilderness areas, the western forests would contain 34 of the 45 largest contiguous areas of strictly protected forests in the United States (DeVelice and Martin 2001).

Table 4. Comparison of the degree of overlap between inventoried roadless areas (IRAs) and quadranges containing threatened or endangered (TE) species or quadranges containing TE species that are also ranked as highly imperiled (G1–G2) by the IUCN. The mean number of TE or TE/G1–G2 species present in each IRA is given.

Region	Total no. of IRA units†	No. of IRA units with TE species quadrangles (% of total)	Mean no. of species‡	No. of IRA units with TE/G1–G2 species quadrangles (% of total)	Mean no. of species‡
Eastern United States	286	201 (70.3)	2.1	228 (79.7)	4
Western United States	2159	1317 (61.0)	1.6	1692 (78.3)	2.9
Alaska	150	2 (1.3)	1	88 (58.6)	1.3

†Units are defined by each named inventoried roadless area.
‡Where multiple quadrangles occurred in a single IRA unit, we used only the quadrangle with the greatest number of species.

Threatened, endangered, and imperiled species

Of the 2595 IRA units, approximately 58% of them overlap with TE species quadrangles (Table 4). When separated into geographic regions, the IRAs in the eastern and western United States demonstrate overlaps of 70.3 and 61.0%, respectively. Of the IRAs that contain TE species, the mean number of TE species found in IRAs is highest in the east (2.1 species) and lowest in Alaska (1.0 species).

When G1–G2 species are included in the analysis, both the number of IRAs that contain TE/G1–G2 species and the mean number of species of concern found in each IRA increase (Table 4). In sum, approximately 77% of the IRAs overlap with quadrangles that contain species at risk. The Alaska region contains the largest increase in IRAs when G1–G2 species are included, increasing to 58.6 from 1.3%. The west increases to 78.3%, and the east increases to 79.7%. However, the east shows the largest increase in mean number of TE/G1–G2 species found in IRAs, increasing from 2.1 to 4.0 species (Table 4).

The IRAs could also contribute a significant amount of land area to existing conservation reserves for both TE and TE/G1–G2 species in all geographic regions (Table 5). The largest increase in area and the greatest percent increase in conservation reserves are found in the western United States, with the exception of the 100% increase from the single quadrangle in Alaska. IRAs would contribute to a 96% increase in available habitat in conservation reserves for TE species, whereas the inclusion of G1–G2 species expands that increase to 210%. Although the eastern region would see similar but more modest gains, habitat in conservation reserves in the Alaska region would increase 113% for TE/G1–G2 species (Table 5). Overall, IRAs would increase the conservation reserve network containing TE, G1, or G2 species by 156%.

Table 5. The concordance of occurrences of threatened or endangered (TE) species or of TE species that are also classified as highly imperiled (G1–G2) by the IUCN with the existing conservation reserve network (IUCN I–III) and inventoried roadless areas (IRAs).

Region	No. of TE species quadrangles in IUCN I–III conservation reserves	No. of TE species quadrangles in IRAs	Percent increase	No. of TE/G1–G2 species quadrangles in IRAs	No. of TE/G1–G2 species quadrangles in IUCN I–III conservation reserves	Percent increase
Eastern United States	995	217	22	1027	431	42
Western United States	1752	1679	96	2200	4627	210
Alaska	0	1	100	38	43	113

Grizzly bear case study

As seen in Fig. 5, the inclusion of IRAs in the existing system of conservation reserves in Washington, Idaho, Montana, and Wyoming shows a strong concordance with the grizzly bear recovery zones of the U.S. Fish and Wildlife Service, as well as bear habitat range (Martin et al. 2000, USDA Forest Service 2000). In total, the six grizzly bear recovery zones include approximately 15,300 km^2 of IRAs. Approximately 24,750 km^2 of almost contiguous IRAs surround the Salmon-Selway (Bitterroot) Recovery Zone (SSRZ), which has already been designated a wilderness area and assigned to IUCN category I.

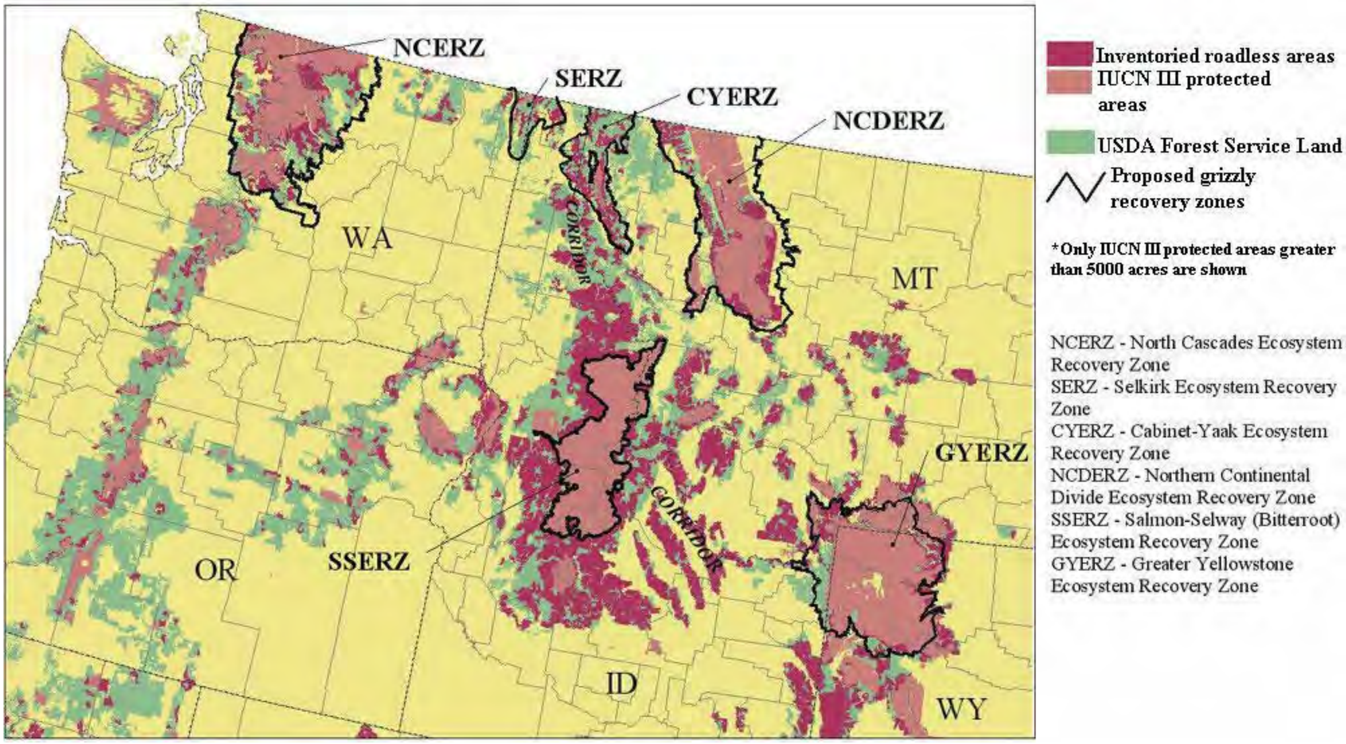
DISCUSSION

Our analyses found that one-quarter of the inventoried roadless ares (IRAs) are found in globally or regionally outstanding ecoregions, and that they have the potential to provide important habitat for numerous species, including threatened, endangered, and imperiled species. This conclusion is further illustrated by an investigation of the potential benefit of IRAs to grizzly bear conservation.

Based on these findings, the assignment of IRAs to IUCN category III or higher could increase the area of conservation reserves in the United States from 4.8 to 8.5%. This broad national conclusion has different implications depending on geographic region. For example, whereas fewer than 3% of the IRAs are found in the eastern United States, the vast majority of eastern IRAs are found in the ecoregions with the greatest amount of biodiversity and the least amount of existing protection. In addition, despite the fact that western forests currently have some of the highest existing protection levels in the United States, Scott et al. (2001) found that many existing reserves in the United States are concentrated in areas of high elevation and low soil productivity. Therefore, despite the current levels of perceived protection, the nation's biological diversity may be under-represented in the current system, particularly in the mountainous west (Scott et al. 2001). DeVelice and Martin (2001) have shown that approximately 40% (about 91,300 km^2) of the IRAs are at an elevation below 1500 m and that 35% of the total IRAs are adjacent to designated wilderness areas. The combination of increased protection of forest habitat and the potential increase in size of conservation reserves would have a positive effect on the conservation of large mammals in the western United States.

The purpose of the *Endangered Species Act* is to " ... provide a means whereby the ecosystems upon which endangered and threatened species depend may be conserved ... " (U.S. Fish and Wildlife Service 1973). The act further directs that " ... all Federal departments shall seek to conserve endangered species and threatened species." In this regard, many IRAs function as biological refugia for terrestrial and aquatic species, including numerous threatened, endangered, and imperiled species. The maintenance of natural values in IRAs could contribute to their long-term viability (Brown and Archuleta 2000). IRAs contain more than 220 TE species, i.e., approximately 25% of listed or proposed animal species and 13% of listed plant species (USDA Forest Service 2000).

Fig. 5. Overlap of USDA Forest Service inventoried roadless areas and grizzly bear recovery zones.



Among TE species, 88% are imperiled by habitat destruction and degradation (Wilcove et al. 1998). Dobson et al. (1997) found that, if the habitats of TE species were more extensively protected, a large number of them would be efficiently conserved. Our analysis showed that the vast majority of IRAs hosted TE or G1–G2 imperiled species and that, by adding the IRAs to the existing conservation reserve system, the conservation of species at risk and their habitat could be better realized. Although we recognize that not all threatened, endangered, or imperiled species require lands free of active land management to survive, limiting the human footprint by placing IRAs off limits to road construction and maintenance, resource extraction, and other development activities could provide a counterpoint to the multiple-use activities taking place elsewhere within the National Forest System.

Furthermore, although there may be duplicate species populations within IRAs or existing conservation reserves, the high level of endangerment of these species should predicate that we conserve as many populations as possible. Therefore, the potential issues of complementarity or duplication of species across IRAs should not diminish the contribution that IRAs could make to conserving species at risk. Our analyses have shown that, despite the small size and extent of IRAs in the eastern United States, they contain a greater number of endangered or imperiled species across more IRAs than do the west and Alaska. However, many of the western IRAs are missing data or have not been surveyed. This error of omission serves only to emphasize that our findings are a conservative estimate of potential species endangerment particularly in IRAs in Alaska and the western United States.

Top carnivore species, such as the grizzly bear, often have the largest species-level area requirements in an ecosystem and maintain ecological structures and resilience by top-down trophic interactions. They need large, contiguous habitat blocks to persist, and there must be landscape connectivity among core areas to ensure sufficient habitat for viable populations (Soulé and Noss 1998, Carroll et al. 2001). As a result of these requirements, large reserves are necessary to maintain populations of these wide-ranging species. Woodroffe and Ginsberg (1998) recently estimated that habitats of 20,000 km^2 are needed to provide a 90% chance for the long-term survival of the grizzly bear in the wild. Indeed, only those wilderness areas that were 20,000 km^2 or larger in 1920 still support grizzly bears today (Mattson and Merrill 2002). The 40,000 km^2 of IRAs in and near designated grizzly recovery zones in the northern Rockies will help improve the long-term habitat viability for grizzly bears in the region (Martin et al. 2000, USDA Forest Service 2000).

Carroll et al. (2001) proposed the need for a comprehensive conservation strategy for carnivores in the Rocky Mountains that considers the requirements of several species, including grizzly bear, wolverine, fisher, and lynx. The regions where these four species overlap show a strong concordance with grizzly bear recovery zones. IRAs may benefit all of these species by providing expanded and buffered habitat and, in turn, secure the ecological integrity of those ecosystems (Terborgh and Soulé 1999, Conner et al. 2000, Martin et al. 2000). If grizzly bear populations remain limited by the size and configuration of current conservation reserves, their long-term survival in the conterminous United States cannot be assured (Mattson and Merrill 2002).

Bruner et al. (2001) found a clear relationship between the existence of a viable and well-connected system of conservation reserves and biodiversity conservation. Because of the stable long-term ownership tenure associated with USDA Forest Service lands, as opposed to privately held forests, many of these forested areas contain a wealth of biological diversity. Historically, land within the Forest Service has been managed under a multiple-use strategy, with timber extraction being a main component of many of these plans. However, multiple-use management may not ensure the protection of the full range of biodiversity, because anthropogenic habitat degradation and destruction are the primary causes of biodiversity loss (Ehrlich 1988, Myers 1988, Wilcove et al. 1996, Haila 1999, Wood 2000).

Setting aside IRAs for stricter protection from extractive or economically driven activities may indeed meet many biological objectives, e.g., integration of fish and wildlife values and watershed and forest health, consistent with the agency's multipurpose agenda. In addition, IRAs may also contribute invaluable benchmarks to gauge ecological changes on managed U.S. Forest Service lands. A representative system of natural habitats, set aside from active management, would allow natural ecological processes, including a full suite of existing native species, to survive free of human activities. Without strict conservation areas that represent all forest habitat types, it will be difficult to make objective assessments on the sustainability of forest management (Noss and Cooperrider 1994, Norton 1999, Noss et al. 1999). Based upon our analyses, we conclude that IRAs support many at-risk species and thereby greatly contribute to the conservation of biodiversity throughout the United States. For some species with only a few remaining populations, the strict and permanent protection of IRAs may represent the final, critical refuge.

Responses to this article can be read online at:
http://www.consecol.org/vol7/iss2/art5/responses/index.html

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fire & fuels management

Constraints on Mechanized Treatment Significantly Limit Mechanical Fuels Reduction Extent in the Sierra Nevada

Malcolm North, April Brough, Jonathan Long, Brandon Collins, Phil Bowden, Don Yasuda, Jay Miller, and Neil Sugihara

With air quality, liability, and safety concerns, prescribed burning and managed wildfire are often considered impractical treatments for extensive fuels reduction in western US forests. For California's Sierra Nevada forests, we evaluated the alternative and analyzed the amount and distribution of constraints on mechanical fuels treatments on USDA Forest Service land. With the use of current standards and guides, feedback from practicing silviculturists, and GIS databases, we developed a hierarchy of biological (i.e., nonproductive forest), legal (i.e., wilderness), operational (i.e., equipment access), and administrative (i.e., sensitive species and riparian areas) constraints. Of the Sierra Nevada Bioregion's 10.7 million acres in USDA Forest Service ownership, 58% contains productive forest and 25% is available to mechanical treatment. National forests in the southern Sierra Nevada have higher levels of constraint due to more wilderness and steeper, more remote terrain. We evaluated different levels of operational constraints and found that increasing road building and operating on steeper slopes had less effect on increasing mechanical access than removing economic considerations (i.e., accessing sites regardless of timber volume). Constraints due to sensitive species habitat and riparian areas only reduced productive forest access by 8%. We divided the Sierra Nevada Bioregion into 710 subwatersheds (mean size of 22,800 acres) with $>$25% Forest Service ownership as an approximation of a relevant management planning unit for fire or "fireshed." Only 20% of these subwatersheds had enough unconstrained acreage to effectively contain or suppress wildfire with mechanical treatment alone. Analysis suggests mechanical treatment in most subwatersheds could be more effective if it established a fuel-reduced "anchor" from which prescribed and managed fire could be strategically expanded. With potential future increases in wildfire size and severity, fire policy and forest restoration might benefit if mechanical thinning is more widely used to leverage and complement managed fire.

Keywords: forest planning, fuels management, mixed conifer, prescribed burning, wildfire

Current rates of fuels treatment on western public lands are far below what is needed to effectively influence landscape-level fire behavior or approximate historic levels of annual area burned (Stephens and Ruth 2005, North et al. 2012). Many issues contribute to this low level of implementation (e.g., limited budgets, shrinking workforce, and other factors), but a significant factor is the challenge of working in landscapes riddled with operational constraints (Collins et al. 2010). With optimal spacing, models suggest that fuels reduction can be effective for reducing fire size and severity when roughly 15–30% of the landscape has been treated (Finney 2001, 2007). In practice, however, the increasing number of rural homes (Theobald 2005, Theobald and Romme 2007), administrative boundaries that restrict management options (Lee and Irwin 2005), and economics of wood harvest and transportation (Hartsough et al. 2008) can result in a default fuels reduction strategy of treating what is left. These constraints can affect what type of treatment is practical in different areas, with treatments broadly divided into three options, mechanical thinning (including mastication), fire (prescribed burning and managed wildfire), or a combination of both (Agee and Skinner 2005). Research has suggested that greater restoration and resilience in forests that historically had low to

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moderate severity frequent fire regimes can be achieved with treatments that include fire (North et al. 2009, Fule et al. 2012, Stephens et al. 2012). Fire, however, can be difficult to use because of smoke impacts, proximate human communities, and liability and cost constraints (Quinn-Davidson and Varner 2012). This is particularly true in densely populated areas such as California, where mechanical thinning is sometimes viewed as the only realistic means of increasing the pace and scale of fuels reduction treatment (Quinn-Davidson and Varner 2012).

Mechanical treatments,[1] however, have their own set of restrictions (Reinhardt et al. 2008). This is particularly true on public lands where legal, operational, and administrative constraints can significantly restrict treatment locations and extent. For example, mechanical thinning is not allowed in wilderness and roadless areas, may not be economical or operationally feasible in remote areas with steep ground and smaller trees, and is constrained in some areas with special administrative designations, such as sensitive species activity centers and riparian forest buffers (Donovan and Brown 2005). Furthermore, the arrangement of constrained lands within potential firesheds (i.e., subwatersheds in which fire spread may be controlled at bordering ridges) (Bahro et al. 2007) also matters because it is the scale at which fuels treatments can most effectively influence fire behavior (Finney et al. 2007). For mechanical treatments to be effective, three questions need to be examined. How extensive are these constraints, which have the greatest impact on limiting treatment extent, and how do they affect the ability to successfully influence landscape-level fire effects?

To investigate these questions, we examined constraints on mechanical operability on US Department of Agriculture (USDA) Forest Service land across the Sierra Nevada Bioregion (SNBR) (Figure 1). The intent was to identify the extent to which mechanical fuel reduction treatments can be used to meet the stated objective of increasing the pace and scale of restoration within the SNBR (USDA Forest Service 2011). In particular, we asked the following questions: (1) What percentage of the total land base has mechanical constraints? (2) How do different operational constraints (i.e., slope, distance from existing road, and economics) affect the amount and distribution of mechanically treatable areas and how does this vary across national forests (NFs) with increasing topographic relief? (3) What impact do special land management restrictions such as sensitive species habitat and riparian zones have on mechanical fuels reduction? and (4) Given the spatial distribution of these constraints, how many SNBR watersheds can be effectively treated with mechanical fuels reduction alone? The Sierra Nevada may be at the forefront for evaluating how these constraints affect forest planning. The Forest Service recently adopted a new planning rule (USDA Forest Service 2012) that initiates the development of new forest plans for most of the 155 NFs. Eight NFs have been identified as "early adopters" for plan development, and three of these (the Sierra, Sequoia, and Inyo) are in the Sierra Nevada.

Methods

We examined the amount and spatial distribution of USDA Forest Service land in the SNBR in which fuels reduction using ground-based equipment is allowed and operationally feasible, considering factors such as legislative restrictions, operational limitations, and administrative constraints.[2] Our analysis used ESRI ArcGIS software and data layers developed by the USDA Forest Service Pacific Southwest Region. Our analysis included the Lake Tahoe Basin Management Unit (for simplicity hereafter grouped with the other NFs) and 9 of the 11 NFs examined in the 2013 USDA Forest Service Sierra Nevada Bioregional Assessment (Figure 1). We excluded the Klamath and Shasta-Trinity NFs because although each has area within the SNBR, they are relatively small areas in the foothills that exclude more mountainous terrain affecting constraint patterns on the other 10 NFs.

We used a hierarchy of constraints that affect mechanical operability on Forest Service land, starting with fixed limitations and moving down to constraints with more flexibility in interpretation and implementation. At the first level (L0: Biological constraint), we started with the total acres in each NF and then removed land identified as nonforest (i.e., rock, water, barren, meadow, and shrub) and with a forest cover <10% (Table 1). Then considering only productive forestland, we removed areas where mechanical equipment is not allowed (i.e., wilderness and roadless) (L1: Legal constraint).

The next two levels of constraint were based on existing standards and guidelines (USDA Forest Service 2004) and current practices (L2: Operational and L3: Administrative) (Table 1). Current practices were identified using expert opinion from one-half dozen NF silviculturists within the SNBR area. Some operational constraints are specifically identified in the standards and guidelines (i.e., mechanical equipment is generally prohibited on slopes >35% with unstable soils), but many give managers some discretion (i.e., thinning is allowed in riparian areas, but mechanical yarders cannot travel within 50–100 ft of streams). Silviculturists gave us a range of operational constraints that were affected by three factors, slope, distance from existing road, and commercial value of the accessed forest. Mechanical equipment generally is allowed on slopes of <35%, whereas some equipment (i.e., self-leveling feller-bunchers) can operate more slowly and at higher cost on slopes up to 50% with suitable soils and more valuable wood. Logging on slopes of >50% re-

Management and Policy Implications

Western US efforts to increase the pace and scale of fuels treatment and forest restoration often rely on mechanical treatment because of limitations on using managed fire. We found that with only 25% of national forestland in the Sierra Nevada available to mechanical treatment, there is limited ability to affect wildfire extent and severity in many areas. Furthermore, when these mechanical constraints are grouped and examined by subwatershed, almost half of these have too little mechanically available acreage to affect potential wildfire behavior. Mechanically treatable areas are often not optimally located for containing wildfire but are well situated as anchors from which prescribed burning and managed wildfire might be expanded. Rather than primarily planning and placing mechanical treatments to contain and suppress wildfire, many treatments could be targeted to facilitate the reintroduction of beneficial fire. After adoption of a new planning rule, three of the first eight National Forests developing new Land and Resource Management Plans ("early adopters") are in the southern Sierra Nevada. Our analysis suggests that new plans consider identifying areas and weather conditions under which fire is allowed to burn. Efforts to increase the pace and scale of fuels reduction and forest restoration are unlikely to succeed without more extensive and innovative use of managed fire.

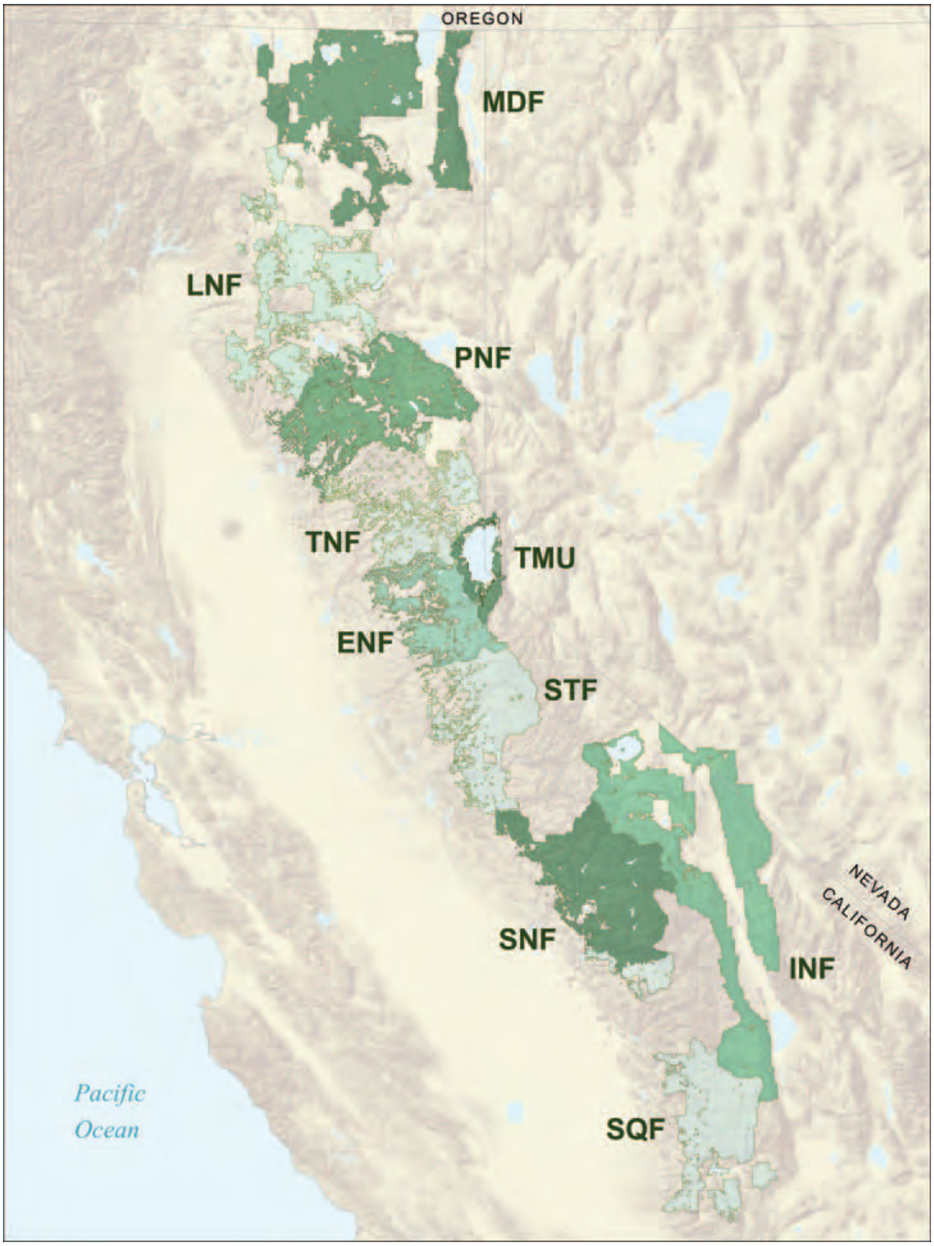

Figure 1. Map of the Sierra Nevada Bioregion used in this analysis. Outlines of the 10 NFs examined are shown and shaded polygons show Forest Service ownership. The NFs are Modoc (MDF), Lassen (LNF), Plumas (PNF), Tahoe (TNF), Eldorado (ENF), Stanislaus (STF), Sierra (SNF), Inyo (INF), and Sequoia (SQF). TMU indicates the Lake Tahoe Basin Management Unit.

quire cable yarding systems, which are not widely used for fuels reduction treatments on Forest Service land in the Sierra Nevada. Distance from existing road impact operations because the Forest Service typically limits construction to temporary roads <1,000 ft long. However, longer access roads may be constructed if there is a resource need and costs can be offset based on timber harvest value. As an indirect measure of economic potential, we used the California Wildlife Habitat Relationship (CWHR) system, a classification widely used by the Forest Service to indicate forest type, average tree size, and canopy cover of different forests (Mayer and Laudenslayer 1988). Forests are classified by a code that indicates the forest type (e.g., SMC for Sierran mixed conifer), size class (1–6, depending on average tree diameter), and canopy cover. We considered forests as having economic potential if they were conifer forest types found in lower to midelevations, with an average tree diameter ≥11 in. and canopy cover ≥40%. In general, forests in the Sierra Nevada that meet these criteria usually have large enough trees to provide merchantable timber, such that fuels reduction treatments could essentially "pay for themselves" (i.e., the value obtained from thinning larger trees could offset the cost of removing smaller, submerchantable trees that often function as ladder fuels). We did not consider small-diameter biomass utilization, as currently there are few facilities to subsidize the costs of removing this material.

Using these factors, we developed four scenarios of operational constraints (A–D). These scenarios capture the range of feasible interpretations of current standards and guides. Scenario A reflects the most strict adherence to current standards and guides where mechanical operations occur on <35% slopes and within 1,000 ft of existing roads (Table 1). Scenario B extends the road building distance to 2,000 ft if more valuable timber is accessed to help defray costs. Scenario C adds working on steeper slopes (35–50%) within 500 ft of existing roads if more valuable timber is accessed. Scenario D accesses all forest (regardless of timber value) on <35% slope within 2,000 ft of existing roads and all forest on 35–50% slope within 1,000 ft of existing roads. Some forests adhere to scenario A constraints particularly if operating in or near riparian areas and sensitive species habitat. Many forests use a combination of scenario B and C, depending on forest and physiographic conditions. Scenario D is rarely used but has been used when there is nontimber, high-resource value to a particular area. As a conservative approach, in some of our analyses we use scenario C to evaluate the effects of less restrictive mechanical constraints on fuels treatment implementation.

Some forestland also limits mechanical treatment through special administrative designation (L3). We included those that are most common in the Sierra Nevada, including riparian zones, California spotted owl (*Strix occidentalis occidentalis*) and northern goshawk (*Accipiter gentilis*) activity centers, and Research Natural Areas (Table 1). Mechanical treatments are not strictly prohibited in these areas, but they are highly restricted and in practice are areas that are often left untreated. For buffer widths on either side of streams we used 100 and 50 ft for perennial and intermittent streams, respectively, following current standards and guidelines (USDA Forest Service 2004). In areas designated as wildland urban interface (Radeloff et al. 2005), restrictions for sensitive species habitat apply to a 500 ft radius around nest or activity center areas. In all other areas, it is 300 and 200 acres for spot-

Table 1. Hierarchy, types, and criteria of mechanical treatment constraints used in our analysis.

Constraint type	Criteria		
L0: Biological			
a. Not timber productive	a. Either nonforest or <10% cover		
b. Water/Barren			
L1: Legal			
a. Wilderness			
b. Recommended wilderness			
c. Inventoried roadless	c. All inventoried roadless except those areas where new road construction is allowed		
L2: Operational	Slope	Road distance	CWHR
A. Existing (most constrained, gentle slope near roads)	<35	<1,000	
B. A plus road distance increase (distance extended for areas with greater economic return)	<35	<1,000	
		<2,000	4, 5 (M and D), 6
C. B plus slope increase (if close to road, slope increased for areas with greater economic return)	<35	<1,000	
		<2,000	4, 5 (M and D), 6
	35–50	<500	4, 5 (M and D), 6
D. C plus all forest types (least constrained by slope, road access and economics)	<35	<2,000	
	35–50	<1,000	
L3: Administrative			
a. Riparian proximity	a. Buffer width: 100 ft perennial; 50 ft intermittent		
b. California spotted owl	b. WUI—500 ft radius; otherwise 300 acres around activity center/nest		
c. Goshawk	c. WUI—500 ft radius; otherwise management identified polygon (mean = 200 acres)		
d. Research natural areas			

The CWHR system is a widely used forest classification with M and D referring to canopy cover of 40–59% and 60–100%, respectively, and 4, 5, and 6 indicating a quadratic mean diameter of 11–24, >24, and >24 in. with a multilayer canopy, respectively. We confined our CWHR forest types to conifers only. WUI, wildland urban interface.

ted owls and goshawks around the nest/activity center, respectively. Although it is an important sensitive species in the Sierra Nevada, we did not include the fisher (*Martes pennanti*) in our analysis because there were no data identifying resting and core activity areas. If their habitat had been included, it would decrease the area available for mechanical treatment, but only on the Sierra and Sequoia NFs where a small (<200 individuals) isolated population of fisher is present (Zielinski et al. 2005).

In an effort to characterize the spatial arrangement of mechanically operable land, we subdivided the SNBR area into discrete geographic units. Earlier analysis of Forest Service managed lands in the SNBR used the concept of "firesheds" to identify meaningful landscape management units. A fireshed has been defined as a contiguous area with similar fire history and problem fire characteristics where a coordinated suppression effort would be most effective (Ager et al. 2006, Bahro et al. 2007). Although this effort was not completed for the entire region, many of the firesheds that were identified followed subwatershed boundaries. As a unit for our landscape analysis, we used sixth level hydrologic units (HUs) enumerated with 12-digit codes, commonly referred to as "subwatersheds." These units represent an imperfect approximation of potential firesheds. They are generally sized at a scale at which fire containment is initially managed (8,000–40,000 acres), and the ridge tops that separate watersheds commonly provide opportunities for wildfire containment. Omernik (2003) has pointed out that many HUs are smaller than entire watersheds, but for our fire-focused analysis, their topographic delineation may serve as an appropriate initial fireshed classification for forestland in the Sierra Nevada.

We excluded HUs that were not entirely within the SNBR and where Forest Service ownership was <25% of the burnable forest area (excluding bare rock and sparsely vegetated areas). We used this cutoff under the assumption that with <25% ownership, Forest Service treatment alone could not substantially affect wildfire behavior across the subwatershed. For the remaining subwatersheds, we calculated the percentage of the subwatershed's total burnable forest that the Forest Service could mechanically treat. Based on model simulations of how much area generally needs to be treated to influence wildfire behavior, we binned the subwatersheds into three classes of mechanical constraint: high (85–100% [i.e., only 0–15% is available for mechanical treatment]), medium (65–84%), and low (<65%). We chose these levels to identify watersheds where fuels treatment would principally need to rely on fire (those with a high level of mechanical constraint), could use a combination of fire and mechanical thinning (medium), and could effectively influence wildfire behavior with mechanical treatment alone (low). We calculated the percentage of subwatersheds in each of these categories for each NF across the SNBR.

Results

Of the SNBR's 10.7 million acres, 4.5 million acres were nonproductive forestland. The NFs with the largest amount of nonproductive forestland are the Modoc with 63% (mostly sagebrush [*Artemisia* spp.]) and the Inyo with 80% (mostly alpine, rock, and some low-elevation sagebrush) (Figure 2). Focusing on just the productive forestland on each NF (Table 2), legal constraints (wilderness and roadless) reduced mechanically available acreage on average by 22.5% (Table 2). On productive forestlands, legal constraints imposed the largest reduction in mechanically available acreage in the southern (the Stanislaus, Sierra, and Sequoia) and eastern (Inyo) NFs of the bioregion (Table 2).

A comparison of the impact of different operational constraints found a much higher range between scenarios A to D in the northern than the southern parts of the SNBR (Figure 3). In the northern NFs (Modoc, Lassen, Plumas, and Tahoe), there is an average increase in mechanically available acreage of 17% between scenario A (current standard and guides) and D (increasing slope and road access to all productive forest) compared with just a 9.5% increase for the southern and eastern NFs (Stanislaus, Sierra, Sequoia, and Inyo) (Figure 3). Changing operational constraints from scenario A to B (greater access distance from existing roads for large trees) increased mechanical acreage on average about 2–3% in the southern part of the range but up to about 6–7% in the northern NFs (Figure 3). The greater increase in northern NFs results because the increased operational "reach" from existing roads tends not to overlap as much with legal constraints such as wilderness and roadless areas, which limit the effect of easing the operational constraints in the southern NFs. There was little increase in available acreage between operational constraint scenarios B and C (adding steeper slopes close to roads for large trees), regardless of NF. The relatively greater increase between scenarios C and D (increased access to *all* productive forests) reflects the limited amount of large-tree forests, particularly in the northern extent of the SNBR. Focusing on scenario C, operational constraints reduced productive forestland available for mechanical treatment on average by 25.6% (Table 2).

The percent reduction of mechanically available acreage with administrative constraints (riparian zones, sensitive species habitat, and Research Natural Areas) varied widely between different NFs. Whereas the overall reduction averaged 8.1% (Table 2), NFs generally fell into two equal-sized classes with either a modest reduction of 1.9–6.3% (Inyo, Modoc, Sequoia, Lassen, and Lake Tahoe Basin Management Unit [TMU]) or a higher reduction of 9.2–13.2% (Sierra, Tahoe, Eldorado, Plumas, and Stanislaus). What drove this difference was the distribution of sensitive species habitat, particularly that of spotted owls, because Research Natural Areas are small and riparian constraints were fairly similar between NFs.

We identified 710 subwatersheds across the SNBR for further analysis using our rule of Forest Service managed area being >25% of the total burnable area. On average, 46, 34, and 20% of the subwatersheds were highly, moderately, and lightly mechanically constrained, respectively (Table 3). The constrained area was determined using L0–L3, scenario C (Table 1). Half of SNBR's NFs (Stanislaus, Modoc, Sierra, Sequoia, and Inyo) have ≥50% of their subwatersheds highly constrained in which mechanical treatment alone is too limited to affect wildfire behavior or containment. Only the Lassen, Plumas, and Tahoe NFs have >25% of their subwatersheds lightly mechanically constrained (Table 3). A range-wide map of subwatersheds shaded by constraint level (Figure 4) indicates mechanically constrained areas tend to be clustered. The Modoc NF and the forests in the southern and eastern Sierra Nevada have large contiguous areas in which mechanical treatments make up a small percentage of each subwatershed's burnable acres.

A closer examination of the subwatersheds on a portion of the Sierra NF demonstrates the wide array of patterns in mechanical operability, ranging from large clusters to highly dispersed numerous small fragments (Figure 5). In highly constrained subwatersheds, mechanical treatment alone probably will have a limited and localized effect on reducing potential fire intensity and size (e.g., subwatersheds in the lower left and lower right of Figure 5 with 13 and 10% mechanically available). In subwatersheds with moderate constraint levels, mechanical treatment alone can affect wildfire for some or most of the subwatershed's area depending on configuration (e.g., upper middle and lower middle subwatersheds with 23 and 20% in Figure 5). Mechanically treatable areas in subwatersheds with only a light constraint level are often large and numerous enough that they can achieve most of the subwatershed's desired fuels reduction with mechanical treatment alone (e.g., center and upper middle subwatersheds with 36 and 57% in Figure 5).

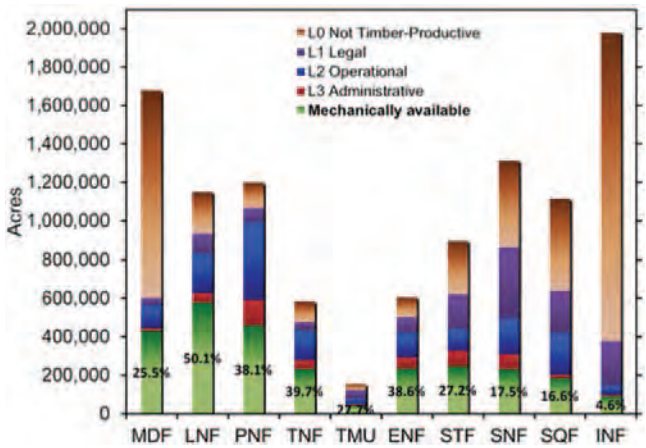

Figure 2. Histogram of how constraints reduce *total* acreage available to mechanical treatment in Sierra Nevada NFs. The height of the bar indicates each NF's total acres, with each constraint designated by a different color. The acreage available for mechanical treatment is what remains in the green portion of each bar and is indicated by the percentage values. Forests are arranged from northern most to southern along the western slope and the Inyo on the eastern slope. The L2 constraint uses scenario C (see Table 1).

Discussion

In California's Sierra Nevada forests, mechanical treatment is often considered the only practical large-scale fuels reduction strategy because there are many limitations on using fire (Williamson 2008, Quinn-Davidson and Varner 2012). Our analysis,

Table 2. Productive forest acreage (L0) of each NF and the percent reduction of different types of constraints on mechanical treatment.

NF	L0: Productive forest (acres)	L1: Legal	L2: Operational	L3: Administration	Total remaining (acres)	% of productive forest
		................(%)	....	....		
Modoc	602,209	−7.1	−18.9	−2.9	428,223	71.1
Lassen	935,571	−11.0	−21.9	−5.5	575,845	61.6
Plumas	1,065,594	−7.0	−37.6	−12.6	456,714	42.9
Tahoe	474,902	−8.9	−32.6	−9.8	231,276	48.7
TMU	121,434	−37.8	−21.5	−6.3	41,882	34.5
Eldorado	499,798	−16.3	−25.2	−11.8	233,448	46.7
Stanislaus	621,032	−28.9	−18.7	−13.2	243,774	39.3
Sierra	864,993	−42.8	−21.4	−9.2	229,502	26.5
Sequoia	639,808	−34.9	−33.2	−3.0	185,156	28.9
Inyo	376,325	−61.6	−12.3	−1.9	91,280	24.3
Total	6,201,666	−22.5	−25.6	−8.1	2,717,100	43.8

Constraints L1–L3 are the percentages of reduction in productive (in contrast to total forest acreage in Figure 2) forest. L2 reduction uses scenario C (see Table 1). Total remaining is the number of productive forest acres that are available for mechanical treatment after all constraints are applied.

lands that may or may not complement Forest Service fuels reduction objectives. These activities may significantly affect the impact of Forest Service mechanical fuels reduction particularly in moderate constraint level subwatersheds (e.g., in Figure 5 the upper left subwatershed with 20%, where much of the private ownership around Shaver Lake has had treatments by a private owner, Southern California Edison). The data sets used in this analysis may also fail to recognize numerous more localized operational constraints based on topography, additional protections (e.g., archeological and cultural sites), and treatment histories. Project plans may justify treatments in areas that are typically constrained (e.g., owl core areas) or have special practices in riparian buffers. Our analysis is intended to operate at a broad scale for planning, not a project-specific one. In general, the three constraint levels provide broad qualitative categories for subwatersheds where mechanical fuels reduction may have limited impact, will need to be strategically examined (considering configuration and other ownership management practices), or can be highly effective.

To check our analysis, we did compare our results with actual treatment plans on several NFs and found that there was a high level of consistency between areas that were not treated and areas that we identified as constrained. Our geographic information system (GIS) analysis may help inform forest planning efforts and serves as a useful communication tool for describing the feasibility of various treatment scenarios to public stakeholders.

Our analysis yielded strikingly different results from a similar analysis undertaken in the ponderosa pine (*Pinus ponderosa*) belt of Central Arizona (Hampton et al. 2011), which found that 78% of that landscape was potentially available to mechanical restoration thinning treatments. Key differences between the two regions include a much higher percentage of wilderness and roadless areas and greater constraints due to steep slopes in the Sierra Nevada. Areas with conditions analogous to the Sierra Nevada, however, are widespread in much of the western United States, particularly in more mountainous areas with productive forests, such as most of the Rocky Mountain lower and midelevation forests.

Economics constrains mechanical operability in the Sierra Nevada more than road building and steep slope limitations (Figure

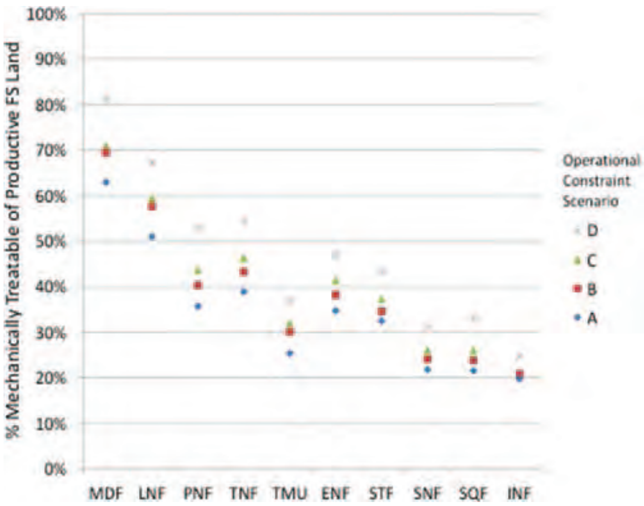

Figure 3. Symbols show the percentages of mechanically available productive forestland left on each NF under four different operational constraint scenarios (i.e., scenarios A–D [see L2 criteria in Table 1]) after all four constraint levels, L0–L3, are applied. The difference between the four scenarios (A is the most restrictive and D is the least constrained) indicates how sensitive the amount of mechanically available acreage is to different road distance, slope, and economic variables.

Table 3. Number of subwatersheds on each NF with ≥25% USDA Forest Service ownership of all burnable acres.

NF	HUs Total	HUs >25% USDA FS	Level of constraint High (85–100%)	Level of constraint Moderate (65–84%)	Level of constraint Light (<65%)
			. . . .	.(%).	. . . .
Modoc	144	96	51.0	32.3	16.7
Lassen	150	98	22.4	39.8	37.8
Plumas	111	87	20.7	44.8	34.5
Tahoe	90	54	24.1	48.1	27.8
TMU	27	16	37.5	50.0	12.5
Eldorado	65	50	26.0	50.0	24.0
Stanislaus	80	53	49.7	30.2	20.1
Sierra	92	77	66.2	15.6	18.2
Sequoia	103	70	72.9	22.8	4.3
Inyo	167	109	91.7	3.7	4.6
Total	1,029	710			
Average			46.2	33.7	20.1

The level of constraint values are the percentages of each NF subwatershed in which mechanical treatment is highly (85–100%), moderately (65–84%), and lightly (<65%) constrained using operational scenario C. The three categories are calculated based on the number of Forest Service acres available to mechanical treatment divided by the total burnable acres (across all ownerships) within the HU.

however, found that in many Forest Service managed areas, there are considerable areal constraints on mechanical treatment, suggesting that mechanical treatment alone may not be able to effectively increase the pace and scale of fuels reduction and forest restoration in much of the Sierra Nevada. The small amount of mechanically treatable acreage in 46% of subwatersheds and the often suboptimal distribution in another 34% of subwatersheds (Table 3), suggests that there is a limited ability to create effective extensive fuels treatments by mechanical methods alone. If mechanically available areas, however, are used as anchors from which to expand fire-based fuels reduction, the pace and scale of fuels treatment and forest restoration might be accelerated in many subwatersheds across the Sierra Nevada. Although our analysis focuses on the Sierra Nevada, other mountainous western US areas with productive forests may have similarly high levels of mechanical constraint due to extensive wilderness and roadless areas and steep terrain limiting access.

Our analysis has several limitations. One clear weakness is that we cannot capture what management may occur on private

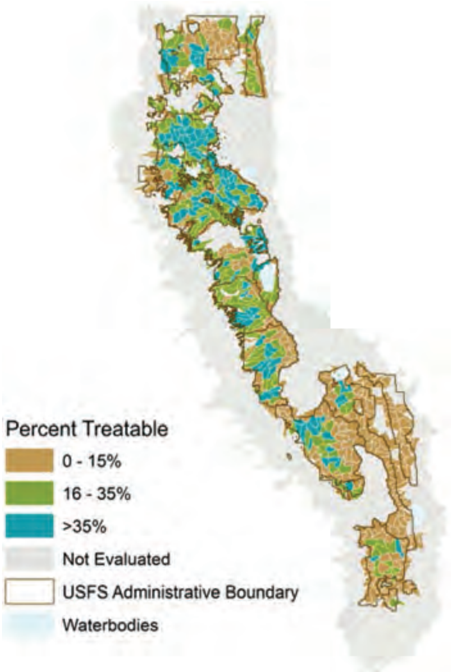

Figure 4. Sierra Nevada Bioregion divided into subwatersheds (HU12). Shadings indicate percentages of the total burnable acres that are available for the Forest Service to mechanically treat: gray, FS ownership <25%; brown, 0–15%; green, 16–35%; and blue, >35%.

3). Relaxing the allowable length of newly constructed road to access merchantable trees improved access more in the northern than in the southern NFs (scenario B). Operating on steeper slopes with large trees (scenario C) only slightly increased access regardless of location (scenarios A and B). This suggests that increasing use of temporary roads or alternative harvesting strategies (i.e., cable yarding) may not substantially ease constraints on mechanical operability. There is a larger increase in accessible acres when longer access roads are built and steeper slopes are treated without considering the need to offset these increased costs with timber revenues (Figure 3, scenario D). Given the current limited budgets for fuels treatment, this scenario is unlikely to be widely used. Over the next few decades, operational constraints may not change much until trees become large enough in less accessible areas to support higher costs of harvest.

Restrictions around sensitive species habitat are often considered a significant constraint on widespread fuels treatments (Keele et al. 2006). However, we found that on only 3 of the 10 NFs did these constraints reduce acreage by >10% (Table 2), after accounting for legal (wilderness and roadless designation) and operational (remote and steep slopes) constraints. In our analysis, the percent area removed (Table 2) was hierarchical from L0 to L3, meaning that for each successive level, we only report the additional area removed. This partially explains the relatively low percentages associated with the administrative constraints (L3, Table 2). Administrative constraints have the largest impact when operational constraints are relaxed to access merchantable trees. Larger trees (i.e., 20–29 in. dbh) are common in the preferred habitat for sensitive species (Berigan et al. 2012, Zielinski et al. 2013). These areas also have some of the highest fuel loads of SNBR forests (Spies et al. 2006), yet are generally left untreated due to concerns over potential resource damage and litigation.

Refinements in fire modeling have improved our understanding of optimal treatment size and location (Finney 2007, Ager et al. 2010, 2013), but extensive fuels treatment effectiveness can be limited when management is focused primarily on mechanical methods. Although the initial models suggested that a herringbone pattern of fuels treated areas is most effective in an idealized landscape (Finney 2001), multiple case studies (Collins et al. 2011, 2013) demonstrate that treating what is available can still be highly effective if the treated area is ≥20% of the landscape and generally perpendicular to prevailing wind and likely fire movement direction. For many SNBR subwatersheds, however, we found that mechanically available acreage may not be strategically oriented (relative to the dominant wind pattern) or arranged (too skewed or clumped) to effectively disrupt landscape-level fire spread and effects (Finney 2001). Furthermore, some areas would remain susceptible to wildfire spread due to untreated "stringers" (Figure 5). Some of these long linear mechanical exclusion zones are remote or steeply sloped areas, but many are riparian areas. Riparian zones often perforate mechanically treatable areas, yet leaving these areas untreated can significantly compromise fuels treatment effectiveness. Many riparian areas in the Sierra Nevada burned as frequently as adjacent upland forests, but given their higher productivity, often now have some of the highest fuels in the Sierra Nevada (Van de Water and North 2010, 2011). For riparian areas, designing treatments to specific characteristics of streams within a landscape may afford protections while reducing the fragmentation associated with standardized buffers (Hunsaker and Long 2014). Current policies appear to provide managers with some flexibility as long as they provide justification for riparian area treatment. If riparian buffers were treated with either thinning and/or fire, their wildfire wicking potential might be significantly reduced, increasing mechanical treatment effectiveness.

The subwatershed maps generated by this analysis indicate that the collective constraints on mechanical treatment may limit opportunities for effective extensive fuel reduction. Although it is possible that hand thinning could be substantially expanded, this is unlikely given the high cost per unit area and overall lack of funding. Recent noncommercial projects in the SNBR area demonstrate the economic limitations associated with removing only nonmerchantable trees (i.e., Cedar Valley and Sugar Pine Projects, Sierra NF). In an analysis conducted in the Stanislaus NF, Finney et al. (2007) noted that once constraints reserved 45% of the area from treatment, strategically placed fuels treatments performed no better than random placement. In our analysis, 8 of the 10 NFs in the SBNA had mechanical constraints on >45% of their productive forestland (Table 2). Some of these constraints are relatively fixed by policy or by nature, but others have been designed as temporary safeguards to minimize impacts to sensitive species and areas through administrative rules. To facilitate landscape-scale restoration, it may be important to relax these constraints in an adaptive management approach, such as within landscape demonstration areas (North et al. 2014). Another alternative is to apply threshold values for disturbance over time (Zielinski et al. 2013) at larger scales to mitigate impacts to sensitive wildlife species.

Our analysis suggests that in many areas a wildfire policy focused on containment and suppression is unlikely to be effective if it relies primarily on mechanical fuels reduction methods. Although fire models can help identify areas with higher burn probabilities (Ager et al. 2010, 2013), effective containment and suppression hinges on treatment placement (Syphard et al. 2011). In many SNBR subwatersheds current constraints rarely optimize mechanical treatment locations. Furthermore, mechanically maintaining reduced fuel loads in treated areas eventually consumes all of the fuels treatment effort, leading to a backlog of forest that never gets treated. By one estimate, >60% of productive forests in the Sierra Nevada will remain in the

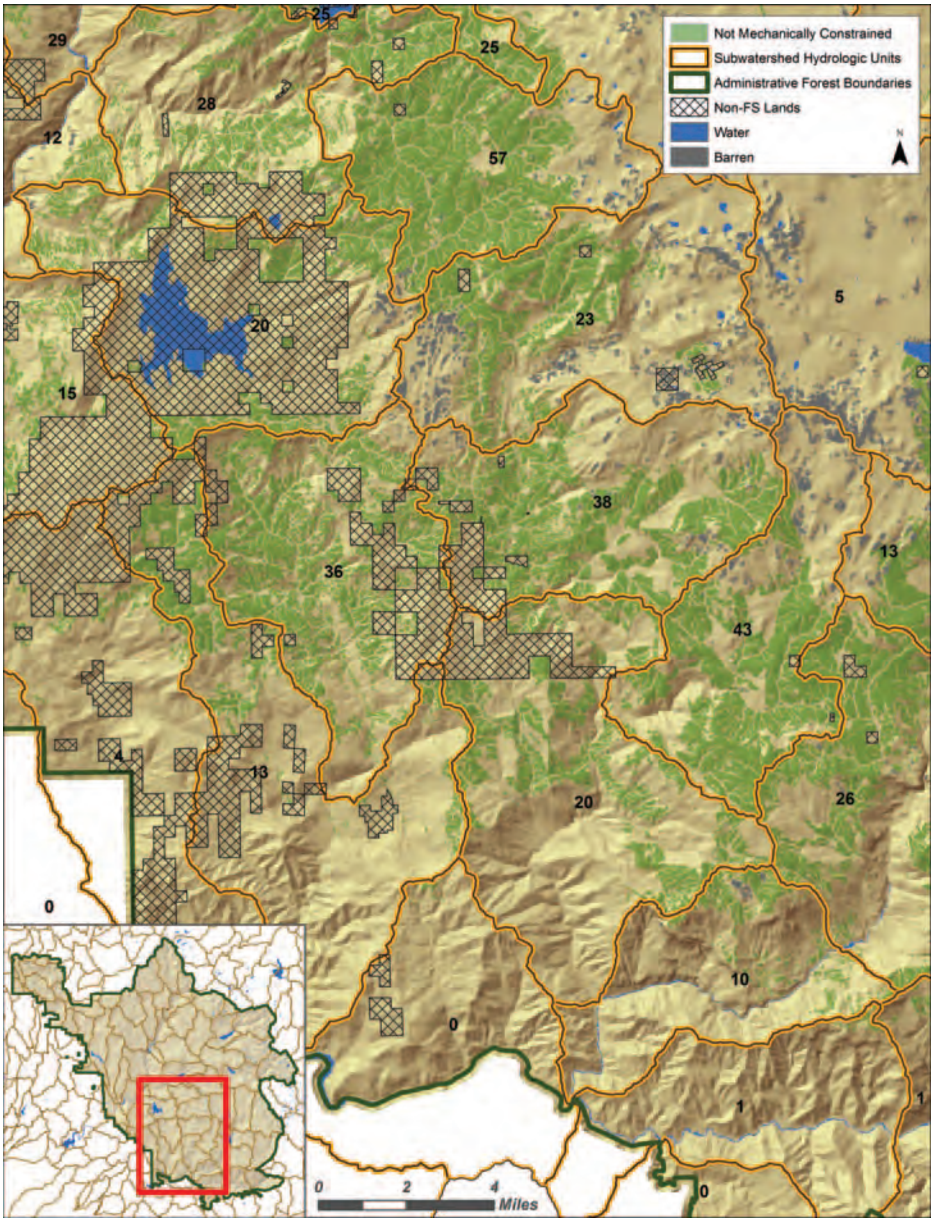

Figure 5. Adjacent subwatersheds (orange outlines) in the Dinkey Creek (middle) and Shaver Lake (upper left) area of the Sierra NF. Green shading indicates areas that can be mechanically treated, and the bold values indicate the percentages of total burnable forest within the subwatershed that can be mechanically treated by the Forest Service under operational constraint scenario C.

backlog of fuel-loaded forests at current treatment rates (North et al. 2012).

In most subwatersheds, the most effective use of mechanically treatable areas may be as "safe zone" anchors for wider reintroduction of fire. For example, the 10% subwatershed in Figure 5 has a few lower slope mechanically treatable areas (near the river), but fire-based fuels reduction is needed to effectively connect these areas to upper slope/ridgetop mechanically unconstrained areas. Large prescribed burns commonly used in western Australia are possible because a network of low-fuel "anchors" (previous burns, rocky areas, and low-fuel forests) allow 6–8% of the forest to be burned annually (Sneeuwjagt et al. 2013). Although the outcomes of fuels reduction by prescribed burn and managed wildfire are less precise or "surgical" at the stand level, across a landscape it can be much more effective than relying on constrained mechanical treatments. Mechanical treatment still is probably the most practical fuels treatment in the wildland urban interface, and opportunities for extensive use of managed fire may be further reduced during extended droughts. However, under moderate weather conditions and in remote locations, prescribed burning may be more efficient, cost-effective, and ecologically beneficial (North et al. 2012) than extensive mechanical treatments. Using machine harvest to establish more anchors for fire reintroduction would also generate forest products that provide economic opportunities for rural communities with processing infrastructure.

Our analysis suggests that the current heavy reliance on mechanical fuels reduction is unlikely to effectively contain or suppress wildfire in many areas of the Sierra Nevada. Too much NF area is unavailable for mechanical treatment and what is available is often too small and scattered to effectively alter landscape-level fire spread and intensity. However, significant increases in treatment pace and scale are possible if mechanical thinning is used to facilitate larger prescribed burns and enable managed wildfire. Wildfire size and intensity are predicted to increase under future projected climate scenarios (Lenihan et al. 2003, Lenihan et al. 2008), suggesting that fire policy and forest restoration might benefit if mechanical thinning is more widely used to leverage and complement managed fire.

Endnotes

1. In this article, we use the term mechanical treatment to refer to machine-based fuels reduction and tree harvest (i.e., use of ground-based heavy equipment such as feller-bunchers and skidders). We did not include hand thinning with chainsaws within our scope of mechanical treatments because high costs and slow pace constrain its effectiveness for reducing fuels in the Sierra Nevada.
2. The GIS analysis, data layers, and more detailed methods are available at https://fs.usda.gov/wps/PA_WIDContribution/widct/previewhtml.jsp?param1=STELPRDB5327833&param2=text/html&param3=1646948.

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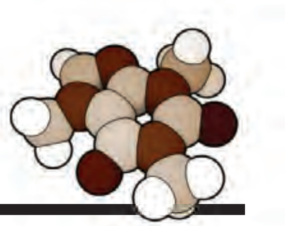

PERSPECTIVES

ENVIRONMENTAL SCIENCE

Reform forest fire management

Agency incentives undermine policy effectiveness

By M. P. North,[1,2*] S. L. Stephens,[3] B. M. Collins,[1,3] J. K. Agee,[4] G. Aplet,[5] J. F. Franklin,[4], P. Z. Fulé[6]

Globally, wildfire size, severity, and frequency have been increasing, as have related fatalities and taxpayer-funded firefighting costs (*1*). In most accessible forests, wildfire response prioritizes suppression because fires are easier and cheaper to contain when small (*2*). In the United States, for example, 98% of wildfires are suppressed before reaching 120 ha in size (*3*). But the 2% of wildfires that escape containment often burn under extreme weather conditions in fuel-loaded forests and account for 97% of fire-fighting costs and total area burned (*3*). Changing climate and decades of fuel accumulation make efforts to suppress every fire dangerous, expensive, and ill advised (*4*). These trends are attracting congressional scrutiny for a new approach to wildfire management (*5*). The recent release of the National Cohesive Wildland Fire Management Strategy (NCWFMS) (*6*) and the U.S. Forest Service's (USFS's) current effort to revise national forest (NF) plans provide openings to incentivize change. Although we largely focus on the USFS, which incurs 70% of national firefighting costs (*7*), similar wildfire policies and needed management reforms are relevant throughout the United States and fire-prone areas worldwide.

POLICY

Accumulated fuels in dry forests need to be reduced so that when fire occurs, rather than "crowning out" and killing most trees, it is more likely to burn along the surface at low-moderate intensity, consuming many small trees and restoring forest resilience to future drought and fire. Mechanical thinning can reduce tree density and some fuels but is often limited by legal (wilderness and park areas), operational (steep or remote ground), and cost constraints (*8*). Fire can also be used to reduce fuels either intentionally (prescribed burning) or opportunistically (letting a natural ignition burn as "managed wildfire") under moderate weather conditions. Although these burns are much less precise than mechanical thinning, in remote locations, fire is usually more efficient, cost-effective, and ecologically beneficial than mechanical treatments (*9*).

Many severe wildfires are due to past fire suppression. Firefighters during the Rim Fire near Yosemite National Park, California, 25 August 2013.

ENTRENCHED DISINCENTIVES. Management reform in the United States has failed, not because of policy, but owing to lack of coordinated pressure sufficient to overcome entrenched agency disincentives to working with fire. Responding to established research, official agency policy now supports a more flexible response to fire than ever before (*6*). Actual wildfire response, however, has changed little because of substantial management impediments. Suppression generally begets larger, more intense wildfires, which in turn intensifies agencies' suppression response (*10*). The alternative, working with fire, is rarely used because of liability and casualty risks and little tolerance for management errors.

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For example, during the most recent decade when data were collected (ending in 2008), only 0.4% of ignitions were allowed to burn as managed wildfires (*7*). For individual NFs, there is little economic incentive to change because fire suppression is steadfastly financed through dedicated congressional appropriations, which are augmented with emergency funding, whereas fuels reduction and prescribed burning costs come out of a limited budget allotted to each NF and is often borrowed to cover wildfire suppression costs. With these deterrents, "battling" fire and "only you can prevent wildfire" campaigns have more traction than recognizing that many severe fires result from accrued management decisions. This skewing of agency motivation also distorts economic, insurance, and local regulatory incentives that influence development in fire-prone regions (*11*).

Although agencies are slow to reform internally, they may more rapidly respond to local stakeholder pressure. The core problem has been the lack of a public constituency that advocates for reform of fire-use practices (*11*). The benefits of greater fire use have been a difficult sell because of public objections to smoke and a negative perception of forest fires. This has begun to change as communities increasingly threatened by large fires are urging land-management agencies to accelerate fuel reduction efforts, including the use of managed fire (e.g., yosemitestanislaussolutions.com and 4FRI.org). Timber companies would also benefit from more fire-resilient landscapes in which their private lands are embedded. There is growing awareness that large, severe fires are inevitable in many dry forests, especially in a warming climate. Smoke, safety threats, fire intensity, and human health risks can be better managed for public benefit with proactive fire use under favorable weather and wind dispersal conditions (*12*).

EFFECTING CHANGE. Public support for expanded fire use could thus be directed toward revision of each NF plan, which provides standards and guidelines for daily management decisions. Plans can divide the landscape into zones for different fire management strategies, an approach used by Parks Canada. U.S. forest plans could zone areas close to homes (wildland-urban interface) as an area where most fuels reduction relies on mechanical thinning and fires are suppressed. Beyond this could be an intermediate area where prescribed fire and mechanical treatment are used to optimize fuels reduction. More remote forests could be intentionally burned with prescribed fire, or lightning ignitions allowed to burn as managed wildfires under moderate weather conditions.

Three of the first eight NFs to develop new plans have proposed that more than half of their area in the southern Sierra Nevada be zoned for prescribed and managed fire use. Over the next decade, most of the 155 NFs will begin writing new plans and holding public forums. Engaged local stakeholders will need to look beyond short-term impacts of fire use (e.g., smoke, limited access, and risk of escape) to support managers working with fire and challenge suppression in remote forest zones.

Public support of NCWFMS may help overcome reform disincentives by stressing national interagency collaboration. In response to decades of problem wildfires, the U.S. Congress passed the FLAME Act in 2009 requesting development of NCWFMS, a coordinated strategy to support landscape restoration and fire-adapted communities. Coordination is essential as large, intense wildfires often cross ownership boundaries. For example, in California's 2013 Rim Fire, large patches of old-growth trees in Yosemite National Park were killed when fuel-loaded forests on nearby NF land generated extreme fire behavior that crossed into the park (*13*). NCWFMS can exert peer pressure between agencies and provide support for tough decisions. To accomplish these changes, some policy and resource-deployment decisions supporting fire use could be made at the national level. In the United States, federal land agencies each fund their own fire crews but the National Interagency Fire Center (NIFC) coordinates resource deployment between agencies and nationally across geographic areas. Dedicated crews could be hired and trained for managed fire use, and NIFC could be charged with deploying them for beneficial burning (*14*). Some local and regional agencies have briefly created such crews, but they were often pulled into fire suppression when wildfire activity increased. By giving NIFC deployment authority, it could ensure that these crews are only used for working with fire and are available to burn when weather conditions are favorable. Optimal weather and smoke dispersal conditions occur even in heavily populated and regulated areas such as California, but many burn windows are missed because crews are at or being held for wildfire deployment (*9*). Air-quality regulations limit prescribed fires, although they have much lower emissions than the inevitable wildfire. The Environmental Protection Agency could consider treating prescribed fire smoke like wildfire, as an unregulated "exceptional event."

National government also has an incentive to reduce wildfire expenses and forest agencies' emergency fire borrowing. In many years, suppression costs consume 50% of agency annual budgets, which, after operating expenses, leaves little money for proactive fuels treatment or forest restoration (*11*). Costs and injuries, however, are much lower on managed fires than on escaped wildfires (*7*, *15*). The estimated cost savings for using managed fire compared with wildfire suppression over the same area (*15*) could be reported to Congress to highlight the economy of using proactive restoration rather than reactive triage.

Increased fire use will necessitate management changes (*16*). Mechanical fuels reduction could also be used not only for fire containment but also to establish safe-zone anchors to facilitate greater fire reintroduction (*8*). Large prescribed burns commonly used in Western Australia are possible because a network of these anchors allows 6 to 8% of the forest to be burned annually (*16*). Australian foresters make substantial efforts to educate the public about the inevitability of fire and its ecological benefits and to build support for fire use and smoke tolerance.

We will not eliminate wildfire, but public support for proactive use of managed fires can help restore millions of hectares of forest ecosystems. ■

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Species Status Assessment Report
for the
Rio Grande Cutthroat Trout

Photo courtesy of Colorado Parks and Wildlife

August 2014

U.S. Fish and Wildlife Service
Region 2
Albuquerque, NM

This document was prepared by Susan Oetker and Nathan Allan with assistance from the rest of the U.S. Fish and Wildlife Service's Rio Grande Cutthroat Trout Listing Team (Creed Clayton, George Dennis, Carey Galst, Chris Kitcheyan, Frank Lupo, Melissa Mata, Wally Murphy, Sarah Quamme, and Justin Shoemaker). We also received assistance from the Service's Greg Breese and from David Smith and Conor McGowan of the U.S. Geological Survey.

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U.S. Fish and Wildlife Service. 2014. Species status assessment report for the Rio Grande cutthroat trout. Albuquerque, NM.

EXECUTIVE SUMMARY

This species status assessment reports the results of the comprehensive status review for the Rio Grande cutthroat trout (*Oncorhynchus clarkii virginalis*) and provides a thorough account of the subspecies' overall viability and thus extinction risk. Rio Grande cutthroat trout (a subspecies of cutthroat trout) inhabit high elevation streams in New Mexico and southern Colorado where they need clear, cold, highly oxygenated water, clean gravel substrates, a network of pools and runs, and an abundance of food (typically aquatic and terrestrial invertebrates) to complete their life history.

The Rio Grande cutthroat trout needs multiple resilient populations widely distributed across its range to maintain its persistence into the future and to avoid extinction. Resilient populations require long continuous suitable stream habitats to support large numbers of individuals and to withstand stochastic events; the populations should be free from the impacts of nonnative trout. The resilient populations should be distributed in each of the four Geographic Management Units (GMUs) where the subspecies currently occurs. This distributional pattern will provide for the needed redundancy and representation to increase the probability that the subspecies will withstand future catastrophic events and maintain future adaptive capacity in terms of genetic and ecological diversity. The population and subspecies-level needs for viability of the Rio Grande cutthroat trout are summarized in column 2 of Table ES-1. The likelihood of the Rio Grande cutthroat trout's persistence depends upon the number of populations, its resilience to threats, and its distribution. As we consider the future viability of the subspecies, more populations with greater resiliency and wider geographic distributions are associated with higher overall subspecies viability.

The Rio Grande cutthroat trout historically occurred in New Mexico and southern Colorado. Its distribution has been divided into GMUs reflecting major hydrologic divisions. The subspecies no longer occurs in one GMU, the Caballo GMU, where only one population was historically known. The remaining four GMUs are managed by the States of Colorado and New Mexico and other agencies as separate units to maintain genetic and ecological diversity within the subspecies where it exists and to ensure representation of the subspecies across its historical range. GMUs were not created to necessarily reflect important differences in genetic variability, although fish in the Pecos and Canadian GMUs do exhibit some genetic differentiation from those in the Rio Grande basin GMUs. From a rangewide perspective, multiple Rio Grande cutthroat trout populations should be dispersed throughout the various GMUs to maintain subspecies viability, reduce the likelihood of extinction, and provide the subspecies with redundancy.

Table ES-1. Overall summary of species status assessment for Rio Grande cutthroat trout. ("RG" = Rio Grande, "pop's" = populations)

3 R's	NEEDS	CURRENT CONDITION	FUTURE CONDITION (VIABILITY)
Resiliency: Population ***(large populations to withstand stochastic events)***	• Large Effective Population Sizes (effective population sizes >500 are best). • Long Streams for Habitat (streams greater than 9.65 km are best). • Free of Nonnative Trout (mainly rainbow and brown trout) and Disease (whirling). • High Quality Habitat (water temps < critical summer maximums).	• 122 Extant Populations across range. * 55 (45%) of populations are currently in the best or good condition *(based on absense of nonnative trout, effective population size, and occuppied stream length)* * 67 (55%) of populations are currently in fair or poor condition.	• Status assessment model estimates probability of persistence for each population based on risks from: * Effective Population Size. * Nonnatives (hybridization, competition) and Disease. * Wildfire and Stream Drying . * Water Temperature Increase. • Included climate change considerations for increased risks.
Resiliency: Subspecies ***(populations to withstand stochastic events)***	• Multiple interconnected resilient populations.	• About 11% of historic range remains occupied due to past impacts from nonnatives. • Populations are isolated (16 populations have some connectedness).	• 2080 model forecasts future populations persisting; results range depending on future management level and severity of climate change: reporting **best to worst (intermediate)** results: * **50 to 132 (69)** populations rangewide. • Limited opportunity to regain interconnectedness of populations (due to pervasive nonnative trout).
Redundancy ***(number and distribution of populations to withstand catostrophic events)***	• Multiple highly resilient populations within each of the 4 Geographic Management Units (GMUs).	• Current total number of populations persisting by GMU: * 41 pop's in RG Headwaters GMU. * 59 pop's in Lower RG GMU. * 10 pop's in Canadian GMU. * 12 pop's in Pecos GMU.	• 2080 model forecasts for future populations persisting by GMU: * **21 to 55 (27)** pop's in RG Headwaters. * **21 to 47 (28)** pop's in Lower RG. * **3 to 14 (6)** pop's in Canadian. * **5 to 16 (8)** pop's in Pecos.
Representation ***(genetic and ecological diverstiy to maintain adaptive potential)***	• Genetic variation exists between 1) Two GMUs in the Rio Grande Basin and 2) Two GMUs in Canadian and Pecos River Basins. • Unknown ecological variation, but we used GMUs as proxy.	• Current total populations persisting by Watershed: * 100 pop's in Rio Grande Basin. * 22 pop's in Candadian and Pecos GMUs.	• 2080 model forecasts for future populations persisting by watershed: * **42 to 102 (55)** pop's in Rio Grande Basin. * **8 to 30 (14)** pop's in Canadian and Pecos GMUs.

Currently the subspecies is distributed in 122 populations across the four extant GMUs (ranging from 10 to 59 populations per GMU), and most of the populations are isolated from other populations. The total amount of currently occupied stream habitat is estimated to be about 11% of the historically occupied range. This large decline in distribution and abundance is primarily due to the impacts of the introduction of nonnative trout. Nonnative rainbow trout (*O. mykiss*) and other nonnative subspecies of cutthroat trout have invaded most of the historical range of the Rio Grande cutthroat trout and resulted in their extirpation because the nonnative trout readily hybridize with Rio Grande cutthroat trout. In addition, brown trout (*Salmo trutta*) and brook trout (*Salvelinus fontinalis*) have also displaced Rio Grande cutthroat trout in some historical habitats through competition and predation pressures. We evaluated the current condition of the 122 populations and categorized the condition of each population based on the absence of nonnative trout, the effective population size, and the occupied stream length. Fifty-five populations were in either the "best" or "good" condition in this categorization. Table ES-2 identifies the number populations placed in each category by GMU (see Chapter 3 for a description of the categories).

Table ES-2. Current status of Rio Grande cutthroat trout showing the number of current conservation populations in 4 categories by GMU. The percentages (%) are the proportion of total populations within each GMU.

Populations per GMU	Best	%	Good	%	Fair	%	Poor	%	Total
Canadian	1	10%	3	30%	5	50%	1	10%	10
Rio Grande Headwaters	5	12%	14	34%	20	49%	2	5%	41
Lower Rio Grande	13	22%	15	25%	20	34%	11	19%	59
Pecos	1	8%	3	25%	7	33%	1	42%	12
Rangewide	20	16%	35	29%	52	43%	15	12%	122

We next reviewed the past, current, and future factors that could affect the persistence of Rio Grande cutthroat trout populations. Seven risk factors were evaluated in detail to estimate their individual and cumulative contributions to the overall risk to the subspecies' viability. We focused on these seven factors because they were found to potentially have population-level effects on the subspecies. The seven factors were:

(1) **Demographic Risk:** Small population sizes are at greater risk from inbreeding, demographic fluctuations, and reduced genetic diversity, and they are more vulnerable to extirpation from other risk factors.

(2) Hybridizing Nonnative Trout: Nonnative rainbow and other cutthroat trout subspecies have historically been introduced throughout the range of Rio Grande cutthroat trout for recreational angling, and they are known to readily hybridize with Rio Grande cutthroat trout. Climate change may exacerbate this risk factor as warmer waters may make high elevation habitats more susceptible to invasion by rainbow trout.

(3) Competing Nonnative Trout: Brook and brown trout compete with Rio Grande cutthroat trout for food and space, and larger adults will prey upon young Rio Grande cutthroat trout.

(4) Wildfire: Ash and debris flows that occur after a wildfire can eliminate populations of fish from a stream, and wildfires within the range of Rio Grande cutthroat trout have depressed or eliminated fish populations. As drought frequency increases due to climate change, dry forests are more likely to burn and burn hotter than they have in the past.

(5) Stream Drying: Drying of streams occupied by Rio Grande cutthroat trout may occur as a result of drought or, in a few cases, water withdrawals. Drought frequency is expected to increase as a result of climate change due to a combination of increased summer temperatures and decreased precipitation.

(6) Disease: Whirling disease damages cartilage, killing young fish or causing infected fish to swim in an uncontrolled whirling motion, making it impossible to avoid predation or feed.

(7) Water Temperature Changes: Changes in air temperature and precipitation patterns expected from climate change could result in elevated stream temperatures that make habitat unsuitable for Rio Grande cutthroat trout to complete their life history.

We considered other potential factors as well, including hydrologic changes related to future climate change, effects to habitat related to land management, and angling. Our review of the best available information did not demonstrate a relationship between hydrologic changes and the effects on the subspecies to allow for reasonably reliable conclusions; therefore, we did not consider that factor further. We found that land management activities are not likely to have a measurable population-level effect on the subspecies, and angling was also not found to be a substantial factor affecting the subspecies. Therefore, these factors were not evaluated further in our analysis.

We included future management actions as an important part of our overall assessment. The Rio Grande Cutthroat Trout Rangewide Conservation Team (Conservation Team) is composed of biologists from Colorado Parks and Wildlife (CPW), New Mexico Department of Game and Fish (NMDGF), U.S. Bureau of Land Management (BLM), U.S. Forest Service (USFS), National Park Service (NPS), Mescalero Apache Nation, Jicarilla Apache Nation, Taos Pueblo, and the Service. The Conservation Team developed the Conservation Agreement and Strategy in 2013 (revised from the previous Conservation Agreements in 2003 and 2009), which formalized many ongoing management actions. The Conservation Agreement and Strategy includes activities such as stream restorations, barrier construction and maintenance, nonnative species removals, habitat improvements, public outreach, and database management. Over the 10-year life of the Conservation Agreement and Strategy, the Conservation Team has committed to restoration of between 11 and 20 previously extirpated Rio Grande cutthroat trout populations to historical habitat. We included these activities in our analysis of the future status of the subspecies over the next 10 years and projected various scenarios of active management beyond that.

We developed a species status assessment model to quantitatively incorporate the risks of extirpation from the seven risk factors listed above (including cumulative effects) in order to estimate the future probability of persistence of each extant population of Rio Grande cutthroat trout. We used this model to forecast the future status of the Rio Grande cutthroat trout in a way that addresses viability in terms of the subspecies' resiliency, redundancy, and representation. As a result we developed two distinct modules. Module 1 estimates the probability of persistence for each Rio Grande trout population by GMU for 3 time periods (2023, 2040, and 2080) under a range of conditions, and Module 2 estimates the number of surviving populations by GMU for three time periods under several scenarios related to future management actions and the effects of climate change. A detailed explanation of the methodology used to the develop the model is provided in Appendix C, and the results are summarized in Chapter 5. The results of the analysis for three scenarios in 2080 are listed in Column 4 of Table ES-1.

We used the results of this analysis to describe the Rio Grande cutthroat trout viability (viability is the ability of a species to persist over time and thus avoid extinction; "persist" means that the species is expected to sustain populations in the wild beyond the end of a specified time period) by characterizing the status of the species in terms of its resiliency, redundancy, and representation.

Resiliency is having sufficiently large populations for the subspecies to withstand stochastic events. We measured resiliency at the population scale for the Rio Grande cutthroat trout by quantifying the persistence probability of each extant population under a range of assumed conditions. As expected, because the status assessment model was developed to forecast linearly increasing risks over time, all of the population persistence probabilities decrease in our three time periods. Our results do not necessarily mean that any one population will, in fact, be extirpated by 2080; they simply reflect the risks that we believe the populations face due to their current conditions and the risk factors influencing their resiliency.

Rangewide, the resiliency of the subspecies has declined substantially due to the large decrease in overall distribution in the last 50 years. In addition, the remnant Rio Grande cutthroat trout populations are now mostly isolated to headwater streams due to the fragmentation that has resulted from the historical, widespread introduction of nonnative trout across the range of Rio Grande cutthroat trout. Therefore, if an extant population is extirpated due to a localized event, such as a wildfire and subsequent debris flow, there is little to no opportunity for natural recolonization of that population. This reduction in resiliency results in a lower probability of persistence for the subspecies as a whole. To describe the remaining resiliency of the subspecies, we evaluated the individual populations in detail to understand the subspecies' overall capacity to withstand stochastic events.

Redundancy is having a sufficient number of populations for the subspecies to withstand catastrophic events. For the Rio Grande cutthroat trout, we measured redundancy based on our forecasting of the number of populations persisting across the subspecies' range. The results suggest that, depending on the particular scenario related to risk factors and restoration efforts, the overall number of populations may decline to some extent by 2080 (see Table ES-1, Column 4). We are focusing on the estimates for 2080, because if the subspecies has sufficient redundancy by 2080, it will also have sufficient redundancy in the more recent time periods. Rangewide there are currently 122 populations, and we forecast between 50 and 132 populations surviving in 2080 (with an intermediate forecast of 68 populations). The wide range in the estimated number of surviving populations is due to the various projections of management and climate change intensity. Some GMUs may decline more than others; for example, our forecasts suggest the Lower Rio Grande GMU may have the largest decline. We estimate the current 59 populations in this GMU could be between 21 and 47 populations by 2080 (with an intermediate forecast of 28 populations). The GMU with the least populations, the Canadian GMU, is forecasted to change from 10 current populations to between 3 and 14 populations by 2080 (with an intermediate forecast of 6 populations).

Representation is having the breadth of genetic and ecological diversity of the subspecies to adapt to changing environmental conditions. For the Rio Grande cutthroat trout, we evaluated representation based on the extent of the geographical range expected to be maintained in the future as indicated by the populations occurring within each GMU for a measure of ecological diversity. For genetic diversity, there are important genetic differences between the Rio Grande basin populations and the populations in the Canadian and Pecos GMUs (though the Pecos and Canadian GMUs are not genetically different from each other). The variation in persistence probabilities is distributed across the GMU so that none of the risk is particularly associated with any particular geographic area within the GMU. Combined, the Canadian and Pecos GMUs are forecasted to have 8 to 30 populations surviving in 2080 (with an intermediate forecast of 14 populations).

We used the best available information to forecast the likely future condition of the Rio Grande cutthroat trout. Our goal was to describe the viability of the subspecies quantitatively in a way that characterizes the needs of the subspecies in terms of resiliency, redundancy, and representation. We considered the possible future condition of the subspecies out to about 65 years from the present. We considered nine different scenarios that spanned a range of potential conditions that we believe are important influences on the status of the subspecies. Our results describe a range of possible conditions in terms of the probability of persistence of individual populations across the GMUs and a forecast of the number of populations surviving in each GMU.

None of our "worst case scenario" forecasts result in a predicted loss of all of the populations within any of the GMUs. Therefore, at a minimum, our results suggest the subspecies will have persisting populations in 2080 across its range. Most of the scenarios generally show a declining persistence and number of populations over time. However, the rate of this decline, or whether it occurs at all, depends largely on the likelihood of future management actions occurring, the most important of which are the future restoration and reintroduction of populations within the historical range and the control of nonnative trout. While other factors are important to each population, the future management actions will probably determine the future viability of the Rio Grande cutthroat trout.

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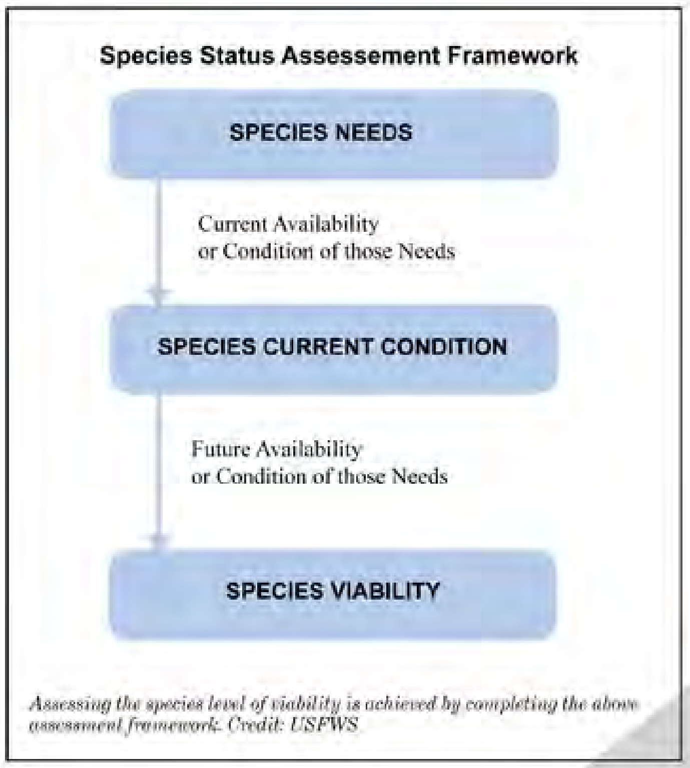
Chapter 1. Introduction

The Rio Grande cutthroat trout (*Oncorhynchus clarkii virginalis*) lives in high elevation, coldwater streams in New Mexico and southern Colorado. It is a subspecies that was made a candidate for listing in 2008 by the U. S. Fish and Wildlife Service (Service) under the Endangered Species Act of 1973, as amended (Act) (73 FR 27900, May 14, 2008). It is now being reviewed for listing as a threatened or endangered species under the Act. This Rio Grande cutthroat trout Species Status Assessment Report (SSA Report) is a summary of the information assembled and reviewed by the Service and incorporates the best scientific and commercial data available. This SSA Report documents the results of the comprehensive status review for the Rio Grande cutthroat trout.

The Service is engaged in a number of efforts to improve the implementation of the Act (see www.fws.gov/endangered/improving_ESA). The priority of the Service is to make implementation of the Act less complex, less contentious, and more effective. As part of this effort, our Endangered Species Program has begun to develop a new framework to guide how we assess the biological status of species (and in this case, subspecies). Because biological status assessments are frequently used in all of our Endangered Species Program areas, developing a single, scientifically sound document is more efficient than compiling separate documents for use in our listing, recovery, and consultation programs. For example, much of the information we gather on species needs within an assessment can provide a basis for recovery criteria during recovery planning. Moreover, we can also use the analysis of risks a species is facing to conduct endangered species consultations, particularly if we determine how conservation measures could be employed to minimize or avoid effects of a proposed action. Therefore, we have developed the following SSA Report that contains summary information regarding life history, biology, and consideration of current and future risk factors facing the Rio Grande cutthroat trout.



Assessing the species level of viability is achieved by completing the above assessment framework. Credit: USFWS

The objective of the SSA is to thoroughly describe the viability of the Rio Grande cutthroat trout. Through this description, we will determine what the subspecies needs to remain viable, its current condition in terms of those needs, and its forecasted future condition. In conducting this analysis we take into consideration the changes that are happening in the environment – past, current, and future – to help us understand what factors drive the viability of the subspecies.

For the purpose of this assessment, we define **viability** as a description of the ability of a species (in this case subspecies) to persist over time and thus avoid extinction. "**Persist**" and "**avoid extinction**" mean that the species is expected to sustain populations in the wild beyond the end of a specified time period. Using the SSA framework, we consider what the species needs to maintain viability by characterizing the status of the species in terms of its **resiliency**, **redundancy**, and **representation**.

• **Resiliency** is having sufficiently large populations for the subspecies to withstand stochastic events. Stochastic events are those arising from random factors such as weather, flooding, or fire. We can measure resiliency based on metrics of population health; in the case of the Rio Grande cutthroat trout, population size, habitat size, and freedom from nonnative trout species are primary indicators of resiliency. Resilient populations are better able to withstand disturbances such as random fluctuations in birth rates (demographic stochasticity), variations in rainfall (environmental stochasticity), or the effects of wildfire.

• **Redundancy** is having a sufficient number of populations for the subspecies to withstand catastrophic events. A catastrophic event is defined here as a rare destructive event or episode involving many populations and occurring suddenly. Redundancy is about spreading risk and can be measured through the duplication and broad distribution of resilient populations across the range of the subspecies. The more resilient populations the subspecies has, distributed over a larger landscape area, the better chances that the subspecies can withstand catastrophic events. For the Rio Grande cutthroat trout, we measure redundancy based on the number of populations persisting across the subspecies' range.

• **Representation** is having the breadth of genetic makeup of the subspecies to adapt to changing environmental conditions. Representation can be measured through the genetic diversity within and among populations and the ecological diversity (also called environmental variation or diversity) of populations across the subspecies' range. The more representation, or diversity, the subspecies has, the more it is capable of adapting to changes (natural or human caused) in its environment. In the case of the Rio Grande cutthroat trout, we evaluate representation based on the extent of the geographical range measured by the populations occurring within Geographic Management Units (GMUs) (see 3.1 Historical Range and Distribution for more information about GMUs) as an indicator of genetic or ecological diversity.

To evaluate the viability of the Rio Grande cutthroat trout both currently and into the future we assessed a range of conditions to allow us to consider the subspecies' resiliency, redundancy, and representation. This SSA Report provides a summary assessment of Rio Grande cutthroat trout biology and natural history and assesses the risks to its future viability. Herein, we summarize biological data and a description of past, present, and likely future risk factors facing the Rio Grande cutthroat trout.

The format for this SSA Report includes: (1) the resource needs of individuals (Chapter 2); (2) the Rio Grande cutthroat trout's historical distribution and a framework for what the subspecies needs in terms of the number and distribution of resilient populations across its range for subspecies viability (Chapter 3); (3) reviewing the likely causes of the current and future status of the subspecies, and determining which of these risk factors affect the subspecies' viability and to what degree (Chapter 4); and (4) concluding with a quantitative description of the viability in terms of resiliency, redundancy, and representation (Chapter 5). This document is a compilation of the best available scientific and commercial information and a description of past, present, and likely future threats to the Rio Grande cutthroat trout.

For a glossary of some of the terms used in this SSA Report, reference Appendix A. The detailed analysis of risk factors summarized in Chapter 4 is found in Appendix B. Finally, we conducted an analysis to quantitatively characterize the viability of the Rio Grande cutthroat trout as described in Appendix C. Our objectives for this Status Assessment Model were twofold: (1) to estimate the probability of persistence of each extant Rio Grande cutthroat trout population over time; and (2) to describe the future persistence of Rio Grande cutthroat trout by forecasting the number of populations expected to persist across the subspecies' range over time. Finally, the literature cited in this SSA Report is in Appendix D[1].

We primarily used information from the Rio Grande cutthroat trout rangewide database (RGCT Database) from 2013, which includes data from 2012. This is the most recent database available (see section 2.5, Management History of Rio Grande Cutthroat Trout, for more information about the RGCT Database). We supplemented information from the RGCT Database based on new information received from various sources, including communications with Rio Grande cutthroat trout biologists from the states of Colorado and New Mexico. We also relied heavily on the prior work completed for the most recent rangewide assessment for the Rio Grande cutthroat trout (Alves *et al.* 2008).

Importantly, this SSA Report does not result in, or predetermine, a decision by the Service on whether the Rio Grande cutthroat trout warrants protections of the Act, or whether it should be proposed for listing as a threatened or endangered species under the Act. That decision will be made by the Service after reviewing this document, along with the supporting analysis, other relevant scientific information, and all applicable laws, regulations, and policies, and the results of the decision will be announced in the *Federal Register*. Instead, this SSA Report provides a strictly scientific review of the available information related to the biological status of the Rio Grande cutthroat trout.

[1] We did not cite every report and information source that was reviewed for this assessment. Only the cited sources are referenced in this SSA Report.

Chapter 2. Individual Needs: Life History and Biology

In this chapter we provide basic biological information about the Rio Grande cutthroat trout, including its taxonomic history, morphological description, and known life history traits. We then outline the resource needs of individuals and populations of the Rio Grande cutthroat trout. There are numerous sources of information on Rio Grande cutthroat trout life history and biology (*e.g.,* Cowley 1993; Behnke 2002; New Mexico Department of Game and Fish (NMDGF) 2002; Pritchard and Cowley 2006). Here we report those aspects of the subspecies' life history that are important to our analysis. Finally, we discuss the management history of the subspecies.

2.1 Taxonomy

In 1541, Francisco de Coronado's expedition in the upper Pecos River discovered Rio Grande cutthroat trout, one of 14 subspecies of cutthroat trout (Behnke 2002, p. 207). Figure 1 shows a generalized range of Rio Grande cutthroat trout, as well as other proximal cutthroat trout subspecies and other native trout.

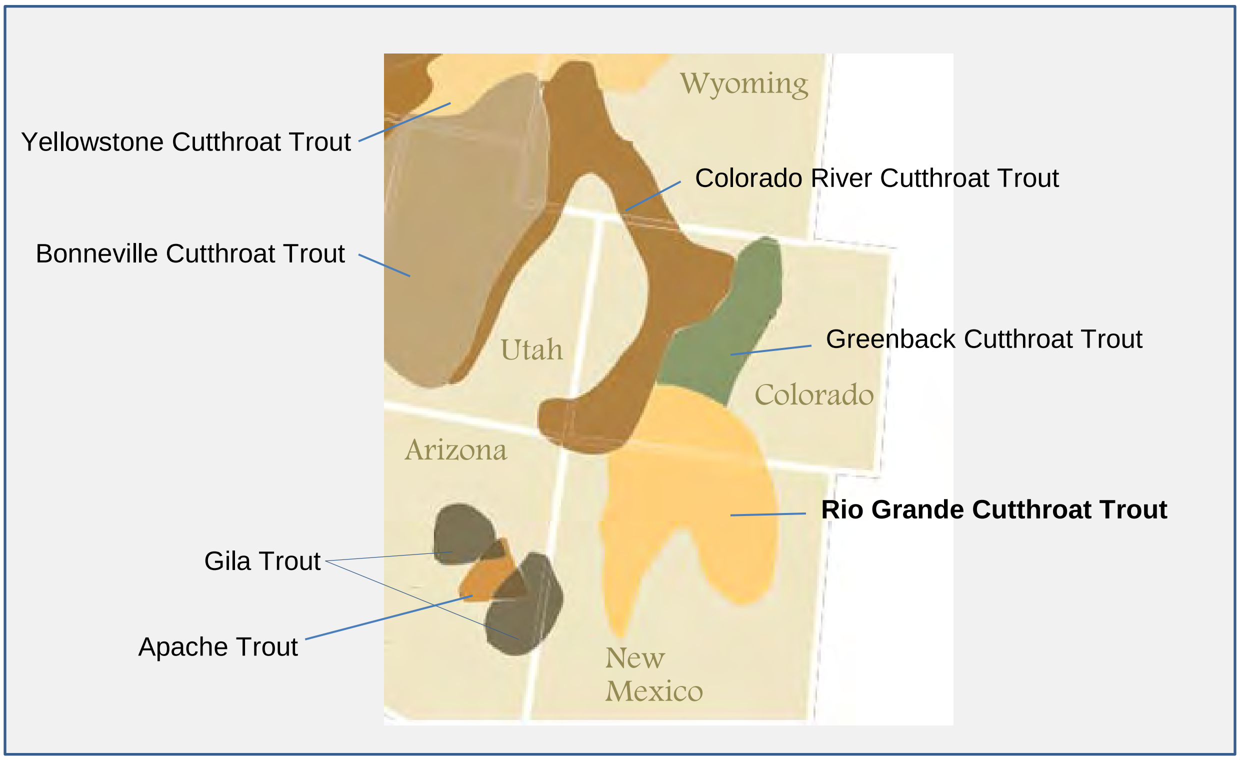


Figure 1. Generalized map of Rio Grande cutthroat trout distribution in relation to other nearby cutthroat trout subspecies, as well as other species of native trout. Map adapted from Western Native Trout Initiative 2008.

The first specimens that were collected for scientific purposes came from Ute Creek in Costilla County, Colorado, in 1853. Rio Grande cutthroat trout was originally described in 1856 (Behnke 2002, p. 210). The currently accepted subspecies classification is:

Class: Actinopterygii
Order: Salmoniformes
Family: Salmonidae
Species: *Oncorhynchus clarkii virginalis* Girard, 1856

2.2 Subspecies Description

Cutthroat trout are distinguished by the red to orange slashes in the folds beneath the lower jaw (Behnke 2002, p. 139) (Figure 2). Rio Grande cutthroat trout have irregular shaped spots that are concentrated behind the dorsal fin, smaller less numerous spots located primarily above the lateral line in front of the dorsal fin, and basibranchial teeth that are minute or absent (Sublette *et al.* 1990, p.53; Behnke 2002, p. 207). Rio Grande cutthroat trout are light rose to red-orange on the sides and pink or yellow-orange on the belly (Behnke 2002, p. 207).

Figure 2. Rio Grande cutthroat trout from the Lake Fork Conejos River, Colorado. Photo courtesy of Colorado Parks and Wildlife.

2.3 Life History

Rio Grande cutthroat trout exhibit a life history similar to other cutthroat trout subspecies. Adults spawn as high water flows from snowmelt recede, which typically occurs from the middle of May to the middle of June (NMDGF 2002, p. 17). Spawning is believed to be tied to day length, water temperature, and runoff (Sublette *et al*. 1990, p. 54; Behnke 2002, p. 141). It is unknown if Rio Grande cutthroat trout spawn every year or if some portion of the population spawns every other year as has been recorded for westslope cutthroat trout (*O. c. lewisi*) (McIntyre and Rieman 1995, p. 1). Likewise, while it is assumed that females mature at age 3, they may not spawn until age 4 or 5 as seen in westslope cutthroat trout (McIntyre and Rieman 1995, p. 3). Individuals greater than 120 millimeters (mm) (4.7 inches (in)) are considered adults (Pritchard and Cowley 2006, p. 25). Adults have been observed as old as 8 years (Pritchard and Cowley 2006, p. 30).

A female constructs the nest (redd) just prior to spawning and deposits 200 – 4,500 eggs in it, which are then fertilized by a male (Cowley 1993, p. 3). Rio Grande cutthroat trout do not exhibit parental care of the redd or young. Depending on water temperature, the eggs hatch within 3 – 7 weeks (Prichard and Cowley 2006, p. 26). The hatchlings remain within the gravel of the redd for several weeks until the yolk sac is absorbed (Pritchard and Cowley 2006, p. 26). Sex ratio also is unknown with certainty, but based on field data, a ratio skewed towards more females might be expected (Pritchard and Cowley 2006, p. 27).

Although Yellowstone (*O. c. bouvieri*) (Gresswell 1995, p. 36), Bonneville (*O. c. utah*) (Schrank and Rahel 2004, p. 1532), and westslope (Bjornn and Mallet 1964, p. 73; McIntyre and Rieman 1995, p. 3) cutthroat trout subspecies are known to have a migratory life history phase, in which the trout will move between lakes and rivers, it is not known if Rio Grande cutthroat trout once had a migratory form when there was connectivity among watersheds. There are no migratory populations today.

Most cutthroat trout are opportunistic feeders, eating both aquatic invertebrates and terrestrial insects that fall into the water (Sublette *et al*. 1990, p. 54). As individuals grow they may exhibit more benthic feeding (Pritchard and Cowley 2006, p. 25). Cutthroat trout subspecies generally become more piscivorous (fish eating) as they mature (Sublette *et al.* 1990, p. 54). Growth of cutthroat trout varies with water temperature and availability of food. Because most populations of Rio Grande cutthroat trout are currently found in high elevation streams, growth may be relatively slow and time to maturity may take longer than is seen in subspecies that inhabit lower elevation, warmer streams.

2.4 Resource Needs (Habitat) of Individuals

As is true of other subspecies of cutthroat trout, Rio Grande cutthroat trout are found in clear, cold, high elevation streams. Much of what is known of Rio Grande cutthroat trout life history is from studies of other cutthroat trout subspecies, and we presume that this knowledge applies to Rio Grande cutthroat trout. Rio Grande cutthroat trout require several types of habitat for survival: spawning habitat, nursery or rearing habitat, adult habitat, and refugial habitat (organized by life stage in Table 1). Rio Grande cutthroat trout spawn as floods from snowmelt runoff recede. Spawning habitat is found in areas exposed to flowing water with clean gravel (little or no fine sediment present) that ranges between 6 – 40 millimeters (mm) (0.24 – 1.6 inches (in)) in diameter (NMDGF 2002, p. 17; Budy *et al.* 2012, p. 437, 447) where redds are formed (Cowley 1993, p. 3). Embryonic development of cutthroat trout within eggs requires flowing water with high oxygen levels (Cowley 1993, p. 3; Budy *et al*. 2012, p. 437). Fry emerge after yolk absorption and at a length of about 20 mm (0.8 in) (McIntyre and Rieman 1995, p. 2).

Following emergence, cutthroat trout fry move to nursery habitat, usually stream margins, backwaters, or side channels where water velocity is low and water temperature is slightly warmer (Pritchard and Cowley 2006, pp. 17–18). Drifting and benthic invertebrates, upon which trout feed, are frequently numerous in such areas (Pritchard and Cowley 2006, p. 18). Fry establish individual territories in these habitats, generally near a source of cover such as aquatic plants or overhanging vegetation, and remain in them for several months (Pritchard and Cowley

2006, p. 18). Juvenile cutthroat trout use stream substrate as cover during winter (McIntyre and Rieman 1995, p. 4). Water temperature is important for juvenile survival; streams with mean daily temperatures in July of less than 7.8 degrees Celsius (ºC) (46 degrees Fahrenheit (ºF)) may not have successful reproduction or recruitment (survival of individuals to sexual maturity and joining the reproductive population) in most years (Harig and Fausch 2002, pp. 542, 543; Coleman and Fausch 2007a, p. 1241; Coleman and Fausch 2007b, p. 651). Recent studies have shown that Rio Grande cutthroat trout have similar thermal tolerances as other subspecies of cutthroat trout. When water temperatures mimic natural daily fluctuations (warmer during the day, cooler at night), Rio Grande cutthroat trout can tolerate up to 25 ºC (77 ºF) (Zeigler *et al.* 2013a, p. 1400). Chronic effects of high temperatures, such as declining growth rates of individuals, have been observed when 30-day average temperatures exceed 18 ºC (64 ºF) (Zeigler *et al.* 2013a, p. 1400).

As Rio Grande cutthroat trout grow, they move back into the main stream channel. Older individuals primarily use pools with cover and riffles for foraging (Pritchard and Cowley 2006, p. 18). Deep pools that do not freeze in the winter and do not dry in the summer or during periods of drought provide refugia. Lack of large pools may be a limiting factor in headwater streams (Harig and Fausch 2002, p. 543). Refugial habitat may also be a downstream reach of stream or a connected adjacent stream that has maintained suitable habitat in spite of adverse conditions that eliminated or reduced habitat from the rest of the stream. For populations to persist, Rio Grande cutthroat trout must be able to disperse to and from these habitats (Fausch *et al.* 2002, p. 494).

Table 1. Known habitat needs of Rio Grande cutthroat trout by life stage.

Life Stage	Resource Needs (Habitat)	References
Eggs – Emergence of Fry - **May to June**	• Flowing water (mean water column velocities between 0.11–0.90 m/sec, with optimal velocities between 0.30 – 0.60 m/sec), with clean gravel (6 – 40 mm diameter) • Water with high dissolved oxygen levels (>7 milligrams per liter (mg/L) at ≤15 °C and ≥9 mg/L at >15 °C). • Water temperature between 6 – 17 °C (43 – 63 °F), optimal 10 °C (50 °F)	NMDGF 2002, p. 17 Budy *et al.* 2012, p. 437, 447 Budy *et al.* 2012, p. 437 Budy *et al.* 2012, p. 437, 446 Zeigler *et al.* 2013a, p. 1399
Fry - **summer through fall**	• Stream margins, backwaters, or side channels. • Benthic invertebrates • Low water velocities • Water temperatures above 7.8 °C (46 °F)	Pritchard and Cowley 2006, pp. 17 – 18 Cowley 1993, p. 3 Sublette *et al.* 1990, p. 4 Harig and Fausch 2002, pp. 542, 543 Coleman and Fausch 2007a, p. 1241 Coleman and Fausch 2007b, p. 651
Juveniles (<120 mm Total Length (TL)) - **Year 1-2**	• Mean water temperatures ideally >7.8 °C (46 °F) and <18 °C (64 °F) • Instream cover for winter	Harig and Fausch 2002, pp. 542, 543 Gard 1963, p. 197 Zeigler *et al.* 2013a, p. 1400 Coleman and Fausch 2007a, p. 1241 Coleman and Fausch 2007b, p. 651
Adults (>120 mm TL) - **Year ~3+**	• Deep water pools (> 30 centimeters (cm) (12 inches (in)) • Prey in the form of invertebrates and in some cases small fish • Mean water temperatures ideally >7.8 °C (46 °F) and <18°C (64 °F)	Harig and Fausch 2002, p. 543 Young *et al.* 2005, p. 2402 Pritchard and Cowley 2006, p. 18 Zeigler *et al.* 2013a, p. 1400

2.5 Management History of Rio Grande Cutthroat Trout

Cooperative efforts between New Mexico, Colorado, Federal agencies, Tribes, and nongovernmental organizations (NGOs) to manage and conserve Rio Grande cutthroat trout have been ongoing for decades. Due in large part to interest in the Rio Grande cutthroat trout for recreational angling, the States of New Mexico and Colorado have long had an interest in managing populations and conducting research on the subspecies, and they have led management efforts for many years to restore populations and improve habitat. In 2003, the first Conservation Agreement was signed, and the Rio Grande Cutthroat Trout Conservation Team was formed. This Team is comprised of representatives of the signatory agencies (including States, Federal agencies, Tribes, and NGOs), as well as members of academia. The Conservation Team developed the RGCT Database, which houses all data collected on populations, including management actions, surveys, and other information. The Conservation Agreement was renewed in 2009 and again in 2013. Further, in 2008 the Conservation Team released its Status Assessment of the Rio Grande Cutthroat Trout (Alves *et al*. 2008), which summarized the current conditions of the trout, based on information from the RGCT Database. We rely on information from this Status Assessment often throughout this SSA Report.

In 2013, the Conservation Team developed the Rio Grande Cutthroat Trout Conservation Strategy (RGCT Conservation Team 2013), which is a signed 10-year commitment to implement ongoing conservation actions. The development of this Strategy was directed by the Conservation Agreement. These actions include reintroduction of 11 – 20 populations, habitat improvement, barrier construction and maintenance, and nonnative fish removals. Annual coordination meetings will continue to occur to review prior actions and to plan upcoming actions. These actions take place rangewide across all GMUs.

Also in 2013, Vermejo Park Ranch signed a Candidate Conservation Agreement with Assurances (Vermejo CCAA) with the Service and the States of Colorado and New Mexico. When completed, this project is expected to increase occupied stream miles by approximately 20% and create a large, interconnected population of over 75,000 individuals throughout over 100 stream miles (Kruse 2013, p. 2). The project is currently 50% completed and is expected to be fully completed by 2020.

Chapter 3. Population and Subspecies Needs and Current Conditions

In this chapter we consider the Rio Grande cutthroat trout's historical distribution and what the subspecies needs in terms of the number and distribution of resilient populations across its range for the subspecies as a whole to be viable. We first review the historical information on the range and distribution of populations of the subspecies. We next review the conceptual needs of the subspecies, including population resiliency, redundancy, and representation to maintain viability and reduce the likelihood of extinction. Finally, we consider the current conditions of all Rio Grande cutthroat trout populations rangewide.

3.1 Historical Range and Distribution

Rio Grande cutthroat trout are generally assumed to have occupied all streams capable of supporting trout in the Rio Grande, Pecos, and Canadian basins (Alves *et al*. 2007, p. 9). The Pecos River is a tributary of the Rio Grande, so a historical connection between the two basins likely existed. Although no early museum specimens document its occurrence in the headwaters of the Canadian River, there is no evidence of human introduction and so it is almost certainly native there as well (Behnke 2002, p. 208; Pritchard *et al.* 2009, p. 1219). The Canadian River, which drains to the Mississippi River basin, has no connection with the Rio Grande. It is possible that through headwater capture (a tributary from one watershed joins with a tributary from another) there may have been natural migration of fish between the Pecos and Canadian headwater streams. Because there are Rio Grande cutthroat trout populations throughout the headwaters of the Rio Grande basin, historically, these fish most likely dispersed through the Rio Grande into the tributary streams.

There is some possibility that Rio Grande cutthroat trout may have occurred in the Pecos River basin in Texas (Behnke 1967, pp. 5, 6; Garrett and Matlock 1991, p. 404) and the Rio Grande basin in Mexico (Behnke 1967, p. 4). However, no specimens were collected to document their presence in these locations with certainty. Their potential occupancy in these locations is based on fluvial connections and on historical articles that describe the presence of trout that could have been Rio Grande cutthroat trout.

The range of the Rio Grande cutthroat trout has been divided by basins into five geographic management units (GMUs) to bring a greater resolution to descriptions of population and habitat distribution and related maintenance and restoration work (Figure 3). These GMUs reflect the hydrologic divisions of the Rio Grande cutthroat trout's historical range by river drainage. The GMUs are managed by the Conservation Team as separate units to maintain genetic and ecological diversity within the subspecies where it exists and to ensure representation of the subspecies across its historical range. However, the GMUs were not created to necessarily reflect important differences in genetic variability in the subspecies based on geography or adaptation to specific environments, although fish in the Pecos and Canadian GMUs do exhibit some genetic differentiation from those in the Rio Grande GMUs (Pritchard *et al*. 2009, p. 1216). Additionally, Rio Grande cutthroat trout are only known from one stream in the Caballo GMU – Las Animas Creek, where a hybridized population currently exists. No other historical locations are known within that GMU.

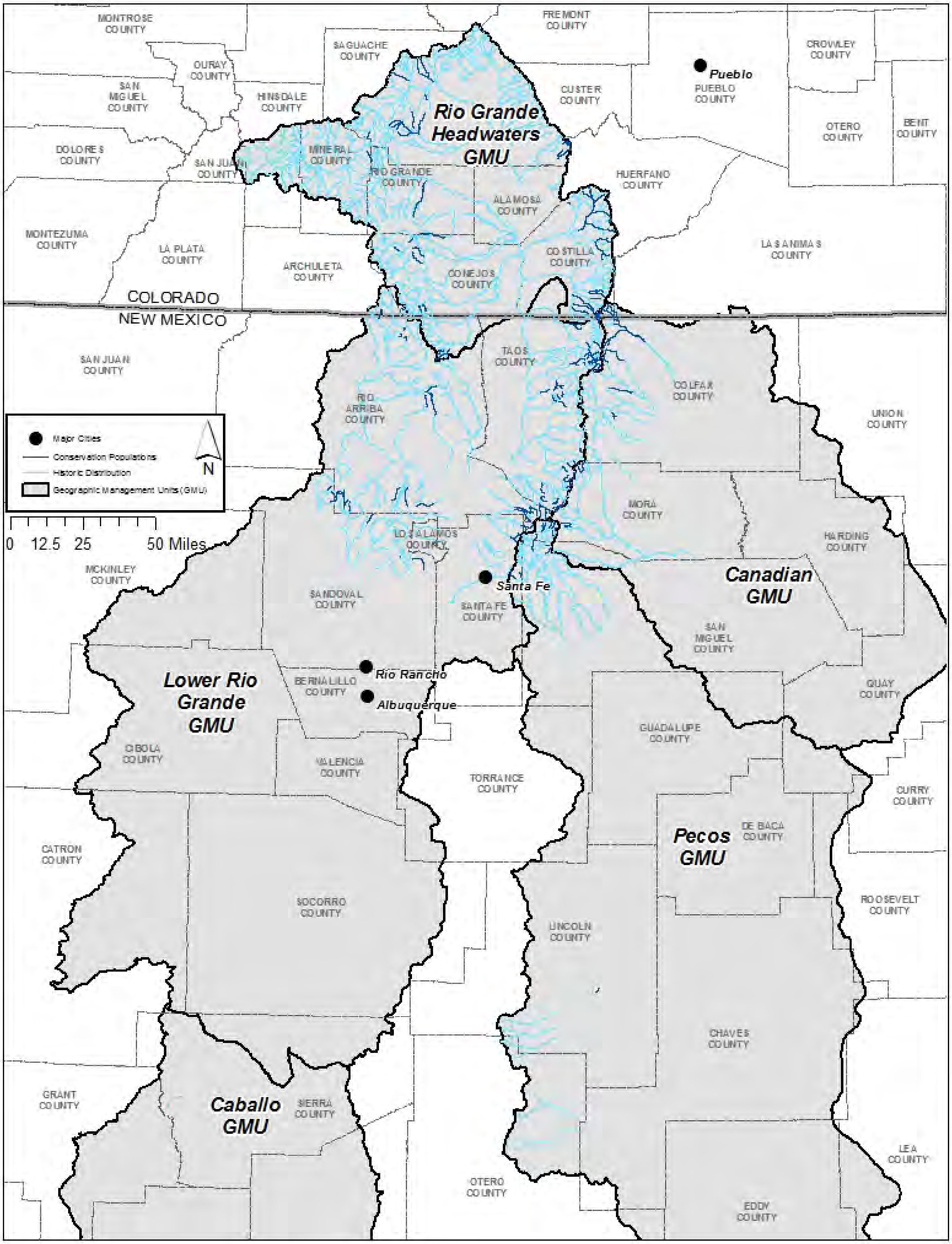


Figure 3. Presumed historical and current ranges of conservation populations of Rio Grande cutthroat trout. Light blue are presumed historically occupied streams, and dark blue streams are currently occupied streams. Map courtesy of New Mexico Department of Game and Fish and U.S. Fish and Wildlife Service.

3.2 Needs of the Rio Grande Cutthroat Trout

As discussed in Chapter 1, for the purpose of this assessment, we define **viability** as the ability of a species to persist over time and thus avoid extinction. Using the SSA framework, we describe the subspecies' viability by characterizing the status of the subspecies in terms of its **resiliency**, **redundancy**, and **representation** (the 3Rs). Using various time frames and the current and projected levels of the 3Rs we thereby describe the subspecies' level of viability over time. To measure these factors, we have created an analysis tool (see Appendix C, Rio Grande Cutthroat Trout Status Assessment Model, and Chapter 5, Viability) that forecasts the subspecies' condition in the future.

3.2.1 Population Resiliency

For the Rio Grande cutthroat trout to maintain viability, its populations, or some portion of its populations, must be resilient. To measure resiliency, we estimated the probability of persistence of each population over three time periods (see Chapter 5, Viability, and Appendix C, Species Status Assessment Model for more information). A number of factors influence the subspecies' viability, including population size and distribution, length of occupied habitat, the potential for nonnative fish invasions, and disease risk. Each of these factors is discussed here.

Resilient Rio Grande cutthroat trout populations must be of sufficient size to withstand demographic effects of low genetic diversity. Larger populations have a higher effective population size, which is a theoretical measure of the number of breeders in the population that contribute to genetic diversity. Populations with a low effective population size are more likely to experience genetic drift and inbreeding and are less likely to adapt to changing environmental conditions. General guidelines for trout have been developed that suggest effective population sizes of 500 and above have a low risk of genetic consequences and retain long term adaptive potential, and those below 50 are highly vulnerable to inbreeding depression and genetic drift (Allendorf *et al.* 1997, pp. 142–143; Rieman and Allendorf 2001, p. 756). Therefore, resilient populations have a sufficient effective population size to avoid adverse genetic consequences on the population.

Resilient Rio Grande cutthroat trout populations also occupy stream reaches long enough to provide the range of habitats needed to complete their life cycle (*i.e.*, spawning habitat, nursery habitat, adult habitat, refugial habitat) (Harig and Fausch 2002, p. 546; Young *et al*. 2005, p. 2406). The longer an unobstructed reach of stream, the more habitat variability is likely to be represented, which increases the likelihood of survival of various life stages (Young *et al.* 2005, p. 2406). In turn, higher likelihood of survival through the life stages supports a higher likelihood of successful recruitment (young individuals joining the breeding population) which supports a larger population size. Further, longer unobstructed stream lengths are more likely to provide habitat during periods of drought (when deep pools provide refugia), over winter (deep pools are less likely to freeze), and longer streams are more likely to provide sufficient complexity (tributaries, stream networking) to allow Rio Grande cutthroat trout populations to survive after stochastic disturbances such as debris flows following wildfire. Streams longer than about 9.65 km (6 miles) are generally assumed to be long enough to encompass the habitat complexity necessary for the population to survive stochastic events (Hilderbrand and Kershner

2000, p. 515; Cowley 2007, p. 9; Peterson *et al.* 2013, p. 10; Roberts *et al.* 2013, p. 12). Streams shorter than 2.8 km (1.7 miles) are unlikely to have enough habitat variability for a population to be able to survive stochastic events (Harig and Fausch 2002, pp. 538–539). Stream reaches smaller than 2.8 km may support populations of Rio Grande cutthroat trout, but local habitat quality is the greatest driver of population occurrence in short segments (Peterson *et al.* 2013, p. 10).

Additionally, resilient Rio Grande cutthroat trout populations are free from hybridization, competition, and predation by nonnative trout. Rainbow trout (*O. mykiss*) and nonnative cutthroat trout subspecies are known to readily hybridize with Rio Grande cutthroat trout (Pritchard and Cowley 2006, p. 3). Once Rio Grande cutthroat trout populations have more than 10% introgression (gene mixing) with nonnative species and subspecies, we no longer consider that population to be a conservation population (Rhymer and Simberloff 1996, pp. 83, 97); this level of introgression has been accepted by the larger cutthroat trout community (Utah Division of Wildlife Resources 2000, p. 4; Alves *et al.* 2008, p. 6). Therefore, resilient Rio Grande cutthroat trout populations must be free of nonnative hybridizing trout.

When brook (*Salvelinus fontinalis*) or brown (*Salmo trutta*) trout invade streams occupied by cutthroat trout, the native cutthroat trout decline over time or are displaced due to competition and predation (Harig *et al*. 2000, pp. 994, 998, 999; Dunham *et al*. 2002, p. 378; Peterson *et al.* 2004, p. 769; Paroz 2005, p. 34; Shemai *et al*. 2007, p. 323). While the use of piscicides (fish toxicants) is the most effective tool to completely eliminate nonnative species, piscicide use is not always feasible (Finlayson *et al.* 2005, pp. 10, 14). Nonnative suppression activities (*i.e.*, electrofishing and removing nonnative species), when occurring annually or nearly annually, can be effective at preventing the displacement of Rio Grande cutthroat trout by brook (Peterson *et al*. 2008b, p. 1861) or brown trout. Because of the high probability of population decline when Rio Grande cutthroat trout co-occur with brook or brown trout, resilient populations should either be free of nonnative trout or have suppression activities occurring regularly.

Finally, resilient Rio Grande cutthroat trout populations are free from disease. Whirling disease, in particular, poses a large risk to salmonid populations in Colorado and New Mexico; once infected, entire year classes are lost, and extirpation of the population is likely (Thompson *et al*. 1999, pp. 312–313). Therefore, resilient populations must be free of whirling disease.

3.2.2 Subspecies Redundancy and Representation

The Rio Grande cutthroat trout needs to have multiple resilient populations distributed throughout its historical range within the four GMUs[2] to provide for rangewide redundancy and representation. The wider the distribution of resilient populations and the larger the number of populations the more redundancy the subspecies will have. This redundancy reduces the risk that a large portion of the subspecies' range will be negatively affected by any catastrophic natural or anthropogenic event at any one time. Species that are well-distributed across their

[2] The Caballo GMU, having only one historical population, cannot have a wider distribution throughout that GMU. While that historical population is currently undergoing restoration (NMDGF *et al.* 2014, entire), if that restoration is unsuccessful it would only marginally affect the subspecies' redundancy and representation rangewide, as it constitutes such a small portion of the historical distribution.

historical range (*i.e*., having high redundancy) are less susceptible to extinction and more likely to be viable than species confined to a small portion of their range (Carroll *et al*. 2012, entire; Redford *et al*. 2011, entire). From a rangewide perspective, multiple Rio Grande cutthroat trout populations should be dispersed throughout the four GMUs to provide for redundancy and subspecies' viability.

Maintaining representation in the form of genetic or ecological diversity is important to keep the capacity of the Rio Grande cutthroat trout to adapt to future environmental changes. Rio Grande cutthroat trout populations vary in the amount of genetic diversity they contain (Pritchard *et al.* 2007, p. 614; Pritchard *et al.* 2009, p. 1216). The Canadian and Pecos GMUs represent significant genetic differentiation from those in the Rio Grande Headwaters and Lower Rio Grande GMUs (Pritchard *et al.* 2009, p. 1219). The Rio Grande cutthroat trout needs to retain populations in the Canadian and Pecos GMUs to maintain the overall potential genetic and life history attributes that can buffer the subspecies' response to environmental changes over time (Moore *et al.* 2010, pp. 340–341; Schindler *et al.*2010, p. 612). Although the GMU boundaries were not generated to represent genetic differences, they encompass the historical range of the Rio Grande cutthroat trout and, therefore, provide a picture of representation of the genetic diversity among populations and the ecological diversity across the subspecies' range. The GMUs serve as a proxy for geographic variation that may represent natural variation in the subspecies' genetic diversity.

To measure representation and redundancy, we estimated the number of persisting populations by GMU for three time periods to provide a geographical estimate of where the Rio Grande cutthroat trout populations will persist into the future (see Chapter 5, Viability, and Appendix C, Species Status Assessment Model for more information).

3.2.3 Subspecies Current Conditions

The current conditions of the Rio Grande cutthroat trout can be summarized based on the number, status, and distribution of the current conservation populations. Conservation populations are those populations of Rio Grande cutthroat trout with less than 10% hybridization with nonnative trout. A single conservation population can include multiple reaches of a stream through which a population may move, or it may encompass only a single reach. As a snapshot of the current condition of the subspecies, we categorized the current 122 conservation populations into four categories (Table 2) based on four main factors affecting their current condition: effective population size, occupied stream length, presence of competing nonnative trout, and presence of hybridizing nonnative trout. Each population was placed in a category of current condition (Best, Good, Fair, and Poor) based on the combination of the four factors as defined in Table 3. For example, a population with an effective population size of 400, a stream length of 6 km, and no competing or hybridizing nonnatives would sort into the "Good" category. Additionally, all populations with hybridizing nonnative trout or effective population sizes of less than 50 would sort into the "Poor" category. Our discussion and analysis of these factors is found in Appendix C.

Table 2. Current status of Rio Grande cutthroat trout showing the number of current conservation populations in four categories by GMU. The percentages (%) are the proportion of total populations within each GMU.

Populations per GMU	Best	%	Good	%	Fair	%	Poor	%	Total
Canadian	1	10%	3	30%	5	50%	1	10%	10
Rio Grande Headwaters	5	12%	14	34%	20	49%	2	5%	41
Lower Rio Grande	13	22%	15	25%	20	34%	11	19%	59
Pecos	1	8%	3	25%	7	33%	1	42%	12
Rangewide	20	16%	35	29%	52	43%	15	12%	122

Overall, we found 20 populations across the range of the Rio Grande cutthroat trout that were in the “Best” condition—that is, they have a long occupied stream reach (>9.65 km), large effective population size (>500), and no nonnative trout present (see Table 3). We found 15 populations rangewide that were in “Poor” condition—that is, either hybridizing nonnative trout were present or the effective population size was less than 50 individuals. The remaining 35 and 52 populations sorted as “Good” and “Fair,” respectively (Table 3).

Table 3. Definitions of four categories (Best, Good, Fair, Poor) used to represent the current condition of conservation populations of Rio Grande cutthroat trout. Each population was placed in a category based on the combination of metrics as indicated by the highlighted colors. See Appendix C for additional information on the factors used to assess the status of the Rio Grande cutthroat trout. A population is placed in a category by meeting any one set of conditions identified.

Categories	Sets of Conditions			
BEST CATEGORY	**Set 1**			
1. Effective Population Size	**>500** / 500-201 / 200-50 / <50			
2. Occupied Stream Length (KM)	**>= 9.65** / **9.64-7.1** / 7.09-2.8 / <2.8			
3. Hybridizing Nonnative Trout Present	**No** / Yes			
4. Competing Nonnative Trout Present	**No** / Yes			
GOOD	**Set 1**	**Set 2**	**Set 3**	
1. Effective Population Size	**>500** / 500-201 / 200-50 / <50	**>500** / 500-201 / 200-50 / <50	>500 / **500-201** / 200-50 / <50	
2. Occupied Stream Length (KM)	**>= 9.65** / **9.64-7.1** / 7.09-2.8 / <2.8	>= 9.65 / 9.64-7.1 / **7.09-2.8** / <2.8	**>= 9.65** / **9.64-7.1** / **7.09-2.8** / <2.8	
3. Hybridizing Nonnative Trout Present	**No** / Yes	**No** / Yes	**No** / Yes	
4. Competing Nonnative Trout Present	No / **Yes**	**No** / Yes	**No** / Yes	
FAIR	**Set 1**	**Set 2**	**Set 3**	**Set 4**
1. Effective Population Size	**>500** / **500-201** / 200-50 / <50	>500 / 500-201 / **200-50** / <50	>500 / **500-201** / **200-50** / <50	**>500** / 500-201 / 200-50 / <50
2. Occupied Stream Length (KM)	>= 9.65 / 9.64-7.1 / 7.09-2.8 / **<2.8**	**>= 9.65** / **9.64-7.1** / **7.09-2.8** / **<2.8**	**>= 9.65** / **9.64-7.1** / **7.09-2.8** / **<2.8**	>= 9.65 / 9.64-7.1 / **7.09-2.8** / **<2.8**
3. Hybridizing Nonnative Trout Present	**No** / Yes	**No** / Yes	**No** / Yes	**No** / Yes
4. Competing Nonnative Trout Present	**No** / Yes	**No** / Yes	No / **Yes**	No / **Yes**
POOR	**Set 1**	**Set 2**		
1. Effective Population Size	**>500** / **500-201** / **200-50** / **<50**	>500 / 500-201 / 200-50 / **<50**		
2. Occupied Stream Length (KM)	**>= 9.65** / **9.64-7.1** / **7.09-2.8** / **<2.8**	**>= 9.65** / **9.64-7.1** / **7.09-2.8** / **<2.8**		
3. Hybridizing Nonnative Trout Present	No / **Yes**	**No** / Yes		
4. Competing Nonnative Trout Present	**No** / **Yes**	**No** / **Yes**		

Another way to view the current status of the subspecies and compare it to historical conditions is using total stream lengths occupied by Rio Grande cutthroat trout. Alves *et al.* (2008, p. 13) estimated the total occupied stream lengths historically based on assumed occupancy for streams that would have likely supported the subspecies, based on the likelihood of suitable habitat being available. Historically it is estimated that the subspecies occurred in about 10,696 stream km, and currently we estimate it occurs in about 1,149 stream km, or about 11% of its historical distribution, and throughout four of the GMUs.

Table 4. Historical and current estimated stream kilometers occupied by Rio Grande cutthroat trout. Historical estimate is from Alves *et al.* (2008, p. 13).

Geographic Management Unit	Historically Occupied (km)	Percent of Historical Total		Currently Occupied (km)	Percent of Current Total
Canadian	1024	9.6%		147	12.8%
Rio Grande Headwaters	5265	49.2%		494	43.0%
Lower Rio Grande	3389	31.7%		446	38.8%
Caballo	17	0.2%		0	0.0%
Pecos	1001	9.4%		62	5.4%
Rangewide Total	10,696	100.0%		1,149	100.0%

Chapter 4. Risk Factors

> **Note:** This chapter contains **summaries** of the risk factors. For further information, see the tables in **Appendix B. Appendix C** contains detailed information about their application to the future condition of the subspecies.

In this chapter we review the past, current, and future risk factors that are affecting what the Rio Grande cutthroat trout needs for long term viability. We analyzed these risk factors in detail using the tables in Appendix B in terms of causes and effects to the subspecies. These tables analyze the pathways by which each stressor affects the subspecies, and each of the causes is examined for its historical, current, and potential future effects on the viability of the Rio Grande cutthroat trout. Each risk factor will be briefly reviewed here; for further information, refer to the tables in Appendix B. The most important factors affecting the future condition of the Rio Grande cutthroat trout were carried forward and analyzed in our Status Assessment Model (see Appendix C).

4.1 Demographic Risk

Small population sizes are at greater risk from reduced genetic diversity, which decreases a population's ability to adapt to environmental changes. Estimating the effective population size (a theoretical measure of the number of breeders in the population that contribute to genetic diversity) of a population is one way to measure the risk of a population experiencing those negative genetic effects. Effective population size is generally lower than census population size due to unequal sex ratios, variable probability of reproductive success, and nonrandom mating (Baalsrud 2011, p. 1). General guidelines developed for trout suggest effective population sizes of 500 and above have a low risk of genetic consequences and retain long term adaptive potential, and those below 50 are highly vulnerable to inbreeding depression and genetic drift (Allendorf *et al.* 1997, pp. 142–143; Rieman and Allendorf 2001, p. 756). To our knowledge, no populations of native trout have been extirpated due to demographic risk alone; instead, it is a factor that can make the population more vulnerable to extirpation from other factors. See p. B-3 for more analysis of demographic risk.

4.2 Hybridizing Nonnative Trout

The introduction of nonnative trout species (including those that hybridize with Rio Grande cutthroat trout and those that compete with them) into Rio Grande cutthroat trout habitat accounts for the majority of the 89% range loss of the subspecies. Nonnative rainbow trout and other cutthroat trout subspecies have historically been introduced throughout the range of Rio Grande cutthroat trout for recreational angling, and they are known to readily hybridize with Rio Grande cutthroat trout (Alves *et al.* 2008, p. 15). Hybrids can have reduced fitness, and even when fitness is increased, hybridization may disrupt important long-term adaptations of native populations (Allendorf *et al*. 2004, p. 1203). The genetic distinctiveness of Rio Grande cutthroat trout can be lost through hybridization (Allendorf *et al*. 2004, p. 1205), and once Rio Grande cutthroat trout populations have more than 10% introgression (gene mixing) with nonnative species and subspecies, that population is no longer considered a conservation population (Rhymer and Simberloff 1996, pp. 83, 97; Utah Division of Wildlife Resources 2000, p. 4).

Populations are not immediately affected after nonnative trout are introduced; it can take years (or decades) for Rio Grande cutthroat trout to be affected at the population level. In some cases it can take even longer for the genetic mixing from hybridization to exceed 10% introgression and for the population to no longer be considered a conservation population and, therefore, extirpated. Fisheries managers throughout the range of Rio Grande cutthroat trout have worked to eradicate nonnative trout from stream reaches historically occupied by Rio Grande cutthroat trout. In general, all hybridizing nonnative trout must be completely removed from the stream system in order to prevent hybridization from occurring. The eradication of nonnative trout includes removing all fish from the reach through the use of piscicides, installing fish passage barriers in streams to prevent future invasion by nonnative trout, and repatriating those reaches with pure Rio Grande cutthroat trout. Currently, 72 conservation populations (of 122; 59%) are protected by complete barriers to upstream fish movement and 14 conservation populations (of 122; 11%) are protected by partial barriers to upstream fish movement (RGCT Database). These barriers reduce the risk of future invasions by hybridizing, nonnative trout.

Rio Grande cutthroat trout continue to be vulnerable to the negative effects of hybridization with nonnative rainbow and Yellowstone cutthroat trout. We expect that accidental or intentional illegal introductions of rainbow trout may continue to occur, though infrequently, so nonnative trout will continue to pose some risk to Rio Grande cutthroat trout populations in the future. Once an invasion occurs, sympatry (co-occurrence) with nonnative hybridizing trout is a high risk to Rio Grande cutthroat trout population persistence. See p. B-6 for more analysis of nonnative hybridizing trout.

4.3 Competing Nonnative Trout

Other species of nonnative trout have historically been stocked throughout the range of the Rio Grande cutthroat trout, as well. Brook and brown trout compete with Rio Grande cutthroat trout for food and space, and larger adults are likely to predate upon young Rio Grande cutthroat trout (Dunham *et al.* 2002, p. 378; Fausch *et al*. 2006, pp. 9–10)[3]. While no stocking of brook or brown trout is currently ongoing in New Mexico or Colorado, both species are found throughout historical Rio Grande cutthroat trout waters. Water temperature, fine sediment, and the abundance of pools and woody debris influence the degree of brook and brown trout invasion (Shepard 2004, p. 1096). Currently, approximately 41% of Rio Grande cutthroat trout conservation populations (50 of 122) are known to co-occur with brook or brown trout. In general, over time native cutthroat trout populations will diminish and may become extirpated when they co-occur with brook and brown trout (Peterson and Fausch 2003, p. 769). As with the introduction of rainbow trout into a conservation population, Rio Grande cutthroat trout populations are not immediately affected after brook or brown trout are introduced; it can take many years for populations to decline and then become extirpated. Unlike with hybridizing trout species, managers can implement mechanical suppression (catching and removing nonnative trout species on a regular basis) within streams where Rio Grande cutthroat trout populations are sympatric with brook and brown trout in areas where complete eradication using piscicides is not feasible. However, eradication of all fish and repatriation with Rio Grande cutthroat trout

[3] Throughout this SSA Report, we refer to brook and brown trout as "competing nonnative trout." We recognize that predation is also a stressor when these species co-occur with Rio Grande cutthroat trout, and this stressor is included in our analysis.

remains the most effective method of decreasing the risk of extirpation due to sympatry with nonnative competing trout species. Currently, 86 conservation populations are protected by complete or partial barriers to upstream fish movement (RGCT Database), reducing the risk of competing nonnative species invasions.

Rio Grande cutthroat trout continue to face pressure from competition with nonnative brook and brown trout. We expect that, as infrequent accidental and/or intentional illegal introductions occur (Johnson *et al.* 2009, p. 389), nonnative trout will continue to pose a risk to Rio Grande cutthroat trout populations in the future. See p. B-9 for more analysis of competing nonnative trout.

4.4 Wildfire

Wildfires are a natural disturbance in forested watersheds, particularly in the Southwest. However, since the mid-1980s, wildfire frequency in western forests has nearly quadrupled compared to the average frequency during the period 1970 – 1986 (Westerling *et al.* 2006, p. 941), and this increase is widely attributed to climate change (McKenzie *et al.* 2004, p. 893; Westerling *et al.* 2006, p. 942; IPCC 2007a, p. 15). Risk of wildfires can be affected by forest management activities; fire suppression or a lack of thinning or prescribed burns can enhance conditions suitable for high-intensity wildfires (Schoennagel *et al*. 2004, p. 669). Although Rio Grande cutthroat trout may survive after a fire burns through a watershed, ash and debris flows that occur after a fire can eliminate populations of fish from a stream (Rinne 1996, p. 654; Brown *et al*. 2001, p. 142). In the past, this was likely not a significant factor affecting Rio Grande cutthroat trout, as interconnected populations provided a source for repatriation of extirpated areas. However, the fragmentation experienced by most Rio Grande cutthroat trout populations prevents recolonization after extirpation. Wildfires within the range of Rio Grande cutthroat trout have depressed or eliminated fish populations (Japhet *et al*. 2007, p. 20; Patten *et al.* 2007, pp. 33, 36; RGCT Database). The amount of ash flow from a fire depends on the severity of the fire, proximity to the stream habitat, stream channel morphology, timing, and amount of rainfall following the fire (Rinne 1996, p. 656; Rieman and Clayton 1997, p. 9).

The extent of one or more populations being affected by wildfire depends on the location of the fire, the length and amount of stream networking of the occupied stream reach, and the extent of stream networking (Roberts *et al*. 2013, p. 6). For example, Polvadera Creek, in the Lower Rio Grande GMU, burned during the South Fork Fire in 2010, and ash flows following that fire nearly eliminated the subspecies from the stream. However, during subsequent fish surveys, young-of-year Rio Grande cutthroat trout were found in the headwaters of the stream (RGCT Database), indicating suitable habitat remained and the population survived in low numbers. The presence of stream reaches that provide refugia during and after fires plays a large role in the ability of the population to repatriate affected areas (Rieman and Clayton 1997, p. 10).

Wildfires may also provide opportunities for Rio Grande cutthroat trout restoration. Just as ash and debris flows following wildfires can eliminate Rio Grande cutthroat trout populations, they can also eliminate nonnative trout. Once the stream has been confirmed to be fishless, and the habitat has regained stability, Rio Grande cutthroat trout can be repatriated to the affected stream reach. This situation has occurred in Pinelodge Creek (Pecos GMU) and Capulin Creek (Lower

Rio Grande GMU) in the past with successful re-establishment of Rio Grande cutthroat trout (NMDGF 2013, p. 3). The recent Las Conchas Fire in New Mexico (Lower Rio Grande GMU) has resulted in the elimination of nonnative trout from 5 stream reaches that NMDGF is planning to restock with Rio Grande cutthroat trout (NMDGF 2013, p. 3).

As drought frequency increases due to climate change, dry forests are more likely to burn and burn hotter than they have in the past (Glick 2006, p. 8). Wildfire risk analysis rangewide (Miller and Bassett 2013, entire) shows that if a wildfire is ignited, all of the watersheds supporting Rio Grande cutthroat trout populations have a high risk of burning and of resulting in high levels of debris flow. The only exceptions are for some populations in the Rio Grande Headwaters GMU, which have a moderate risk of fire and debris flow. This risk analysis evaluated the potential behavior of a fire if it started, based on flame length and crown fire potential. Fuels management may be done on a local scale to reduce some risks; however, given that climate change will increase the likelihood of large, hot fires throughout the Southwest, we expect that the effects of wildfire will continue to result in loss of Rio Grande cutthroat trout populations in the future. See p. B-13 for more analysis of wildfire.

4.5 Stream Drying

Stream drying within Rio Grande cutthroat trout populations may occur as a result of drought or, in a few cases, water withdrawals. As streams begin to dry, the amount of habitat available for Rio Grande cutthroat trout is reduced; streams may become more narrow and intermittent. Drought frequency is expected to increase as a result of climate change due to a combination of increased summer temperatures and decreased precipitation (Nash and Gleick 1993, p. ix; IPCC 2007a, p. 15; Ray *et al.* 2008, p. 37; Haak and Williams 2012, p. 388). Stream intermittency may cause water quality declines (increased temperature, decreased oxygen), lack of access to breeding, feeding, and sheltering areas, and stranding of fish (Lake 2000, p. 577). In the past, this was likely not a significant factor affecting Rio Grande cutthroat trout, as interconnected populations provided a source for repatriation of extirpated areas. However, the fragmentation experienced by most Rio Grande cutthroat trout populations prevents recolonization after extirpation, in most cases. Streams with drought refugia (pools or other areas that remain wetted during dry times) within the occupied reaches can increase the chances of populations surviving if stream drying occurs.

Climate change is expected to increase the frequency and severity of drought, which will result in streams continuing to become intermittent and risking loss of Rio Grande cutthroat trout populations. Reduced summer streamflows have already been observed throughout the range of Rio Grande cutthroat trout (Zeigler *et al.* 2012, p. 1050), and population extirpations have been observed in a few cases (Japhet *et al.* 2007, pp. 42–45; J. Alves, CPW, 2014 pers. comm.). We expect that stream drying as a result of drought and, in some cases, water withdrawals will continue to result in population effects and risk of extirpations throughout the subspecies' range. See p. B-17 for more analysis of stream drying.

4.6 Disease

Whirling disease is caused by a nonnative parasite (*Myxobolus cerebralis*), which requires two separate hosts to complete its life cycle: a salmonid fish and an aquatic worm (*Tubifex tubifex*). Spores of the parasite are released when infected fish die; these spores are ingested by the *T. tubifex* worm, where they undergo transformation in the gut to produce actinosporean triactionomyxons (TAMs). Trout are infected either by eating the worms (and TAMs) or through contact with TAMs after they have been released from the worms into the water. The myxosporean parasite became widely distributed in Colorado in the early 1990s through the stocking of millions of catchable size trout from infected hatcheries (Nehring 2007, p. 1). Parasites damage cartilage, killing young fish or causing infected fish to swim in an uncontrolled whirling motion, making it impossible to avoid predation or feed (Hiner and Moffett 2001, p. 130). Mortality rates of 85% or more may occur within 4 months of exposure (Thompson *et al.* 1999, p. 312). Once *M. cerebralis* is present, total year class failure of Rio Grande cutthroat trout can occur (Nehring 2008, p. 2), and precipitous population declines may result (Thompson *et al.* 1999, p. 313).

NMDGF policies and regulations prohibit the stocking of any whirling disease positive fish in the State of New Mexico (Patten and Sloane 2007, p. 10). In Colorado, stocking of whirling disease-positive fish in protected habitats, which include native cutthroat trout waters, is prohibited (Japhet *et al*. 2007, p. 12).

We expect Rio Grande cutthroat trout populations will occasionally become infected with whirling disease in the future. Risk of disease is gauged by the distance of the population to known locations of whirling disease. No conservation populations are currently determined to be infected or at high risk of infection, and only 7% of conservation populations (9 of 122) have been determined to be at moderate risk of whirling disease infection (i.e., they are within 10 km (6.2 mi) of known whirling disease locations) (Alves *et al.* 2008, p. 38). Because fish movement barriers help guard populations again infection by preventing the invasion of infected trout, and whirling disease has affected very few Rio Grande cutthroat trout populations to date, whirling disease poses extremely low risks to the majority of Rio Grande cutthroat trout populations because of the low likelihood of infection. See p. B-21 for more analysis of disease.

4.7 Water Temperature Changes

Stream warming due to climate change has been observed throughout salmonid habitat in the west, and summer high water temperatures may become a key bottleneck for many species of trout (Isaak *et al.* 2012a, p. 514). Stream warming trends induced by climate change can cause some streams to become too warm for Rio Grande cutthroat trout populations to thrive, while several streams that are currently colder than optimal will warm and become more suitable (Zeigler *et al*. 2013a, p. 1400; Zeigler *et al.* 2013b, pp. 6–9). Air temperatures in the last 45 years throughout the range of Rio Grande cutthroat trout have increased an average of 0.29 °C (0.5 °F) per decade (Zeigler *et al*. 2012, p. 1049). The extent to which streams will warm varies with elevation, slope, and aspect.

As with Colorado River cutthroat trout (*O. c. pleuriticus*) (Roberts *et al.* 2013, p. 13), Rio Grande cutthroat trout populations are currently restricted to higher elevations due to nonnative trout interactions, and the effects of warming temperatures do not appear to be as stark as previously thought. No populations throughout the range of Rio Grande cutthroat trout are currently experiencing acute effects (mortality) due to high temperature; and one population may be experiencing chronic effects (such as reduced growth) due to current stream temperatures (Rogers 2013, pp. 18–21; Zeigler *et al.* 2013a, p. 1400; Zeigler *et al.* 2013b, pp. 6–9). In the future, climate change may cause summer water temperatures to increase, potentially putting future populations at risk from chronic and acute temperature effects. We found that the majority of the high elevation headwater streams where Rio Grande cutthroat trout are currently found are not expected to experience significant temperature increases; therefore, most Rio Grande cutthroat trout populations have an extremely low risk of extirpation over the next 65 years due to water temperature increases. See p. B-23 for more analysis of water temperature changes.

4.8 Changes in Flood Timing and Magnitude

Changes in precipitation and air temperature expected from climate change (becoming drier and warmer) will likely lead to changes in the magnitude, frequency, timing, and duration of spring snowmelt runoff patterns, as well as water temperature changes in streams occupied by Rio Grande cutthroat trout (Poff *et al.* 2002, p. 4; Isaak *et al.* 2012b, p. 544). The life history of salmonids is closely tied to flow regime, runoff in particular (Fausch *et al*. 2001, p. 1440). An increase in magnitude of floods (perhaps due to rain on snow events) can scour streambeds, destroy eggs, or displace recently emerged fry downstream (Erman *et al*. 1988, p. 2199; Montgomery *et al*. 1999, p. 384). Climate warming is also causing snowmelt runoff to peak approximately 10 days earlier in the spring than 45 years ago (Clow 2010, p. 2297; Zeigler *et al.* 2012, p. 1050). The environmental cues for Rio Grande cutthroat trout spawning are most likely tied to increasing water temperature, increasing day length, and possibly flow, as it has been noted that they spawn when runoff from snowmelt has peaked and is beginning to decrease (Behnke 2002, p. 141; Pritchard and Cowley 2006, p. 25). Earlier runoff could disrupt spawning cues because peak flow would occur when the days are shorter in length and, therefore, water temperatures are colder (Stewart *et al*. 2005, p. 1137). This earlier snowmelt, which leads to less flow in the spring and summer, could either benefit Rio Grande cutthroat trout or be detrimental. The benefit could come because the young-of-year would have a longer growing season before winter. However, as discussed above, a longer season of lower flows would lead to increased stream temperatures and increased probability of intermittency and drying.

In summary, it is difficult to project how changes in the hydrograph as a result of climate change will affect Rio Grande cutthroat trout populations. If the growing season is increased because of changes in flood timing and magnitude, they could be beneficial to Rio Grande cutthroat trout by increasing recruitment rates thanks to a longer summer growing season. However, if spawning cues are disrupted or egg and fry survival is reduced because of large magnitude floods during spawning or rearing times, it would negatively affect populations. However, because the large uncertainty regarding the extent and effects of these hydrological changes on Rio Grande cutthroat trout populations makes it difficult to draw reasonably reliable conclusions, and because the effects of hydrological changes that may result in stream drying are captured in the

Stream Drying discussion (see section 4.5, above), the effects of hydrological changes are not carried forward as an analyzed risk in the Status Assessment Model (Appendix C). See p. B-25 for more on changes in flood timing and magnitude.

4.9 Land Management

Cattle grazing, timber harvest, non-angling recreation, road building, and mining all occur within watersheds occupied by Rio Grande cutthroat trout, and all of these activities may lead to stressors that can affect the subspecies. While each activity can reduce riparian vegetation (eliminating cover and potentially resulting in water temperature increases), increase sedimentation (reducing instream habitat quality), increase erosion (reducing stream stability and cover), reduce food availability (overgrazing results in a reduction of terrestrial insects, which generally represent about half the diet of trout) (Saunders and Fausch 2007, p. 1224; 2012, p. 1525), and negatively affect habitat occupied by Rio Grande cutthroat trout, these practices have decreased in severity in recent decades (USFS 2005 (70 FR 68264); Poff *et al.* 2011, p. 2). Some land management activities are occurring throughout the range of the subspecies. Locally, land management activities may still be having some effects on aquatic habitat resulting in limited effects on Rio Grande cutthroat trout. However, the intensity of grazing and other activities is generally light because most of the streams Rio Grande cutthroat trout populations currently occupy are in high elevation, remote areas. We do not expect this to change in the future, given the ruggedness of the landscape and that the land management agencies are party to the Conservation Agreement and Strategy. Therefore, we do not think that land management activities will have measureable population-level effects in the future. See p. B-27 for more analysis of land management.

4.10 Angling

Recreational angling occurs on approximately 84% of Rio Grande cutthroat trout conservation populations (Alves *et al.* 2008, p. 47). Fishing regulations in New Mexico and Colorado appropriately manage recreational angling. For example, many of the streams with Rio Grande cutthroat trout are "catch and release." Those that are not have a 2 (New Mexico) or 4 (Colorado) fish limit. While even catch and release angling can have some effects on individual fish (*i.e.*, handling stress, swallowing hooks) (Bartholomew and Bohnsack 2005, p. 140), many conservation populations of Rio Grande cutthroat trout are in very remote areas and angling pressure is light (Alves *et al.* 2008, p. 47). For these reasons, we do not expect angling is affecting or will affect Rio Grande cutthroat trout populations in the future. See p. B-31 for more analysis of angling.

4.11 Management Actions

The Rio Grande Cutthroat Trout Rangewide Conservation Team developed the Conservation Agreement and Strategy in 2013 (revised from the previous Conservation Agreements in 2003 and 2009). The Conservation Strategy formalized many of the management actions that have been ongoing for the subspecies for decades. Activities such as stream restorations, barrier construction and maintenance, nonnative species removals, habitat improvements, public outreach, database management, and many other activities are described in detail. Over the 10-

year life of the Agreement and Strategy, the Conservation Team has committed to restoration of between 11 and 20 new Rio Grande cutthroat trout populations to historical habitat. If the Agreement and Strategy are implemented as planned, the result would be at least 11 new highly resilient Rio Grande cutthroat trout conservation populations throughout the range of the subspecies. Because of the history of active management of this subspecies by the states of Colorado and New Mexico as well as land management agencies, we expect that even in the absence of the Agreement and Strategy beyond the time period of the current agreement, many management activities would continue to occur. Therefore, for projections after 2023, we analyzed the viability of the subspecies under varying management scenarios. Refer to Appendix C for additional details. See p. B-33 for more analysis of management actions.

4.12 Climate Change

Climate change has already begun, and continued greenhouse gas emissions at or above current rates will cause further warming (IPCC 2007a, p. 13). Warming in the Southwest is expected to be greatest in the summer (IPCC 2007b, p. 887), and annual mean precipitation, length of the snow season, and snow depth are very likely to decrease in the Southwest (IPCC 2007b, p. 887; Ray *et al.* 2008, p. 1). Effects of climate change, such as air temperature increases, drought, and timing and magnitude of flood flows, have been shown to be occurring throughout the range of Rio Grande cutthroat trout (Zeigler *et al*. 2012, pp. 1051–1052), and these effects are expected to exacerbate several of the stressors discussed above, such as water temperature, stream drying, and wildfire (Wuebbles *et al.* 2013, p. 16). We also considered changes in hydrological patterns, although due to the uncertainty in the extent and effects on populations, we did not carry that risk factor forward in our model. In our analysis of the future condition of the Rio Grande cutthroat trout, we added an assessment of how climate change is likely to exacerbate the stressors of hybridizing nonnative trout, stream temperature, stream drying, and the effects of wildfire (see Appendix C for detailed information of how this was assessed).

4.13 Synthesis

Our analysis of the past, current, and future factors that are affecting what the Rio Grande cutthroat trout needs for long term viability revealed that seven of these factors are having the largest influence on future viability of the subspecies. These factors are demographic risk, nonnative hybridizing trout, nonnative competing trout, wildfire risk, stream drying risk, water temperature risk, and disease risk. Other factors, such as land management, recreational angling, and hydrological changes, may be having local effects on populations but do not appear to be affecting the subspecies at a population scale. Therefore, our Status Assessment Model (Appendix C) included these seven factors when examining risks to Rio Grande cutthroat trout populations.

Chapter 5. Viability

We have considered what the Rio Grande cutthroat trout needs for viability and the current condition of those needs (Chapters 2 and 3), and we reviewed the risk factors that are driving the historical, current, and future conditions of the species (Chapter 4 and Appendix B). We now consider what the subspecies' future conditions are likely to be. We analyzed the future conditions based on a Status Assessment Model that allowed us to quantitatively forecast the future status of the subspecies based on our understanding of the risks faced by the Rio Grande cutthroat trout. We apply the results of our model to the concepts of resiliency, redundancy, and representation to describe the viability of the Rio Grande cutthroat trout.

Note: This chapter contains **summaries** of the analysis of viability. For further information, see **Appendix C** which contains detailed information about how we modeled the future conditions of the subspecies.

5.1 Introduction

The Rio Grande cutthroat trout has undergone a precipitous decline in overall distribution and abundance, as is evidenced by the currently occupied stream habitat being on the order of 11% of the presumed historical range. The resulting remnant populations are small compared to presumed historical populations, and, for the most part, they are isolated from other populations in high elevation, headwater streams. The primary reason for this reduction in range and abundance was the introduction of nonnative trout species. Rainbow trout and other subspecies of cutthroat trout had the most obvious impact by hybridizing with Rio Grande cutthroat trout, and, secondarily, brown trout and brook trout also impacted the native trout through competition and predation.

While the future impacts from nonnative species are still a concern to the extant populations, the risk of additional introductions has been largely curtailed due to aggressive and sustained management actions by State management agencies and Federal, Tribal, and private land managers. The main management activities used to reduce the risk of future nonnative invasions are: 1) the cessation of stocking additional nonnative trout in waters near extant Rio Grande cutthroat trout conservation populations, 2) conversion to only stocking triploid rainbow trout (trout possessing three sets of chromosomes instead of two, and are therefore unable to reproduce) in New Mexico waters in Rio Grande cutthroat trout watersheds, 3) the removal of nonnative trout from occupied habitat, and 4) the construction and maintenance of fish barriers in streams that reduce the chance of future invasions of nonnative trout through dispersal to upstream Rio Grande cutthroat trout populations.

Because the remaining populations of Rio Grande cutthroat trout are generally small (compared to historical populations) and isolated, they are likely less resilient than in the past. Now a single stochastic event such as wildfire, and subsequent ash-laden floods, could eliminate an entire population of Rio Grande cutthroat trout. The impacts at the subspecies level are heightened by the isolated nature of the populations because natural recolonization of lost stream segments, which may have been likely historically, now are no longer possible because nearby or connected populations do not exist in most cases. We expect that the frequency and intensity of wildfire is likely to only become greater as the landscape gets warmer and drier from ongoing climate change.

Another source of stress to Rio Grande cutthroat trout, which may not have been significant historically because of the broad distribution of the subspecies, is the loss of populations due to stream drying. Obviously as stream flows decline due to anthropogenic factors of water use (either surface or groundwater), or due to drought, which may be heightened by climate change, then populations can be lost. Therefore, because populations are isolated, lost populations cannot be naturally recolonized.

In addition to those factors that have affected the subspecies in the past (such as wildfire and stream drying), there are several relatively new factors affecting the subspecies. Whirling disease was introduced in the 1990s, and when a population is infected it generally cannot recover. Additionally, climate change is expected to result in warmer stream temperatures, potentially further restricting the range of the subspecies.

Any of these stressors, alone or in combination, could result in the extirpation of populations which would decrease the overall redundancy and representation of the subspecies. Historically the subspecies, with a large range of interconnected populations, would have been resilient to stochastic events such as drought and wildfire because even if some populations were extirpated by such events, they could be recolonized over time by dispersal from nearby surviving populations. This connectivity would have made for a highly resilient subspecies overall. However, under current conditions, restoring that connectivity on a large scale is not feasible due to the wide-ranging presence of nonnative trout species. In fact, rather than increasing stream connectivity, in most locations managers are maintaining fish barriers to keep out nonnative trout rather than building connectivity (see exception on Vermejo Park Ranch, where nearly 161 stream km (100 stream miles) are being restored and reconnected (Vermejo Park Ranch *et al.* 2013, entire)).

As a consequence of these current conditions, the viability of the subspecies now primarily depends on maintaining as many as possible of the remaining isolated populations and restoring new populations where feasible. Management actions to expand existing populations where possible, to remove nonnative trout from occupied habitat, to maintain nonnative fish barriers where needed, and to restore new populations of Rio Grande cutthroat trout are now imperative to the long-term viability of the subspecies. The resiliency of the subspecies has been reduced at the subspecies level, but how is this reduction affecting the overall viability of the subspecies as we consider the future status of the Rio Grande cutthroat trout? We developed a Status Assessment Model to help address this question.

5.2 Forecasting Future Conditions

5.2.1 Status Assessment Model

We undertook an analysis (Appendix C) to quantitatively forecast what the future condition of the Rio Grande cutthroat trout in a way that characterizes viability in terms of the subspecies' resiliency, redundancy, and representation (Figure 4). The purpose of this analysis was to quantitatively reflect our understanding of the future viability of this subspecies by explicitly considering all the factors we found to be potentially affecting population persistence and by using our professional judgment to apply the best available information to assess the status of the Rio Grande cutthroat trout. Our objectives were twofold: 1) to estimate the probability of persistence of each extant Rio Grande cutthroat trout population over time; and 2) describe the future persistence of Rio Grande cutthroat trout by forecasting the likely number of populations expected to persist across the subspecies' range over time. As a consequence we developed two separate, but related, modules that:

1. Estimate the probability of persistence for each Rio Grande trout population by GMU for 3 time periods under a range of conditions; and
2. Estimate the number of surviving[4] populations by GMU for 3 time periods under several scenarios.

For the first module, we used seven risk factors to estimate the probability of persistence of each Rio Grande cutthroat trout population (Figure 4). For each risk factor, we used one or more population metrics that contribute to the risk of extirpation of the populations. We used our expert judgment to develop risk functions for each population metric. These judgments were based on our understanding of these risk factors as explained in Appendix B and Chapter 4. We only considered the risk factors that we deemed are likely to have population level impacts based on analysis of the causes and effects of those risk factors (Chapter 4 and Appendix B). For four of the risk factors, we accelerated the rate of risk increase over time because we believe that environmental changes associated with global climate change will likely increase the risks associated with those factors (see Appendix C, p. C-8 for more discussion of the risk associated with climate change). We summed all the risk functions for each population and subtracted that sum from 1 to calculate a probability of persistence for each population. We did this calculation for each population for future timeframes of 2023, 2040, and 2080. We also calculated the probability of persistence with and without suppression management activities for controlling competing nonnative trout for the 10 populations where suppression is currently occurring. And we did the analysis under two climate change conditions with moderate and severe effects of climate change. These forecasts resulted in a description of the resiliency of the populations in terms of probability of persistence of the current populations. By analyzing the resulting persistence probabilities by GMU, the results also provide a picture of representation and redundancy.

For the second module, we conducted a survival simulation based on the output of persistence probabilities from module 1 to forecast the number of populations that may survive over time (Figure 4). To do this, we used a randomization process to simulate whether a population remains extant or goes extinct based on our modeled probability of persistence. The simulation compares a random number (simulating a possible extirpation event), drawn from a uniform distribution between 0 and 1, to the estimated probability of persistence. If the random number is greater than the probability of persistence, for that iteration that population gets a 0 and is extirpated. If the random number is less than the probability of persistence, for that iteration,

[4] For this report, the terms "persisting" and "surviving" are used interchangeably when referring to populations sustaining themselves beyond the end points evaluated.

that population gets a 1 and survives. We summed the number of extant populations for each replication, and, after running the simulation 100 times, we calculated a mean number of surviving populations by GMU with a 95% confidence interval. We then added to those simulated number of surviving populations an estimate of the number of populations that may be restored over time by proactive management. Forecasting future restoration efforts has a large amount of uncertainty beyond the next 10 years, so we used a range of possibilities to include in the model output. For the overall population survival model, we considered 9 possible scenarios including the 3 time intervals that produce a best case, worse case, and intermediate case. The scenarios represent different combinations of assumptions based on: 1) level of climate change effects (moderate or severe); 2) whether or not suppression of nonnatives occurs; 3) the output of the population simulation model (mean and $\pm$ 95 % confidence interval); and 4) the projected level of future population restorations (low, mid, or high). The results from this analysis provides an assessment of future redundancy and representation based on the number of forecasted surviving populations rangewide and an assessment of representation as we report the results by GMU over time.

We also estimated the potential number of stream kilometers that are forecasted to be occupied in the future using our future population simulation. We did this in order to compare the current and future status of the Rio Grande cutthroat trout to the historical status in terms of total amount of occupied habitat. This estimation is not very precise, however, because we had to make large assumptions in estimating the future amount of occupied stream kilometers by population. Therefore, we only use these results as a general guide to compare the possible total occupied habitat in the future to what was present historically and currently.

For a detailed description of the methodology used in this analysis, as well as a discussion of the strengths and limitations of this analysis, please refer to Appendix C, Rio Grande Cutthroat Trout Status Assessment Model.

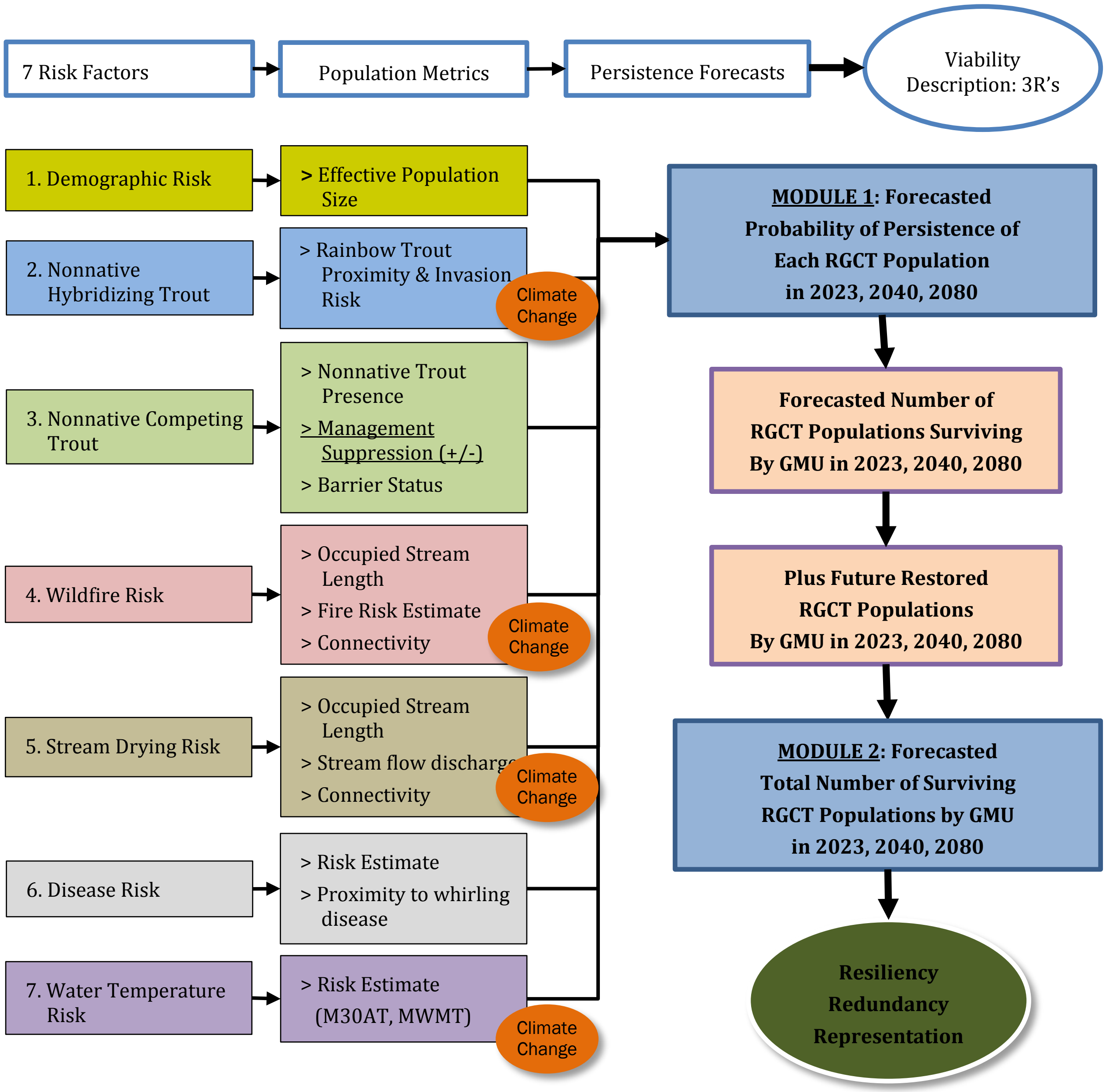


Figure 4. Conceptual diagram of Rio Grande cutthroat trout status assessment model.

5.3 Results: Module 1, Probability of Persistence

An overview of the resulting probability of persistence (on a scale from 0 to 1) for each population is shown in Figures 5 and 6. These scatter plots display 6 of the possible 10 conditions (see Appendix C, Figure C2 for each condition analyzed), but they demonstrate the range of results for each population.

For Figure 7 (and following Figures 8, 10, 12, and 14) we display the results of the population persistence analysis as frequency histograms, similar to Roberts *et al.* (2013, p. 1393). These figures display the probability of persistence over time under various conditions. For 2023, we analyzed the conditions with and without suppression activities and no climate change effects. For the 2040 and 2080 time periods, we show the results with no management suppression and with moderate and severe climate change effects. For these results, we used persistence probability categories of high (greater than 0.9), mod (moderate between 0.75 and 0.9), low (0.5 to 0.75) and minimal (less than 0.5). Figures 8, 10, 12, and 14 also show frequency distributions for the same conditions for each of the four GMUs. Figures 9, 11, 13, and 15 geographically show the location of the populations with the persistence probabilities for 3 sets of conditions over the 3 timeframes. For the 2023 maps we used the condition with nonnative trout suppression. For the 2040 and 2080 maps we used the condition without nonnative trout suppression and with moderate climate change effects.

Although management by the States of Colorado and New Mexico is likely to continue in the future beyond 2023, we are unable to predict when or where the efforts may occur that far into the future. Therefore, to show a conservative estimate of the probability of persistence of the populations in 2040 and 2080, we did not include in these results the nonnative suppression efforts on the streams that are currently being suppressed. Those conservation efforts currently affect the results of 10 of the 122 populations analyzed; therefore, it would not make a substantial difference in the overall results. Furthermore, the Vermejo CCAA will add over 160 stream kilometers of occupied Rio Grande cutthroat trout habitat when it is completed, approximately 50% of which have been restored to date. Our status assessment model is not able to take this into account for future forecasting. While we have found that the Vermejo CCAA satisfies our PECE criteria and may be considered for future analysis, our model does not currently reflect these anticipated increases in population size and resiliency. If the results did include these conservation efforts the overall probabilities of persistence would be higher than forecasted.

5.3.1 Rangewide Probability of Persistence by Population

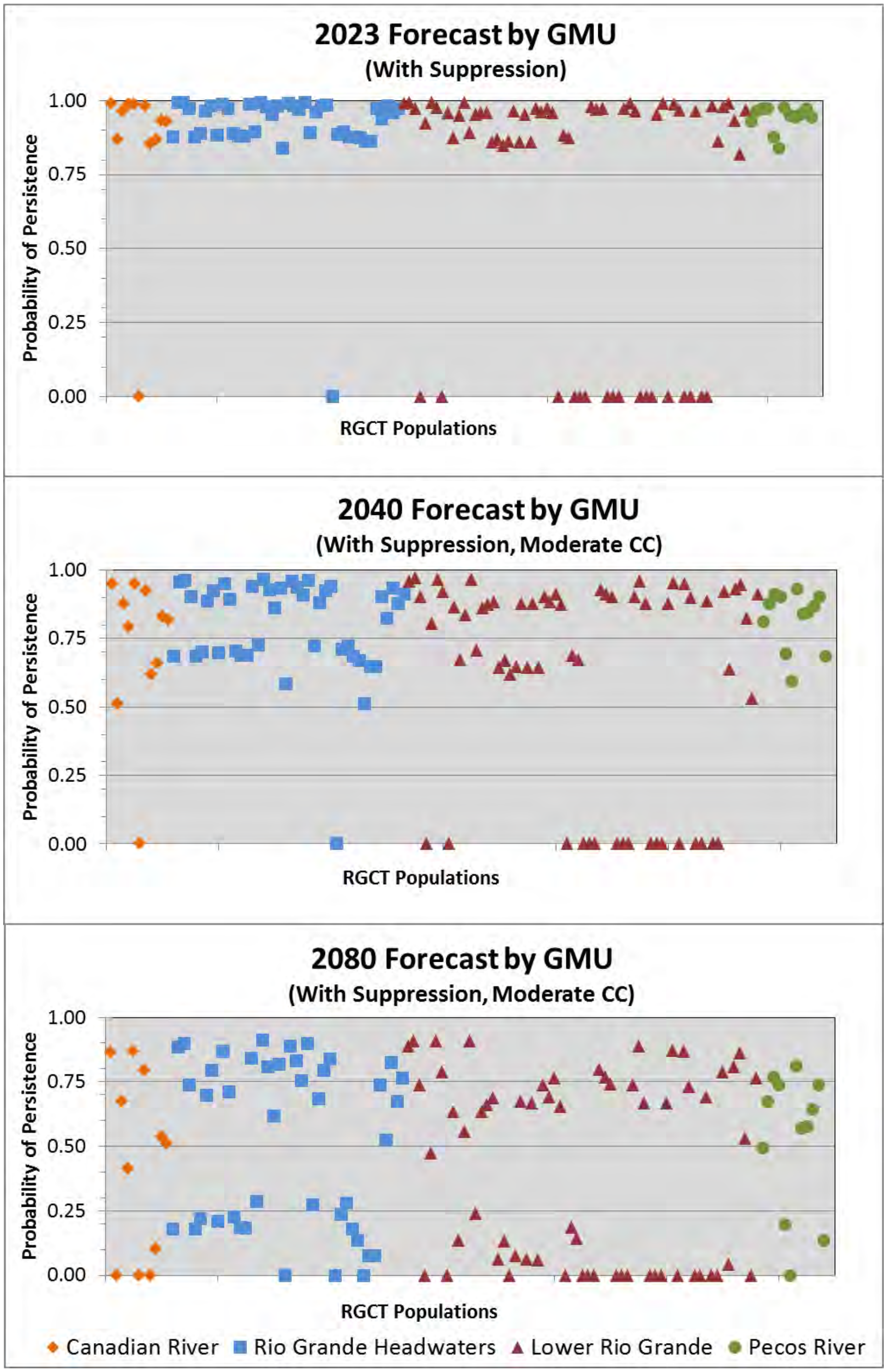


Figure 5. Probability of persistence for each Rio Grande cutthroat trout population. The forecasts include suppression of competing nonnative trout and the 2040 and 2080 forecasts include moderate climate change conditions.

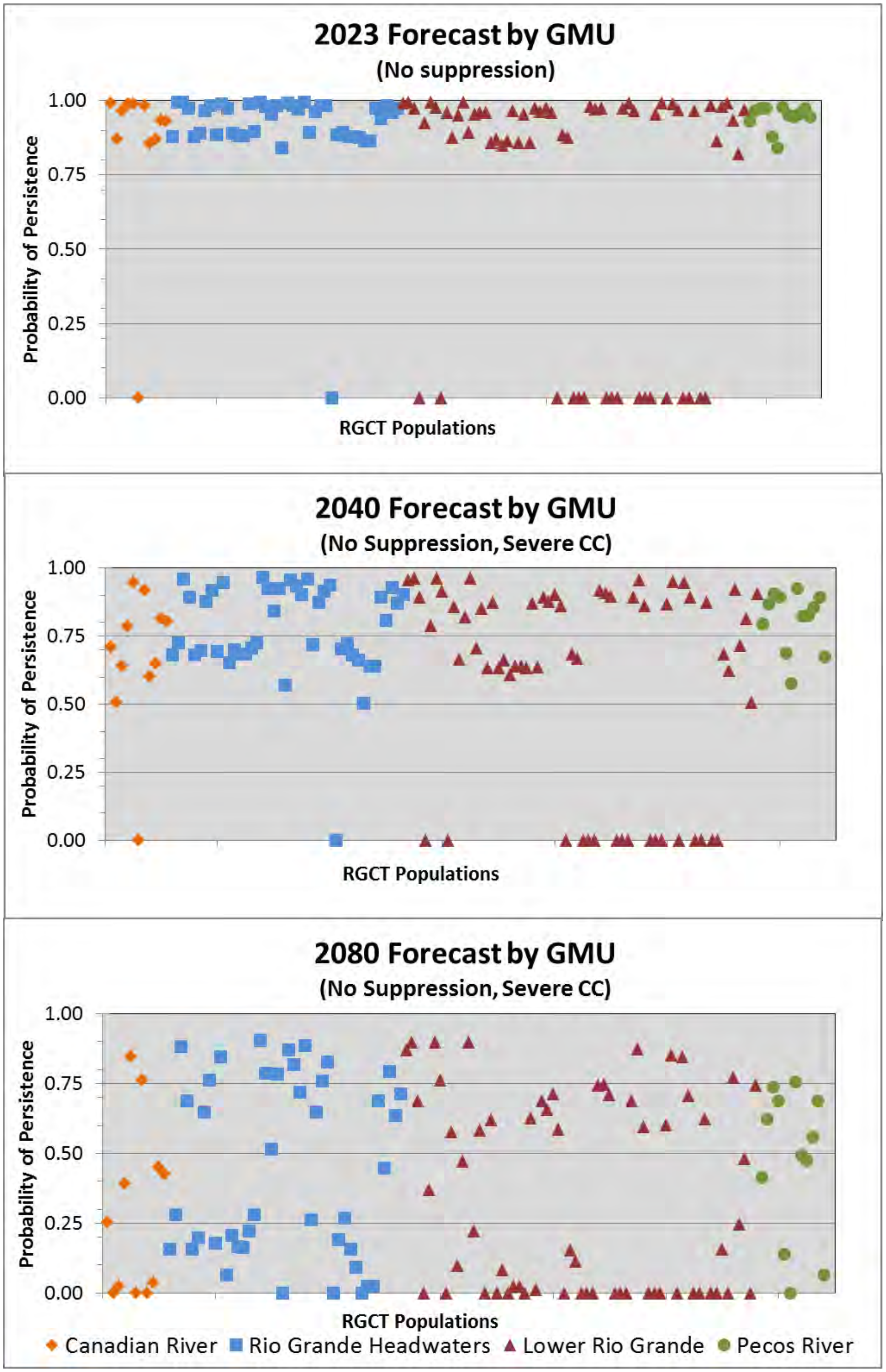


Figure 6. Probability of persistence for each Rio Grande cutthroat trout population. The forecasts include no suppression of competing nonnative trout and the 2040 and 2080 forecasts include severe climate change conditions.

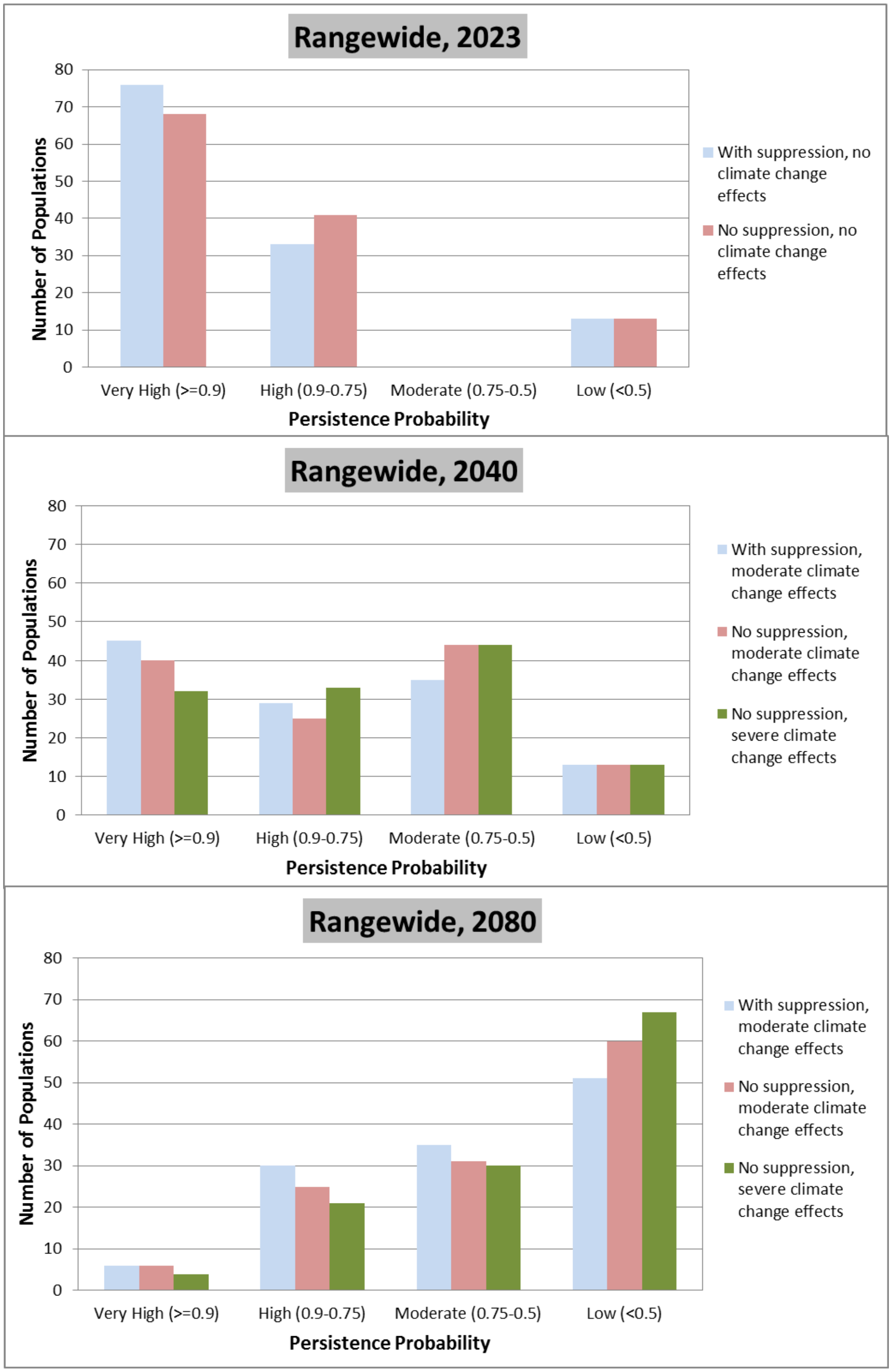


Figure 7. Frequency distributions of Rio Grande cutthroat trout populations rangewide based on their probability of persistence in 2023 (top graph), 2040 (middle graph), and 2080 (bottom graph).

5.3.2 Canadian GMU Populations, Probability of Persistence

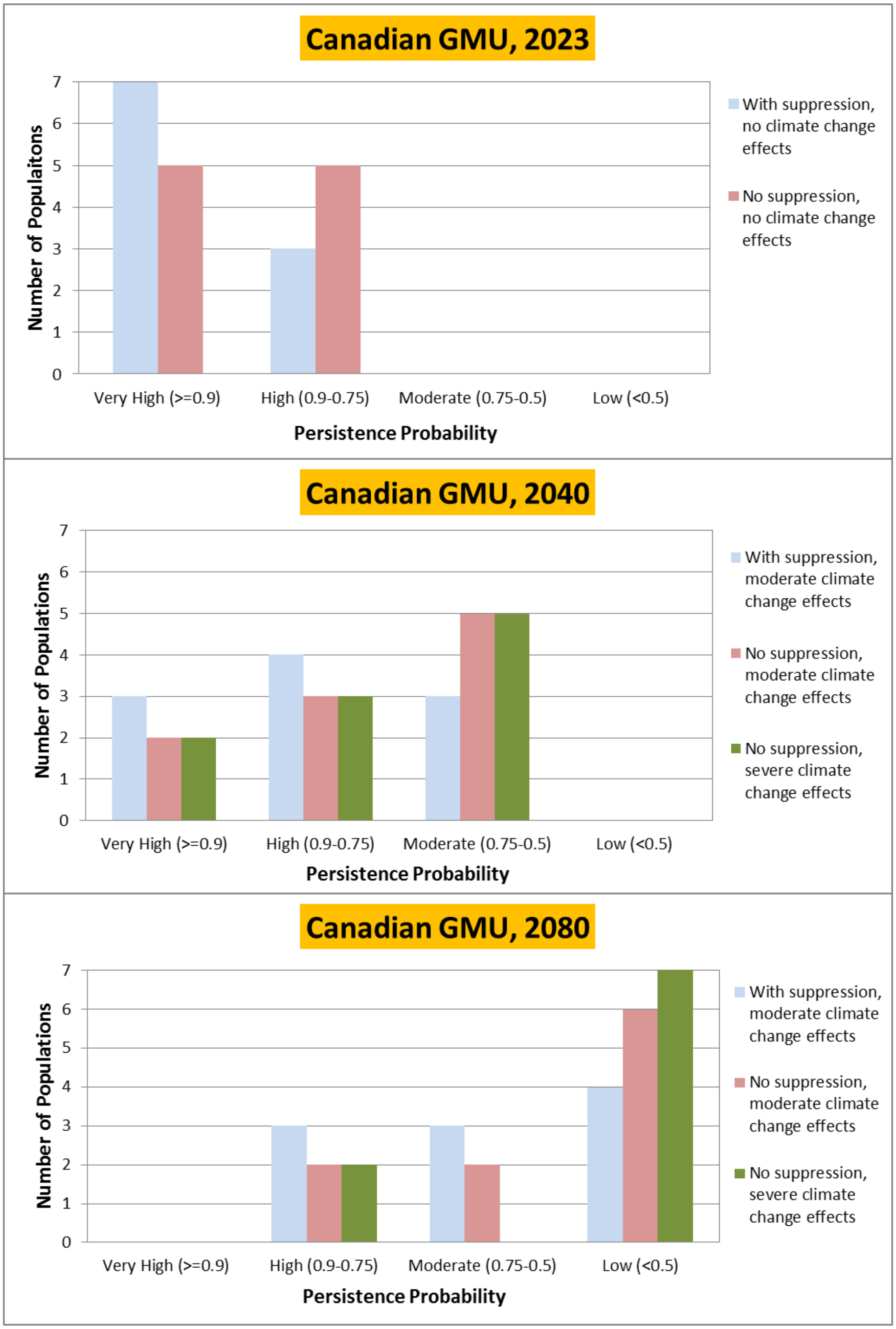


Figure 8. Frequency distributions of Rio Grande cutthroat trout populations in the Canadian GMU based on their probability of persistence in 2023 (top graph), 2040 (middle graph), and 2080 (bottom graph).

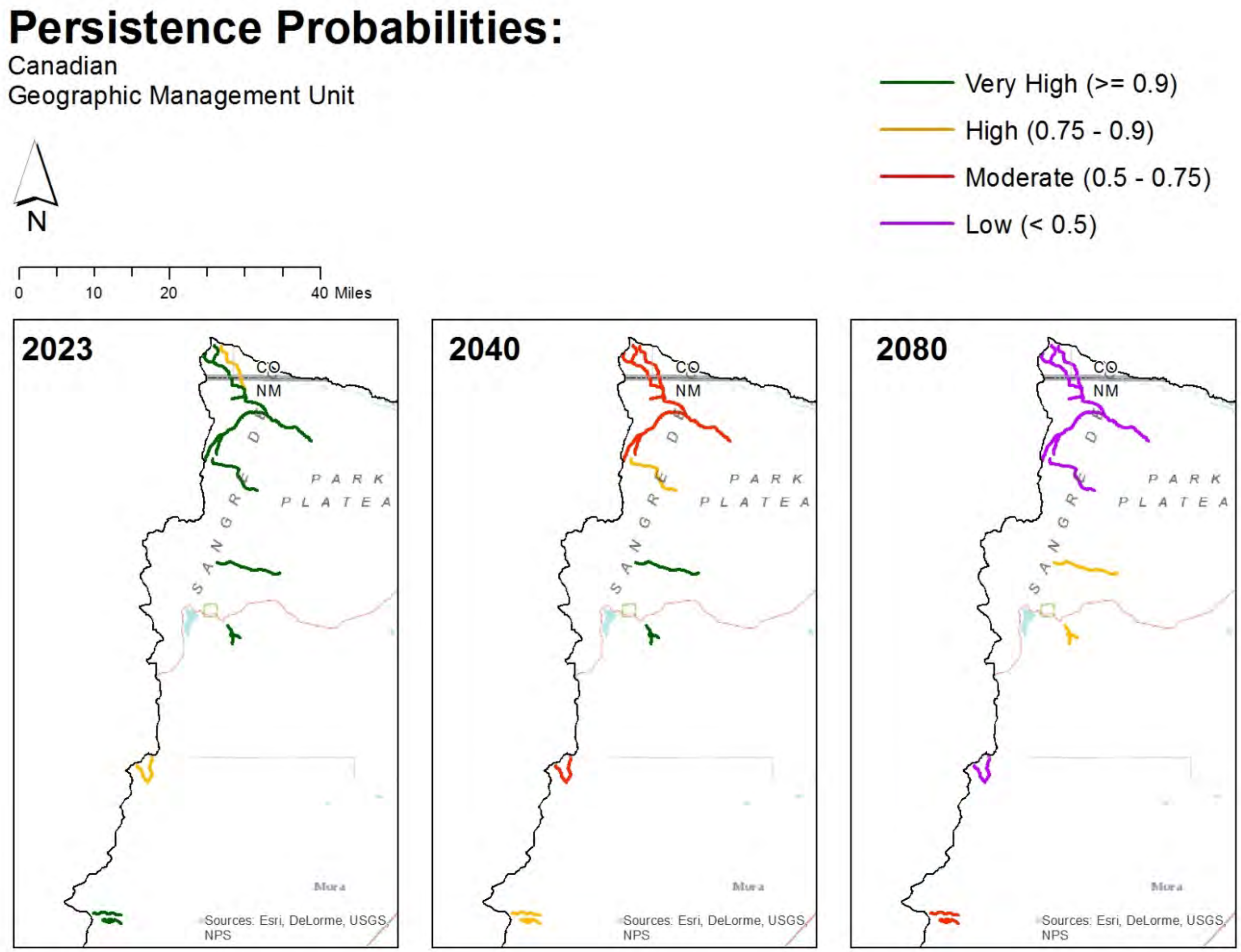


Figure 9. Locations of Rio Grande cutthroat trout populations in the Canadian GMU based on their probability of persistence in 2023, 2040, and 2080. The 2023 map reflects results with competitive nonnative trout suppression. The 2040 and 2080 maps reflect results with no competitive nonnative trout suppression and moderate climate change effects.

5.3.3 Rio Grande Headwaters GMU Populations, Probability of Persistence

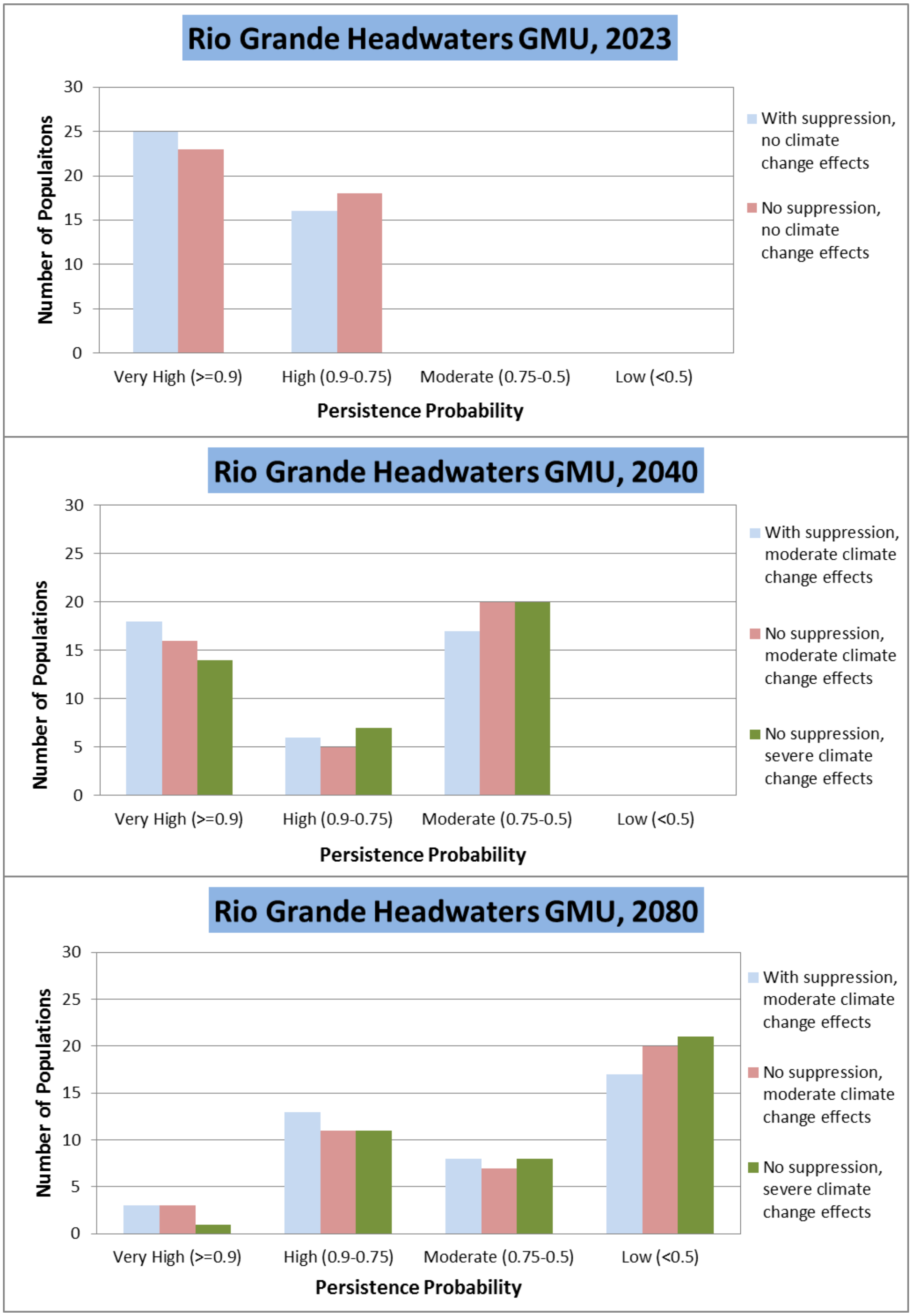


Figure 10. Frequency distributions of Rio Grande cutthroat trout populations in the Rio Grande Headwaters GMU based on their probability of persistence in 2023, 2040, and 2080.

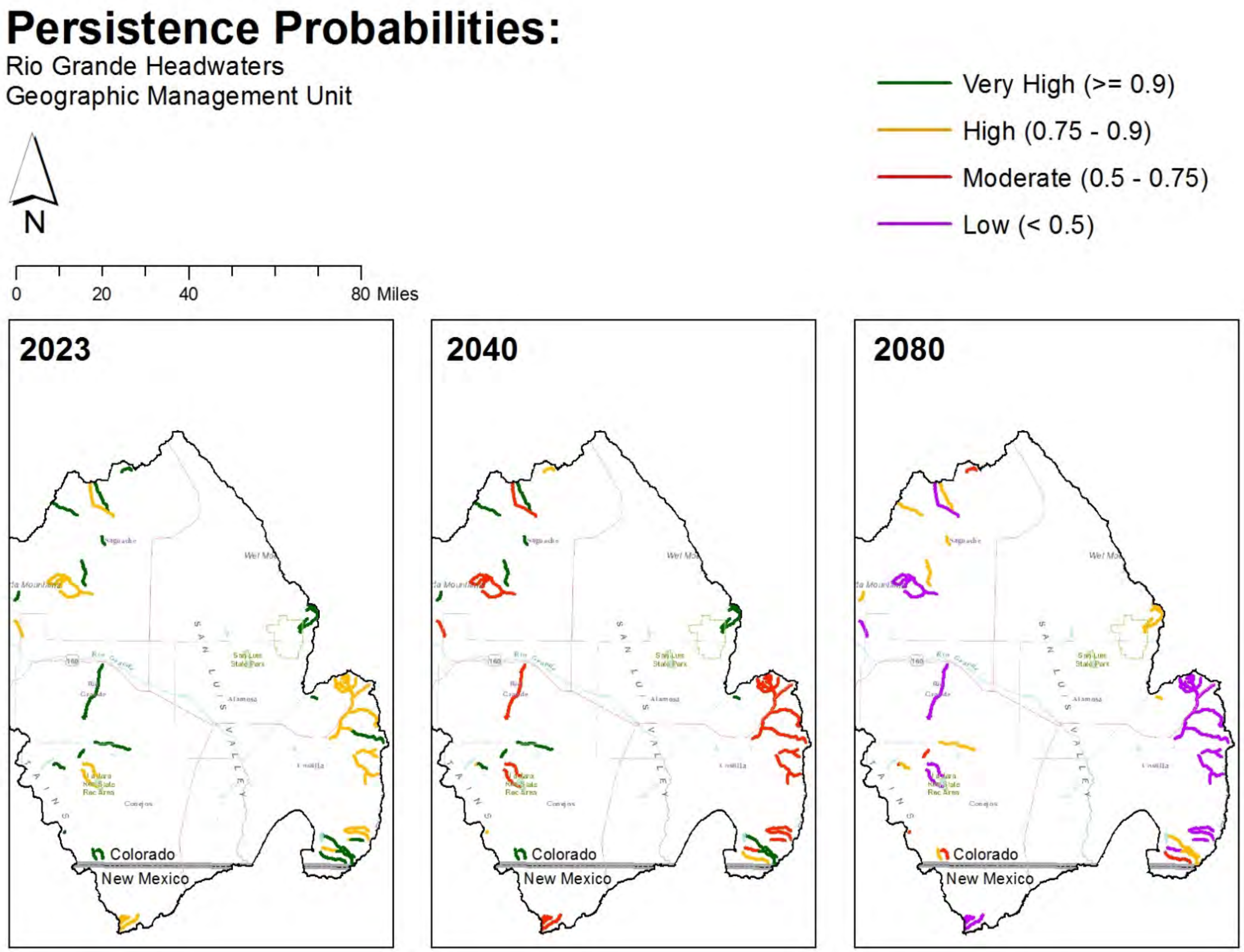


Figure 11. Locations of Rio Grande cutthroat trout populations in the Rio Grande Headwaters GMU based on their probability of persistence in 2023, 2040, and 2080. The 2023 map reflects results with competitive nonnative trout suppression. The 2040 and 2080 maps reflect results with no competitive nonnative trout suppression and moderate climate change effects.

5.3.4 Lower Rio Grande GMU Populations, Probability of Persistence

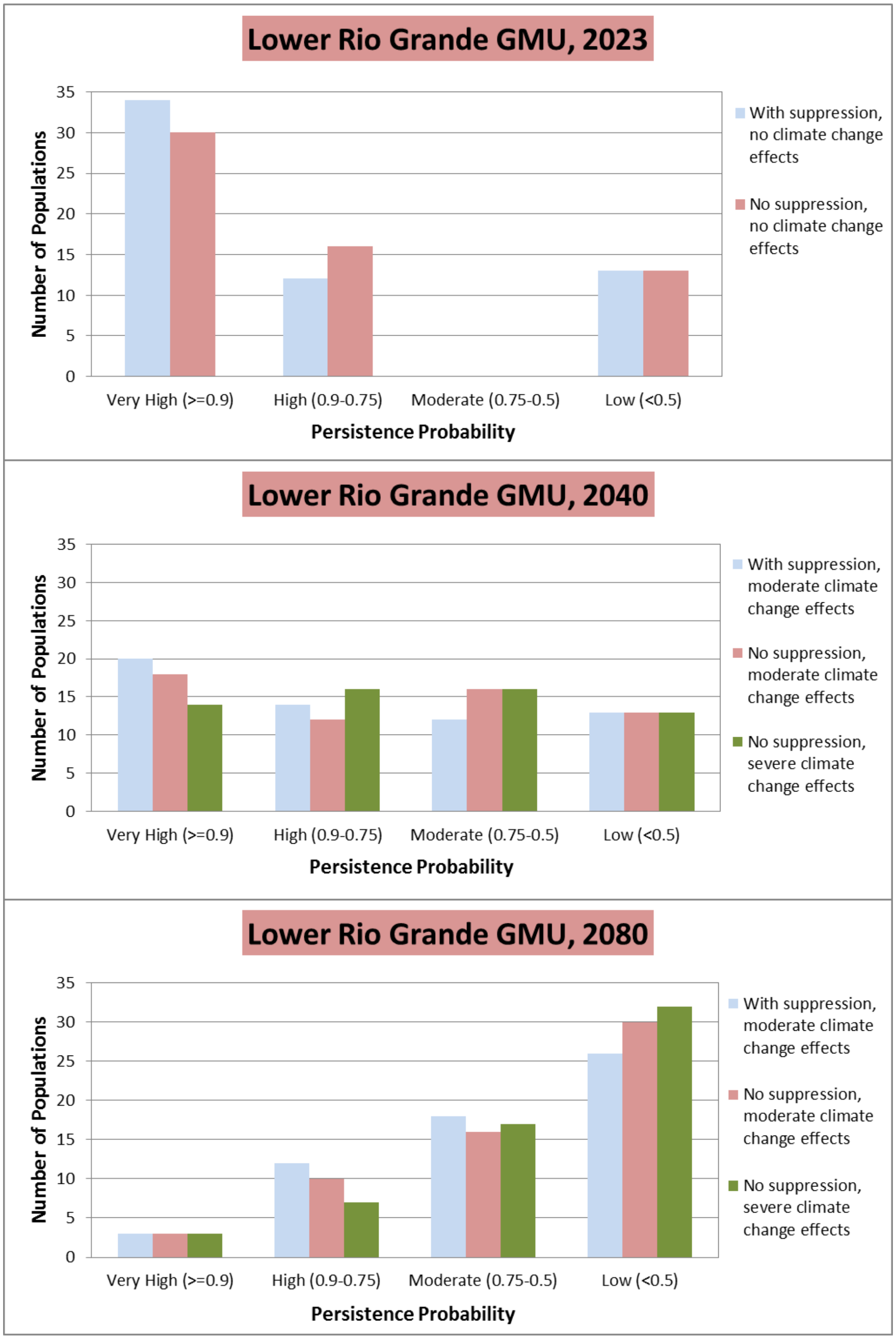


Figure 12. Frequency distributions of Rio Grande cutthroat trout populations in the Lower Rio Grande GMU based on their probability of persistence in 2023, 2040, and 2080.

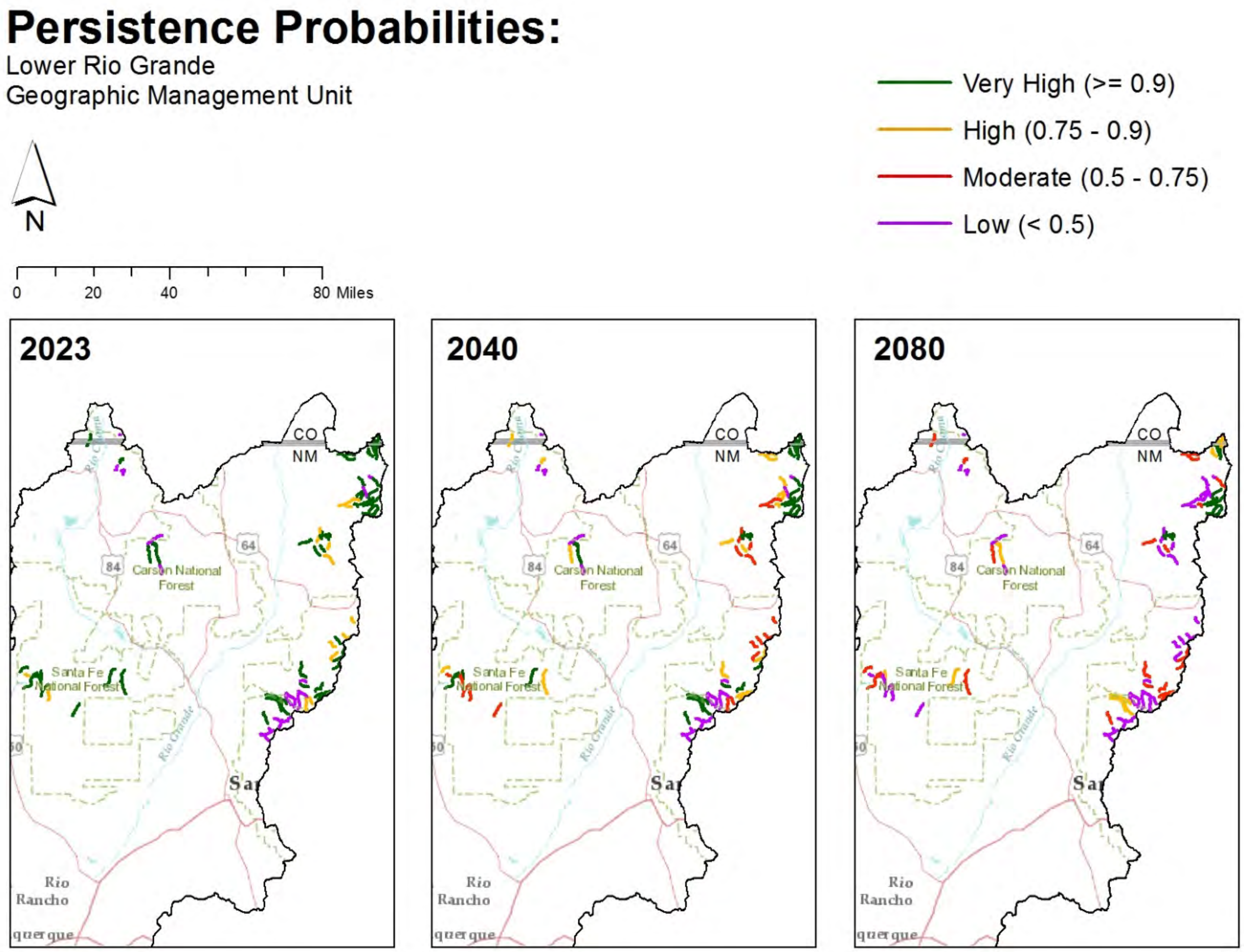


Figure 13. Locations of Rio Grande cutthroat trout populations in the Lower Rio Grande GMU based on their probability of persistence in 2023, 2040, and 2080. The 2023 map reflects results with competitive nonnative trout suppression. The 2040 and 2080 maps reflect results with no competitive nonnative trout suppression and moderate climate change effects.

5.3.5 Pecos GMU Populations, Probability of Persistence

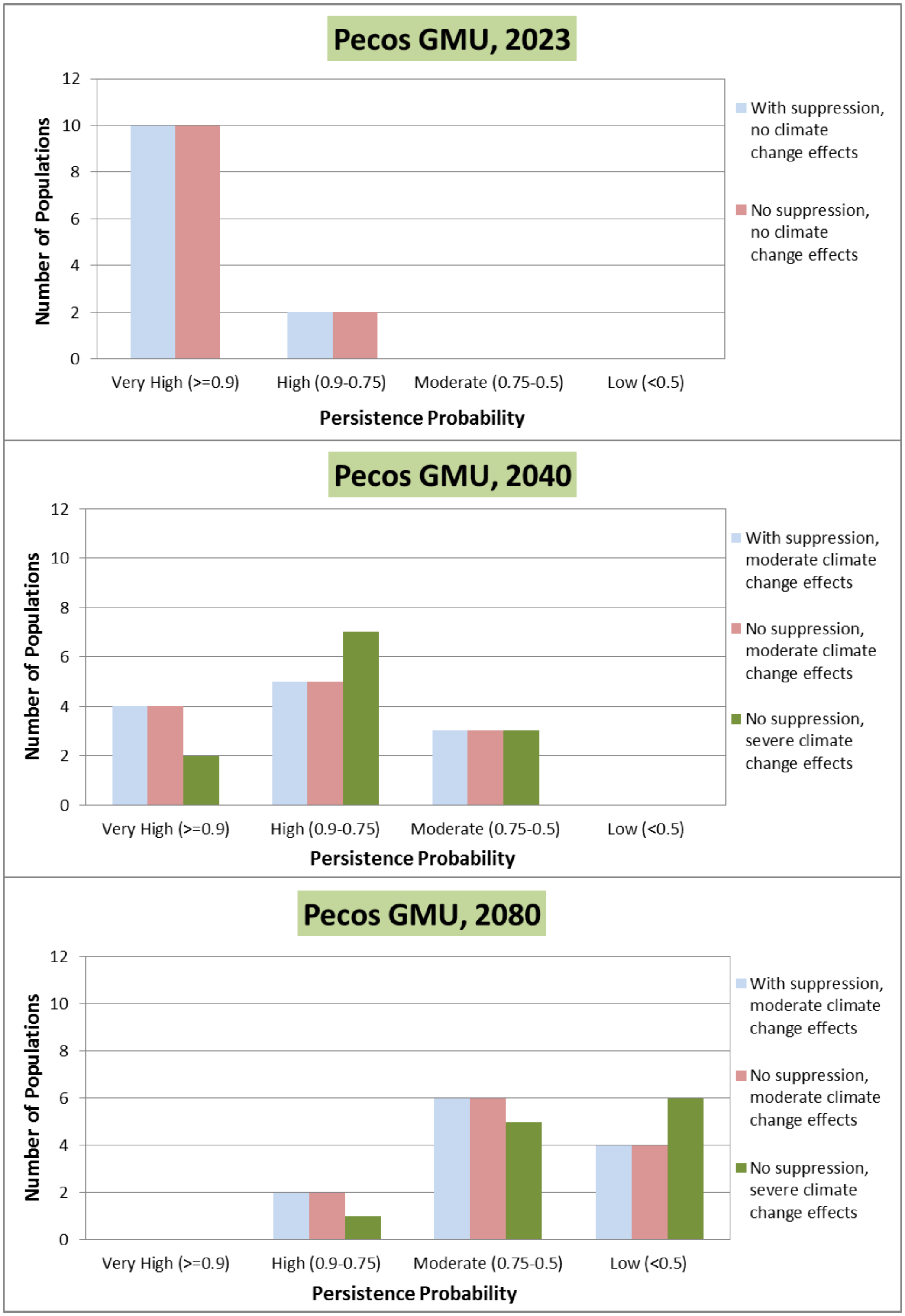


Figure 14. Frequency distributions of Rio Grande cutthroat trout populations in the Pecos GMU based on their probability of persistence in 2023, 2040, and 2080.

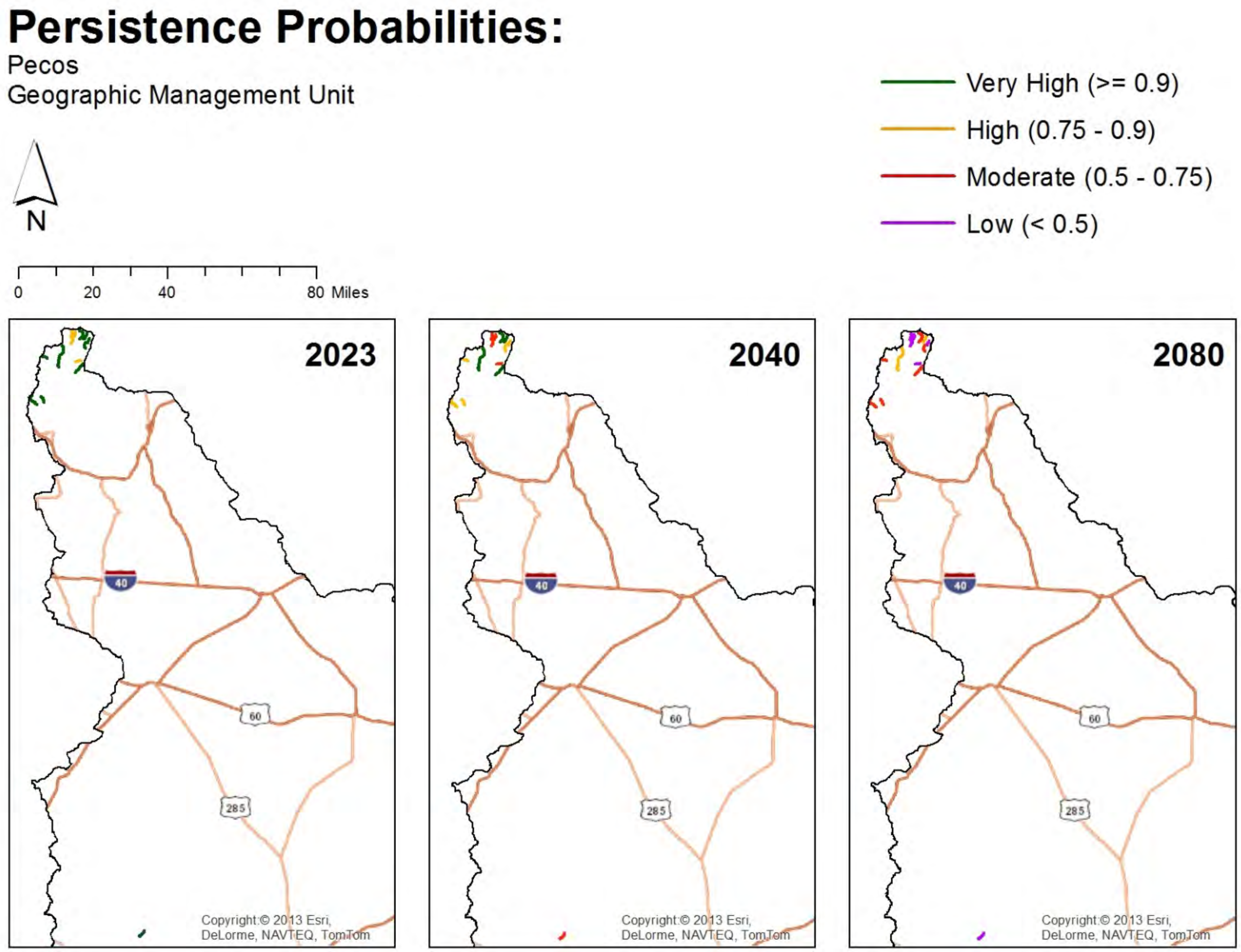


Figure 15. Locations of Rio Grande cutthroat trout populations in the Pecos GMU based on their probability of persistence in 2023, 2040, and 2080. The 2023 map reflects results with competitive nonnative trout suppression. The 2040 and 2080 maps reflect results with no competitive nonnative trout suppression and moderate climate change effects.

5.4 Results: Module 2, Population Survival

The rangewide results of the population survival estimate are provided in Figure 16 for each of the 9 scenarios identified in Table C12 of Appendix C. There are currently 122 extant conservation populations as of 2013. Our analysis suggests that by 2023 the number of populations surviving (that is, forecasted to be persisting and not extirpated) ranges between 104 and 131; by 2040 the range is between 86 and 148; and by 2080 the range is between 50 and 132 populations surviving (Figure 16).

The same results are broken down geographically by GMU in Figures 17 and 18 and Table 4. We displayed the output based on 3 of our scenarios to show a range of estimates (Table 3). The low estimate is scenario 2 (worst case estimate with low management and severe climate change effects) (Appendix C, Table C12). The high estimate is scenario 7 (best case with high management and moderate climate change effects) (Appendix C, Table C12). The Canadian GMU currently has 10 extant populations and by 2080 is forecasted to have between 3 (worst case) and 14 (best case) populations surviving (intermediate case, 6) (Figure 17, Table 4). The Pecos GMU currently has 12 extant populations and by 2080 is forecasted to have between 5 and 16 populations surviving (intermediate, 8) (Figure 17, Table 4). The Rio Grande Headwaters GMU currently has 41 extant populations and by 2080 is forecasted to have between 21 and 55 populations surviving (intermediate, 27) (Figure 18, Table 4). The Lower Rio Grande GMU currently has 59 extant populations and by 2080 is forecasted to have between 21 and 47 populations surviving (intermediate, 28) (Figure 18, Table 4).

Table 3. Summary of three population survival scenarios. Results are displayed in the population survival module, below. These three scenarios represent the overall best, intermediate, and worst cases evaluated in the model.

Scenarios		Climate Change	Nonnative Suppression	Population Simulation	Population Restoration
2	Worst Case	Severe	No	Lower 95% Conf. Interval	Low
6	Intermediate Case	Moderate	No	Mean	Low
7	Best Case	Moderate	Yes	Upper 95% Conf. Interval	High

5.4.1 Rangewide Forecasts

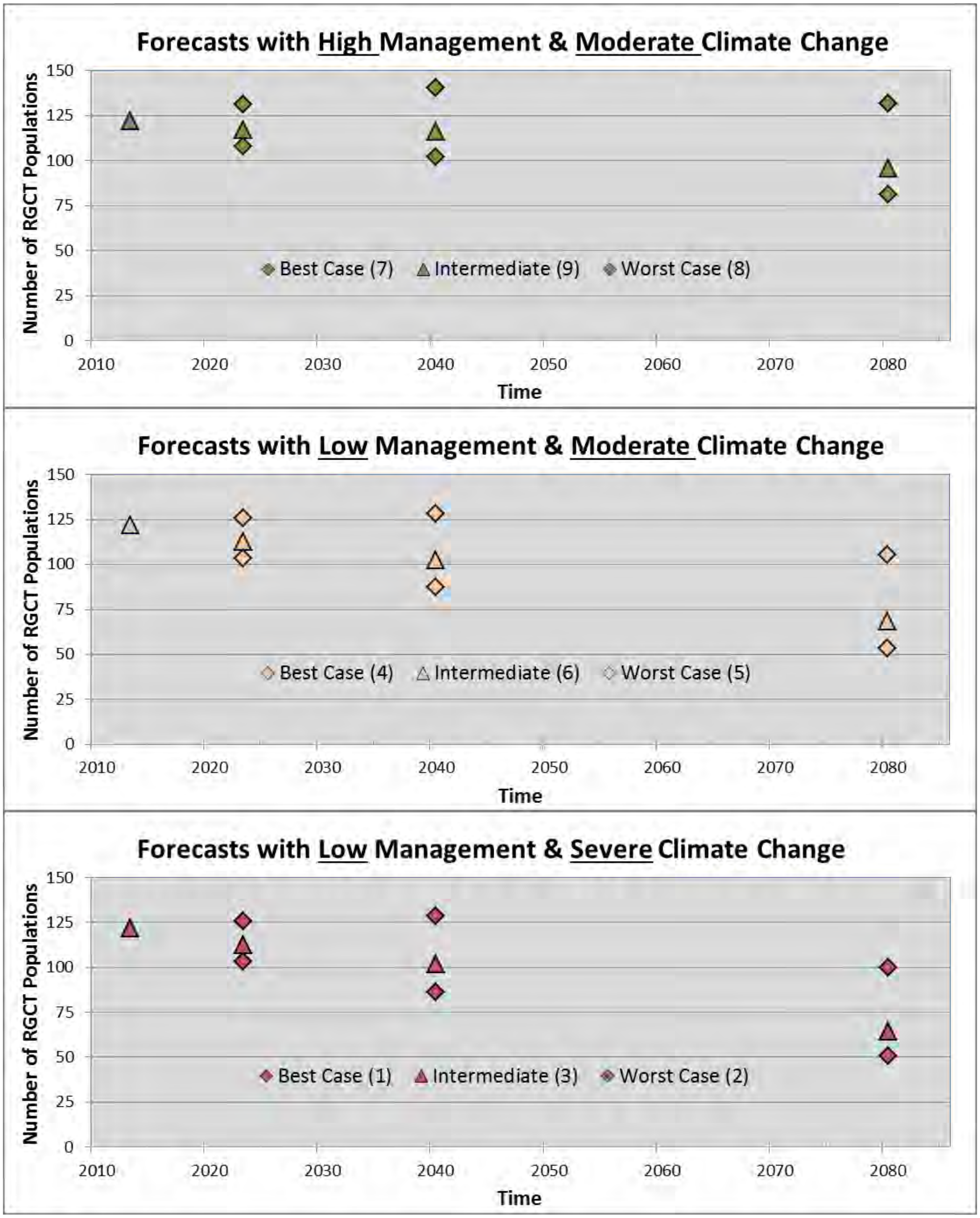


Figure 16. Range of forecasted number of surviving Rio Grande cutthroat trout populations in 2023, 2040, and 2080. Top graph contains scenarios 7-9; center graph contains scenarios 4-6; and bottom graph contains scenarios 1-3 (Appendix C, Table C12). Each graph represents the best, intermediate, and worst cases for the specified level of management and climate change.

5.4.2 Forecasts by GMU

Table 4. Range of forecasted number of surviving Rio Grande cutthroat trout populations in 2023, 2040, and 2080 by GMU. Scenarios represented are found in Appendix C, Table C12.

Canadian GMU			
	Scenarios		
Year	Best (7)	Intermediate (6)	Worst (2)
2013	10	10	10
2023	13.8	10.2	8.5
2040	14.5	9.5	6.6
2080	13.5	5.6	3.1

Pecos GMU			
	Scenarios		
Year	Best (7)	Intermediate (6)	Worst (2)
2013	12	12	12
2023	15.8	12.3	10.9
2040	16.9	11.8	9.5
2080	15.6	8.3	4.8

Rio Grande Headwaters GMU			
	Scenarios		
Year	Best (7)	Intermediate (6)	Worst (2)
2013	41	41	41
2023	49.2	44.2	41.0
2040	55.7	39.5	34.5
2080	55.3	26.8	21.1

Lower Rio Grande GMU			
	Scenarios		
Year	Best (7)	Intermediate (6)	Worst (2)
2013	59	59	59
2023	51.4	45.9	43.0
2040	52.6	41.5	35.5
2080	46.6	27.9	21.4

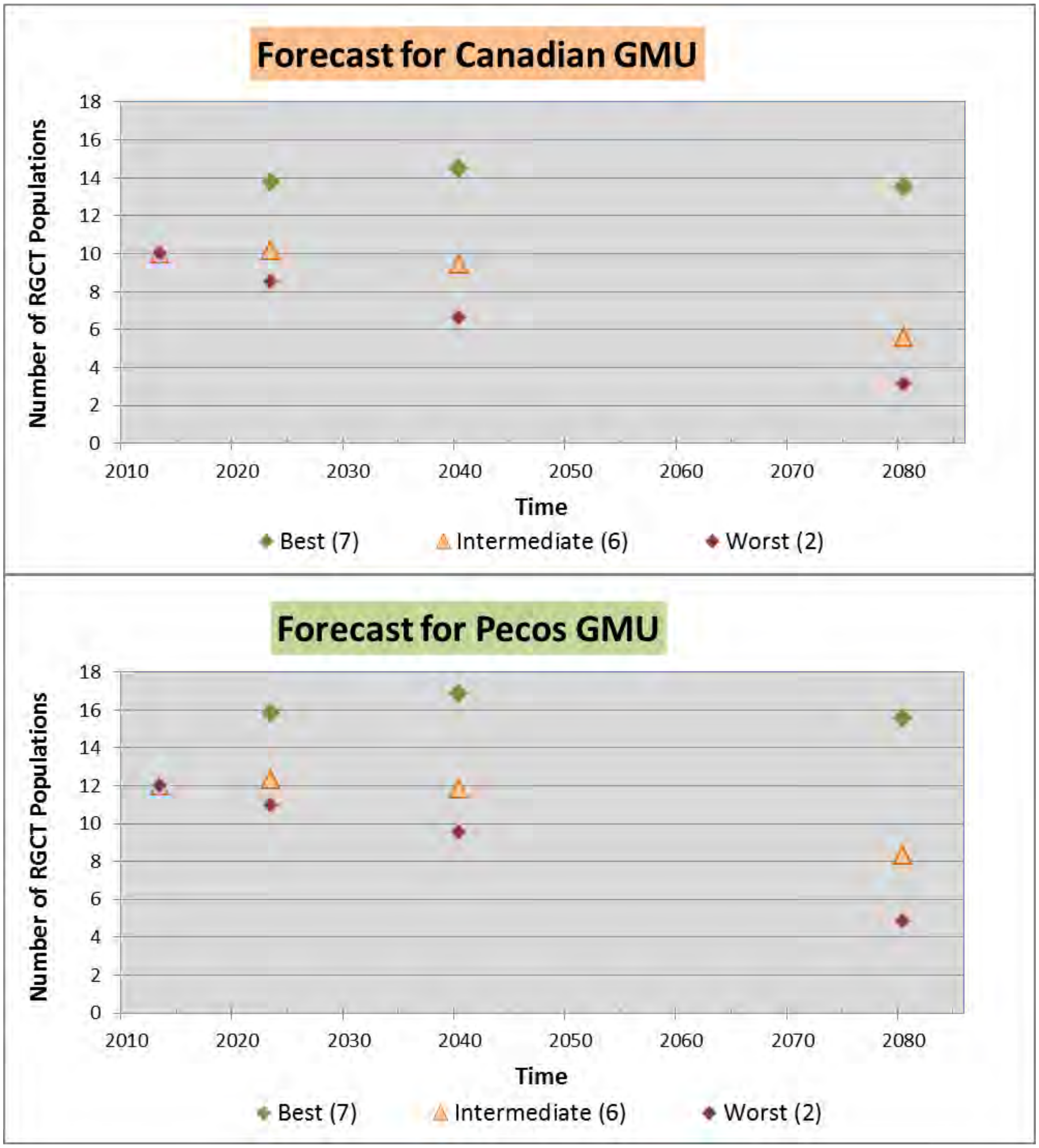


Figure 17. Range of forecasted number of surviving Rio Grande cutthroat trout populations in 2023, 2040, and 2080 in Canadian (top graph) and Pecos (bottom graph) GMUs. Best, intermediate, and worst estimates are from scenarios 7, 6, and 2, respectively (Appendix C, Table C12).

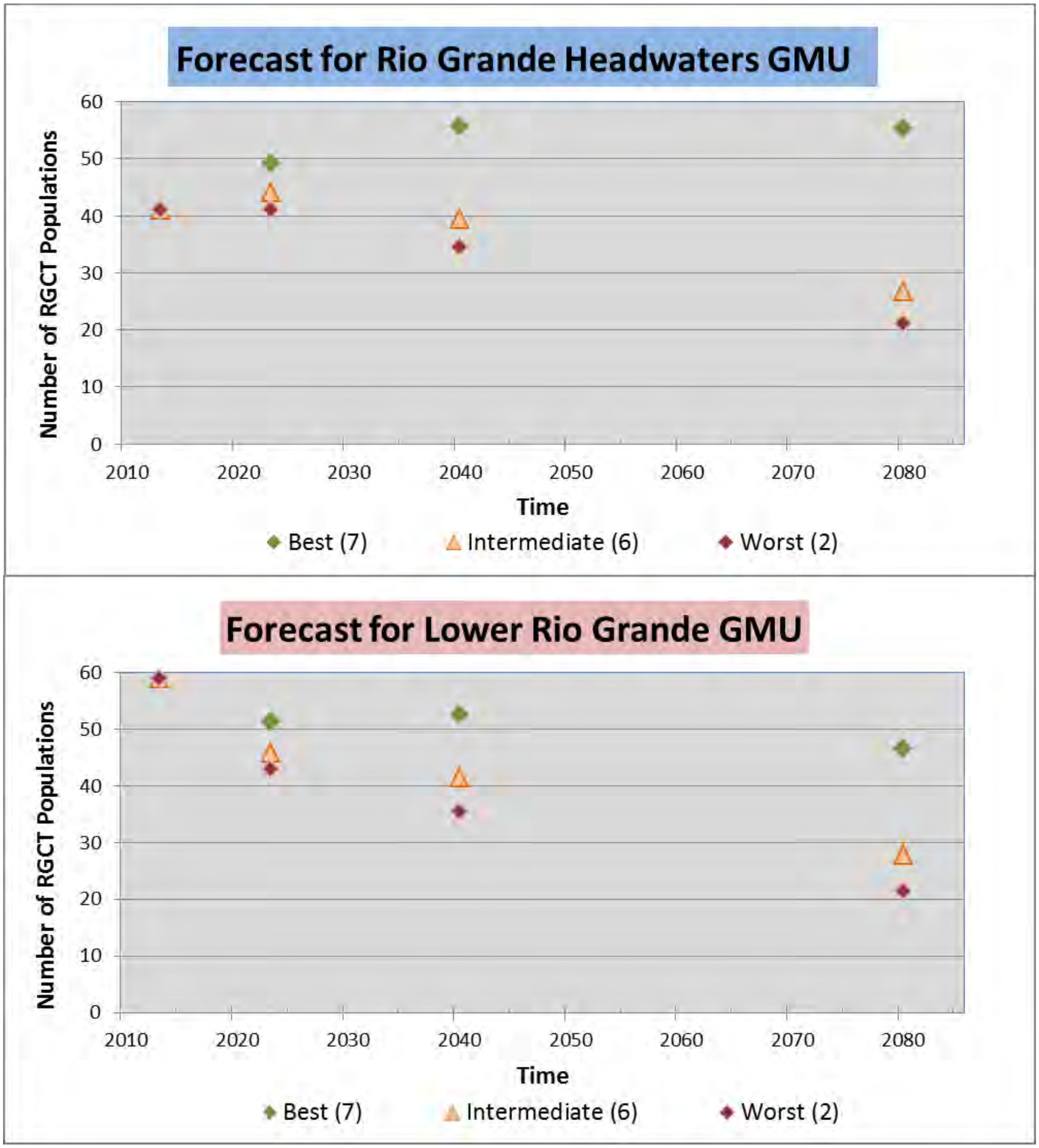


Figure 18. Range of forecasted number of surviving Rio Grande cutthroat trout populations in 2023, 2040, and 2080 in Rio Grande Headwaters (top graph) and Lower Rio Grande (bottom graph) GMUs. Best, intermediate, and worst estimates are from scenarios 7, 6, and 2, respectively (Appendix C, Table C12).

5.5 Results: Stream Length Forecasting

For the results of forecasting the total occupied stream lengths, we plotted the historical, current, and forecasted stream lengths over time (Figure 19). The historical data (estimated 10,696 stream km) was plotted as 1905 just to provide a temporal context on the graph (Alves *et al.* 2008, p. 8, indicates historical was circa 1800). We displayed the output based on 2 of our scenarios to show a range of estimates (worst case, scenario 3, and best case, scenario 9).

The current (2013) estimate for total stream kilometers occupied by Rio Grande cutthroat trout is 1,149 km (about 11% of historical totals). By 2040, we estimate the range of occupied stream kilometers (based on the estimated number and length of surviving populations) to be between 1,076 and 1,292 km (10.1% to 12.1% of historical totals). By 2080, we estimate the range of occupied stream kilometers to be between 722 and 1,186 km (6% to 11.1% of historical totals)[5].

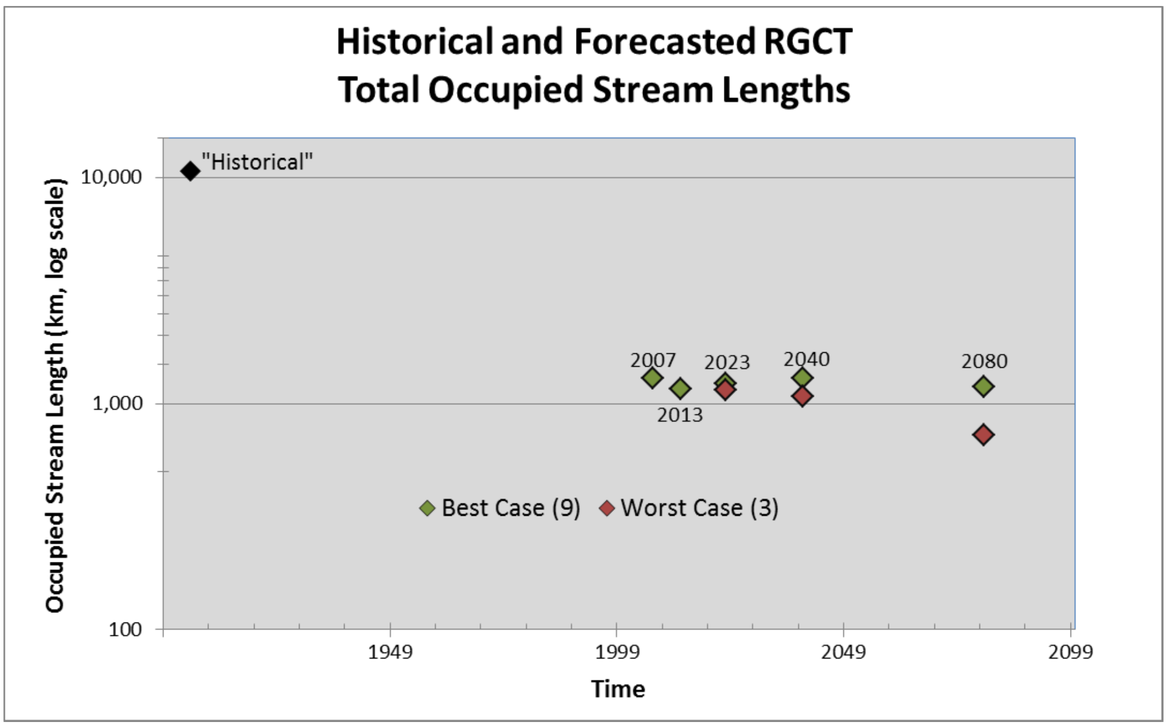


Figure 19. Historical, current, and forecasted total stream lengths estimated to be occupied by Rio Grande cutthroat trout. Historical estimate is plotted as the year 1905 just for display purposes. Low estimate and high estimate use scenarios 3 and 9 (Appendix C, Table C12).

[5] Note the discussion in *Appendix C, Methods, Occupied Stream Length Forecasting* regarding the large uncertainties and low confidence in these forecasted estimates of occupied stream lengths.

5.6 Viability Discussion

We defined viability as a description of the ability of a species to persist over time and thus avoid extinction. "Persist" and "avoid extinction" mean that the subspecies is expected to sustain populations in the wild beyond the end of a specified time period. We are defining the Rio Grande cutthroat trout viability by characterizing the status of the subspecies in terms of its resiliency, redundancy, and representation. Assessing these conditions does not result in a threshold determination (i.e., the subspecies is or is not resilient), but instead we present the results as a risk analysis that reflects our understanding of the relationship between the subspecies' condition, the risk factors it faces, and a range of forecasted possible outcomes in terms of the probability of persistence in the future at the population and subspecies, rangewide, level.

To evaluate the viability Rio Grande cutthroat trout we first determined conceptually what the subspecies needs for viability. We have summarized these needs in Table 5 (Column 2) beginning with what populations need for resiliency. We then assessed the current condition of the subspecies based on how those needs currently are or are not being met at the population and rangewide scales (Table 5, Column 3). Finally, we used our status assessment model (Appendix C) to forecast the possible future conditions of the subspecies based on the number of populations expected to persist given our understanding of the risks faced by each of the current populations and the expectations for future restoration of populations (Table 5, Column 4). The following discusses our results organized around each of the 3Rs.

Table 5. Overall summary of species status assessment for Rio Grande cutthroat trout. ("RG" = Rio Grande, "pop's" = populations)

3 R's	NEEDS	CURRENT CONDITION	FUTURE CONDITION (VIABILITY)
Resiliency: <u>Population</u> ***(large populations to withstand stochastic events)***	• Large Effective Population Sizes (effective population sizes >500 are best). • Long Streams for Habitat (streams greater than 9.65 km are best). • Free of Nonnative Trout (mainly rainbow and brown trout) and Disease (whirling). • High Quality Habitat (water temps < critical summer maximums).	• 122 Extant Populations across range. * 55 (45%) of populations are currently in the <u>best or good</u> condition *(based on absense of nonnative trout, effective population size, and occuppied stream length)* * 67 (55%) of populations are currently in <u>fair or poor</u> condition.	• Status assessment model estimates probability of persistence for each population based on risks from: * Effective Population Size. * Nonnatives (hybridization, competition) and Disease. * Wildfire and Stream Drying . * Water Temperature Increase. • Included climate change considerations for increased risks.
Resiliency: Subspecies ***(populations to withstand stochastic events)***	• Multiple interconnected resilient populations.	• About 11% of historic range remains occupied due to past impacts from nonnatives. • Populations are isolated (16 populations have some connectedness).	• 2080 model forecasts future populations persisting; results range depending on future management level and severity of climate change: reporting **best to worst (intermediate)** results: * **50 to 132 (69)** populations rangewide. • Limited opportunity to regain interconnectedness of populations (due to pervasive nonnative trout).
Redundancy ***(number and distribution of populations to withstand catostrophic events)***	• Multiple highly resilient populations within each of the 4 Geographic Management Units (GMUs).	• Current total number of populations persisting by GMU: * 41 pop's in RG Headwaters GMU. * 59 pop's in Lower RG GMU. * 10 pop's in Canadian GMU. * 12 pop's in Pecos GMU.	• 2080 model forecasts for future populations persisting by GMU: * **21 to 55 (27)** pop's in RG Headwaters. * **21 to 47 (28)** pop's in Lower RG. * **3 to 14 (6)** pop's in Canadian. * **5 to 16 (8)** pop's in Pecos.
Representation ***(genetic and ecological diverstiy to maintain adaptive potential)***	• Genetic variation exists between 1) Two GMUs in the Rio Grande Basin and 2) Two GMUs in Canadian and Pecos River Basins. • Unknown ecological variation, but we used GMUs as proxy.	• Current total populations persisting by Watershed: * 100 pop's in Rio Grande Basin. * 22 pop's in Candadian and Pecos GMUs.	• 2080 model forecasts for future populations persisting by watershed: * **42 to 102 (55)** pop's in Rio Grande Basin. * **8 to 30 (14)** pop's in Canadian and Pecos GMUs.

5.6.1 Resiliency

Resiliency is having sufficiently large populations for the subspecies to withstand stochastic events. Stochastic events are those arising from random events such as severe weather or wildfire. We measured resiliency at the population scale for the Rio Grande cutthroat trout by quantifying the persistence probability of each extant population under a range of assumed conditions. The results provide our best estimate of the resiliency of each population. The primary stochastic events facing Rio Grande cutthroat trout include wildfire, drought, and the invasion of nonnative species. The ability of Rio Grande cutthroat trout to withstand these events depends on the severity of the event and the current status of the population, such as the stream size, a surrogate measure of quantity and diversity of habitat. This ability to survive such events, in combination with the likelihood of such events happening, forms the basis of our population resiliency model and the results it produced.

The resiliency of each population is particularly important for the Rio Grande cutthroat trout because of the severe changes it has undergone in recent times. Rangewide, the resiliency of the subspecies has declined substantially due to the large decrease in overall distribution. In addition, the remnant Rio Grande cutthroat trout populations are now mostly isolated to headwater streams due to the fragmentation that has resulted from the historical, widespread introduction of nonnative trout across the range of Rio Grande cutthroat trout. Therefore, if an extant population is extirpated due to a localized event, such as a wildfire and subsequent debris flow, there is little to no opportunity for natural recolonization of that population. This reduction in resiliency results in a lower probability of persistence for the subspecies as a whole. To describe the remaining resiliency of the subspecies, we evaluated the individual populations in detail to understand the subspecies' overall capacity to withstand stochastic events.

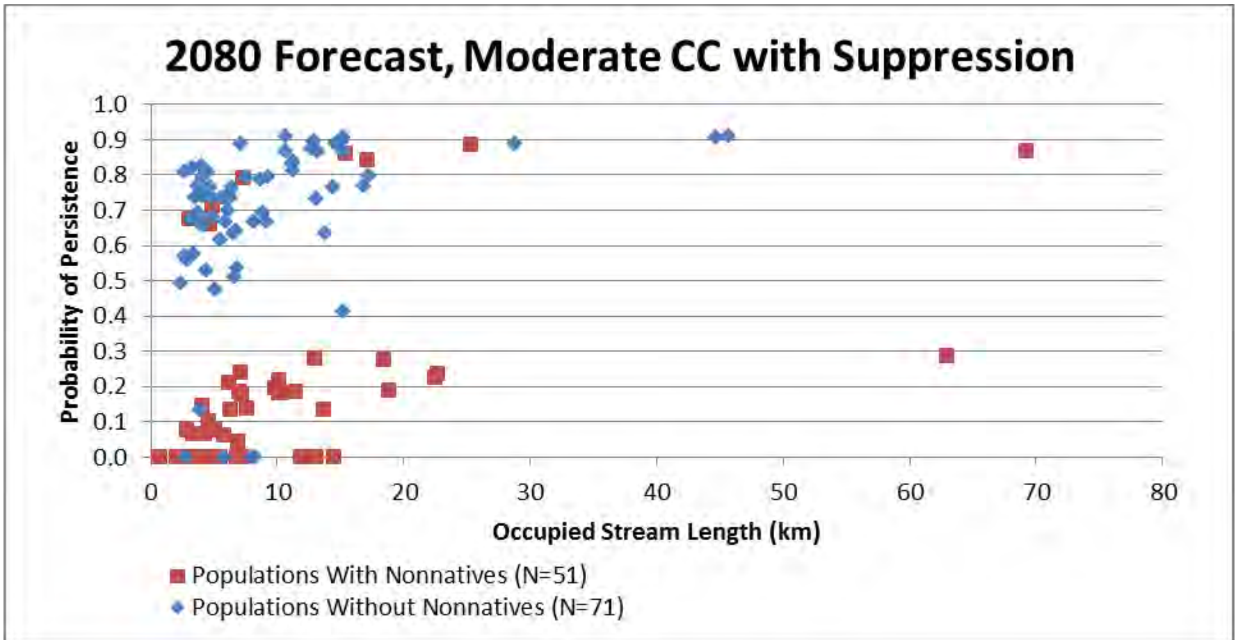
The factors threatening these populations generally have a relatively low risk of occurrence; however, if the stochastic events occur, they potentially have a high risk of resulting in substantial effects to a population, which could possibly result in extirpation (see Chapter 4 and Appendix B for a discussion of these factors). This relationship makes determining the cumulative risk of these stressors particularly difficult to assess and predict the outcome. Additionally, we were not able to quantitatively account for all potential synergistic effects between the risk factors due to the limitations in our analytical process. However, our probability of persistence module incorporates the risks in an explicit way to assess the estimated resiliency of the Rio Grande cutthroat trout.

As expected based on our methodology all of the population persistence probabilities decreased over time (Figures 5–15). This is because we built the model such that the risks associated with each factor increase over time in a linear relationship. As a result there are many populations whose probability of persistence decreases substantially by 2080. These results do not necessarily mean that any one of the populations will, in fact, be extirpated by 2080, but they simply reflect the risks that we believe the populations face due to their current conditions and the factors influencing their resiliency.

One of the most important factors affecting these results is the presence of nonnative trout. We assigned a relatively high risk function to populations with co-occurring populations of brown trout, brook trout, or rainbow trout where no management suppression is happening. Fifty populations of Rio Grande cutthroat trout currently co-occur with competitive nonnative trout populations, and five populations co-occur with rainbow trout, so this factor has a large influence on the overall viability of the subspecies. Figure 21 highlights the difference in the resulting probabilities of persistence for populations with and without nonnative trout, as the results cluster into two groups. In addition, 10 populations where nonnative trout co-occur with Rio Grande cutthroat have higher probabilities of persistence because of the active management suppression that is reducing the risk of extirpation of those populations (Figure 21, top graph).

The other important factor in the population resiliency is the occupied stream length. Our model incorporated this metric into two of the risk factors—wildfire and stream drying. It also indirectly affects demographic risks because longer streams generally have larger effective population sizes and for some streams lacking population size data we used stream length to estimate effective population size. There was not a statistically significant relationship between stream length and probability of persistence, although our results indicate a general trend of increasing probabilities of persistence as the stream length increases (Figure 21). The lack of correlation suggests that this factor alone was not the driving factor in determining overall probabilities of persistence, but other factors were important as well.

One of the main areas of uncertainty in our analysis is the potential effects of climate change, which we incorporated into four of the risk factors (hybridizing nonnative trout, wildfire, stream drying, and water temperature). Even under the case of severe climate change, which we estimated as a 40% increase in the risk factors by 2080, the overall results of the analysis were not substantially different compared to the moderate climate change scenarios (Figure 21). This does not necessarily mean that climate change may not be an important concern for the Rio Grande cutthroat trout, but it does reflect our current understanding of the best available information on the risks to the species from factors that may be influenced by future climate change. Given our current understanding and the best available information, the influence of climate change does not appear to be a dominant factor in the future persistence of Rio Grande cutthroat trout populations.



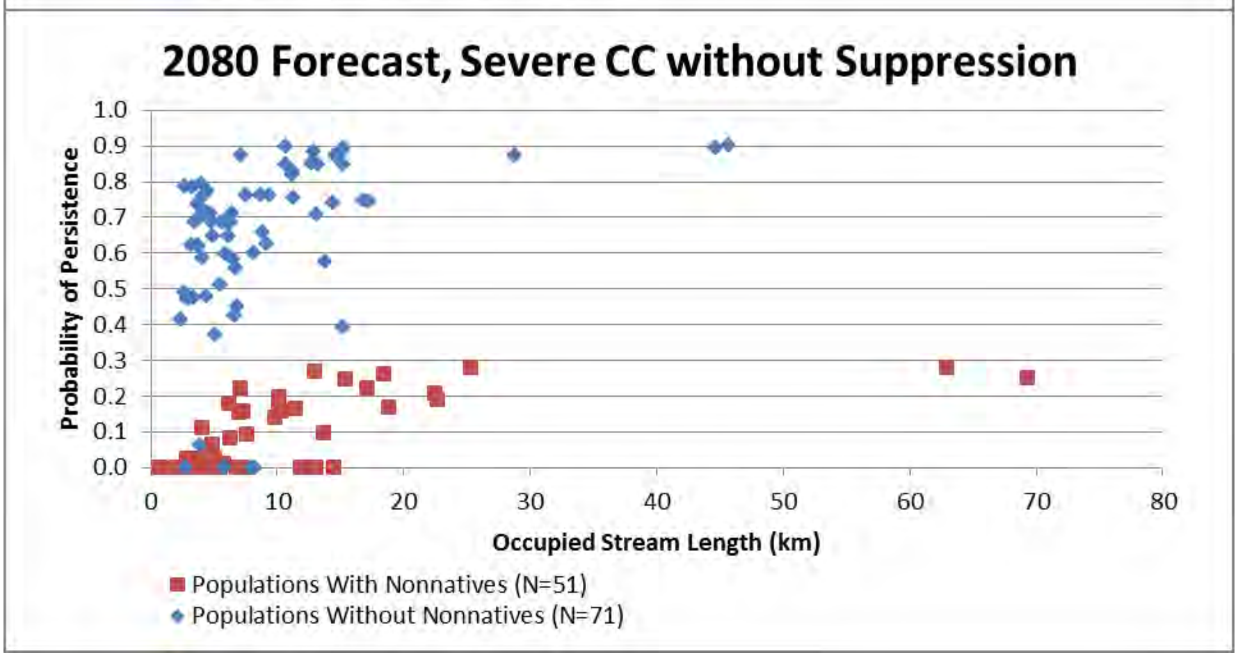


Figure 20. Forecasted probability of persistence of 122 Rio Grande cutthroat trout population compared to occupied stream length under two sets of conditions: moderate climate change effects with suppression of competing nonnative trout (top graph) and severe climate change without suppression of nonnative trout (bottom graph). Populations are designated as those occurring with nonnative trout (blue diamonds) and those not occurring with nonnative trout (red squares). Populations in the upper graph that are co-occurring with nonnative trout and have persistence probabilities greater than 0.6 are those populations with management suppression of nonnative trout.

5.6.2 Redundancy

Redundancy is having sufficient numbers of populations for the subspecies to withstand catastrophic events. A catastrophic event is defined here as a rare destructive event or episode involving many populations and occurring suddenly. The most likely catastrophic event for the Rio Grande cutthroat trout that could affect a substantial portion of the subspecies' range would probably be related to a large-scale hydrologic anomaly, such as an extended drought that changed hydrologic conditions. Wildfire that affected a large portion of the subspecies' range could also result in a catastrophic event. For the Rio Grande cutthroat trout, we measured redundancy by forecasting the number and resiliency of populations distributed across the subspecies' range. The Rio Grande cutthroat trout needs multiple, highly resilient populations across the GMUs to maintain redundancy and high viability. This quality and distribution of populations would provide security to allow the species to withstand future catastrophic events and avoid extinction. The more resilient populations the subspecies has, and the more broadly they are distributed, the greater its redundancy. The subspecies currently has approximately 122 populations distributed across the four GMUs (Table 4), with populations per GMU ranging from 10 to 59.

We used the results of the persistence probabilities along with the number of estimated future restored populations to predict the number and location of future surviving populations by GMU under a range of possible conditions. The results suggest that, depending on the particular scenario considered related to risk factors and restoration efforts, the overall number of populations rangewide surviving by 2080 range from a low of 50 under the worst case scenario to a high of 132 under the best case scenario, with 68 in the intermediate case (Table 4). Some GMUs may decline more than others; for example, our forecasts suggest the Lower Rio Grande GMU could have the largest decline (Figure 16); we estimate the 59 current populations could decline to between 21 and 47 populations by 2080 (Table 4). The GMU with the least populations, the Canadian GMU (with 10 current populations), is forecasted to range between 3 and 14 populations by 2080 (Table 4). Based on our forecasts of persisting populations by 2080, it seems unlikely that a catastrophic event would eliminate the species from an entire GMU, because our forecasts suggest that populations will remain distributed throughout the four GMUs.

5.6.3 Representation

Representation is having the breadth of genetic and ecological diversity of the subspecies to adapt to changing environmental conditions. The only known important genetic structure within the Rio Grande cutthroat trout is between the two GMUs in the Rio Grande basin (Rio Grande Headwaters GMU and Lower Rio Grande GMU) and the other two GMUs (Canadian and Pecos GMUs). Together, the Pecos and Canadian GMUs have some genetic diversity that may be important to maintain for long-term viability. Although we are not aware of any specific ecological diversity across the subspecies' range that might be important for future adaptation, it would be prudent to maintain as much geographic extent of the subspecies range as possible to maintain any potential, but undetected, ecological diversity. To ensure adequate representation, it is important to retain populations in the Canadian and Pecos GMUs to maintain the Rio Grande cutthroat trout's overall potential genetic and life history attributes, buffering the subspecies' response to environmental changes over time. Therefore, we evaluated representation based on the extent of the geographical range as a proxy for considered ecological diversity expected to be maintained in the future as indicated by the populations persisting within each GMU.

We forecasted that the two GMUs in the Rio Grande basin would have between 42 and 102 (intermediate 55) populations continuing to persist in 2080 and that the two GMUs in the Pecos River and Canadian River basins combined would have between 8 and 30 (intermediate 14) populations continuing to persist in 2080 (Table 4). While a potential decline compared to current conditions under the worst and intermediate cases, the important genetic variation across the subspecies range is forecasted by our model to be maintained in 2080. The Canadian and Pecos GMUs together currently have 22 populations of Rio Grande cutthroat trout. Our "worst case scenario" forecast shows a decline in these two GMUs to a total of 8 populations surviving in 2080. This potential decline would be an important trend that indicates an increasing risk to this portion of the range of the subspecies. At the other extreme, with high levels of management actions, the Canadian and Pecos GMUs are forecasted to have as high as 30 populations surviving in 2080. This would represent an increasing trend and a lowering of the overall risk to the Rio Grande cutthroat trout.

In considering the estimated persistence probabilities and their locations, we provide a picture of the future representation of the subspecies potential ecological diversity across its range to 2080 (Figures 5–15). For example, Figures 12 and 19 show the persistence probability of populations in the Lower Rio Grande GMU, where persistence probabilities appear to decline the most over time in our model. The map in Figure 13 would indicate that the variation in persistence probabilities is distributed across the GMU so that none of the risk is associated with any particular geographic area within the GMU. The number of surviving populations by GMU (Figures 15 and 16) also provides an estimate for the future geographic variation that is expected to survive through 2080 and suggests that, even under the worst case scenarios, populations will persist across the range of the subspecies.

5.6.4 Status Assessment Summary

We used the best available information to forecast the likely future condition of the Rio Grande cutthroat trout. Our goal was to describe the viability of the subspecies in quantitative terms that will address the needs of the subspecies in terms of resiliency, redundancy, and representation. We considered the possible future condition of the subspecies out to about 65 years from the present. We considered a range of potential conditions and scenarios that we believe are important influences on the status of the subspecies. Our results describe a range of possible conditions in terms of the probability of persistence of individual populations across the GMUs and a forecast of the number of populations surviving in each GMU.

None of our "worst case scenario" forecasts result in a predicted loss of all of the populations within any of the GMUs. Therefore, at a minimum, our results suggest the subspecies will have persisting populations in 2080 across its range. The most likely scenarios generally show a declining persistence and number of populations over time. However, the rate of this decline, or whether it occurs at all, depends largely on the likelihood of future management actions occurring, the most important of which are the future restoration and reintroduction of populations within the historical range and the control of nonnative trout. While other factors are important to each population, the future management actions will probably determine the future viability of the Rio Grande cutthroat trout.

APPENDIX A - GLOSSARY OF SELECTED TERMS

Anthropogenic– caused or produced by humans.

Basibranchial teeth– teeth found on or at the base of the tongue.

Benthic feeding– eating food found on the stream bottom.

Catastrophic event–a rare destructive event or episode involving many populations and occurring suddenly.

Census population size– the total number of individuals in a population.

Demographic stochasticity–the variability of population growth rates arising from related random events such as birth rates, death rates, sex ratio, and dispersal, which, may increase the risk of extirpation in small populations.

Dorsal fin– fin located on the back of fish

Ecological diversity– the variation in habitats occupied by the species.

Effective population size– a theoretical measure of the number of breeders in the population that contribute to genetic diversity.

Environmental stochasticity–the variation in birth and death rates from one season to the next in response to weather, disease, competition, predation, or other factors external to the population.

Extant–a population that is still in existence.

Extirpation–the loss of a population or a species from a particular geographic region.

Fluvial– of, relating to, or inhabiting flowing water.

Foraging– finding food.

Population fragmentation– a form of population segregation, occurring when populations become separated from other populations of the same species.

Fry– a young, newly hatched fish.

Genetic diversity– the total number of genetic characteristics in the genetic makeup of a species, subspecies, or population.

Genetic drift– the random change in gene frequencies in a population.

Headwaters– a tributary stream of a river close to or forming part of its source.

Headwater capture– a tributary from one watershed joins with a tributary from another.

Hydrology–the movement or distribution of water on the surface and underground, and the cycle involving evaporation, precipitation, and flow.

Inbreeding– the interbreeding of closely related individuals.

Introgression– gene mixing between species.

Lateral line– a system of sense organs along the side of the body of a fish.

Life history– the full range of changes, habits, and behaviors of a living thing over the course of its life.

Morphological–the structure or form of an organism.

Opportunistic feeder– an organism that feeds on whatever food is available.

Persistence– the ability of a population to sustain itself over time.

Piscicide– fish toxicant.

Piscivorous– fish eating.

Predate– to prey upon.

Prescribed burn– the controlled application of fire to a forest to mimic historical wildfire regimes.

Range–the geographic region throughout which a species naturally lives or occurs.

Recruitment– the number of fish growing to maturity in a population.

Redd– a spawning nest built by trout or salmon in the gravel of streambeds.

Redundancy–the ability of a species to withstand catastrophic events.

Repatriation– the process of repopulating an area of historical habitat.

Representation–the ability of a species to adapt to changing environmental conditions.

Resiliency–the ability of the species to withstand stochastic events.

Riffles– a fast flowing, shallow portion of a stream.

Runoff– the flow of water from rain, snowmelt, or other sources over land.

Salmonid– a member of the family Salmonidae, which includes salmon, trout, and whitefish.

Sex ratio– the proportion of males to females in a population.

Spawn– to produce or lay eggs in water.

Stochastic events–arising from random factors such as weather, flooding, or fire.

Sympatry–species occupying overlapping geographic areas.

Taxonomic–the classification of animals and plants.

Thinning– in forestry, the selective removal of trees to improve the health of the forest and reduce wildfire risk.

Viability– a description of the ability of a species to persist over time and thus avoid extinction.

Appendix B

Evaluating Causes and Effects for Rio Grande Cutthroat Trout Species Status Assessment

Template for Cause and Effects Evaluation

THEME: ?			
[ESA Factor(s): ?]	**Analysis**	**Confidence / Uncertainty**	**Supporting Information**
SOURCE(S)	*What is the ultimate source of the actions causing the stressor?*	See next page for confidences to apply at each step.	Literature Citations, with page numbers , for each step.
- Activity(ies)	*What is actually happening on the ground as a result of the action?*		
STRESSOR(S)	*What are the changes in evironmental conditions on the ground that may be affecting the species?*		
- Affected Resource(s)	*What are the resources that are needed by the species that are being affected by this stressor?*		
- Exposure of Stressor(s)	*Overlap in time and space. When and where does the stressor overlap with the resource need of the species (life history and habitat needs)?*		
- Immediacy of Stressor(s)	*What's the timing and frequency of the stressors? Are the stressors happening in the past, present, and/or future?*		
Changes in Resource(s)	*Specifically, how has(is) the resource changed(ing)?*		
Response to Stressors: **- INDIVIDUALS**	*What are the effects on individuals of the species to the stressor? (May be by life stage)*		
POPULATION & SPECIES RESPONSES	*[Following analysis will determine how do individual effects translate to population and species-level responses? And what is the magnitude of this stressor in terms of species viability?]*		
Effects of Stressors: **- POPULATIONS [RESILIENCY]**	*What are the effects on population characteristics (lower reproductive rates, reduced population growth rate, changes in distribution, etc)?*		
- SCOPE	*What is the geographic extent of the stressor relative to the range of the species/populations? In other words, this stressor effects what proportion of the rangewide populations?*		
Effects of Stressors: **- SPECIES (Rangwide) [REDUNDANCY]**	*What are the expected future changes to the number of populations and their distribution across the species' range?*		
Effects of Stressors: **- SPECIES (Rangwide) [REPRESENTATION]**	*What changes to the genetic or ecology diversity in the species might occur as a result of any lost populations?*		
RISK OF EXTIRPATION 2023	*Based on this analysis, how do we characterize the risk of populations being extirpated from this stressor over the next 10 years (by 2023)?*		

This table of Confidence Terminology explains what we mean when we characterize our confidence levels in the cause and effects tables on the following pages.

Confidence Terminology	Explanation
Highly Confident	We are more than **90% sure** that this relationship or assumption accurately reflects the reality in the wild as supported by documented accounts or research and/or strongly consistent with accepted conservation biology principles.
Moderately Confident	We are **70 to 90% sure** that this relationship or assumption accurately reflects the reality in the wild as supported by some available information and/or consistent with accepted conservation biology principles.
Somewhat Confident	We are **50 to 70% sure** that this relationship or assumption accurately reflects the reality in the wild as supported by some available information and/or consistent with accepted conservation biology principles.
Low Confidence	We are less than **50% sure** that this relationship or assumption accurately reflects the reality in the wild, as there is little or no supporting available information and/or uncertainty consistency with accepted conservation biology principles. Indicates areas of high uncertainty.

THEME: Demographic Risk			
[ESA Factor(s): E]	**Analysis**	**Confidence / Uncertainty**	**Supporting Information**
SOURCE(S)	The source of demographic risks comes mainly from the result of having small population sizes. Small population sizes in streams isolated from other populations are a legacy from the loss of areas occupied by Rio Grande cutthroat trout due to the past invasion of nonnative trout.	**Moderately confident**	Rieman and Allendorf 2001 Baalsrud 2011 p. 1
- Activity(ies)	**Historic:** Most small populations received immigrants from other populations, and genetic risk would be small. Isolated populations that were cut off from others may have experienced genetic drift, inbreeding depression, and perhaps local extirpations. **Current:** Nearly all RGCT populations are isolated from one another, and small populations with little genetic diversity are more vulnerable to extirpation by other factors. **Future:** Populations are likely to remain isolated except in areas where large, interconnected populations are being restored (ie, the Costilla system on Vermejo Park Ranch)	**Moderately confident** that historically, interconnected populations rarely experienced strong genetic drift **Highly confident** that populations are very isolated currently and are likely to remain so.	Fausch et al. 2006, p. 8 Peterson et al. 2008a, p. 559 Fausch et al. 2009, p. 861
STRESSOR(S)	Genetic drift and inbreeding depression in small populations can lead to an inability to adapt to changing environmental conditions and put populations at higher risk of extirpation due to other risk factors.	**Highly confident**	Rieman and Allendorf 2001
- Affected Resource(s)	Genetic diversity of populations and population sizes		
- Exposure of Stressor(s)	Where RGCT populations are small (generally with an effective population size of less than 50), the populations are exposed to the stressors associated with demographic risks. Those populations with an effective population size greater than 500 have no exposure to the stressor. Populations with effective population sizes between 50 and 500 have some exposure to the stressor.	**Somewhat confident**	Allendorf et al. 1997, p. 142, 143 Rieman and Allendorf 2001 Cook et al. 2010, p. 1508

THEME: Demographic Risk			
[ESA Factor(s): E]	**Analysis**	**Confidence / Uncertainty**	**Supporting Information**
- Immediacy of Stressor(s)	**Historic:** Small populations were likely rarely exposed to this stressor. **Current:** Those conservation populations that are currently very small and are not being augmented by managers are exposed to the stressor. **Future:** Small populations will continue to be exposed to the genetic effects of small population sizes in the future.	**Historic: Moderately confident** **Current and Future: Highly confident** that small populations may be experiencing genetic drift	Fausch et al. 2006, p. 8 Peterson et al. 2008a, p. 559 Fausch et al. 2009, p. 861
Changes in Resource(s)	Genetic drift and inbreeding depression in small populations can lead to an inability to adapt to changing environmental conditions, although some very small populations have been known to persist for decades.	**Moderately confident**	Rieman and Allendorf 2001 Cook et al. 2010, p. 1508
Response to Stressors: **- INDIVIDUALS**	More inbred individuals with less individual genetic diversity are expected to be less fit than less inbred individuals with more individual genetic diversity.	**Moderately confident**	
POPULATION & SPECIES RESPONSES			
Effects of Stressors: **- POPULATIONS [RESILIENCY]**	Small population sizes are at greater risk from reduced genetic diversity, decreasing a population's ability to adapt to environmental changes, possibly leading to extirpation of the population from other factors. Small populations are also at greater risk from extirpation due to simple demographic processes, accumulation of mildly deleterious mutations, and inbreeding depression. Small populations also have a higher likelihood of extirpation from other risk factors. This is because a population with a low number of individuals is more likely to be completely lost due to a negative event than a population with a larger number of individuals.	**Moderately confident**	Rieman and Allendorf 2001

THEME: Demographic Risk			
[ESA Factor(s): E]	**Analysis**	**Confidence / Uncertainty**	**Supporting Information**
- SCOPE	**Historic:** RGCT populations were rarely isolated from one another and likely only occasionally experienced this stressor. **Current and Future:** See population resiliency model for number of populations with a small effective population size. This stressor can occur rangewide. To our knowledge, no populations of any native trout have been extirpated by demographic risk alone; instead, demographic factors exacerbate the risk of extirpation by other factors.	**Historic: Moderately confident** **Current and Future: Highly confident** in number of populations experiencing a small effective population size.	Alves et al. 2008 RGCT status assessment model
Effects of Stressors: **- SPECIES (Rangwide)** **[REDUNDANCY]**	If populations are lost in the future, then overall redundancy will continue to decline.	**Highly confident**	
Effects of Stressors: **- SPECIES (Rangwide)** **[REPRESENTATION]**	Any future loss of populations will continue to reduce overall genetic and ecological diversity of the species, further limiting the subspecies' representation.	**Moderately confident**	
RISK OF EXTIRPATION 2023	**Very small populations** have a moderate risk of extirpation due to the exacerbating factor of demographic effects by 2023. **Large populations** have no risk of extirpation due to the exacerbating factor of demographic effects by 2023. See Appendix C for projections of extirpation risk over longer time frames.		

THEME: Nonnative Hybridizing Trout			
[ESA Factor(s): C,E]	**Analysis**	**Confidence / Uncertainty**	**Supporting Information**
SOURCE(S)	Historic stocking of nonnative trout for recreational angling throughout western US.	**Highly confident.** Stocking is well documented.	Flebbe 1994, p 657 Dunham et al. 2002, pg 377 Dunham et al. 2004, pp. 6, 7
- Activity(ies)	**Historic** nonnative stocking programs. **Current and future** unauthorized anthroprogenic movement of fish. (Purposeful, authorized nearby current stocking is only of triploid rainbow trout, which are unable to reproduce.) **Current and future** failure of fish barriers. **Future** conservation strategy restores populations, maintains current barriers, and builds new fish barriers.	**Highly confident** about historic stocking and barrier failure. **Low confidence** in the extent of unauthorized movement of nonnative trout. **Moderate confidence** in maintenance of current barriers and construction of new ones.	Young et al. 1997, p. 240 Peterson and Fausch 2003 Conservation Agreement 2013, pp. 7, 8 Conservation Strategy 2013, pp. 24-25
STRESSOR(S)	Nonnative rainbow trout and other subspecies of cutthroat trout mate with RGCT and produce hybridized offspring. The genetic distinctiveness of Rio Grande cutthroat trout can be lost through hybridization.	**Highly confident** that hybridization occurs based on extensive literature and past population responses. The exact extent is site-dependent.	Rhymer and Simberloff 1996 Allendorf et al. 2004, p. 1205 Boyer et al. 2008, p. 666
- Affected Resource(s)	Genetic integrity of RGCT populations.		
- Exposure of Stressor(s)	Overall, where rainbow trout and nonnative subspecies of cutthroat trout occur, RGCT are exposed to these stressors. See population resiliency assessment for stream-by-stream exposure.	Historic: **Highly confident** about past exposure of nonnatives (well documented). Current: **Moderately confident** in current assessment of nonnative distribution from states' field collection and RGCT database. **Somewhat confident** that climate warming will increase rainbow trout invasions	Boyer et al. 2008, p. 666 Muhlfeld et al. 2014 RGCT database

THEME: Nonnative Hybridizing Trout			
[ESA Factor(s): C,E]	**Analysis**	**Confidence / Uncertainty**	**Supporting Information**
- Immediacy of Stressor(s)	**Historic:** Nonnative trout introductions (of both hybridizing and competing species) account for 90% range loss of RGCT. **Current:** Those conservation populations currently coexisting with rainbow are either already hybridized or will be soon and are at high probability of being lost to conservation. **Future:** Invasion risk continues for RGCT populations that do not have a fish barrier preventing natural invasion of nonnative trout. Invasion risk more likely as streams warm and spring floods decrease through climate change. Unauthorized human introduction has a constant, low probability of occurrence. Stressors are contained by management actions (no stocking, barrier maintenance/construction, and population monitoring).	**Historic: Moderately confident** **Current: Moderately confident i**n assessment of the extent of nonnatives overlapping with conservation populations **Future: Highly confident** that stressors will continue to be contained through limiting nonnative stocking and barrier maintenance and construction.	Dunham et al. 2002, p. 374 Alves et al. 2008, pg 26 Muhlfeld et al. 2014 RGCT database
Changes in Resource(s)	Hybridization with rainbow trout results in introgression with RGCT genes and produces non-pure trout populations lost to conservation.	**Highly confident**	Rhymer and Simberloff 1996 Allendorf et al. 2004, p. 1205 Boyer et al. 2008, p. 666
Response to Stressors: - INDIVIDUALS	Genetic introgression of individuals	**Highly Confident**	
POPULATION & SPECIES RESPONSES			
Effects of Stressors: - POPULATIONS [RESILIENCY]	Genetic introgression of individuals results in i) the population becomes ‘swamped’ with nonnative genes and loses its identity as Rio Grande cutthroat trout; ii) nonnative introgression causes loss of local adaptations or maladaptive behavior and therefore increases population extinction risk (outbreeding depression); and iii) nonnative introgression causes reduced fitness due to disruption of locally co-adapted gene complexes, thus increasing population extinction risk. At >10% introgression we do not consider populations to be conservation populations of RGCT. Populations are not immediately affected after nonnative trout are introduced; it can take years (or decades) for RGCT populations to be hybridized, and longer for extirpation to occur.	**Highly confident**	Utah Division of Wildlife 2000 Boyer et al. 2008 Alves et al. 2008 Fausch et al. 2009 Pritchard 2014, pers. comm.

THEME: Nonnative Hybridizing Trout			
[ESA Factor(s): C,E]	**Analysis**	**Confidence / Uncertainty**	**Supporting Information**
- SCOPE	**Historic:** RGCT has been extirpated from about 90% of its historical range primarily due to stressors of nonnatives resulting in the loss of RGCT populations. **Current:** Barriers and stocking of triploid rainbow trout have reduced likelihood of further invasions. Currently 84 conservation populations have complete or partial fish migration barriers, reducing risk of hybridizing species invasion. See population resiliency analysis for geographic locations of RGCT populations related to nonnative trout. **Future:** Continued barrier construction and maintenance wil reduce likelihood of further invasions. Distance from non-triploid rainbow trout populations is a factor in future invasion risk; the farther from a non-triploid rainbow trout (or other nonnative cutthroat trout subspecies) population, the less the risk of future hybridization. Under climate change, rainbow trout are expected to be able to invade further upstream. See RGCT population model for assessment of risk to each population by hybridization. The risk of non-triploid rainbow trout invasion does not vary by GMU.	**Historic: Moderately confident** **Current: Highly confident** **Future: Highly confident**	Alves et al. 2008, pg 26 Muhlfeld et al. 2014 RGCT database RGCT status assessment model
Effects of Stressors: **- SPECIES (Rangwide) [REDUNDANCY]**	If future populations are lost due to nonnatives, overall redundancy will continue to decline.	**Moderately confident**	
Effects of Stressors: **- SPECIES (Rangwide) [REPRESENTATION]**	Any future loss of populations will continue to reduce overall genetic and ecological diversity of the species, further limiting the subspecies' representation.	**Moderately confident**	
RISK OF EXTIRPATION 2023	Populations characterized as **no risk of hybrid invasion** have no risk of extirpation due to hybridization. Populations **sympatric** with rainbow or Yellowstone cutthroat trout have a very high risk of extirpation due to hybridization. See Appendix C for projections of extirpation risk over longer time frames.		

THEME: Nonnative Competing Trout			
[ESA Factor(s): C,E]	**Analysis**	**Confidence / Uncertainty**	**Supporting Information**
SOURCE(S)	Historic stocking of nonnative trout for recreational angling throughout western US.	**Highly confident.** Stocking is well documented.	Flebbe 1994, p 657 Dunham et al. 2002, pg 377 Dunham et al. 2004, pp. 6, 7
- Activity(ies)	**Historic** nonnative stocking programs: mainly brown trout and brook trout. **Current and future** unauthorized anthroprogenic movement of fish. No purposeful nearby current stocking is occuring. **Current and future** failure of fish barriers can allow new invasions into RGCT populations. **Future** conservation strategy restores populations, maintains current barriers, and builds new fish barriers.	**Highly confident** about historic stocking and barrier failure. **Low confidence** in the extent of unauthorized movement of nonnative trout. **Moderate confidence** in maintenance of current barriers and construction of new ones.	Flebbe 1994, p 657 Harig et al. 2000b Dunham et al. 2002, pg 377 Dunham et al. 2004, pp. 6, 7 Johnson et al. 2009, p. 389 Conservation Agreement 2013 Conservation Strategy 2013
STRESSOR(S)	Nonnative trout compete with and predate on RGCT: 1) COMPETITION. Brown trout and brook trout outcompete RGCT for food and space. 2) PREDATION. Brown trout (and likely brook trout) will eat young RGCT.	**1) Highly confident** that these stressors occur based on extensive literature and past population responses. The exact extent is site dependent. **2) Moderately confident** that brown and brook trout predate upon young RGCT.	Dunham et al. 2002, p. 378 Peterson et al. 2004 Fausch et al. 2006, pp. 9-10
- Affected Resource(s)	1) COMPETITION. Food (insects and small fish) and space (sheltering/feeding habitat). 2) PREDATION. Predator avoidance.		Paroz 2005, p. 34 Shemai et al. 2007, pp. 315, 320, 321 Peterson et al. 2004, pp. 768, 769
- Exposure of Stressor(s)	Overall, where nonnative trout occur, RGCT are exposed to these stressors. Water temperature, fine sediment, and abundance of pools and woody debris may influence nonnative trout invasion. (See RGCT population model for stream-by-stream exposure and risk to competing nonnative species.)	Historic: **Highly confident** about past exposure of nonnatives (well-documented). Current: **Moderately confident** in current assessment of nonnative distribution from states' field collection and trout database.	Shepard 2004, p. 1096 RGCT database RGCT status assessment model

THEME: Nonnative Competing Trout			
[ESA Factor(s): C,E]	**Analysis**	**Confidence / Uncertainty**	**Supporting Information**
- Immediacy of Stressor(s)	**Historic:** Nonnative trout introductions (of both hybridizing and competing species) account for 90% range loss of RGCT. **Current:** Although the majority of range contraction was due to hybridizing nonnative species, competing nonnative trout cooccur with approximately 40% of current populations. Stressors of competition and predation persist for RGCT populations that are currently coexisting with nonnative brown or brook trout. **Future:** Invasion risk continues for RGCT populations that do not have a fish barrier preventing natural invasion of nonnative trout. Brown trout may be able to invade further upstream as stream temperatures warm under climate change, and brook trout may be adversely impacted by the earlier peak flows due to climate change. Both of these effects of climate change on competing nonnative trout are highly uncertain. Unauthorized human invasion has a constant, low probability of occurrence. See "Management Actions" worksheet for a description of how stressors are being contained.	**Historic: Moderately confident** **Current: Moderate Confidence i**n assessment of the extent of nonnatives overlapping with conservation populations **Future: Highly confident** that stressors will continue to be contained through limiting nonnative stocking and barrier maintenance and construction, but low confidence in rate of nonnative invasions.	Dunham et al. 2002, p. 374 Alves et al. 2008, pg 26 RGCT database RGCT status assessment model Fausch 2014, pers. comm.
Changes in Resource(s)	1) COMPETITION. Reduction in availability of food and space, harassment by large competitors. Young RGCT are consistently outcompeted by brook and brown trout. 2) PREDATION. Increased rates of predation of young RGCT.	1) **Highly confident** 2) **Moderately Confident**	Paroz 2005, p. 34 Shemai et al. 2007, pp. 315, 320, 321 Peterson et al. 2004, pp. 768, 769
Response to Stressors: **- INDIVIDUALS**	1) COMPETITION. Competition for food will lower fitness of RGCT individuals because less food causes smaller sizes of individuals and potential for less reproductive output. Competition for space will result in higher mortality and lowered reproductive rates of RGCT. Indviduals may spend more energy competing for food and sheltering space (and avoiding harrassment from nonnatives) and less energy in reproduction, which may cause individuals to be more susceptible to predation or disease. 2) PREDATION. Results in death of individuals of smaller sizes.	1) **Highly confident** 2) **Highly confident**	Paroz 2005, p. 34 Shemai et al. 2007, pp. 315, 320, 321 Peterson et al. 2004, pp. 768, 769

THEME: Nonnative Competing Trout			
[ESA Factor(s): C,E]	**Analysis**	**Confidence / Uncertainty**	**Supporting Information**
POPULATION & SPECIES RESPONSES			
Effects of Stressors: **- POPULATIONS [RESILIENCY]**	1) COMPETITION. Decreased fitness results in lower reproductive success and lower population growth rates. When brook and brown trout invade streams occupied by cutthroat trout, the native cutthroat trout decline or are displaced. Cutthroat trout condition declines in the presence of brook and brown trout. Age-0 cutthroat trout survival is 13 times higher when brook trout are removed, and age-1 survival is twice as high. 2) PREDATION. Higher mortality rates and lower recruitment of RGCT leads to overall decrease in population size by removing smaller individuals and preventing recruitment from subadults to reproductive adults. It is unknown how quickly populations are affected after nonnative competing trout are introduced; it may take years (or decades) for RGCT populations to be affected, and longer for extirpation to occur, or it could happen more quickly.	**1) Highly confident** **2) Moderately confident** about effects of predation on RGCT. **Low confidence** in how quickly populations are affected.	Peterson et al. 2004, p. 761 Paroz 2005, p. 34 Shemai et al. 2007, pp. 315, 320, 321 Peterson et al. 2004, pp. 768, 769
- SCOPE	**Historic:** RGCT has been extirpated from about 90% of its historic range primarily due to stressors from nonnatives, resulting in the loss of RGCT populations; most of this range reduction was due to hybridizing nonnative trout. **Current:** Barriers and nonnative removals have reduced likelihood of further invasions. Currently 84 conservation populations have complete or partial fish migration barriers, eliminating or reducing risk of competing nonnative species invasion. See population resiliency analysis for geographic locations of RGCT populations related to nonnative trout. Cutthroat trout may occupy headwater streams and brook and brown trout occupy downstream reaches because of the influence of temperature on competitive abilities. Mechanical suppression of nonnative species is occurring on 10 streams by states of Colorado and New Mexico, as well as Vermejo Park Ranch. **Future:** Continued barrier construction and maintenance and nonnative suppression will reduce likelihood of further invasions. Brown trout may be able to move further upstream as stream temperatures become warmer, although we do not have any data supporting this to date. Brook trout may become less pervasive due to increased temperatures and winter flood frequency (cutthroat trout are less susceptible than brook trout.) See RGCT population model for assessment of risk to each population by competition and predation. The risk of nonnative competing trout invasion does not vary rangewide.	**Historic: Moderately confident** **Current: Highly confident** **Future: Highly confident** in rates of barrier construction and maintenance. **Moderately confident** in the effects warming temperatures and changing flood frequencies may have on nonnative trout.	Jager et al. 1999 pp. 232, 235 McCullough 1999, p. 156 IPCC 2002 p 32 Alves et al. 2008 p. 26 Peterson et al. 2008b Wenger et al. 2011a, pp. 1000-1001 Wenger et al. 2011b, pp. 14176 Kruse 2013, p. 4 RGCT Database RGCT status assessment model

THEME: Nonnative Competing Trout			
[ESA Factor(s): C,E]	**Analysis**	**Confidence / Uncertainty**	**Supporting Information**
Effects of Stressors: - SPECIES (Rangwide) [REDUNDANCY]	If future populations are lost due to nonnatives, then overall redundancy will continue to decline.	**Moderately confident**	
Effects of Stressors: - SPECIES (Rangwide) [REPRESENTATION]	Any future loss of populations will continue to reduce overall genetic and ecological diversity of the species, further limiting the subspecies' representation.	**Moderately confident**	
RISK OF EXTIRPATION 2023	Populations with **no nonnative trout** and with a **complete or partial barrier** to fish movement have no risk of extirpation by 2023 due to competition and predation. Populations **sympatric** with brown or brook trout with **no mechanical suppression** have a high risk of extirpation due to competition and predation. See Appendix C for projections of extirpation risk over longer time frames.		

THEME: Wildfire			
[ESA Factor(s): A,E]	**Analysis**	**Confidence / Uncertainty**	**Supporting Information**
SOURCE(S)	Wildfire frequency and intensity is increasing due to climate change (drier, warmer regional climate). Wildfire frequency is locally influenced by forest management.	**Highly confident** that fire is a natural, regular part of the ecosystem and that the incidence of large, hot fires has increased. **Moderately confident** that climate change will exacerbate the rate of burning even further.	Schoennagel et al. 2004 p. 666 Westerling et al. 2006 p. 941 Bachelet et al. 2007 IPCC 2007a (pg 15)
- Activity(ies)	Risk of wildfires can be affected by forest management activities; fire suppression and lack of thinning or prescribed burns can enhance conditions suitable for high-intensity wildfires.	**Highly confident** that management influences fire frequency and intensity	Ferrell 2002, pp. 11-12 Schoennagel et al. 2004 p. 669
STRESSOR(S)	When natural or human-caused catostrophic wildfires burn within watersheds upstream of RGCT populations, subsequent rainstorms produce ash and debris-laden runoff of water from the burned forest into streams occupied by RGCT. Stormwater runoff following wildfire results in highly sedimented and ash-laden waters and very unstable stream channels. Additionally, fire retardant is often dropped in wildfire areas, and those chemicals (such as surfactant foams and fire retardants) can cause fish mortality.	**Highly confident**	Rinne 1996 p. 654 Buhl and Hamilton 2000, pp 410-416 Brown et al. 2001 pp 140-141 Backer et al. 2004, pg 942, 943 USFS 2006 p. 32
- Affected Resource(s)	High quality water and stable stream channels.	**Highly confident**	
- Exposure of Stressor(s)	A wildfire event can happen at any time, but forest condition of some areas makes the probability of high-intensity wildfire greater. Wildfires may be patchy and burn hotter in some places than in others, allowing some portions of the population to survive and recolonize downstream reaches after ash flow effects have been ameliorated. The amount of ash flow from a fire depends on the severity of the fire, proximity to the stream habitat, stream channel morphology, timing, and amount of rainfall following the fire. The extent of one or more populations being affected depends on the location of the fire relative to the stream reaches occupied by RGCT.	**Highly confident**	Schoennagel et al. 2004, p. 669 Miller and Bassett 2013 Roberts et al. 2013 pg 6

THEME: Wildfire			
[ESA Factor(s): A,E]	**Analysis**	**Confidence / Uncertainty**	**Supporting Information**
- Immediacy of Stressor(s)	**Historic:** Wildfires have resulted in at least 5 documented extirpations of RGCT populations in the past 10 years, with increasing fire severity in modern times due to forest management practices. **Current:** Wildfires are contuing to occur. Several large fires have occured in recent years resulting in populations of RGCT being extirpated. **Future:** Climate change is predicted to cause southwestern forests to be hotter and drier in coming decades, resulting in higher risks of catostrophic fires. Land managers are making efforts to reduce fire risks. Fish managers are committed to respond with restoration activities following wildfires, which in some cases create opportunities for restoration when nonnative trout are eliminated from stream reaches historically occupied by RGCT.	**Historic: Highly confident** **Current: Highly confident** **Future: Moderately confident**	Schoennagel et al. 2004 p. 666 Westerling et al. 2006 p. 941 Bachelet et al. 2007 IPCC 2007a (pg 15) Extirpations: pers. comm. with B. Bakevich and J. Alves, 2014
Changes in Resource(s)	Ash-filled flood waters make stream habitat unhabitable and can kill all fish in the stream. Stream channel changes and water quality impacts can make streams unsuitable for years following the fire and flood event. Extent of the impact of a particular event depends on the local conditions and nature of the fire and flood relative to RGCT habitat. If a stream is sufficiently long, fish may survive in an unburned upstream reach or tributary, then recolonize the burned reach when habitat becomes suitable.	**Highly confident**	Rinne 1996 p. 655 Brown et al. 2001 pp. 140-141
Response to Stressors: **- INDIVIDUALS**	All life stages of RGCT in the reach exposed to significant ash flow are killed and elimnated.	**Highly confident**	Rinne 1996 p. 654
POPULATION & SPECIES RESPONSES			
Effects of Stressors: **- POPULATIONS** **[RESILIENCY]**	The RGCT population can be eliminated from the area impacted by the ash flow.	**Highly confident**	Rinne 1996 p. 654

THEME: Wildfire			
[ESA Factor(s): A,E]	**Analysis**	**Confidence / Uncertainty**	**Supporting Information**
- SCOPE	**Historic:** Wildfire is a part of the ecosystem in the southern Rocky Mountains. Wildfires have always occurred, and, historically, RGCT populations extirpated in one area would be eventually repatriated by nearby populations. **Current and Future:** The frequency and intensity of wildfire is increasing rangewide. As drought frequency increases due to climate change, dry forests will be more likely to burn and burn hotter than in the past. In the past 10 years, at least 5 populations have been extirpated due to the effects of wildfire, representing about 4% of existing populations. Any one stream has a low likelihood of experiencing wildfire during any single year. We expect wildfire to occur, although we are unable to predict the location. The networking of the stream system influences whether a population is extirpated or eventually repatriates the ash flow area; tributaries may provide refuges from ash flows where some portion of a population may survive (example: Polvadera Creek). TNC has provided a risk assessment of fire for RGCT. In general, populations in the Rio Grande Headwaters GMU have less risk of wildfire (categorized as moderate fire risk) than those in the rest of the range (categorized as high fire risk).	**Historic: Highly confident** **Future: Moderately confident**	Westerling et al. 2006, pp. 940-941 Miller and Bassett 2013 Roberts et al. 2013 p 6 Wuebbles et al. 2013, p. 16 RGCT database
Effects of Stressors: **- SPECIES (Rangwide)** **[REDUNDANCY]**	If future populations are lost due to wildfire, then overall redundancy will continue to decline. The number of populations experiencing wildfire is expected to increase due climate change, but this may be ameliorated if land managers can reduce forest fuels. In some cases, the population elimination resulting from ash flows can provide restoration opportunities where nonnative species had been sympatric with RGCT populations.	**Highly confident**	NMDGF 2013, p. 3
Effects of Stressors: **- SPECIES (Rangwide)** **[REPRESENTATION]**	Any future loss of populations can reduce overall genetic and ecological diversity of the species, further limiting the subspecies' representation, although this is dependent on the timing and location of fires and ash flows.	**Moderately confident**	

THEME: Wildfire			
[ESA Factor(s): A,E]	**Analysis**	**Confidence / Uncertainty**	**Supporting Information**
RISK OF EXTIRPATION 2023	Populations with a **moderate wildfire risk, long occupied stream lengths, and some stream connectivity** have a very low risk of extirpation due to the effects of wildfire by 2023. Populations with **high fire risk, short occupied stream lengths, and no stream connectivity** have a very high risk of extirpation due to the effects of wildfire by 2023. See Appendix C for projections of extirpation risk over longer time frames.		

THEME: Stream Drying			
[ESA Factor(s): A,E]	**Analysis**	**Confidence / Uncertainty**	**Supporting Information**
SOURCE(S)	Drought and, in some cases, water withdrawals	**Highly confident**	Pritchard and Cowley 2006, p. 36
- Activity(ies)	Streamflows may decline, particularly in summer, due to drought (reduced precipitation, snowmelt runoff, and groundwater recharge) in combination with hot summer temperatures, and also from instream and groundwater withdrawals. Drought, hot summer temperatures, and water withdrawals may become more severe due to climate change. Water withdrawals can occur from stream diversions (acequias) or groundwater pumping for agriculture or solar projects.	**Highly confident**	Nash and Gleick 1993 p. ix Barnett et al. 2008, p. 1082 IPCC 2007a p. 15 Ray et al. 2008 p. 37
STRESSOR(S)	Stream drying (the significant reduction or loss of streamflow) reduces or eliminates habitat available for all life stages of RGCT. Stream intermittency may cause water quality declines (increased temperature, decreased oxygen), lack of access to breeding, feeding, and sheltering areas, and stranding of fish.	**Highly confident**	Elliott 2000, pp 938, 945 Lake 2000, p. 577
- Affected Resource(s)	Aquatic habitat (providing breeding, feeding, and sheltering areas). Water with cool temperatures and high dissolved oxygen content.		
- Exposure of Stressor(s)	Stream drying typically occurs in the late spring or early summer timeframe, after snowmelt runoff but prior to summer monsoon rains. If monsoon rains fail to produce precipitation, the drying trend can extend into fall. Reproduction and recruitment may be reduced due to a lack of spawning habitat and habitat for eggs and young of year.	**Highly confident**	Elliott 2000

THEME: Stream Drying			
[ESA Factor(s): A,E]	**Analysis**	**Confidence / Uncertainty**	**Supporting Information**
- Immediacy of Stressor(s)	**Past:** This was not likely a significant ecological factor in the past, due to expanded range of the fish occuring in varying elevations. Under historic conditions, local drying of streams would have been short term and effects would have been offset by recolonization from nearby populations when the stream rewetted. **Current:** Stream drying has been shown to depress populations, particularly after drought in 2002. However, in NM, virtually all populations remained stable through 2007 despite drought of early 2000s. In Colorado, the population in Medano Creek has survived 2 drought periods, although several other populations were either exirpated or populations reduced to low levels. **Future:** The stressor is expected to increase in frequency and intensity due to the effects of climate change making the region hotter and drier (and with earlier cessation of spring runoff).	**Past: Moderately confident** **Current: Somewhat confident** **Future: Highly confident**	Japhet et al. 2007, pg 42-44 Patten et al. 2007, p 13, 104 Isaak et al. 2012b, p. 548 Great Sand Dunes NP 2013, p. 1 Wuebbles et al. 2013, p. 16 RGCT database
Changes in Resource(s)	Habitat is reduced as shallow streams become intermittent or dry. Individuals must retreat into higher elevation, cooler steam reaches, springfed stream reaches, or lower elevation steam reaches with more pools (deeper water with lower temperatures). Pools in an intermittant section of stream will eventually reach higher temperatures during summer, potentially causing stress to individuals.	**Highly confident**	Elliott 2000, pp 938, 945 Lake 2000, p. 577
Response to Stressors: **- INDIVIDUALS**	**Adults:** More competition for scarce resources in available pools. Heat stress or death can occur. **Juveniles:** Heat stress. Higher mortality if in pools with adults, where predation may occur.	**Highly confident**	

THEME: Stream Drying			
[ESA Factor(s): A,E]	**Analysis**	**Confidence / Uncertainty**	**Supporting Information**
POPULATION & SPECIES RESPONSES			
Effects of Stressors: - POPULATIONS [RESILIENCY]	Demographic: Loss of individuals results in reduction of population sizes. If drought persists for only 1-2 years and sufficient refugia exist, the population can likely rebound. If drought is longer and/or there is a lack of refugia, extirpation of the population is likely. Historically, drought has occurred and streams have dried, but populations were able to be recolonized from other reaches. Recently, North Fork Carnero Creek, in Colorado, appears to have been extirpated after the drought of 2011 and 2012. We don't have any examples where streams have been affected only by water withdrawal, but this may be an exarcerbating factor.	**Highly confident**	RGCT Database J. Alves, pers. comm.
- SCOPE	Stream drying from drought can affect streams throughout the range. Water withdrawals are localized by stream. Streams on the Rio Grande National Forest (Rio Grande Headwaters GMU) are afforded some protection from stream drying (from water withdrawals) via the water rights settlement agreement of 2000, in which water rights were reserved for instream flow. Streams in the southern extent of the subspecies' distribution (ie Caballo GMU, southern portion of Pecos GMU) are more vulnerable to stream drying as these streams tend to be in hotter and drier areas. Further, south-facing streams across the distribution and those with less riparian vegetation are more vulnerable than north-facing streams or those with shading riparian vegetation. Riparian management can decrease the vulnerability of a stream to drying. Summer streamflow has decreased rangewide by 5.3% per decade over the last 45 years, increasing the risk of stream drying. Frequency of drought is expected to increase due to climate change.	**Moderately confident**	Zeigler et al. 2012, pp. 1049-1050 Rio Grande Water Conservation District 2014, pp. 3-4 RGCT database

THEME: Stream Drying			
[ESA Factor(s): A,E]	**Analysis**	**Confidence / Uncertainty**	**Supporting Information**
Effects of Stressors: **- SPECIES (Rangwide)** **[REDUNDANCY]**	Losses of populations will reduce redundancy.	**Highly confident**	
Effects of Stressors: **- SPECIES (Rangwide)** **[REPRESENTATION]**	Any future loss of populations can reduce overall genetic and ecological diversity of the subspecies, further limiting the subspecies' representation.	**Highly confident**	
RISK OF EXTIRPATION **2023**	Populations with **long occupied stream lengths, some stream networking within the occupied reach, and moderate to high baseflow discharges** have an extremely low risk of extirpation due to stream drying by 2023. Populations with **short occupied stream lengths, no stream networking, and very low baseflow discharges** have a low risk of extirpation due to stream drying by 2023. Although short stream lengths reduce the ability of the population to seek refuge and rebound after periods of drought, we have very few instances where populations were extirpated due to stream drying. (North Fork Carnero Creek appears to have been extirpated after the 2011-2012 drought; Medano Creek, which was thought to have been extirpated from drought (Japhet *et al.* 2007), was not extirpated, although numbers were quite low.) See Appendix C for projections of extirpation risk over longer time frames.		

THEME: Disease			
[ESA Factor(s): C]	**Analysis**	**Confidence / Uncertainty**	**Supporting Information**
SOURCE(S)	Whirling Disease: Caused by the nonnative myxosporean parasite *Myxobolus cerebralis*	**Highly confident**	DuBey et al. 2007, p. 1411
- Activity(ies)	**Historic:** Parasite introduced into US from Europe in 1950s. Disease transmitted by translocation of affected fish and worms. **Current/Future:** NMDGF policies and regulations prohibit the stocking of any whirling disease positive fish in the states of New Mexico. In Colorado stocking of whirling disease positive fish in protected habitats, which include native cutthroat trout waters, is prohibited. Testing for whirling disease involves collecting and sacrificing 60 fish (nonnatives are preferred over RGCT for testing, but some RGCT are usually collected)	**Highly confident**	Japhet et al. 2007, p. 12 Patten and Sloane 2007, p. 10 Nehring 2007, 2008
STRESSOR(S)	Parasites damage cartilage, killing young fish or causing infected fish to swim in an uncontrolled whirling motion, making it impossible to avoid predation or feed. Total year class failure can occur.	**Highly confident**	Koel et al. 2006
- Affected Resource(s)	Young-of-year and juvenile trout.		Nehring 2007, p. 1
- Exposure of Stressor(s)	Trout infected by eating the worms (*Tubifex tubifex*) carrying the parasite (specifically, the actinosporean triactionomyxons (TAMs) produced in gut of worms) or through contact with water in which TAMs are present. See population resiliency model for the assessment of disease risk for each population.	**Highly confident**	Koel et al. 2006 RGCT database RGCT status assessment model
- Immediacy of Stressor(s)	Whirling disease has affected Columbine Creek in NM and Placer Creek in CO in the past. No other known infections of RGCT populations.	**Moderately confident** that our knowledge of the incidence of whirling disease represents all of the affected streams.	Japhet et al. 2007, p 27 Patten and Sloane 2007, p. 5
Changes in Resource(s)	Infected fish die.	**Highly confident**	Hiner and Moffett 2001, p. 130 DuBey et al. 2007, p. 1411
Response to Stressors: **- INDIVIDUALS**	Infected fish die.	**Highly confident**	Hiner and Moffett 2001, p. 130 DuBey et al. 2007, p. 1411

THEME: Disease			
[ESA Factor(s): C]	**Analysis**	**Confidence / Uncertainty**	**Supporting Information**
POPULATION & SPECIES RESPONSES			
Effects of Stressors: - POPULATIONS [RESILIENCY]	Total year class failure can result from whirling disease infections. Within 4 months of exposure, 85% of population can die. Repeated year class loss can result in population loss. Whirling disease is the source of many major population declines of rainbow trout. To recover from whirling disease infection, all fish in the stream must be killed and the stream must remain fishless for three years.	**Highly confident**	Thompson et al. 1999, pp. 312-313 Nehring 2007, p. 2
- SCOPE	Whirling disease is found in NM and CO, but not in RGCT conservation populations at this time. 84% of the conservation populations are judged to have very limited risk from whirling disease or other potential diseases because the pathogens are not known to exist in the watershed or a barrier blocks upstream fish movement. 5% are at minimal risk because they are greater than 10 km (6.2 mi) from the pathogen or they are protected by a barrier, but the barrier may be at risk of failure. 7% were identified as being at moderate risk because whirling disease had been identified within 10 km of occupied habitat. No protection from being in high elevation headwater streams has been documented.	**Moderately confident** that our assessment of the risk of whirling disease is correct.	Nehring 2007, pg 10 Alves et al. 2008 RGCT database
Effects of Stressors: - SPECIES (Rangwide) [REDUNDANCY]	The loss of year classes will result in the loss of populations over time, which will result in a loss of redundancy. Populations known to have whirling disease are killed, left fishless for 3 years, and repatriated.	**Highly confident**	
Effects of Stressors: - SPECIES (Rangwide) [REPRESENTATION]	The loss of year classes will result in the loss of populations over time, which will result in a loss of representation. Populations known to have whirling disease are killed, left fishless for 3 years, and repatriated.	**Highly confident**	
RISK OF EXTIRPATION 2023	Populations identified as a **limited risk of infection** have no risk of extirpation due to disease by 2023. Populations identified as having a **moderate risk of infection** have a very low risk of extirpation due to disease by 2023. See Appendix C for projections of extirpation risk over longer time frames.		

THEME: Water Temperature			
[ESA Factor(s): A,E]	**Analysis**	**Confidence / Uncertainty**	**Supporting Information**
SOURCE(S)	Climate Change	**Highly confident**	IPCC 2007a,b
- Activity(ies)	Changes in air temperature and precipitation will likely lead to changes in water temperature	**Highly confident**	Poff et al. 2002, p. 4
STRESSOR(S)	Changes in air temperature and water temperature	**Highly confident**	Battin et al. 2007 Zeigler et al. 2012, pp. 1045-1046.
- Affected Resource(s)	Thermal suitability	**Highly confident**	
- Exposure of Stressor(s)	RGCT are exposed to the temperature and water changes wherever they occur.	**Highly confident**	
- Immediacy of Stressor(s)	**Past:** This was not likely a significant ecological factor in the past, due to expanded range of the fish occuring in varying elevations. Under historic conditions, streams with less than optimal water temperature conditions would have fewer RGCT until conditions improved, and effects would have been offset by recolonization from nearby populations. **Current/Future:** The stressor is expected to increase in frequency and intensity due to the effects of climate change making the region hotter and drier.	**Moderately confident**	Regonda et al. 2005, p. 373 Battin et al. 2007 Lenart et al. 2007, p 2 Barnett et al. 2008 Ray et al. 2008 p 1, 2,10 Clow 2010, p. 2297 Isaak et al. 2012b, p. 544 Llewellyn and Vaddey 2013, p. S-iv
Changes in Resource(s)	**Temperature:** Stream warming can cause some streams to become too warm for RGCT populations to thrive. Conversely, several streams that are currently colder than is optimal will warm and become more suitable.	**Moderately confident**	Rogers 2013 Zeigler et al. 2012 Zeigler et al. 2013a Zeigler et al. 2013b RGCT status assessment model
Response to Stressors: **- INDIVIDUALS**	Individuals in warmer than optimal water will be stressed, have lower fecundity, and could die if water is warm enough.	**Highly confident**	
POPULATION & SPECIES RESPONSES			
Effects of Stressors: **- POPULATIONS [RESILIENCY]**	Demographic: Loss of recruitment results in reduction of population sizes. If conditions persist for only 1-2 years and sufficient refugia exist, the population can likely rebound. If water temperatures increase by more than 2 degrees (currently expected), more streams than the few that are currently expected could become unsuitable.	**Moderately confident**	Roberts et al. 2013 Rogers 2013 Zeigler et al. 2013a Zeigler et al. 2013b

THEME: Water Temperature			
[ESA Factor(s): A,E]	**Analysis**	**Confidence / Uncertainty**	**Supporting Information**
- SCOPE	Temperature changes could occur rangewide, as climate change is expected to affect the southwest. However, streams in the southern extent of the subspecies' distribution (ie Caballo GMU, southern portion of Pecos GMU) are more vulnerable to temperature increases as these streams tend to be in hotter and drier areas. Further, south-facing streams across the distribution and those with less riparian vegetation are more vulnerable to temperature increases than north-facing streams or those with shading riparian vegetation. Riparian management can lessen temperature increases. Also, smaller streams are more affected by temperature changes than larger ones, which buffer temperature swings. Some temperature changes have been observed throughout the range of RGCT, including increased air temperatures of 0.29 degrees C per decade over the last 45 years.	**Moderately confident**	Smith and Lavis 1975, p. 229 Isaak et al. 2012a Isaak et al. 2012b, p. 544 Zeigler et al. 2012, pp. 1049-1050 Llewellyn and Vaddey 2013 Roberts et al. 2013 Zeigler et al. 2013b
Effects of Stressors: **- SPECIES (Rangwide) [REDUNDANCY]**	Loss of populations would result in a loss of redundancy.	**Highly confident**	
Effects of Stressors: **- SPECIES (Rangwide) [REPRESENTATION]**	Loss of populations would result in a loss of representation.	**Highly confident**	
RISK OF EXTIRPATION 2023	No populations throughout the range of Rio Grande cutthroat trout have currently been identified as having any risk of extirpation by 2023 due to water temperature effects. By 2040, those populations with **low risk** of chronic water temperature effects have no risk of extirpation due to the effects of increased water temperature. Those populations with **predicted acute effects** have a low risk of extirpation due to the effects of increased water temperature. See Appendix C for projections of extirpation risk over longer time frames.		

THEME: Changes in Flood Timing and Magnitude			
[ESA Factor(s): A,E]	**Analysis**	**Confidence / Uncertainty**	**Supporting Information**
SOURCE(S)	Climate Change	**Highly confident**	IPCC 2007a,b
- Activity(ies)	Changes in air temperature and precipitationwill likely lead to changes in the magnitude, timing, and duration of spring runoff floods and higher magnitude summer rainstorm floods.	**Highly confident**	Poff et al. 2002, p. 4 Barnett et al. 2008
STRESSOR(S)	Changes in timing and amount of floods	**Somewhat confident** that the change in flood timing and amount is a stressor to the subspecies.	Archer and Predick 2008, p. 23 Battin et al. 2007
- Affected Resource(s)	Water timing and amount		
- Exposure of Stressor(s)	RGCT are exposed to the changes wherever they occur. If hydrological changes result in different spring runoff, this is the time of year when the subspecies is preparing for spawning. Changes in summer floods are when eggs are in the gravel or when fry are emerging from the gravels.	**Moderately confident**	
- Immediacy of Stressor(s)	**Past:** This was not likely a significant ecological factor in the past, due to the large range of the subspecies occuring in varying elevations. Under historic conditions, streams with less than optimal flooding conditions would have fewer RGCT until conditions improved, and effects would have been offset by recolonization from nearby populations. **Current/Future:** The stressor is expected to increase in frequency and intensity due to the effects of climate change making the region hotter and drier (and with earlier cessation of spring runoff).	**Moderately confident**	Regonda et al. 2005, p. 373 Battin et al. 2007 Lenart et al. 2007, p 2 Barnett et al. 2008 Ray et al. 2008 p 1, 2,10 Clow 2010, p. 2297 Isaak et al. 2012b, p. 544 Llewellyn and Vaddey 2013, p. S-iv
Changes in Resource(s)	**Timing:** A change in timing or magnitude of floods can scour the streambed, destroy eggs, or displace recently emerged fry downstream. Change in the timing of runoff from spring to winter could disrupt spawning cues because peak flow would occur when the days are still short in length and water temperatures cold. Conversely, earlier spawning that may result from earlier floods may lead to a longer growing season for the fry, benefiting the subspecies.	**Low confidence** that a change in timing or magnitude of flooding will have a largely negative effect on RGCT populations.	Erman et al. 1988, pg 2199 Montgomery et al. 1999 Stewart et al. 2004, p. 1154 Stewart et al. 2005, p. 1137 Isaak et al. 2012b, p. 544 RGCT status assessment model
Response to Stressors: **- INDIVIDUALS**	Nests and eggs can be destroyed if flood changes cause scour after spawning. Some individuals may not reproduce if spawning cues are disrupted due to timing changes.	**Somewhat confident**	Erman et al. 1988, pg 2199 Montgomery et al. 1999

THEME: Changes in Flood Timing and Magnitude			
[ESA Factor(s): A,E]	**Analysis**	**Confidence / Uncertainty**	**Supporting Information**
POPULATION & SPECIES RESPONSES			
Effects of Stressors: - POPULATIONS [RESILIENCY]	Demographic: Loss of recruitment results in reduction of population sizes. If conditions persist for only 1-2 years and sufficient refugia exist, the population can likely rebound. If flood timing changes are dramatic and/or there is a lack of refugia, extirpation of the population is possible.	**Moderately confident**	Montgomery et al. 1999 Isaak et al. 2012b, p. 545 Roberts et al. 2013
- SCOPE	Hydrologic changes could occur rangewide, as climate change is expected to affect the Southwest. The seasonality of flows is projected to change. Anticipated changes include earlier snowmelt runoffs as well as increased variability in the magnitude, timing, and spatial distribution of streamflow and other hydrologic variables. Some hydrological changes have been observed throughout the range of RGCT, including increased air temperatures of 0.29 degrees C per decade over the last 45 years and snowmelt runoff occurring 10.6 days earlier than 45 years ago.	**Moderately confident**	Clow 2010, p. 2297 Isaak et al. 2012b, p. 544, 545 Llewellyn and Vaddey 2013, p. S-iv
Effects of Stressors: - SPECIES (Rangwide) [REDUNDANCY]	We do not expect an effect of changed hydrology on the subspecies' redundancy because of the uncertainty surrounding many of these relationships and how they may affect the subspecies. We expect that there may some negative effects (increased scouring) and some positive effects (longer growing season). We are uncertain about whether the net effect of these changes will be positive or negative.	**Moderately confident** that there will not be largely negative effects on subspecies' redundancy.	
Effects of Stressors: - SPECIES (Rangwide) [REPRESENTATION]	We do not expect an effect of changed hydrology on the subspecies' representation because of the uncertainty surrounding many of these relationships and how they may affect the subspecies. We expect that there may some negative effects (increased scouring) and some positive effects (longer growing season). We are uncertain about whether the net effect of these changes will be positive or negative.	**Moderately confident** that there will not be largely negative effects on subspecies' representation.	
RISK OF EXTIRPATION 2023	We have not identified any populations at risk of extirpation due to the effects of changed hydrology. Changed hydrology is too uncertain of a risk to the subspecies to add to the model as a risk factor.		

THEME: Land Management			
[ESA Factor(s): A]	**Analysis**	**Confidence / Uncertainty**	**Supporting Information**
SOURCE(S)	Land management actions	**Highly confident.**	
- Activity(ies)	Land management and uses. 1. Cattle Grazing. 2. Recreation (ie camping, hiking, ATV use). 3. Timber Harvest. 4. Road building. 5. Mining	**Highly confident.**	Behnke 1979, p 102 Alves et al. 2008
STRESSOR(S)	**1. Cattle Grazing:** Cattle grazing reduces riparian vegetation, increases sediment inputs, and alter hydrologic regimes. Land management that removes or degrades natural riparian or updland vegetation can impact the quality of water and stream channels in downstream reaches. This occurs through runoff of sediment or physical alteration of stream through stream bank erosion. Grazing within riparian areas can result in soil compaction, damage or elimination of plants, reduction in terrestrial insects (which fall into the water and are about half the trout diet), and changes in fluvial processes. Improper grazing can cause adverse impacts (e.g., loss of cover, increased sedimentation, loss of riparian vegetation) to some individual RGCT populations, especially during drought conditions when the cattle tend to concentrate in riparian areas. The effects of excessive grazing can also result in long-term impacts that change hydrology and soils, leading to downcutting or headcutting. **2. Recreation:** Heavy recreational use can result in damage such as reducing density of herbaceous plants, eliminating seedlings and younger trees, and increasing tree diseases. Additionally, recreation can increase sediment inputs to streams with road and trail construction. **3. Timber Harvest:** Logging affects riparian ecosystems through tree falling, log skidding, road construction, and direct removal of vegetation, all of which add sediment to streams. **4. Road building:** Road construction contributes significant sediment to streams as land is disturbed, and existing roads can collect add sediment to streams. Additionally, culverts and bridges constrict the channel, changing the channel morphology, leading to ponding upstream of the structure and erosion and bankcutting downstream. Culverts under roads may serve as migration barriers, which can be positive (preventing nonnative trout invasions) or negative (fragmenting RGCT habitat). **5. Mining:** Mining as well as sand and gravel operations can alter flow and sediment regimes.	**Highly confident.**	Behnke 1979 p 102 Armour et al. 1994, p. 10 Trimble and Mendel 1995 Fausch et al. 2006, p. 19 Saunders and Fausch 2007, p. 1224 Saunders and Fausch 2012, p. 1525 Poff et al. 2011, p. 2, 6

THEME: Land Management			
[ESA Factor(s): A]	**Analysis**	**Confidence / Uncertainty**	**Supporting Information**
- Affected Resource(s)	Aquatic habitat (providing breeding, feeding, and sheltering areas). Food availability is reduced when riparian area is overgrazed, resulting in a less heterogeneous riparian zone. Sediment-free gravels and cobbles on stream bottom are vital for producing aquatic insects for food and serving as spawning areas for egg incubation.	**Highly confident.**	Young et al. 2005, p 2400 Pritchard and Cowley 2006, p 25 Saunders and Fausch 2007, pp. 1221, 1224 Budy et al. 2012 p 437 Saunders and Fausch 2012, p. 1525
- Exposure of Stressor(s)	Land management changes that affect RGCT stream conditions, where they occur, would represent long-term changes in the stream conditions that could affect all life stages of RGCT.	**Low Confidence** that the stressors are actually exposured to RGCT populations.	
- Immediacy of Stressor(s)	Some land management activities have occurred in the past, present, and future. Past practices were likely more severe than current practices, due to implementation of best management practices and, for example, more restrictive travel management rules on Forest Service lands. Grazing has decreased overall in the last 20 years and has been better managed.	**Moderately confident.**	USFS 2005 Poff et al. 2011, p. 2
Changes in Resource(s)	Decrease in food availability and decrease in adequate spawning areas due to siltation in substrates.	**Moderately confident.**	Young et al. 2005, p 2400 Pritchard and Cowley 2006, p 25 Saunders and Fausch 2007, p. 1224 Budy et al. 2012 p 437 Saunders and Fausch 2012, p. 1525
Response to Stressors: - INDIVIDUALS	Reduced fitness of individuals if food supply is limited. Reduced survival of young and juvenile stages. Reduced reproductive success due to limited spawning areas.	**Moderately confident.**	
POPULATION & SPECIES RESPONSES			
Effects of Stressors: - POPULATIONS [RESILIENCY]	Possible reduced fitness, reduced survival, and reduced reproduction rates for affected populations. Reduced trout biomass when riparian area is overgrazed (resulting in less available terrestrial insects) Review of "habitat quality" of RGCT streams (Alves 2007, p. 20), found 56.8 % had good or excellent quality.	**Low Confidence** that land management activities are having significant population-level effects.	Alves et al. 2007 Saunders and Fausch 2007, p. 1224 Alves et al. 2008 Saunders and Fausch 2012, p. 1525

THEME: Land Management			
[ESA Factor(s): A]	**Analysis**	**Confidence / Uncertainty**	**Supporting Information**
- SCOPE	Land use activities occur at some level across the range of RGCT (percentages represent percent of occupied habitat experiencing these activities) (Alves 2008): 1. Grazing 87% 2. Recreation 90% 3. Timber harvest 19% 4. Roads 58% 5. Mining 3% The intensity of each activity as related to potential effects on RGCT habitat, individuals, and populations depends on the specific level of activities and the conditions at each site. Overall, land management practices have improved and have less direct impact on Rio Grande cutthroat trout streams, and some streams are still recovering from past land management practices. 1) GRAZING: Specific information on grazing impacts to Rio Grande cutthroat trout habitat on a rangewide basis is not available. We have no information that leads us to conclude that improper grazing is significantly affecting RGCT rangewide. 2) RECREATION: ATV use off of designated routes has been prohibited, reducing the impact of off road vehicles on the landscape. Camping and hiking have minimal effect on RGCT. 3) TIMBER HARVEST: Timber harvest in the National Forests has declined appreciably in the last 20 years. While the effects of past logging practices may still be evident on the landscape in some locations, we have no information to conclude that timber harvest is significantly affecting RGCT populations. 4) ROADS: Roads have been identified as an area of concern for some streams (e.g., Tio Grande, Rio Grande del Rancho). Culverts serve as migration barriers on certain streams but may also be fragmenting habitat in other locations. The USFS Travel Management Plan directs road building and includes guidance to minimize effects on aquatic resources. Although there have been some local effects of roads, they are not affecting the subspecies rangewide. 5) MINING: Occurs within 3% of RGCT streams. Not a significant factor.	**Moderately confident** that this represents the scope of land use activities.	USFS 2005 (70 FR 68264) Alves et al. 2008 Peterson et al. 2013b, p. 5

THEME: Land Management			
[ESA Factor(s): A]	**Analysis**	**Confidence / Uncertainty**	**Supporting Information**
Effects of Stressors: - SPECIES (Rangwide) [REDUNDANCY]	We do not expect an effect of land management on the subspecies' redundancy because of lack of response expected at the population level.	**Moderately confident**	Alves et al. 2008
Effects of Stressors: - SPECIES (Rangwide) [REPRESENTATION]	We do not expect an effect of land management on the subspecies' representation because of lack of response expected at the population level.	**Moderately confident.**	
RISK OF EXTIRPATION 2023	We have not identified any populations at risk of extirpation due to the effects of land management. Land management is not a high enough risk to the subspecies to analyze further.		

THEME: Angling			
[ESA Factor(s): B]	**Analysis**	**Confidence / Uncertainty**	**Supporting Information**
SOURCE(S)	Recreational Anglers		
- Activity(ies)	Fishing for RGCT.	**Highly confident**	
STRESSOR(S)	Mortality of those fish kept by anglers; occasional mortality of fish caught and released due to handling stress or damage from hooks.	**Highly confident**	Bartholomew and Bohnsack 2005, p. 140
- Affected Resource(s)	Individual fish die or may experience stress for a period of time.		
- Exposure of Stressor(s)	Those fish caught are exposed to the stressor. Those kept die. Those released may experience stress or occasionally death.	**Highly confident**	Bartholomew and Bohnsack 2005, p. 140
- Immediacy of Stressor(s)	**Past:** Angling for RGCT has occurred for at least a century, likley more. **Current/Future:** Angling is regulated by the state wildlife agencies. In NM, reduced bag limit of 2/day, in CO a bag limit of 4/day; some streams in both states are catch-and-release only. Special angling regulations occur on 85% of conservation populations. Fishing is likely to continue to occur at these same levels.	**Highly confident**	NMDGF 2002, p. 22 Alves et al. 2008, p. 47, 48
Changes in Resource(s)	Fish kept by anglers die. Fish released experience stress from handling and may die from injuries sustained, although this is expected to be rare.	**Highly confident**	Bartholomew and Bohnsack 2005, p. 140
Response to Stressors: **- INDIVIDUALS**	Individual fish die or may experience stress for a period of time.	**Highly confident**	Bartholomew and Bohnsack 2005, p. 140
POPULATION & SPECIES RESPONSES			
Effects of Stressors: **- POPULATIONS [RESILIENCY]**	Because conservation populations of RGCT are remote and RGCT are small, angling pressure on the populations is not expected to have a population-level effect.	**Highly confident**	Alves et al. 2008, p. 47
- SCOPE	Angling occurs in 84% of conservation populations. Many of the streams with pure populations of Rio Grande cutthroat trout are remote (*e.g.* populations in the upper Pecos GMU) and angling pressure is light.	**Highly confident**	Alves et al. 2008, p. 47
Effects of Stressors: **- SPECIES (Rangwide) [REDUNDANCY]**	Because no population-level effect of angling is expected, we do not expect there to be an effect on redundancy.	**Highly confident**	

THEME: Angling			
[ESA Factor(s): B]	**Analysis**	**Confidence / Uncertainty**	**Supporting Information**
Effects of Stressors: - SPECIES (Rangwide) [REPRESENTATION]	Because no population-level effect of angling is expected, we do not expect there to be an effect on representation.	**Highly confident**	
RISK OF EXTIRPATION 2023	We have not identified any populations at risk of extirpation due to the effects of angling.		

THEME: Management Actions			
[ESA Factor(s): E]	**Analysis**	Confidence / Uncertainty	Supporting Information
SOURCE(S)	Fisheries and land managers removing nonnatives, constructing and maintaining barriers, reintroducing RGCT, conducting riparian restoration, and improving habitat.	**Highly confident**	Conservation Strategy 2013 Conservation Agreement 2013
- Activity(ies)	**Nonnatives:** chemical removal, physical suppression, barrier construction and maintenance; **Riparian restoration:** restricting grazing, reducing roads and timber harvest in riparian area; **Habitat improvement:** reducing sediment inputs, improving pool ratio	**Highly confident**	Vermejo Park Ranch et al. 2013 Conservation Strategy 2013 Conservation Agreement 2013 Colorado Parks and Wildlife 2013 NMDGF 2013
STRESSOR(S)	Reduces stressors related to nonnative species, stream drying, land management, and water temperature.	**Highly confident**	
- Affected Resource(s)	RGCT populations		
- Exposure of Stressor(s)	**Nonnatives:** reducing and eliminating nonnatives reduces their exposure to RGCT; **Riparian restoration** reduces exposure of RGCT to stream drying and water temperature changes; **Habitat improvement** reduces exposure of RGCT to stream drying and water temperature changes.	**Highly confident**	Vermejo Park Ranch et al. 2013 Conservation Strategy 2013 Conservation Agreement 2013 Colorado Parks and Wildlife 2013 NMDGF 2013
- Immediacy of Stressor(s)	N/A		
Changes in Resource(s)	Populations affected by management actions will remain stable or grow, as management actions reduce the stressors to the population (ie, nonnative trout removals, barrier maintenance, riparian management)	**Highly confident**	Vermejo Park Ranch et al. 2013 Conservation Strategy 2013
Response to Stressors: - INDIVIDUALS	Individuals will be exposed to fewer stressors, although the primary response will be at the population level.	**Highly confident**	Vermejo Park Ranch et al. 2013 Conservation Strategy 2013

THEME: Management Actions			
[ESA Factor(s): E]	**Analysis**	Confidence / Uncertainty	Supporting Information
POPULATION & SPECIES RESPONSES			
Effects of Stressors: - POPULATIONS [RESILIENCY]	Population resiliency will increase, and new populations with high resiliency will be added. Vermejo Park Ranch project expected to result in 20% increase in occupied stream miles for species, and likely to support a large interconnected population of over 75,000.	**Highly confident**	RGCT status assessment model Kruse 2013, p. 2 Vermejo Park Ranch et al. 2013
- SCOPE	Resiliency will likely increase wherever management actions occur	**Highly confident**	RGCT status assessment model
Effects of Stressors: - SPECIES (Rangwide) [REDUNDANCY]	The more reslient populations throughout RGCT range, the more redundancy will increase.	**Highly confident**	
Effects of Stressors: - SPECIES (Rangwide) [REPRESENTATION]	Representation will increase as populations are restored and rehabilitated.	**Highly confident**	
RISK OF EXTIRPATION 2023	We considered the management actions in the Vermejo Ranch CCAA and the CA/CS in future population projections. Additionally, we considered that, due to the importance of the species to both states, the states would continue to manage the species at some level as they have in the past. In future projections (past 2023) we examined both high and low levels of management to incorporate the range of management intensity that may occur. See Appendix C for specific information on how management was incorporated into our analysis.		

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APPENDIX C
RIO GRANDE CUTTHROAT TROUT
STATUS ASSESSMENT MODEL

Note: This is an appendix to the 2014 Rio Grande Cutthroat Trout Species Status Assessment Report. It provides only a summary of the methodology used in the Rio Grande Cutthroat Trout Status Assessment Model. The Model results are presented in Chapter 5 of the Report.

INTRODUCTION

As a part of the species status assessment (SSA) for the Rio Grande cutthroat trout, we conducted an analysis to quantitatively characterize the viability of the Rio Grande cutthroat trout. Our objectives were twofold: 1) to estimate the probability of persistence of each extant Rio Grande cutthroat trout population over time; and 2) describe the future persistence of Rio Grande cutthroat trout by forecasting the likely number of populations expected to survive[1] across the subspecies' range over time.

The purpose of this analysis is to quantitatively reflect our understanding of the future viability of this subspecies by using our professional judgment to apply the best available information to assess the status of the Rio Grande cutthroat trout. Like all models, ours is an oversimplification of the real world, and we do not claim that this analytical tool provides highly certain predictive outcomes. Instead it is designed to explicitly portray our understanding of how the status of the Rio Grande cutthroat trout may look in the future given our assumptions about the factors that we believe most influence the viability of the subspecies. The assignment of numerical values to reflect our best professional judgment of the risks to the subspecies provides an explicit way to communicate our understanding, but it does not mean the model is an overall objective assessment. To the contrary, it is a quantitative tool to show clearly the results of our subjective assessment of the future risks faced by the subspecies. This effort may represent a novel approach for the Fish and Wildlife Service in using this kind of numerical system to evaluate the status of a species by quantitatively forecasting the future resiliency, redundancy, and representation of the subspecies. This Appendix describes the analysis used in the accompanying Rio Grande Cutthroat Trout Species Status Assessment Report (SSA Report).

This analysis was conceived in large part based on the ongoing modeling work being conducted by a group of scientists using a Bayesian Network (BN) model to more comprehensively estimate the probability of persistence for Rio Grande cutthroat trout populations (funded by the State of Colorado). The effort was referenced in the 2013 Rio Grande cutthroat trout Conservation Strategy (p. 18). This new BN model for Rio Grande cutthroat trout is intended to provide both an assessment to measure population persistence and to provide a management planning tool for decisions about alternative future management actions. We had hoped to use the outcome from BN model in our status assessment for the Rio Grande cutthroat trout. However, the BN model was still under development at the time we needed to move forward in our analysis to support upcoming decisions related to the status of Rio Grande cutthroat trout (under the Endangered Species Act). We recognize that the work this group is doing is expected to be much more robust compared to our effort described here because it is planning to: 1) include many more factors; 2) incorporate the outputs of other modeling efforts; 3) use Bayesian statistics that allow for cumulative and synergistic relationships to be considered; and 4) include broader expert judgment input into the probability tables.

Nevertheless, to the extent possible, we attempted to incorporate many of the ideas and concepts from the ongoing BN modeling effort. Both efforts are intended to produce results that estimate the probability of

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[1] For this report, the terms "persisting" and "surviving" are used interchangeably when referring to populations sustaining themselves beyond the end points evaluated.

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persistence of each Rio Grande cutthroat trout population in 2040 and 2080. Therefore, the outputs from our analysis should be directly comparable to the future output of the BN model. We appreciate that the authors developing the BN model shared preliminary descriptions with us so that we could craft much or our work in a similar fashion with consistent assumptions where possible. We also gained inspiration from the work of Roberts *et al.* (2013) where they used a simpler BN model to estimate probability of persistence for Colorado River cutthroat trout and from the unpublished work of Rogers (2013) who also used a simpler BN model to estimate probability of persistence for Rio Grande cutthroat trout. These examples were very helpful in our development of this analysis.

MODEL SUMMARY

This report documents the analysis that we undertook to quantitatively forecast the Rio Grande cutthroat trout's future condition in a way that addresses viability in terms of the resiliency, redundancy, and representation (Figure C1). As a consequence we developed two separate, but related, modules that:

1. Estimate the probability of persistence for each Rio Grande trout population by GMU for 3 time periods under a range of conditions; and
2. Estimate the number of surviving populations by GMU for 3 time periods under several scenarios.

For the first module, we used 7 risk factors to estimate the probability of persistence of each Rio Grande cutthroat trout population (Figure C1). For each risk factor, we used one or more population metrics that contribute to the risk of extirpation of the populations. We used our own expert judgment to develop risk functions for each population metric. These judgments were based on our understanding of these risk factors as explained in Appendix B and Chapter 4 of the draft SSA Report. We only considered the risk factors that we deemed are likely to have population level impacts based on our cause and effects analysis. For 4 of the risk factors, we accelerated the rate of risk increase over time because we believe that environmental changes associated with global climate change will likely increase the risks associated with those factors. We summed all the risk functions for each population and subtracted that sum from 1 to calculate a probability of persistence for each population[2]. We did this calculation for each population for future timeframes of 2023, 2040, and 2080. We also ran this model with and without suppression management activities for controlling competing nonnative trout at 10 populations where suppression is currently occurring. And we did the analysis under two conditions of moderate and severe effects of climate change. These forecasts resulted in a description of the resiliency of the populations in terms of probability of persistence of the current populations. By analyzing the resulting persistence probabilities by GMU, the results also provide a picture of representation and redundancy.

For the second module, we conducted a survival simulation based on the output of persistence probabilities from module 1 to forecast the number of populations that may survive over time (Figure C1). We used a randomization process to simulate whether a population remains extant or goes extinct based on our modeled probability of persistence. After running the simulation 100 times, we calculated an average number of surviving populations with a 95% confidence interval by GMU under variable conditions. To that simulated number of surviving populations we added an estimate of the number of populations that may be restored over time by proactive management. Forecasting future restoration efforts has a large amount of uncertainty beyond the next 10 years, so we used a range of possibilities to include in the model output. For the overall population survival model, we considered 9 possible scenarios including the 3 time intervals that produces a best case, worse case, and intermediate case projection. The results from this analysis provides an assessment of future redundancy and representation based on the number of forecasted surviving populations rangewide and an assessment of representation as we report the results by GMU over time.

We also calculated the potential number of stream kilometers that are forecasted to be occupied in the future using our future population simulation. We did this in order to compare the current and future status of the Rio

[2] If a population's probability of persistence fell below zero, then the population was given a zero for the remainder of the analysis.

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Grande cutthroat trout to the historical status in terms of total occupied habitat. This estimation is not very precise, however, because we had to make large assumptions[3] in estimating the future amount of occupied stream kilometers by population. Therefore, we only use these results as a general guide to compare the possible total occupied habitat in the future to historical and current levels.

For a description of the results of this analysis, please refer to Chapter 5 of the SSA Report.

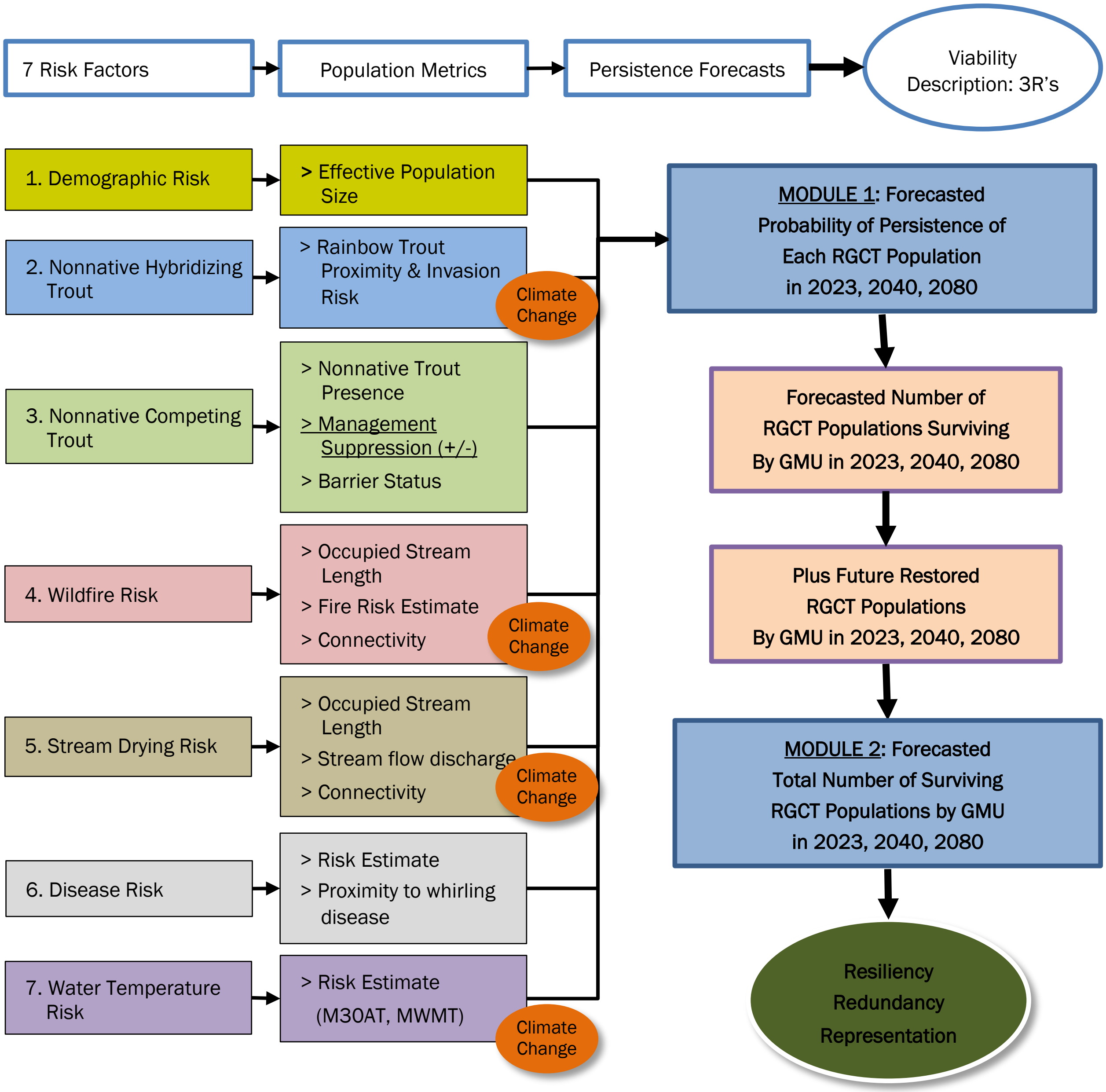

Figure C1. Conceptual diagram of Rio Grande cutthroat status assessment model.

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[3] The assumptions were related to using average stream lengths for future persisting populations because the model does not predict which streams will be persisting in the future.

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Information Sources

We primarily used information from the Rio Grande cutthroat trout rangewide database (RGCT Database) from 2012, the most recent database available (see section 2.5, Management History of Rio Grande Cutthroat Trout, for more information about the database). We supplemented information from the RGCT Database based on new information received from various sources, included communications with Rio Grande cutthroat trout biologists from the states of Colorado and New Mexico. We also relied heavily on the prior work done for the most recent rangewide assessment for the Rio Grande cutthroat trout (Alves *et al.* 2008).

As a starting point, we used the 128 conservation populations [4] as tabulated from the RGCT Database by Rogers (2013, pp. 5–6, 18–21). These populations are from the database, with the exception that 6 of the Rio Grande cutthroat trout populations in the database were split into 2 populations to take into account the presence of a fish barrier that alters the condition of the populations upstream and downstream of the barrier. In some instances we consulted the Rio Grande cutthroat trout biologists to determine which conditions in the database applied to both upstream and downstream populations and which conditions were different between the 2 reaches. The 128 populations include these split populations. See Table C14 for a list of conservation populations evaluated in this analysis.

Six of the 128 populations that were in the initial version of the database we used were effectively removed from the analysis as they are presumed to be currently extirpated. Four of these populations (all from Lower Rio Grande GMU) were extirpated due to fire. One other population (also from the Lower Rio Grande GMU) was removed because after being separated into lower and upper segments based on the presence of a fish barrier, the lower segment does not contain Rio Grande cutthroat trout. One population (from the Canadian GMU) is not considered conservation population because it is currently more than 10% introgressed with rainbow trout genes. Our evaluation then used 122 as the number of extant conservation populations.

Time Frames Analyzed

We considered the current condition of the Rio Grande cutthroat trout as the status in 2013. We then forecasted the probability of persistence and survivability for Rio Grande cutthroat trout populations at three future time intervals: 2023, 2040, and 2080.

2013. This is considered the current condition of populations based when the Conservation Strategy was signed and one year from the latest data (2012) from the RGCT Database (see the discussion under Information Sources above). All the forecasting for future time intervals related to analysis of the risk factors and risk functions are largely based on the current conditions of the populations.

2023. This is approximately 10 years from current. This relatively short time period corresponds with the 10-year Rio Grande cutthroat trout Conservation Strategy to be implemented as part of the Rio Grande cutthroat trout Conservation Agreement signed in 2013. It also represents about two to three Rio Grande cutthroat trout generations (assuming generation time is between 3 and 5 years). Based on our understanding of recent environmental conditions and our ability to forecast over the next 10 years, we have high confidence (more than 90% sure) in our ability to forecast future conditions in 2023 related to the risk factors evaluated and to the responses of Rio Grande cutthroat trout populations.

[4] "Conservation populations" refers to populations of Rio Grande cutthroat that are less than 10% introgressed with nonnative trout genes. Throughout this document, references to populations of Rio Grande cutthroat trout refer to conservation populations.

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2040. This is approximately 25 years from current[5]. This time frame represents about five to eight Rio Grande cutthroat trout generations. We chose this time frame to correspond with available downscaled climate change models. Although we were not able to include climate change directly in our models, the BN model in development is planning to use climate change models and produce output of probability of persistence of Rio Grande cutthroat trout populations in 2040. We desired for the output from our model to be comparable to the developing BN model, and therefore, we used the same time intervals for our forecasting. This timeframe also represents a reasonable time from present when we have moderate confidence (70 to 90% sure) in our ability to forecast future environmental conditions related to the risk factors evaluated and to the responses of Rio Grande cutthroat trout populations.

2080. This is approximately 65 years from current. This time frame represents about 13 to 21 Rio Grande cutthroat trout generations. As with the 2040 time interval, this relatively long time frame of 65 years also corresponds with the output of downscaled climate change models and the developing BN model, so we chose 2080 for similar purposes as the 2040 time frame. It represents our outermost estimate for forecasting, where our confidence naturally decreases to somewhat confident (50 to 70% sure) in our ability to forecast future environmental conditions related to the risk factors evaluated and to the responses of Rio Grande cutthroat trout populations.

METHODS: POPULATION PERSISTENCE[6]

Risk Factors and Risk Functions

To accomplish our first objective to estimate the resiliency of current Rio Grande cutthroat trout populations, we developed a model to estimate the probability of persistence for each current population. After reviewing the causes and effects of factors that could have population-level effects to Rio Grande cutthroat trout, we chose seven risk factors to include in our analysis. For additional discussion of these factors, beyond the discussion here of how they were used in the analysis model, please refer to Chapter 4, Vulnerabilities, and Appendix B, Evaluating Causes and Effects for Rio Grande Cutthroat Trout Species Status Assessment, in the accompanying Draft SSA Report.

Seven Risk Factors

1. Demographic Risk
2. Hybridizing Nonnatives
3. Competing Nonnatives
4. Wildfire Risk
5. Stream Drying Risk
6. Disease Risk
7. Water Temperature Risk

For each risk factor (described in more detail below) we chose one or more Rio Grande cutthroat trout population metrics available to consider how that factor affects the risk of extirpation of Rio Grande cutthroat trout populations. For each state[7] of the population metric we assigned a risk function to that state for that risk factor (see Table C14 for a list of the population metrics used in this analysis). The first risk function for the 2023 forecast represented the probability (assigned as a number from 0 to 1) that that risk factor could result in the extirpation of a Rio Grande cutthroat trout population in that state (Table C1). The risk levels listed in Table C1 provide our valuation of the qualitative risk assessment we considered for each risk function and risk factor. We predicted these risk functions based on our best professional judgment as explained below under each risk function. To further

Key Uncertainty

Assigning risk functions is a fundamental assumption of this analysis. We cannot claim any particular level of accuracy related to the assignment of these risk functions. However, we are confident that the risk functions represent our best understanding of the risks to Rio Grande cutthroat trout related to the risk factors.

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[5] We recognize that 2040 and 2080 are not exactly 25-year and 65-year forecasts from "current" (these dates are actually 27 and 67 years from 2013, which we are considering current), but it was more convenient to consider and calculate. We also realize that none of these forecasts are precise enough that +/- 2 years will make a substantial difference in the results in the model.

[6] All of the calculations and simulations for this model were conducted using Microsoft Excel 2010.

[7] "State" means the state of that metric, whether it is a category based on a natural scale, such as effective population size, or a condition such as Yes or No if competitive nonnative trout are present in the population.

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predict the risk functions for 2040 and 2080 time intervals, we scaled up the risk functions proportional to the length in the time interval. In other words, assuming we were considering 10-year, 25-year, and 65-year forecasts, we multiplied the 10-year risk function by 2.5 and 6.5 to determine the 2040 and 2080 risk functions, respectively (see below for explanation of increasing risks due to climate change). Because we had no information that the risks would change at a different rate over time, we increased the risks at a similar rate for all risk factors.

So, as a hypothetical example, for risk factor X (numbered 1 through 7) in state X.1 of the population metric, we might assign the risk function of 0.1, which means we predict that the population in that state has a 10% chance of extirpation by 2023 as a result of that risk factor. For this example, the 2040 risk function of risk factor X would be 0.25 in 2040 (a 25% chance of extirpation by 2040) and 0.65 in 2080 (a 65% chance of extirpation by 2080).

In assigning our risk functions we considered two components of risk: 1) the likelihood that the factor will actually occur over the given time frame; and, if it should occur, 2) the likelihood that the factor will result in the extirpation (as opposed to only some effects to individuals) of the population. We included both of these ideas in our judgments to assign risk functions. These risk functions reflect our perception about the potential impacts of these risks on the probability of persistence at the population scale.

Table C1. Categories of risk predictions used in the Rio Grande cutthroat trout status assessment model. Chance of extirpation is our predicted chance that a single population of Rio Grande cutthroat trout will become extirpated due to the effects of a given risk factor by 2023 (over about 10 years).

Risk Level by 2023	Risk Function	Chance of Extirpation
No risk	0.0	0%
Extremely low risk	0.001	0.1%
Very low risk	0.005	0.5%
Low risk	0.010	1.0%
Moderate risk	0.020	2.0%
High risk	0.100	10%
Very high risk	0.200	20%

Climate Change Considerations

One of the important factors to consider in the future status of Rio Grande cutthroat trout relates to the potential effects of climate change. Climate change represents a future source of environmental changes that can exacerbate a number of different stressors to Rio Grande cutthroat trout populations. Our assessment found that climate change is likely to influence four of the seven risk factors evaluated in this model (Figure C1): 2. Hybridizing Species Risk; 4. Wildfire Risk; 5. Stream Drying Risk; and 7. Water Temperature Risk. Figure C2 is an overview diagram of the conceptual relationships of the cause and effects pathway relating future climate change to potential impacts on populations of Rio Grande cutthroat trout.

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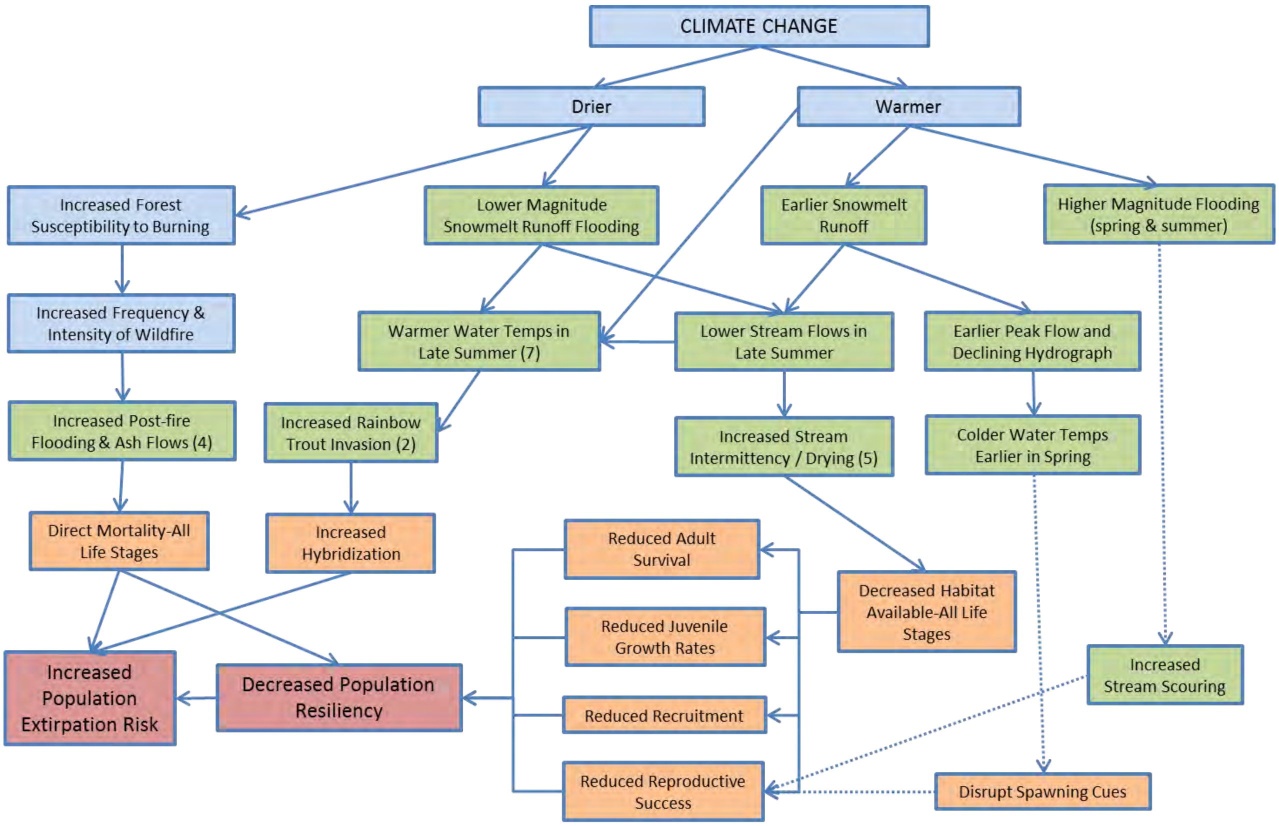

Figure C2. Diagram of conceptual relationships of climate change effects on Rio Grande cutthroat trout populations. Sources of effects are in blue; stressors are in green; responses by individuals are in orange; and population response is in red. Numbers in parentheses correspond to risk factors used in the status assessment model. Dashed lines indicate high uncertainty about the relationships and so those factors were not used in the status assessment model.

In addition to the four risk factors included in our analysis, we also considered two other potential stressors associated with possible hydrological changes that could be influenced by climate change. One is earlier peak spring snowmelt runoff flows and an earlier declining hydrograph. These changes could occur as a result of earlier warming temperatures in the spring, which could disrupt spawning cues and cause reduced reproductive success. Conversely, the longer growing season the fry may experience could also enhance juvenile survival through the following winter and increase recruitment success (this positive influence is not depicted in Figure C2). The other change is related to overall increase in flooding magnitude that could result from rain on snow events in the spring or increased large-scale flash flood events during the summer. These hydrological changes could result in increased stream scouring and alter stream habitats for the fish, particularly during spawning when eggs are in the gravel or fry have emerged; both are susceptible to being displaced and lost in flood events. In considering these two situations, we determined that the uncertainty of these relationships were too great to incorporate further into our analysis. Although these effects are possible, with our current level of understanding we could not adequately account for how these changes might result in specific population-level effects to the Rio Grande cutthroat trout, therefore we did not include these risk factors in the status assessment model.

We can conceptually understand that a warming climate can exacerbate four of these stressors; however, the magnitude of the increase in these stressors due to climate change is difficult to project and quantify. Therefore, to address this uncertainty we considered two different levels of climate change influence in our model. We incorporated these influences by changing the rate of increase in risk over time for those risk factors identified to be influenced by climate change (Table C2).

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In calculating the probability of persistence of Rio Grande cutthroat trout populations over time we used three different multipliers to scale the risk from 2023 (a 10-year forecast) to 2040 (a 25-year forecast) and 2080 (a 65-year forecast). First, for those risk factors without a consideration for climate change (risk factors 1, 3, and 6), we assumed the risk would increase over time in a linear relationship proportional to the amount of time in the forecast. In other words, the 25-year risk function[8] (in 2040) is 2.5 higher than the 10-year risk function (25/10), and the 65-year risk function is 6.5 times higher than the 10-year risk function (65/10) (Table C2).

Dealing with Uncertainty

To address the uncertainty of climate change we ran the model with two climate change scenarios: one with a 5% and 10% increased risk in 2040 and 2080, respectively, and a second scenario with a 20% and 40% increased risk in 2040 and 2080, respectively.

However, for those risk factors that we believe are likely to be influenced by climate change (risk factors 2, 4, 5, and 7), we increased the risk over time such that the risk function increases more than the proportional time interval. To account for these increases in risk, we multiplied the risk functions used in the risk factors without climate change influences to reflect larger increases in risk of extirpation over time. We used our best professional judgment to estimate the multipliers that correspond with increasing risks. In addition, because of the high uncertainty associated with climate change we considered a "moderate" and a "severe" effect of climate change. For the moderate climate change effect, we increased the risk function over time by 5% for the 2040 forecast and 10% for the 2080 forecast (Table C2). The resulting moderate climate change 2040 multiplier was 2.625 ([25/10]*1.05), and the multiplier for 2080 was 7.15 ([65/10]*1.1). For the severe climate change effects, we increased the risk function over time by 20% for the 2040 forecast and 40% for the 2080 forecast (Table C2). The resulting severe climate change 2040 multiplier was 3.12 ([25/10]*1.2), and the multiplier for 2080 was 9.1 ([65/10]*1.4). These multipliers were our best judgment of the potential effects of climate change on the risk factors, and using two multipliers provided us the opportunity to view the model results under two different climate change scenarios.

Table C2. Multipliers used to scale risk functions over time from the 2023 forecast to the 2040 and 2080 forecasts for Rio Grande cutthroat trout risk factors without climate change effects and with "moderate" and "severe" climate change effects.

	Risk Multiplier Over Time		
Risk Factors	**2023**	**2040**	**2080**
Not considered affected by climate change (Risk Factors 1,3,6)	1	2.500	6.500
Affected by climate change (Risk Factors 2,4,5,7):			
"Moderate" Effects (5% and 10% increase)	1	2.625	7.150
"Severe" Effects (20% and 40% increase)	1	3.120	9.100

[8] The discussion of the specific risk factors and risk functions are in the sections immediately following this discussion of climate change considerations.

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Risk Factors

The following descriptions explain the 7 different risk factors we analyzed, the metrics we included, and the risk functions that we assigned.

1. Demographic Risk

The risk factor associated with demographic effects is associated with the vulnerabilities related to small population sizes. We assume that small population sizes can lead to loss of genetic diversity and increased inbreeding depression, and the larger the population size the less likely deleterious genetic effects will be. These genetic effects increase the population's likelihood of extirpation from other risks. See Chapter 4 and Appendix B of the Draft SSA Report for discussion of the cause and effects of small population sizes on Rio Grande cutthroat trout.

Metric: Effective Population Size

We used an estimate of the effective population size for each Rio Grande cutthroat trout conservation population as the metric to determine the risks from small population sizes. The majority of the populations in the RGCT Database (101 of 127) had a reported density of Rio Grande cutthroat trout based on collected field data. The standard metric used in the database is an estimate of the number of adult fish (individuals greater than 120 mm TL (total length)) (Alves *et al.* 2008, p. 7) per mile of occupied stream length, usually estimated through three-pass depletion sampling. Although many populations have estimates over multiple sampling years, for our analysis we used the most recent survey data available. The year these estimates were made ranged by population from 2001 to 2012 (with one estimate from 1990). For these populations with density estimates, we multiplied the density of adult fish by the occupied stream length (see discussion of occupied stream length below) to reach a total estimate of adult fish for each population.

Some conservation populations are made up of multiple stream reaches identified in the RGCT Database with unique population density estimates. In these cases (5 populations) we used the estimate for each stream reach multiplied by that occupied stream length and then summed the products for each stream reach to generate a total number of fish for the population.

In some other cases there were no data available on the estimate of the number of fish in the population. For some of these populations, the database contained an estimate of the range of fish densities that were judged in the field to occur in those populations[9]. In these cases (7 populations), we used the midpoint of the density range as the density for those populations as the population estimate.

In a few other cases there was no data available for fish density, and there was also no density category included in the database. For these cases (14 populations) we used a calculation to estimate the total population size based on the occupied length of stream. This relationship was developed for other species of cutthroat trout by Young *et al.* (2005, p. 2404) and has been applied to Colorado River cutthroat trout modeling efforts (Roberts *et al*. 2013, p. 1388) and to Rio Grande cutthroat trout (Rogers 2013, p .5).

$$\sqrt{N_i} = 0.00508 \times (l_i + 5.148)$$

Where N_i is the census population size of cutthroat trout >75mm TL for population *i,* and l_i is the length of stream (meters) occupied by cutthroat trout.

The effective population size is an important metric to measure for assessing population persistence as it is considered a surrogate metric for genetic variation within a population. In general, the larger an effective population size the more genetic variation it should have. Low effective population sizes can result in loss of

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[9] The categories of fish density estimates from the Rio Grande cutthroat trout database are in the following ranges (and the midpoint we used in our calculations of total population estimate): 0 to 50 fish (25); 50 to 150 fish (100); 150 to 400 fish (no populations missing in this category); >400 fish (no populations missing in this category).

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genetic diversity via genetic drift and inbreeding. To our knowledge, no populations of native trout have been extirpated due to demographic risk alone; instead, it is a factor that can make the population more vulnerable to extirpation from other factors.

To calculate the effective population size (Ne) as a proportion of the total population size of individuals greater than 120 mm TL (N), we used 0.25 as the N/N_e ratio.

$$N_e = N_i \times 0.25$$

Where N_e is the effective population size estimate, and N_i is the census population size of cutthroat trout population *i*.

Key Assumption

For those population calculated from stream lengths, we used an N/N_e ratio of 0.25 to estimate effective population size. For those populations estimated from census data, we used an N/Ne ratio of 0.375 to estimate effective population size.

One limitation of using this relationship for our model is that it was based on estimating population sizes that included all age-1 and greater fish that are equal to 75 mm TL or longer, while the Rio Grande cutthroat trout database uses 120 mm TL as the standard for adult fish. As a result, using the data from the Rio Grande cutthroat trout database requires an adjusted N/Ne ratio to account for the fish in the 75-120 mm TL range. There is some debate in the literature about the appropriate ratio to apply; we followed the rationale used by Roberts *et al.* (2013, p. 1388) to use 0.25. However, we recognize that Roberts was assuming N was a measure of all individuals greater than 75 mm TL, rather than 120 mm TL. After analysis of Rio Grande cutthroat trout collection data in Colorado and New Mexico, Rogers (2014, pers. comm.) determined that 0.375 is the appropriate N/Ne ratio for census data in the RGCT database.

Risk Functions

Table C2 lists the risk functions we used for this risk factor. The larger the population, the lower the risk that a stochastic event associated with demographic risks will result in the extirpation of a population. We followed Roberts *et al.*'s (2013, p. 1387) method to use 500, 200, and 50 as states to predict potential genetic effects from small effective population sizes. At least 500 individuals were considered adequate to ensure long-term persistence, and less than 50 would be considered in danger of immediate inbreeding effects (Cowley 2007, p. 3). Populations between 50 and 200 are at risk of genetic consequences over the short term, and populations less than 500 but greater than 200 are at some risk over the long term. As long as populations are greater than 500, we do not expect demographic-related effects over any time interval (0% chance of demographic effects exacerbating other risk factors). Over the short term (to 2023) we would not expect any demographic risk for populations greater than 200. For populations less than 200 but greater than 50 we predict there is a low risk (1% over 10 years) of loss due to demographic effects exacerbating other risk factors. We predict a moderate risk (2% over 10 years) for populations less than 50.

The risk functions for demographic risk increase proportionally as the time interval lengthens (in other words the 2040 risk function is 2.5 times greater than the 2023 predicted risk, and the 2080 prediction is 6.5 times greater than the 2023 predicted risk) because we do not foresee any effects of climate change, or other sources, increasing the risk over time.

Table C3. States of population metrics and risk functions for demographic risks to Rio Grande cutthroat trout populations. Risks indicate the increased probability of extirpation from other risk factors.

		Predicted Risk of Extirpation		
	State of Population Metric	**2023**	**2040**	**2080**
1. Demographic Risk	1.1 Effective Population Size (N_e) > 500	0	0	0
	1.2 Effective Population Size (N_e) = 201 - 500	0	0.010	0.026
	1.3 Effective Population Size (N_e) = 50 - 200	0.010	0.025	0.065
	1.4 Effective Population Size (N_e) < 50	0.020	0.050	0.130

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2. Hybridizing Nonnative Trout

This risk factor associated with hybridizing nonnative trout describes the chance that a population will be extirpated or become hybridized at a level greater than 10%[10] over a given time frame. See Chapter 4 and Appendix B of the Draft SSA Report for discussion of the causes and effects related to hybridizing nonnative trout.

Metric: Proximity to Hybridizing Nonnative Trout

We evaluated the risk from hybridizing nonnative trout introduction as related to the proximity of hybridizing nonnative trout, mainly rainbow trout populations. The closer a population of (non-triploid) rainbow trout is to the Rio Grande cutthroat trout population, the greater the opportunity for either human-caused introduction or dispersal from a nearby stream. Additionally, those populations with secure stream barriers that prevent upstream fish dispersal have low risk of rainbow trout invasion.

Risk Functions (+Climate Change)

Table C3 lists the risk functions we used for this risk factor. We have assumed that populations identified in the Rio Grande cutthroat trout database as not tested but suspected to be unaltered (12 populations) actually are Rio Grande cutthroat trout conservation populations (less than 10% introgressed). For the purposes of this analysis, Rio Grande cutthroat trout populations greater than 10% introgressed are considered extirpated populations—there was one population in the database identified as greater than 10% introgressed. Populations identified in the Rio Grande cutthroat trout database as suspected hybridized (12 populations) were assumed hybridized and given a risk function of extirpation of 1 (already extirpated). Rio Grande cutthroat trout populations with less than 10% introgression of rainbow or other cutthroat trout genes are considered conservation populations of Rio Grande cutthroat trout.

> **Key Assumption**
>
> For this model, we assumed the genetic status of untested populations was as presumed in the Rio Grande cutthroat trout database.

The risk functions for the pure Rio Grande cutthroat trout populations were based on the presence of a secure stream barrier and whether the nearest wild rainbow trout population is located more or less than 10 km from the Rio Grande cutthroat trout population. This metric from the Rio Grande cutthroat trout database is consistent with that used by Alves *et al*. (2008, p. 35) to evaluate the risk of invasion by hybridizing nonnatives. For some populations, the Rio Grande cutthroat trout database identifies that there are no risks of hybridization. For those populations we used a 0% chance of extirpation by 2023, but we included a low risk (1% in 2040 and 2.6% in 2080 with no climate change multiplier) to account for the possibility of human-caused introduction. We assigned the risk of extirpation as very low (0.5% by 2023) for Rio Grande cutthroat trout populations where the nearest hybridizing nonnative trout population is greater than 10 km away. We believe this risk is very low because there are few examples of accidental introductions that are known. Rio Grande cutthroat trout populations located less than 10 km from hybridizing nonnative trout populations are at increased risk of extirpation, but still at a relatively low risk by 2023 (we predicted 1%, with moderate climate change effects), increasing proportionally in 2040 (2.6%) and 2080 (7.5%). Rio Grande cutthroat trout populations that are already invaded by rainbow trout (6 populations) but so far still persisting as conservation populations of Rio Grande cutthroat trout are considered at a very high risk of extirpation due to hybridization (10% in 2023 scaling up to 65% by 2080). We did not assign a 100% chance of extirpation by 2080 to sympatric populations because there are rare cases in which conditions for rainbow trout are not ideal, and Rio Grande cutthroat trout may continue to persist and not become introgressed more than 10%.

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[10] Rio Grande cutthroat trout populations with less than 10% genetic introgression are considered Rio Grande cutthroat trout conservation populations (Alves *et al.* 2008, p. 6). Rio Grande cutthroat trout populations with greater than 10% genetic introgression are not considered conservation populations.

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The risk functions for hybridizing nonnative trout increase proportionally as the time interval lengthens (in other words the 2040 risk function is 2.5 times greater than the 2023 predicted risk, and the 2080 prediction is 6.5 times greater than the 2023 predicted risk). We foresee that ongoing and future climate change could increase the risk of hybridization into the future because an expected drier, hotter climate could result in greater rainbow trout recruitment (Muhlfeld *et al.* 2014, p. 2). Therefore, we increased the rate of risk over time by 5% in 2040 and 10% in 2080 (for an predicte of moderate climate change effects, Table C4a) and by 20% in 2040 and 40% in 2080 (for an predicte of severe climate change effects, Table C4b). We used these two scenarios throughout the model to capture some of the uncertainty due to future climate change (see earlier discussion of Climate Change for more information).

Dealing with Uncertainty

How will climate change affect the risk of hybridization of Rio Grande cutthroat trout? To address this uncertainty we ran the model with moderate and severe levels of effects of climate change on the risk of invasion and hybridization by nonnative trout.

Table C4a. States of population metrics and risk functions for hybridizing nonnative trout risks to Rio Grande cutthroat trout populations under “moderate” climate change effects. Risks indicate the probability of an entire population being extirpated by hybridizing nonnative trout.

		Predicted Risk of Extirpation		
	State of Population Metric	**2023**	**2040**	**2080**
2a. Hybridizing Nonnative Trout (<u>moderate</u> climate change)	2a.1 <10% introgressed, or suspected unaltered, and no hybridization risk	0	0.010	0.026
	2a.2 <10% introgressed, or suspected unaltered, and > 10 km from rainbow trout	0.005	0.013	0.038
	2a.3 <10% introgressed, or suspected unaltered, and < 10 km from rainbow trout	0.010	0.026	0.075
	2a.4 Sympatric with rainbow trout	0.250	0.625	0.950
	2a.5 >10% introgressed or suspected hybridized	1	1	1

Table C4b. States of population metrics and risk functions for hybridizing nonnative trout risks to Rio Grande cutthroat trout populations under “severe” climate change effects. Risks indicate the probability of an entire population being extirpated by hybridizing nonnative trout.

		Predicted Risk of Extirpation		
	State of Population Metric	**2023**	**2040**	**2080**
2b. Hybridizing Nonnative Trout (<u>severe</u> climate change)	2b.1 <10% introgressed, or suspected unaltered, and no hybridization risk	0	0.010	0.026
	2b.2 <10% introgressed, or suspected unaltered, and > 10 km from rainbow trout	0.005	0.015	0.055
	2b.3 <10% introgressed, or suspected unaltered, and < 10 km from rainbow trout	0.010	0.030	0.109
	2b.4 Sympatric with rainbow trout	0.250	0.625	0.950
	2b.5 >10% introgressed or suspected hybridized	1	1	1

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3. Competing Nonnative Trout

This risk factor associated with competing nonnative trout describes the chance that a population will be extirpated over a given time frame as a result of the impacts associated with the presence of other nonnative trout, mainly brown and brook trout. While the majority of the impacts from nonnative trout (other than rainbow trout) comes from competition for space and resources, this risk factor also includes impacts from predation (particularly adult brown trout preying on young Rio Grande cutthroat trout). See Chapter 4 and Appendix B of the Draft SSA Report for discussion of the causes and effects related to competing nonnative trout. We considered three population metrics for this risk factor: presence of competing nonnative trout; management suppression; and barrier presence.

Metric: Presence of Competing Nonnative Trout

We evaluated this risk based on whether or not the competing nonnative trout are currently co-occurring with the Rio Grande cutthroat trout populations. Rio Grande cutthroat trout populations occurring with competing nonnative trout are at an increased risk of extirpation as a result. We used the information from the Rio Grande cutthroat trout database as to whether brown trout or brook trout are currently present. In some cases we supplemented this information with updated information from biologists familiar with the status of the populations. For five Rio Grande cutthroat trout populations in the database it was unknown whether competing nonnative trout are currently present. For these populations we assumed that the competing nonnative trout were not present.

Key Assumption

Where it was unknown if competing nonnative trout are co-occurring with 5 Rio Grande cutthroat trout populations, we assumed no competing nonnatives are present.

Metric: Management Suppression

A second metric we evaluated for those populations where competing nonnative trout are already present was the ongoing management suppression. For six Rio Grande cutthroat trout populations, fisheries managers are routinely (every few years) mechanically removing nonnative trout to suppress their populations temporarily (Alves *et al.* 2008, p. 48; RGCT Database). This suppression reduces the impacts on Rio Grande cutthroat trout populations and reduces their risk of extirpation. It is unknown whether these suppression activities will continue into the future. While it is likely that they may, at least for the 10-year duration of the Conservation Agreement, we chose to run our model with and without this suppression continuing. In this way we can weigh the benefits of these actions on the status of the Rio Grande cutthroat trout and evaluate this uncertainty in our model outputs. For the metric (described below) with suppression continuing, we assume that nonnatives will be mechanically removed on a regular basis through 2080 for those six populations where suppression is currently occurring.

Dealing with Uncertainty

Will current fisheries management activities continue to suppress nonnative trout? To address this uncertainty we ran the model with and without continued suppression actions.

Metric: Barrier Presence

Barriers to fish movement (either natural or man-made) are an important component for protecting Rio Grande cutthroat trout populations from invasion by nonnative trout and reducing their risk of extirpation (Alves *et al.* 2008, p. 5). The Rio Grande cutthroat trout database assesses the type of barrier present for each Rio Grande cutthroat trout population. We converted the information in the database to either a complete, partial, or no barrier reference for our metric (Table C5). We assumed that complete barriers are providing a high level of protection to prevent dispersal of nonnative trout upstream into a Rio Grande cutthroat trout conservation population. We assumed partial barriers were providing some limited protection from nonnative trout and populations with no barriers are not protected from nonnative trout.

Table C5. Categorizing references of fish barriers in the Rio Grande cutthroat trout database into a barrier metric for the status assessment model.

Database Barrier Reference	Barrier Metric	Database Barrier Reference	Barrier Metric
Manmade temporary	Complete	Water diversion/partial	Partial
Temperature		Waterfall/partial	
Bedrock		Culvert/partial	
Water diversion		Manmade dam/unknown	
Insufficient flow		Water diversion/unknown	
Waterfall		Pollution/partial	
Manmade dam		Manmade temporary/partial	
Culvert		Unknown/partial	
Manmade complete			
Debris		None	None
		NA	

Risk Functions

Table C6 lists the risk functions we used for this risk factor. While impacts of co-occurring nonnatives do not necessarily result in extirpation of Rio Grande cutthroat trout populations in the near term, they do increase the risk of extirpation of over time. For populations where nonnative trout are already present and no mechanical suppression of the nonnative trout are occurring, we considered those populations to be at a relatively high risk of extirpation by 2023 (10% chance of extirpation, scaling up to 65% chance of extirpation by 2080). For populations where the nonnative trout are already present but suppression actions are occurring, we reduced the risk of extirpation to none by 2023 and to a moderate risk by 2040 (2% chance or extirpation, scaling up to a 5.2% chance by 2080). For populations without nonnative trout present and with a complete or partial barrier, we predicted there was no risk of extirpation by 2023 and a low and moderate risk for complete and partial barriers, respectively. For populations without nonnative trout present and with no barrier in place, we predicted a moderate risk of extirpation by 2023 (2% chance, scaling up to 13% chance of extirpation by 2080).

The risk functions for competing nonnative trout increase proportionally as the time interval lengthens. Brown trout may be able to range farther upstream than in the past (due to warming streams affected by climate change) (K. Fausch, CSU, pers. comm. 2014). Conversely, brook trout are expected to be negatively affected by climate warming and concurrent flow regime changes more than cutthroat trout (Wenger *et al.* 2011a, pp. 1000–1001; Wenger *et al.* 2011b, p. 14176). Because the effects of climate change may increase brown trout populations but decrease brook trout populations, we did not change the rate of increase associated with this risk factor due to climate change.

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Table C6. States of population metrics and risk functions for competing nonnative trout risks to Rio Grande cutthroat trout populations. Risks indicate the probability of an entire population being extirpated by competing nonnative trout.

		Predicted Risk of Extirpation		
	State of Population Metric	**2023**	**2040**	**2080**
3. Competing Nonnative Trout	3.1 Nonnatives present, no management suppression	0.100	0.250	0.650
	3.2 Nonnatives present, with continuing management suppression	0	0.020	0.052
	3.3 Not present, complete barrier	0	0.010	0.026
	3.4 Not present, partial barrier	0	0.020	0.052
	3.5 Not present, no barrier	0.020	0.050	0.130

4. Wildfire

This risk factor associated with wildfire describes the chance that a population will be extirpated over a given time frame as a result of the impacts associated with wildfire in the contributing watershed and the resulting floods and changes in the stream. Wildfire is an inherent risk to the persistence of Rio Grande cutthroat trout populations that has increased in recent times due to the relatively small and isolated nature of the currently remaining populations of Rio Grande cutthroat trout. See Chapter 4 and Appendix B of the Draft SSA Report for discussion of the causes and effects related to wildfire. We considered two population metrics for this risk factor: fire risk as estimated in The Nature Conservancy's (TNC) modeling effort on fire risk, and the occupied stream length.

Metric: Fire Risk

We predicted wildfire risk using the output from a newly developed model by TNC (Miller and Bassett 2013). It reviewed the watershed conditions contributing to each Rio Grande cutthroat trout conservation population. It characterized the impacts of wildfire for each population as moderate or high based on models evaluating fire behavior and debris flow. Their analysis is limited in application because it does not take into account the probability of fire occurrence in a given location. Factors affecting risk of ignition include location, fire return interval, snow pack, snow melt, average number of snow free days, slope, aspect, geographic orientation and average number of lightning strikes per year. Despite these limitations, this model provides a good indication of the risks of wildfire. The output of the model resulted in all the populations being at a "High" wildfire risk with the exception of 17 populations in the Rio Grande Headwaters GMU being at a "Moderate" risk.

Metric: Occupied Stream Length

We evaluated the potential impacts of wildfire risk based on the occupied stream length of the Rio Grande cutthroat trout populations. The longer an occupied stream length, the more likely that the population will survive a wildfire and debris flow event (Roberts *et al*. 2013, p. 1388) because it is more likely to have some stream reaches not affected by the fire, and it is more likely to have sufficient habitat diversity in the stream to provide refugia for individuals to survive (Isaak *et al.* 2012b, p. 551). We used four stream lengths to evaluate the effects of wildfire (Tables C7a and C7b). Streams longer than about 9.65 km (6 miles) are generally assumed to be long enough to encompass the habitat complexity necessary for the population to survive stochastic events (Hilderbrand and Kershner 2000, p. 515; Cowley 2007, p. 9). Streams between 9.65 km and 7.1 km are considered robust to stochastic risks (Roberts *et al*. 2013, p. 6), but may not have as much resiliency as longer streams. Streams shorter than 2.8 km (1.7 miles) are unlikely to have enough habitat variability for a population to be able to survive stochastic events (Harig and Fausch 2002, pp. 538–-539).

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Stream reaches smaller than 2.8 km may support populations of Rio Grande cutthroat trout, but local habitat quality is the greatest driver of population occurrence in short segments (Peterson *et al.* 2013, p. 10).

Metric: Stream Network

We also evaluated the potential impacts of wildfire risk based on the network (number of tributaries) of the streams occupied by Rio Grande cutthroat trout populations. Similar to overall stream length, the more networking an occupied stream has (in other words the more tributary branches in the stream), the more likely that the population will survive a wildfire and debris flow event. This is because these stochastic events are patchy in space or limited in extent, and networked streams are more likely to have some stream reaches not affected by the fire and are more likely to have sufficient habitat diversity in the stream to provide refugia for individuals to survive (Isaak *et al.* 2012b, p. 551; Roberts *et al*. 2013, p. 1388)). According to the RGCT Database, the vast majority of Rio Grande cutthroat populations has no stream tributaries and is categorized as isolated populations (109 of 122). Five streams are moderately networked (having 2 to 5 tributaries), and 11 streams are weakly networked (having 1 tributary) (Alves *et al.* 2008, p. 29). To account for the decreased risks from wildfire by streams with some connectivity, we reduced the risk functions for those streams. We used our best judgment of the benefits of networked streams to reduce the wildfire risk function of weakly connected streams by 25% and reduced moderately connected streams by 50%. This adjustment allows us to account for the benefits of connected streams that have a higher likelihood of surviving a wildfire event.

Risk Functions (+ Climate Change)

Tables C7a and C7b list the risk functions we used for this risk factor. We predicted that the risks for populations with moderate wildlife risk and long occupied stream lengths (>9.65 km) were at very low risk of extirpation due to wildfire by 2023 (0.5% chance of extirpation, scaling up to a 3.6% chance of loss by 2080). We chose 9.65 km (6 miles) as the threshold for the best stream length condition for several reasons. Hilderbrand and Kershner (2000, p. 515) estimated 8.3 km (5.1 mi) were required to maintain a population of 2,500 cutthroat trout when fish abundance was high (0.3 fish/m (0.09 fish/ft)). Adding a 10 percent loss rate to account for emigration and mortality increased the length up to 9.3 km (5.8 mi) in order to maintain 2,500 fish. Young *et al.* (2005, p. 2405) found that to maintain a population of 2,500 cutthroat trout, 8.8 km (5.5 mi) of stream were needed. Other studies have recommended stream lengths of 11 km (6.8 mi) and above (Cowley 2007, p. 10) based on stream widths. We chose stream lengths greater than 9.65 km (6 mi) to be the condition for best occupied stream lengths because this is a reasonable midpoint of those stream lengths recommended to support robust trout populations.

For the remaining conditions, we increased the risks by 25% for the next two states (moderate wildfire risk and 9.65 to 7.1 km stream length, and moderate wildfire risk and 7.1 to 2.8 km stream length) by 2023. For populations with a moderate fire risk and short occupied stream length (< 2.8 km) we predicted the risk to be low by 2023 (1% chance of extirpation, scaling up to 7.2% chance by 2080 with moderate climate change effects). For populations with high wildfire risk and long stream lengths (> 9.65 km), we predicted the fire risk to be low by 2023 (1% chance of extirpation, scaling up to 7.2 % chance with moderate climate change effects). We increased the risks by 100% for the next 2 states (high wildfire risk and 9.65 to 7.1 km stream length, and high wildfire risk and 7.1 to 2.8 km stream length) by 2023. For populations with a high fire risk and short occupied stream length (< 2.8 km) we predicted the risk to be low by 2023 (2% chance of extirpation, scaling up to 14.3% chance by 2080 with moderate climate change effects).

We foresee that ongoing and future climate change could increase the risk of wildfire impacts into the future because an expected drier, hotter climate would result in more frequent and larger intensity wildfires (Isaak *et al.* 2012b, p. 548). Therefore, we increased the rate of risk over time by 5% in 2040 and 10% in 2080 (for an estimate of moderate climate change effects, Table C7a) and by 20% in 2040 and 40% in 2080 (for an estimate of severe climate change effects, Table C7b).

Dealing with Uncertainty

How will climate change affect the risk of wildfire on Rio Grande cutthroat trout? To address this uncertainty we ran the model with moderate and severe levels of effects of climate change on the risk of wildfire.

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Table C7a. States of population metrics and risk functions for wildlife risks to Rio Grande cutthroat trout populations under "moderate" climate change effects. Moderate and high risks are from TNC's wildfire risk assessment and the stream lengths are the lengths occupied by Rio Grande cutthroat trout. Risks indicate the probability of an entire population being extirpated by the effects of wildfire. Risks were reduced by 25% for weakly networked streams and by 50% for moderately networked streams.

		Predicted Risk of Extirpation		
	State of Population Metric	**2023**	**2040**	**2080**
4a. Wildfire Risk (moderate climate change)	4a.1 Moderate Risk, stream length >9.65 km	0.005	0.013	0.036
	4a.2 Moderate Risk, stream length 9.65-7.1 km	0.006	0.016	0.045
	4a.3 Moderate Risk, stream length 7.1-2.8 km	0.008	0.021	0.056
	4a.4 Moderate Risk, stream length <2.8	0.010	0.026	0.072
	4a.5 High Risk, stream length >9.65 km	0.010	0.026	0.072
	4a.6 High Risk, stream length 9.65-7.1 km	0.013	0.033	0.089
	4a.7 High Risk, stream length 7.1-2.8 km	0.016	0.041	0.112
	4a.8 High Risk, stream length <2.8 km	0.020	0.053	0.143

Table C7b. States of population metrics and risk functions for wildfire risks to Rio Grande cutthroat trout populations under "severe" climate change effects. Moderate and high risks are from TNC's wildfire risk assessment and the stream lengths are the lengths occupied by Rio Grande cutthroat trout. Risks indicate the probability of an entire population being extirpated by the effects of wildfire. Risks were reduced by 25% for weakly networked streams and by 50% for moderately networked streams.

		Predicted Risk of Extirpation		
	State of Population Metric	**2023**	**2040**	**2080**
4b. Wildfire Risk (severe climate change)	4b.1 Moderate Risk, stream length >9.65 km	0.005	0.015	0.046
	4b.2 Moderate Risk, stream length 9.65-7.1 km	0.006	0.019	0.057
	4b.3 Moderate Risk, stream length 7.1-2.8 km	0.008	0.023	0.071
	4b.4 Moderate Risk, stream length <2.8	0.010	0.030	0.091
	4b.5 High Risk, stream length >9.65 km	0.010	0.030	0.091
	4b.6 High Risk, stream length 9.65-7.1 km	0.013	0.038	0.114
	4b.7 High Risk, stream length 7.1-2.8 km	0.016	0.047	0.142
	4b.8 High Risk, stream length <2.8 km	0.020	0.060	0.182

5. Stream Drying

This risk factor associated with stream drying describes the chance that a population will be extirpated over a given time frame as a result of the impacts associated with drought in the contributing watershed and the resulting loss in stream flow. Stream drying is an inherent risk to the persistence of Rio Grande cutthroat trout populations that has increased in recent times due to the relatively small and isolated nature of the currently remaining populations of Rio Grande cutthroat trout. See Chapter 4 and Appendix B of the Draft SSA Report for discussion of the causes and effects related to stream drying. We considered three population metrics for this risk factor: occupied stream length; stream discharge; and stream networking.

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Metric: Occupied Stream Length

We evaluated the potential impacts of stream drying based on the occupied stream length of the Rio Grande cutthroat trout populations. The longer an occupied stream length the more likely that the population will survive a stream drying event (Roberts *et al*. 2013, p. 1388) because it is more likely to have some stream reaches not affected by the loss of flow, and it is more likely to have sufficient habitat diversity in the stream to provide refugia for individuals to survive if some reaches become uninhabitable for some time period (Isaak *et al.* 2012b, p. 551). We used four stream lengths to evaluate the effects of stream drying (Tables C8a and C8b). Streams longer than about 9.65 km (6 miles) are generally assumed to be long enough to encompass the habitat complexity necessary for the population to survive stochastic events (Hilderbrand and Kershner 2000, p. 515; Cowley 2007, p. 9). Streams between 9.65 km and 7.1 km are considered robust to stochastic risks (Roberts *et al*. 2013, p. 6) but may not have as much resiliency as streams longer than 9.65 km (Hilderbrand and Kershner 2000, p. 515; Cowley 2007, p. 10). Streams shorter than 2.8 km are unlikely to have enough habitat variability for a population to be able to survive stochastic events (Harig and Fausch 2002, pp. 538–539).

Metric: Stream Discharge

We also evaluated the potential impacts of stream drying based on the predicted size of the stream as measured by stream discharge. The larger the discharge of streams during the summer and early fall critical time period, the less likely that stream will undergo substantial stream drying during drought events because it is more likely to maintain streamflow and habitat even when hydrological conditions decline. We used four stream discharge levels to evaluate the risks of stream drying (Tables C8a and C8b). The states for the stream discharge metric was based on the minimum 7-day average discharge between June 15 and September 30 (Zeigler *et al.* 2013b, p. 13). Streams predicted to have high discharge were those with a greater 0.1779 cubic meters per second (cms). Streams estimated to have moderate discharge were those with discharge ranging from 0.0291 to 0.1779 cms. Low discharge streams had discharges between 0.0017 to 0.0291 cms, and very low discharge streams had discharages less than 0.0017 cms.

Using data provided by Zeigler *et al.* (2013b, pp. 6–9), we were able to obtain discharge data for streams at 73 of 122 populations. For streams without data, we estimated the discharge based on the stream width as reported in RGCT database. Streams were categorized as less than 5 feet wide, 5 to 10 feet wide, 10 to 15 feet wide, and 15 to 20 feet wide (no streams were wider than 20 feet). We used an unpublished regression model to estimate discharge based on stream width which was derived from 267 field sites within the range of Rio Grande cutthroat trout (Zeigler, pers. comm. 2014). The resulting regression was: $\ln(\text{wetted width}) = 2.453 + (0.383 * \ln(\text{discharge}))$[11]. The results were that streams categorized as less than 5 feet wide in the RGCT Database were considered to have very low discharge, streams 5 to 10 feet wide were estimated to have low discharge, and streams between 10 and 20 feet wide were estimated to have moderate discharge.

Metric: Stream Network

We also evaluated the potential impacts of stream drying on the population based on the amount of networking (number of tributaries) of the streams occupied by Rio Grande cutthroat trout populations. Similar to overall stream length, the more networking an occupied stream has (in other words, the more tributary branches in the stream), the more likely that the population will survive a stream drying event (Dunham *et al.* 1997, p. 1130). This is because it is more likely to have some stream reaches not affected by the event, and it is more likely to have sufficient habitat diversity in the stream to provide refugia for individuals to survive (Isaak *et al.* 2012b, p. 551). According to the RGCT Database, the vast majority of Rio Grande cutthroat populations has no stream networking and is categorized as isolated populations (109 of 122). Four streams are moderately networked (having 2 to 5 tributaries), and nine streams are weakly networked (having 1 tributary) (Alves *et al.* 2008, p. 29). To account for the decreased risks from stream drying by streams with some connectivity, we reduced the risk functions for those streams. We used our best judgment of the benefits of

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[11] "Wetted width" is in meters and "discharge" is in cubic meters per second. The R^2 value of this regression was 0.44.

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networked streams to reduce the stream drying risk function of weakly connected streams by 25% and reduced moderately connected streams by 50%. This adjustment allows us to account for the benefits of connected streams that have a higher likelihood of surviving a stream drying event.

Risk Functions (+ Climate Change)

Tables C8a and C8b list the risk functions we used for this risk factor. Under moderate climate change conditions (Table C8a), we predicted that the risks for populations with large or moderate stream discharge, regardless of stream length, were at no risk of extirpation by 2023. Those populations in streams with moderate discharge we predicted to be at extremely low risk of extirpation by 2040 (0.1% chance of extirpation, scaling up to 0.29% chance of loss by 2080), regardless of stream length. Populations in low discharge streams and with long occupied stream lengths (>9.65 km) were predicted to be at extremely low risk of extirpation due to stream drying by 2023 (0.1% chance of extirpation, scaling up to a 0.7% chance of loss by 2080). For populations in low discharge streams and with 9.65 to 7.1 km occupied stream lengths we increased the risk to a 0.5% chance of extirpation due to stream drying by 2023 (scaling up to a 3.6% chance by 2080). For populations in low discharge streams and with 7.1 to 2.8 km occupied stream length we increased the risk to a 1% chance of extirpation due to stream drying by 2023 (scaling up to a 7.2% chance by 2080). And for the shortest streams (<2.8 km) with low discharge we predicted the risk as moderate for extirpation due to stream drying (2% chance of loss by 2023, scaling up to a 14.3% chance by 2080).

Dealing with Uncertainty

To address uncertainty from climate change we ran the model with moderate and severe levels of effects of climate change on the risk of stream drying.

We foresee that ongoing and future climate change could increase the risk of stream drying into the future because an expected drier, hotter climate would result in more frequent and larger intensity droughts with decreased precipitation and increased evaporation and evapotranspiration (Archer and Predick 2008, p. 29; Isaak *et al.* 2012b, p. 549). Therefore, we increased the rate of risk over time by 5% in 2040 and 10% in 2080 (for an estimate of moderate climate change effects, Table C8a) and by 20% in 2040 and 40% in 2080 (for an estimate of severe climate change effects, Table C8b). In the same way we used two climate change scenarios to calculate wildfire risk and hybridization risk, we used these two scenarios to capture some of the uncertainty of stream drying risk due to future climate change (see earlier discussion in Climate Change section for more information).

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Table C8a. States of population metrics and risk functions for stream drying risks to Rio Grande cutthroat trout populations under "moderate" climate change effects. Risks indicate the probability of an entire population being extirpated by stream drying. Risks were reduced by 25% for weakly networked streams and by 50% for moderately networked streams.

		Predicted Risk of Extirpation		
	State of Population Metric	**2023**	**2040**	**2080**
5a. Stream Drying Risks (moderate climate change)	5a.1 Any stream length, high discharge	0	0	0
	5a.2 Any stream length, moderate discharge	0	0.001	0.003
	5a.3 Stream length >9.65 km, low discharge	0.001	0.003	0.007
	5a.4 Stream length 9.65-7.1 km, low discharge	0.005	0.013	0.036
	5a.5 Stream length 7.1-2.8 km, low discharge	0.010	0.026	0.072
	5a.6 Stream length <2.8 km, low discharge	0.020	0.053	0.143
	5a.7 Stream length 9.65-7.1 km, very low discharge	0.002	0.005	0.014
	5a.8 Stream length 7.1-2.8 km, very low discharge	0.010	0.026	0.072
	5a.9 Stream length <2.8 km, very low discharge	0.020	0.053	0.143
	5a.10 Stream length <2.8 km, very low discharge	0.050	0.131	0.358

Table C8b. States of population metrics and risk functions for stream drying risks to Rio Grande cutthroat trout populations under "severe" climate change effects. Risks indicate the probability of an entire population being extirpated by stream drying. Risks were reduced by 25% for weakly networked streams and by 50% for moderately networked streams.

		Predicted Risk of Extirpation		
	State of Population Metric	**2023**	**2040**	**2080**
5b. Stream Drying Risks (severe climate change)	5b.1 Any stream length, high discharge	0	0	0
	5b.2 Any stream length, moderate discharge	0	0.001	0.004
	5b.3 Stream length >9.65 km, low discharge	0.001	0.003	0.009
	5b.4 Stream length 9.65-7.1 km, low discharge	0.005	0.015	0.046
	5b.5 Stream length 7.1-2.8 km, low discharge	0.010	0.030	0.091
	5b.6 Stream length <2.8 km, low discharge	0.020	0.060	0.182
	5b.7 Stream length 9.65-7.1 km, very low discharge	0.002	0.006	0.018
	5b.8 Stream length 7.1-2.8 km, very low discharge	0.010	0.030	0.091
	5b.9 Stream length <2.8 km, very low discharge	0.020	0.060	0.182
	5b.10 Stream length <2.8 km, very low discharge	0.050	0.150	0.455

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6. Disease Risk

This risk factor associated with disease describes the chance that a population will be extirpated over a given time frame as a result of the impacts associated with a future disease introduction, primarily whirling disease, but others as well. Although the risks of infection from whirling disease are relatively low for Rio Grande cutthroat trout populations, if a population is infected the risk of extirpation is high. See Chapter 4 and Appendix B of the Draft SSA Report for discussion of the causes and effects related to disease. We considered one population metric for this risk factor: the risk of infection, which is based on the distance of the Rio Grande cutthroat trout population to the nearest infection source.

Metric: Proximity to Infection Source

We evaluated the risk from disease as related to the proximity of disease causing pathogens to the Rio Grande cutthroat trout population. These metrics follow the rationale by Alves *et al.* (2008, p. 38). Populations where disease and pathogens are not known to exist in the watershed or a barrier provides complete protection to upstream fish movement are considered at limited risk of infection. Populations that have minimal risk are those with disease or pathogens in the watershed, but are more than 10 km away, or the barrier protecting the population may be at risk of failure. Populations at moderate risk are those where disease or pathogens have been identified within 10 km of the population.

Risk Functions

Table C9 lists the risk functions we used for this risk factor. We assumed that those populations with limited risk had no chance of infection by 2080. For populations identified at minimal risk of infection, we predicted the risk of extirpation as extremely low by 2023 (0.1% chance of extirpation, scaling up to a 0.65% chance by 2080). For populations identified at moderate risk of infection, we predicted the risk of extirpation as very low by 2023 (0.5% chance of extirpation, scaling up to a 3.25% chance in 2080). Populations already infected by disease are highly likely to be extirpated; there are no conservation populations considered currently infected.

The risk functions for disease increase proportionally as the time interval lengthens. We do not foresee any effects of climate change on the rate of increase associated with this risk factor.

Table C9. States of population metrics and risk functions for disease risks to Rio Grande cutthroat trout populations. Risks indicate the probability of an entire population being extirpated by disease.

		Predicted Risk of Extirpation		
	State of Population Metric	**2023**	**2040**	**2080**
6. Disease Risk	6.1 Limited risk	0	0	0
	6.2 Minimal risk	0.001	0.003	0.007
	6.3 Moderate risk	0.005	0.013	0.033
	6.4 Population is infected	0.8	0.9	0.95

7. Water Temperature Risk

This risk factor associated with increasing water temperatures describes the chance that a population will be extirpated over a given time frame as a result of the impacts associated with a future water temperature increase from a warming climate. These risks have been evaluated through field work monitoring seasonal stream temperatures (Zeigler *et al.* 2013b) and laboratory analysis to determine effects of increased water temperatures on individuals of Rio Grande cutthroat trout (Zeigler *et al.* 2013a). See Chapter 4 and Appendix B of the Draft SSA Report for discussion of the causes and effects related to disease. We considered one

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population metric for this risk factor: the risk of effects of increased water temperature, which is a combined risk analysis of expected increased water temperature.

Metric: Temperature Risk Analysis

We evaluated the risk from future water temperature rise using one metric that assesses the risk of water temperature increases on Rio Grande cutthroat trout populations. These metrics follow the rationale by Rogers (2013, pp. 4, 6, and 11). We used stream water temperature data from two sources, all of which are field monitoring data: one was published by Zeigler *et al.* (2013b) which had data for 48 streams; the second was unpublished data from an additional 23 streams (Zeigler *et al.* 2013c, unpublished data), as reported in Rogers (2013, pp. 18–21). The Zeigler *et al.* (2013b) data was reported as 2-hour maximum water temperatures (2-hr Max), which we converted to the mean weekly maximum temperature (MWMT) using the formula: MWMT=(0.9377 * 2-hr Max) + 0.0447, as suggested by Zeigler (2014, pers. comm.). The Zeigler *et al.* (2013b) data was also reported as mean weekly average temperatures (MWAT, maximum average temperature over a continuous 30 days of daily average temperatures), which we converted to the mean 30-day average temperature (M30AT) using the formula: M30AT=(0.9342 * MWAT) + 0.2622, as suggested by Zeigler (2014, pers. comm.). In all we had water temperature information for 71 of the 122 total populations.

The results of Zeigler *et al.* (2013a p. 1400) showed Rio Grande cutthroat trout juveniles in water greater than 25 degrees Celsius (°C) (under fluctuating temperature conditions similar to cooler nights and warmer days) could experience mortality due to water temperatures exceeding thermal tolerances. Zeigler *et al.* (2013a p. 1400) also showed that sublethal effects (such as decreased growth, malformations, and fungal growth) occurred at water temperatures above 18 °C. Therefore, we used these temperatures to evaluate the risks of streams having either "acute" (possible lethal effects) or "chronic" (possible sublethal effects) risks due to elevated water temperatures. In addition, to account for the potential effects of climate change to increase summer water temperatures, we followed the method of Rogers (2013, p. 9) and increased the reported temperatures (both MWMT and M30AT) for each stream by 2°C[12]. The results were that six populations where data were available had potential for acute effects (one of which also had potential chronic effects), and one population had potential for chronic effects. Populations with neither an acute nor a chronic risk of extirpation due to water temperature increases were considered at low risk. Because the vast majority of populations where data were available were found to be at low risk due to water temperature increases (7 out of 71 streams, 10%), we assumed that the populations not monitored for water temperatures were also at low risk[13] of extirpation from water temperature increases. Rogers (2013, pp. 9) made a similar assumption. This assumption would overestimate the probabilities of persistence in our analysis if other populations are actually going to be affected by increasing water temperatures.

> **Key Assumption**
> We assumed that the populations in streams not monitored for water temperature were at low risk of extirpation due to climate change.

Risk Functions (+ Climate Change)

Tables C10a and 10b list the risk functions we used for this risk factor. Under <u>moderate</u> climate change conditions (Table c10a), we predicted that the risks for populations identified as low risk for water temperature effects (including those not monitored) are at no risk of extirpation due to this risk factor by 2040 and at extremely low risk of extirpation by 2080 (0.1% chance of extirpation). For populations with predicted chronic effects on growth due to increasing water temperatures, we also predicted no risk of extirpation by 2040 and slightly higher risk of extirpation by 2080 (0.2% chance of extirpation). For populations with predicted acute

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[12] Although Rogers (2013, p. 9) did not include a reference for this application of a 2°C increase as a way to consider future effects of climate change, it is consistent with Robert *et al*. (2013, p. 1384) where they reference Ray *et al*. (2008, p. 29) that average summer air temperatures in the Southern Rocky Mountains are predicted to increase by ~2.7°C. So it is reasonable, as a rough estimate, to add 2°C to approximate climate change effects.

[13] Efforts are currently underway as part of the developing BN model for Rio Grande cutthroat trout to use actual down-scaled climate change models to predict water temperature changes at Rio Grande cutthroat trout populations, to include additional field data for stream temperatures, and to model the expected stream water temperatures for populations not monitored. Unfortunately this analysis has not yet been completed and was not available for use in our assessment.

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effects due to increasing water temperatures, we predicted no risk of extirpation by 2023 and a low risk of extirpation by 2040 (1% chance, scaling up to 2.86% chance of extirpation by 2080).

Dealing with Uncertainty

To address uncertainty from climate change we ran the model with a moderate and a severe level of effects of climate change on the risk of water temperature increases.

We foresee that ongoing and future climate change could increase the risk of extirpation due to water temperature increase into the future because an expected drier, hotter climate would result in increased water temperatures. Therefore, as we did with wildfire and stream drying risks, we increased risk over time by 5% in 2040 and 10% in 2080 (for an estimate of moderate climate change effects, Table C10a) and by 20% in 2040 and 40% in 2080 (for an estimate of severe climate change effects, Table C10b). In the same way we used these two scenarios to calculate nonnative hybridizing trout, wildfire, and stream drying risks, we used these two scenarios throughout the model to capture some of the uncertainty due to future climate change (see earlier discussion of Climate Change for more information).

Table C10a. States of population metrics and risk functions for water temperature rise risks to Rio Grande cutthroat trout populations under "moderate" climate change effects. Risks indicate the probability of an entire population being extirpated by water temperature.

		Predicted Risk of Extirpation		
	State of Population Metric	**2023**	**2040**	**2080**
10a. Water Temperature Risk (moderate climate change)	10a.1 Low Risk or Not Monitored	0	0	0.001
	10a.2 Chronic Effects on Growth Predicted	0	0.02	0.057
	10a.3 Acute Effects (lethal) Predicted	0	0.15	0.429

Table C10b. States of population metrics and risk functions for water temperature rise risks to Rio Grande cutthroat trout populations under "severe" climate change effects. Risks indicate the probability of an entire population being extirpated by water temperature.

		Predicted Risk of Extirpation		
	State of Population Metric	**2023**	**2040**	**2080**
10b. Water Temperature Risk (severe climate change)	10b.1 Low Risk or Not Monitored	0	0	0.001
	10b.2 Chronic Effects on Growth Predicted	0	0.02	0.073
	10b.3 Acute Effects (lethal) Predicted	0	0.15	0.546

Forecasting Persistence

Once we determined the risk functions for all of the risk factors being considered, we next calculated the probability of persistence for each current conservation population for each of our three time frames being considered. In this calculation we included two options related to competing nonnatives (with and without management suppression actions). We also included two different climate change scenarios (moderate and severe) which were included as alternative risk functions for four of the seven risk factors. This resulted in 10 different probabilities of persistence for each population (Figure C3).

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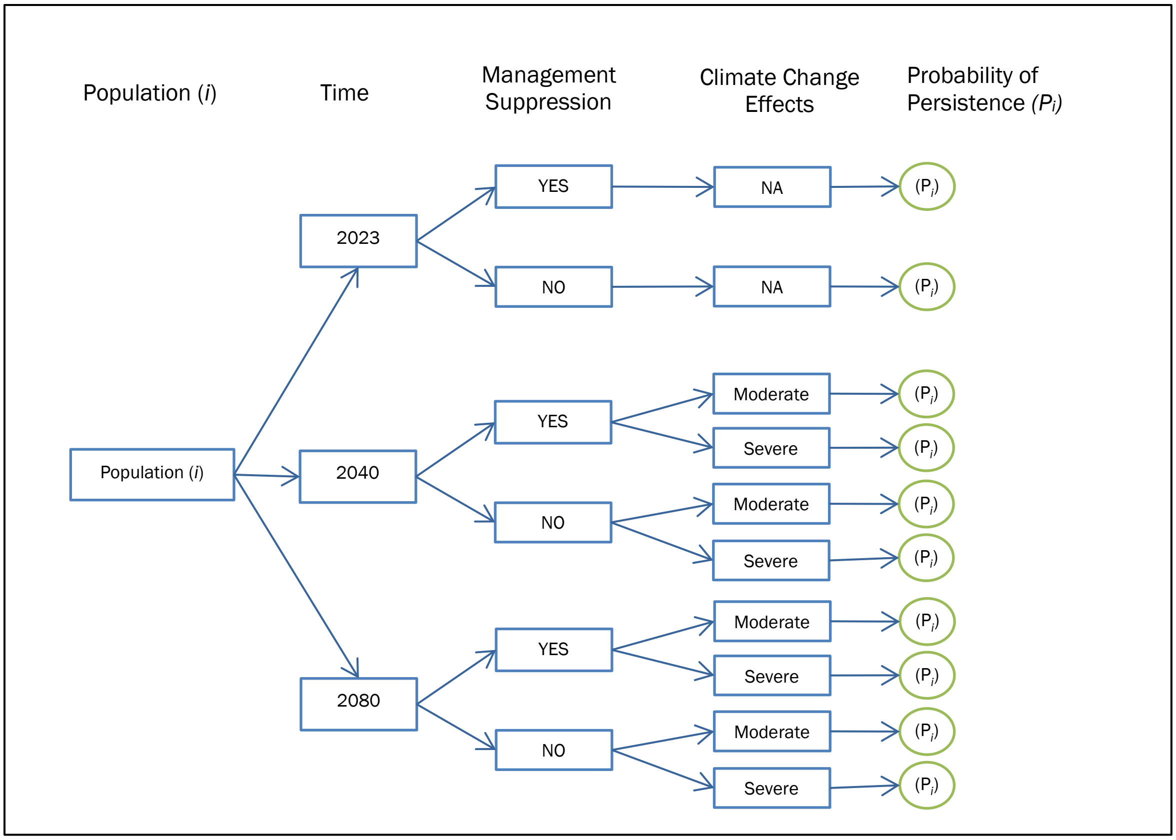

Figure C3. List of variable conditions for calculating probability of persistence for Rio Grande cutthroat trout populations.

For each of these 10 conditions we calculated an expected probability of persistence for each population by summing the individual risk functions for each risk factor and subtracting from 1. By summing the independent risk functions we can calculate a total risk of extirpation for each population that represents the cumulative chance of extirpation for each population for each scenario[14].

$$P_i = 1 - \sum R_{i,j}$$

Where P_i is the probability of persistence for population *i* and $R_{i,j}$ is risk factor *j* for population *i*. The risk factors are the probability of extirpation, and they are summed and subtracted from 1 to represent the probability that the population will persist (i.e., will not be extirpated). This approach assumes that in the absence of these ecological or environmental risk factors the probability of extinction is 0 (the probability of persistence is 1.0).

[14] Our model is not sufficiently sophisticated to take into account the potential for synergistic effects between the risk factors, which could occur in nature as multiple factors affect a species' viability. This is a recognized limitation in this approach. However, the cumulative nature of the approach and the linearly increasing risks over time provide a robust and conservative approach to assessing the viability of these populations.

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METHODS: POPULATION SURVIVAL

To estimate redundancy and representation of Rio Grande cutthroat trout we forecasted the number of populations that may persist in the future by simulating survival of the current populations and adding the number of populations projected to be restored over time. The sum of these two estimates resulted in a range of forecasted populations surviving under different conditions in each GMU for three time periods analyzed.

Population Survival Simulation

We constructed a survivability simulation model to estimate the number of RGCT populations that may be persisting in the future. To forecast the surviving populations, we used the results of the population persistence model and summed the probabilities of each population within each GMU and rangewide under each of the 10 conditions (Table C2). We then used a randomization process in MS Excel to simulate whether a specific population goes extinct under each of the 10 conditions. The simulation generates a random number (simulating a possible extirpation event) for a uniform distribution between 0 and 1.0. Then, if the randomly generated number is less than the persistence probability, the population remains extant (gets a value of 1) for the specified condition, if the random number is greater than the persistence probability, then the population goes extinct (gets a value of 0) for the specified condition. This model was developed as an *if* statement to mimic a binomial probability draw simulating population persistence.

$$T_{i,j} = \begin{cases} 1, if\ runif(0,1) < P_i \\ 0, if\ runif(0,1) > P_i \end{cases}$$

Where $T_{i,j}$ is a variable indicating whether trout population *i* in replicate *j* remains extant (1) or goes extinct (0). The persistence probability (P_i) derived from the risk functions serves as a threshold and allows us to convert the uniform random function into a binomial function of persistence estimate for each population. We ran the simulation 100 times for each population under each condition. We then calculated a mean number of populations surviving for each GMU and rangewide under each condition along with the 95% confidence interval around those estimates.

$$\overline{N_g} = \frac{\left(\sum T_{i,j,g}\right)}{100}$$

Where $\overline{N_a}$ is the average number of extant populations in GMU *g*, and *Ti,j,g* is the status of each population *i* (1= extant, 0 = extinct) in replicate *j*, in GMU *g*. The 95% C.I. was calculated as follows

$$95\%\ C.I. = \overline{N_g} \pm 1.96 \times S.D.(\overline{N_g})$$

Where S.D. is the standard deviation of the mean.

Restoration Forecasting

An important aspect of assessing the future status of the Rio Grande cutthroat trout is considering the future management actions that are likely to occur. State, Federal, Tribal, and private organizations have a number of past and ongoing activities both to protect, maintain, and enhance maintain current populations, but also to restore Rio Grande cutthroat trout to streams where they have been formerly extirpated. These population restoration efforts are a key component to the long-term viability of Rio Grande cutthroat trout.

Dealing with Uncertainty

To address uncertainty related to forecasting future population restorations, we estimated a range of the number of populations that may be restored in the future.

We incorporated a range of estimates ("High," "Low", and "Mid") of the potential numbers of populations that could be restored in the future into the population survival forecast. We assumed that those restored populations would be highly resilient (long stream lengths, large effective population sizes, and protected from nonnative trout). We based our estimates on the past and near future expectations for the number of

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populations that may be restored (Table C11). We generally took a conservative approach to estimating future population restorations past 2023.

For the 2023 population restoration forecast (Table C11), we used the number of populations planned to be restored by GMU from the 2013 Conservation Strategy. In that Strategy a range of populations are projected to be restored, which we used as the “High” (HE_{23}) and “Low” (LE_{23}) estimate for 2023, and we used the mean of the two for the “Mid” estimate.

For the 2040 population restoration forecast, the high estimate (HE_{40}) assumes that the high estimate from the 2023 forecast (HE_{23}) is completed and that future restorations continue at the same rate as the 2023 low estimate (LE_{23}) for the next 15 years [$HE_{40}=HE_{23}+(LE_{23}*1.5)$]. For the 2040 low estimate (LE_{40}) we used the 2023 low estimate for 2023 (LE_{23}), plus one additional population for each GMU [$LE_{40}= LE_{23}+1$]. The 2040 "Mid" estimate is an average of the 2040 low and high estimates.

For the 2080 population restoration forecast (Table C11), the high estimate (HE_{80}) is the sum of the 2023 high estimate plus the populations that would be restored at the same 2023 low estimate rate for the next 15 years plus additional populations restored at half of that rate over the next 40 years [$HE_{80}=HE_{23}+(LE_{23}*1.5)+(LE_{23}*2)$]. The 2080 low estimate is equivalent to the 2040 low estimate, which assumes no additional populations are restored for the 40 years between 2040 and 2080. The 2080 "Mid" estimate is an average of the 2080 low and high estimates.

Table C11. Estimates of the number of future populations of Rio Grande cutthroat trout restored by Geographic Management Unit. Totals for each time period are the cumulative totals.

	2023			2040			2080		
Populations per GMU	**High**	**Low**	**Mid**	**High**	**Low**	**Mid**	**High**	**Low**	**Mid**
Canadian	3	1	2	5	2	3	7	2	4
Rio Grande Headwaters	8	6	7	17	7	12	29	7	18
Lower Rio Grande	5	3	4	10	4	7	16	4	10
Caballo[15]	1	0	0	1	0	1	1	0	1
Pecos	3	1	2	5	2	3	7	2	4
Rangewide	**20**	**11**	**15**	**37**	**15**	**26**	**59**	**15**	**37**

Population Survival Scenarios

We used the various conditions, range of population survival simulations, and the ranges of possible population restorations to develop possible scenarios through which to forecast the range of total surviving Rio Grande cutthroat trout populations for the three time periods analyzed. We created 9 scenarios using different combinations of these conditions and results for each of the three time periods. Table C12 outlines these 9 scenarios.

For example, scenario 2 (“Worst Case with Severe CC, Low Mgt”) includes the population simulation based on a severe estimate of climate change effects, low management (with no suppression of competing nonnative

[15] We include Caballo GMU here because efforts are underway to restore a population in this GMU. However, we did not incorporate this population in the rest of the model output. See Chapter 3 of the SSA Report for a discussion of the Caballo GMU.

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trout), an estimate of the population simulation based on a negative 95% confidence interval of the mean, and a low estimate of populations being restored (from Table C11). This scenario represents the overall "worst case" scenario we considered.

On the other extreme, scenario 7 ("Best Case with Moderate CC, High Mgt") represents the overall "best case" scenario we considered. This scenario is based on moderate climate change effects, high management (with suppression of competing nonnative trout), an estimate of the population survival simulation with a positive 95% confidence interval of the mean, and a high estimate of populations being restored (from Table C11).

The results of both the population survival simulation and the forecasted number of populations restored are essentially a generated estimate of the number and distribution of Rio Grande cutthroat trout populations we predict will be surviving in the future. As such, we cannot predict exactly where these surviving populations might be, other than within the particular GMUs. We also cannot predict the actual condition (or resiliency) of these populations in the future. We presume that the surviving populations in the future would be in a range of conditions similar to those populations across the range today (expected to range from best to poor conditions).

Occupied Stream Length Forecasting

Dealing with Uncertainty

We recognize the high uncertainty in forecasting the future amount of occupied stream miles, due to the lack of confidence in forecasting the length of future populations. Therefore, we use the results only as a general measurement of the trend in relation to historic and current amounts of occupied habitat.

We also estimated the potential number of stream kilometers that are forecasted to be occupied in the future using our future population simulation. We did this in order to compare the current and future status of the Rio Grande cutthroat trout to the historical status in terms of total occupied habitat. We do not have a historical measure of the number of Rio Grande cutthroat trout populations that existed before modern impacts began to occur[16]. However, we do have estimates of historically occupied amount of stream kilometers (Alves *et al.* 2008, p. 13), the amount occupied in 2007 (Alves *et al.* 2008, p. 13), and an estimate of the current occupied stream miles from the Rio Grande cutthroat trout database. We estimated the future number of potential stream kilometers occupied based on our forecasted number of populations surviving. This estimation is not very precise, however, because we had to make large assumptions in estimating the future amount of occupied stream kilometers by population. For the population survival model we used an average occupied stream length (9.5 km) of currently occupied streams multiplied by the number of simulated streams surviving. For the estimate of restored populations we used a median stream length by GMU from an unpublished list of potential streams that could be restored[17]. We multiplied these stream lengths (ranging from 15 to 22 km) by the estimated number of streams to be restored. We summed the totals for the population survival simulation and populations restored to arrive at an estimate of future occupied stream kilometers. We have low confidence in the precision of these estimates because large assumptions[18] were made to approximate the average occupied stream lengths in the future. Therefore, we only use these results as a general guide to compare the possible total occupied habitat in the future to what was present historically and currently.

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[16] A measure of historical population numbers would not a reasonable assessment, given that many populations would have been large and interconnected compared to current populations.

[17] These lists were from the states of New Mexico and Colorado and were preliminary planning documents. We used the median rather than the mean stream length because some of the streams were extremely long and the likelihood of restoring the enter stream lengths seems low.

[18] We assumed future stream lengths would be equal to the average currently occupied stream lengths.

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Table C12. Range of scenarios used in forecasting survival of Rio Grande cutthroat trout populations.

Scenarios		Climate Change	Nonnative Suppression	2023 Forecast		2040 Forecast		2080 Forecast	
				Population Simulation	Population Restoration	Population Simulation	Population Restoration	Population Simulation	Population Restoration
1	Best Case with Severe Climate Change and Low Management	Severe	No	Upper 95% C.I.	Mid	Upper 95% C.I.	Mid	Upper 95% C.I.	Mid
2	Worst Case with Severe Climate Change and Low Management			Lower 95% C.I.	Low	Lower 95% C.I.	Low	Lower 95% C.I.	Low
3	Intermediate Case with Severe Climate Change and Low Management			Mean	Low	Mean	Low	Mean	Low
4	Best Case with Moderate Climate Change and Low Management	Moderate	No	Upper 95% C.I.	Mid	Upper 95% C.I.	Mid	Upper 95% C.I.	Mid
5	Worst Case with Moderate Climate Change and Low Management			Lower 95% C.I.	Low	Lower 95% C.I.	Low	Lower 95% C.I.	Low
6	Intermediate Case with Moderate Climate Change and Low Management			Mean	Low	Mean	Low	Mean	Low
7	Best Case with Moderate Climate Change and High Management	Moderate	Yes	Upper 95% C.I.	High	Upper 95% C.I.	High	Upper 95% C.I.	High
8	Worst Case with Moderate Climate Change and High Management			Lower 95% C.I.	Mid	Lower 95% C.I.	Mid	Lower 95% C.I.	Mid
9	Intermediate Case with Moderate Climate Change and High Management			Mean	Mid	Mean	Mid	Mean	Mid

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Results: Population Persistence & Population Survival

The results of our status assessment model will be used in the Rio Grande cutthroat species status assessment as a way to measure resiliency, redundancy, and representation. These measures allow us to describe the viability of the subspecies. Our results are displayed in *Chapter 5. Viability* of the SSA Report.

For a description of the results of this analysis, please refer to Chapter 5 of the SSA Report.

Discussion

Model Strengths and Limitations

Like all models, this Status Assessment Model is only a quantitative reflection of our understanding of the way the ecological system works and influences the future status of the Rio Grande cutthroat trout. The analysis contains lots of uncertainty, assumptions, and professional judgment that affect the overall confidence of the results. However, this kind of forecasting is always fraught with uncertainties. We think it is worthwhile to be explicit about our understanding of the system in quantitative terms and be transparent about the uncertainties. The uncertainties in this kind of forecasting are inherently present whether we are explicit in quantitative terms or only describe the assessment in qualitative terms. The use of this kind of model is a somewhat novel approach to conducting status assessments and may or may not prove useful in future efforts to conduct species status assessments.

Risk Factors and Risk Functions

The most important part of this analysis is the quantification of the risk factors by assigning the risk functions to each state of the population metrics. These relationships form the foundation of the forecasts for the rest of the analysis. We used our best professional judgment to determine these risk functions, and, although subjective, they represent our understanding of the relationships between what the species currently has, in terms of the population metrics, and what the species is likely to have in the future as measured by the probability of extirpation. We explain these relationships elsewhere in the Draft SSA Report (Chapter 4 and Appendix B). As new information becomes available, or new, more robust models are conceived, these relationships are open to revision.

Cumulative and Synergistic Effects

Our approach to quantitatively assess the risks faced by Rio Grande cutthroat trout analyzes the cumulative effects of multiple risk factors in evaluating the viability of the species. By summing the risk functions for each population, we obtain a measure of the cumulative effects of all seven risk factors considered. However, the model is not sufficiently sophisticated to take into account the potential for synergistic effects and interactions between the risk factors, which undoubtedly could occur in nature as multiple factors affect a species' viability. This is a recognized limitation in this approach. However, the cumulative nature of the analysis and the linearly increasing risks over time provides a robust and conservative approach to assessing the viability of these populations.

Ongoing Management Actions

Ongoing and future management actions provide a critical contribution to the viability of the Rio Grande cutthroat trout. State, Federal, and Tribal agencies and private organizations are heavily engaged in the conservation of this subspecies, and these efforts provide substantial benefits to the subspecies. However, our modeling effort here could only directly incorporate limited aspects of this active management (suppression of competing nonnative trout and future restoration of populations). Because the species is so closely managed, some risks can be reduced by continued monitoring, maintenance of barriers, and other efforts to prevent nonnative trout invasions. For example, our risk functions assume that some passive management will continue into the future (for example, not stocking viable rainbow trout populations near Rio Grande cutthroat trout populations). These sorts of efforts are reflected in our risk functions for those factors

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that can be influenced by management. Nevertheless other management efforts, such as constructing new barriers, connecting existing populations, and responding to nonnative invasions, were beyond our ability to predict in the model, although we are confident they are likely to continue and will provide ongoing benefits to the Rio Grande cutthroat trout.

Key Assumptions and Uncertainty

Inherent in any effort to use a simple model like this one to reflect a complex ecological system are numerous assumptions about missing information and about how the system works. While we could not call out every assumption, throughout the model development we identified areas where key assumptions were made. In some cases, we had missing data and had to assume the population was in a particular state based on our best judgment considering the most likely state. In other instances, we made assumptions about the relationship between some important factors. Table C13 lists some of the key assumptions we made along with the possible effects of those assumptions on the model results.

Table C13. List of key assumptions. Effect of assumption on our model results is: "–" if the assumption is expected to result in a lower probability of persistence compared to an unknown reality; and "+" when the assumption is expected to result in a higher probability of persistence compared to an unknown reality; and "+/–" when the effect of the assumption on the model could be positive or negative.

Selected Key Assumptions	Number of Populations Affected	Effect of Assumption on Model Results
Effective population size, assumed N/Ne ratio of 0.375 for streams with population census data	108	+/–
Calculated population estimates using stream length formula	14	+
Populations with genetic status untested, assumed as presumed in database	24	+/–
Populations with unknown competing nonnatives present, assumed absent	5	+
Populations with no water temperature data, assumed no water temperature effects	57	+
Numbers of future populations restored, erred on low estimates	~11 - 59	–
Future occupied stream miles, assumed mean stream length for current and median stream length for future restored	All	+/–

Suppression of Competing Nonnative Trout

Suppression of competing of nonnative trout is an important management action that can help Rio Grande cutthroat trout populations continue to persistence even when the nonnative trout are present. Suppression activities (mechanically removing nonnatives) typically take place every few years and are currently occurring at 6 populations. One source of uncertainty was whether or not we should assume that this management action will continue for these populations into the future. To account for this uncertainty we ran the full analysis both with and without this suppression. The suppression activities make a large difference in our judgment about the risks of extirpation on each population where the suppression is occurring (Table C6). However, in relationship to all the population this distinction makes only a small difference in the overall outcome

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describing viability.   This is because the uncertainty only addresses 6 populations (out of a total of 122), and those populations may be facing other risks that lower their probability of persistence.

Climate Change

As described above under *METHODS: POPULATION PERSISTENCE, Climate Change Considerations* we conducted the analysis under two different conditions consider the possible effects of climate change. Overall, the difference between the moderate and severe conditions did not appear to be particularly large. This is likely because the risk factors we associated with climate change we judged to have relatively low risk functions on a single population basis, reflecting our understanding of the risk associated with the these factors. In addition our multiplying factor for risk functions with severe climate change effects of 20% and 40% still did not significantly increase the overall risks associated with these factors. We think that this is a fair representation of the effects of climate change, but future input on these factors, their risks, and the potential impacts of climate change could adjust these parameters.

Population Restoration

As described above under *METHODS: POPULATION SURVIVAL, Restoration Forecasting*, we used a range of estimates to accommodate for the uncertainty related to forecasting the number of populations that may be restored in the future. We think our methods predict a reasonable range of outcomes; however, these estimates have a substantial impact on the results of the total population survival estimates. In an attempt not to overestimate the potential for population restorations, we used some low estimates in our development of scenarios evaluated (Table C12). Had we used higher estimates in these scenarios, the results of the high estimates for population survival would have been considerably larger.

Forecasting

As discussed under Time Frames Analyzed above, there is an inverse relationship between the confidence we have in our results and the length of time for our forecasting. In many areas of our model there are large uncertainties related to our ability to forecast the future. Obviously any forecast, particularly those of ecological systems, is rife with uncertainties. We identified a number of these uncertainties and addressed them through calculating a range of possibilities so that our results reflect a range of outcomes. Obviously the confidence in our results decreases with increasing length of the forecast timeframe, which are reflected in our results where the ranges of outcomes increase between 2023, 2040, and 2080. We estimated our confidence level in the range of our results as high for the 2023 forecast (greater than 90% certain), moderate for the 2040 forecast (70 to 90% certain) and somewhat confident for the 2080 forecast (50 to 70% certain). We understand this is an inherent characteristic for forecasting any future events, particularly when related to complex ecological systems. We recognize this is a necessary shortcoming in our ability to forecast.

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Table C14. Summary data used in the Rio Grande cutthroat trout status assessment model: index to columns. More detailed explanations are provided in the report above. Most data is from the RGCT Database unless otherwise noted.

Column Header	Explanation
Conservation_Population	Names of the stream(s) or stream segment(s) included within the conservation population.
GMU	Geographic Management Unit.
CP_Pop_ID	Conservation Population Identifier. "L" designates the lower population for populations split into upper and lower stream segments because of a fish barrier.
EXTANT	Is the Rio Grande cutthroat trout population considered still extant?
Ne_Est	Effective population size estimate.
PopEst_Year	Year of the estimate of effective population size. "Est" indicates estimates based on the density category. "Calc" indicates estimates calculated using stream length. "na" indicates there was not a year associated with the population size estimate.
Genetic status	Genetic status of the population in terms of percent introgressed with rainbow trout. "NT-Sus_Un"= Not Tested – Suspected Unaltered. "NT-Sus_Hyb" = Not Tested – Suspected Hybridized.
RBT_Pres	Is rainbow trout present in the population?
Hyb_Risk	Estimated risk of hybridization based on proximity of rainbow trout.
Com_Nnat	Are competing nonnative fish present in the population?
Supp	Is suppression of competing nonnative fish occurring?
Barrier	Status of a fish barrier?
Str_Lth	Length of stream, in kilometers, occupied by Rio Grande cutthroat trout.
Networked	Level of stream network for the population.
Fire_Risk	Estimated fire risk from Miller and Bassett (2013).
Flow_Cat	Category of stream discharge.
Disease_Risk	Estimated risk of disease based on proximity to disease.
Temp_Risk	Estimated risk to water temperature increases (temperature data from Zeigler *et al.* 2013b and Zeigler *et al.* 2013c).

Table C14. Summary data used in the Rio Grande cutthroat trout status assessment model.

Conservation_Population	GMU	CP_Pop_ID	EXTANT	Ne_Est	Ne_Yr	Genetic status	RBT_Pres	Hyb_Risk	Com_Nnat	Supp	Barrier	Str_Lth	Networked	Fire_Risk	Flow_Cat	Disease_Risk	Temp_Risk
Ricardo, Elk, Gold, Leandro, Vermejo	Canadian	11080001cp001	YES	4,018	2006	Unaltered (< 1%)	NO	None	YES	YES	Complete	69.3	Weakly	High	Moderate	Limited	LowRisk
Little Vermejo Creek	Canadian	11080001cp002	YES	39	2003	Unaltered (< 1%)	NO	None	YES	NO	Complete	11.9	Isolated	High	Low	Limited	Acute
Leandro Creek	Canadian	11080001cp003	YES	126	2009	Unaltered (< 1%)	NO	None	YES	YES	Complete	3.1	Isolated	High	Low	Limited	LowRisk
McCrystal, North Ponil	Canadian	11080002cp001	YES	460	2009	Unaltered (< 1%)	NO	None	NO	NO	Complete	15.2	Isolated	High	Low	Limited	Acute
South Ponil Creek	Canadian	11080002cp002	YES	503	2009	Unaltered (< 1%)	NO	None	NO	NO	Complete	15.2	Isolated	High	Low	Limited	LowRisk
Middle Ponil Creek	Canadian	11080002cp003	**NO**	630	2004	>10% and <=20%	NO	<10km	NO	NO	Complete	9.6	Isolated	High	Low	Moderate	LowRisk
Clear Creek, Hdwtr Trib to Clear Creek	Canadian	11080002cp005	YES	337	2006	Unaltered (< 1%)	NO	None	NO	NO	Complete	7.5	Isolated	High	Low	Limited	LowRisk
East Fork Luna Creek	Canadian	11080004cp001	YES	394	calc	>1% and <=10%	NO	<10km	YES	NO	None	6.8	Isolated	High	Very Low	Limited	LowRisk
West Fork Luna Creek	Canadian	11080004cp002	YES	237	2004	Unaltered (< 1%)	NO	>10km	YES	NO	Partial	4.6	Isolated	High	Low	Limited	LowRisk
Rito Morphy, Hdwtr Trib to Rito Morphy	Canadian	11080004cp003	YES	158	est	Unaltered (< 1%)	NO	<10km	NO	NO	None	6.8	Weakly	High	Very Low	Limited	LowRisk
Santiago Creek	Canadian	11080004cp004	YES	154	est	>1% and <=10%	NO	>10km	NO	NO	None	6.6	Isolated	High	Very Low	Limited	LowRisk
West Alder Creek	RG Headwaters	13010001cp002	YES	89	2005	Unaltered (< 1%)	NO	None	YES	NO	Partial	7.2	Isolated	Moderate	Low	Limited	LowRisk
East Trib Middle Fk, West Trib, San Francisco Creek	RG Headwaters	13010002cp001	YES	1,432	2004	Unaltered (< 1%)	NO	None	YES	YES	Complete	25.3	Weakly	Moderate	Moderate	Moderate	LowRisk
Cat Creek	RG Headwaters	13010002cp002	YES	605	2007	Unaltered (< 1%)	NO	None	NO	NO	Complete	15.1	Isolated	Moderate	Very Low	Limited	LowRisk
Rhodes Gulch	RG Headwaters	13010002cp003	YES	236	2006	>1% and <=10%	NO	None	NO	NO	Complete	3.5	Isolated	High	Low	Limited	LowRisk
Torsido Creek	RG Headwaters	13010002cp004	YES	94	2005	NT-Sus_Un	NO	None	YES	NO	None	10.4	Isolated	High	Low	Limited	LowRisk
Jim Creek	RG Headwaters	13010002cp005	YES	480	2004	Unaltered (< 1%)	NO	None	YES	NO	None	10.2	Isolated	High	Low	Limited	LowRisk
Cuates Creek	RG Headwaters	13010002cp006	YES	141	2012	Unaltered (< 1%)	NO	None	NO	NO	Complete	6.1	Isolated	High	Low	Limited	LowRisk
Jaroso Creek	RG Headwaters	13010002cp007	YES	316	2012	Unaltered (< 1%)	NO	None	NO	NO	Complete	9.3	Isolated	High	Low	Limited	LowRisk
Jaroso Creek	RG Headwaters	13010002cp008	YES	739	2005	NT-Sus_Un	NO	None	YES	NO	Complete	6.2	Isolated	High	Moderate	Limited	LowRisk
Torcido Creek	RG Headwaters	13010002cp009	YES	1,054	2012	Unaltered (< 1%)	NO	None	NO	NO	Complete	13.2	Isolated	High	Low	Limited	LowRisk
Alamosito Creek	RG Headwaters	13010002cp010	YES	333	2005	Unaltered (< 1%)	NO	None	YES	YES	Complete	4.9	Isolated	High	Low	Limited	LowRisk
Vallejos Creek, North Vallejos	RG Headwaters	13010002cp011	YES	338	2005	Unaltered (< 1%)	NO	None	YES	NO	None	22.5	Isolated	High	High	Limited	LowRisk
Deep Canyon, South Fk Trinchera, Trinchera	RG Headwaters	13010002cp012	YES	92	2012	Unaltered (< 1%)	NO	None	YES	NO	None	18.9	Isolated	High	High	Limited	LowRisk
North Fork Trinchera Creek	RG Headwaters	13010002cp014	YES	83	2006	Unaltered (< 1%)	NO	None	YES	NO	Complete	11.5	Isolated	High	Moderate	Limited	LowRisk
South Fork West Indian Creek, West Indian	RG Headwaters	13010002cp015	YES	716	2008	Unaltered (< 1%)	NO	None	YES	YES	Partial	17.1	Isolated	High	Low	Limited	LowRisk
Lower Placer, Sangre De Cristo, Wagon	RG Headwaters	13010002cp016	YES	6,719	2011	Unaltered (< 1%)	NO	None	YES	NO	Partial	63	Weakly	Moderate	Moderate	Moderate	LowRisk
Upper Placer, Grayback	RG Headwaters	13010002cp021	YES	4,491	2013	>1% and <=10%	NO	None	NO	NO	Complete	45.7	Weakly	Moderate	Moderate	Moderate	LowRisk
Little Ute Creek	RG Headwaters	13010002cp017	YES	143	2004	Unaltered (< 1%)	NO	None	NO	NO	Complete	2.7	Isolated	Moderate	Moderate	Limited	LowRisk
Cuates Creek	RG Headwaters	13010002cp018	YES	274	calc	NT-Sus_Un	NO	<10km	Unknown	NO	Complete	5.5	Isolated	High	Very Low	Limited	LowRisk
Torcido Creek	RG Headwaters	13010002cp019	YES	660	2005	Unaltered (< 1%)	NO	None	NO	NO	Complete	3.3	Isolated	Moderate	Low	Limited	LowRisk
Alamosito Creek	RG Headwaters	13010002cp020	YES	15	2012	Unaltered (< 1%)	NO	None	YES	NO	None	0.8	Isolated	High	Low	Limited	LowRisk
Medano Creek	RG Headwaters	13010003cp001	YES	2,354	2011	Unaltered (< 1%)	NO	None	NO	NO	Complete	28.8	Weakly	High	Low	Limited	LowRisk
East Pass Creek	RG Headwaters	13010004cp002	YES	138	2005	Unaltered (< 1%)	NO	None	NO	NO	Complete	11.2	Isolated	Moderate	Very Low	Limited	LowRisk
Whale Creek	RG Headwaters	13010004cp001	YES	138	2003	Unaltered (< 1%)	NO	None	NO	NO	Complete	4.2	Isolated	Moderate	Low	Limited	LowRisk
Cross Creek	RG Headwaters	13010004cp003	YES	1,382	2005	Unaltered (< 1%)	NO	None	NO	NO	Complete	12.9	Isolated	Moderate	Low	Moderate	LowRisk
Jacks Creek	RG Headwaters	13010004cp003L	YES	1,818	2005	Unaltered (< 1%)	NO	None	YES	NO	Complete	18.5	Isolated	Moderate	Low	Moderate	LowRisk
East Middle Creek	RG Headwaters	13010004cp004	YES	193	2011	>1% and <=10%	NO	None	NO	NO	Complete	4.9	Isolated	Moderate	Low	Minimal	LowRisk
Big Springs Creek	RG Headwaters	13010004cp006	YES	240	2007	Unaltered (< 1%)	NO	None	NO	NO	Complete	4.1	Isolated	Moderate	Low	Limited	LowRisk
Middle Fork Carnero Creek	RG Headwaters	13010004cp007	YES	129	2005	Unaltered (< 1%)	NO	None	NO	NO	Complete	11.3	Isolated	Moderate	Low	Limited	LowRisk
North Fork Carnero Creek	RG Headwaters	13010004cp008	**NO**	-	2005	Unaltered (< 1%)	NO	None	NO	NO	Complete	13	Isolated	Moderate	Very Low	Limited	LowRisk
South Carnero Creek	RG Headwaters	13010004cp010	YES	1,406	2006	Unaltered (< 1%)	NO	<10km	YES	NO	None	22.7	Isolated	Moderate	Moderate	Limited	LowRisk
Miners Creek, Prong Creek	RG Headwaters	13010004cp011	YES	678	2006	>1% and <=10%	NO	None	YES	NO	None	13	Isolated	Moderate	Low	Limited	LowRisk
Cave Creek	RG Headwaters	13010004cp012	YES	154	2001	Unaltered (< 1%)	NO	None	YES	NO	None	10.2	Isolated	High	Low	Limited	LowRisk
Tio Grande	RG Headwaters	13010005cp001	YES	392	2004	Unaltered (< 1%)	NO	None	YES	NO	Complete	7.6	Isolated	High	Very Low	Limited	LowRisk
Tio Grande	RG Headwaters	13010005cp002	YES	436	2004	NT-Sus_Un	NO	None	YES	NO	Complete	4.5	Isolated	High	Low	Limited	Acute
Tanques Creek	RG Headwaters	13010005cp003	YES	188	2010	Unaltered (< 1%)	NO	None	YES	NO	Complete	2.9	Isolated	High	Low	Limited	LowRisk
Rio Nutritas	RG Headwaters	13010005cp004	YES	86	2001	Unaltered (< 1%)	NO	None	YES	NO	None	5.1	Isolated	High	Low	Limited	LowRisk

Table C14. Summary data used in the Rio Grande cutthroat trout status assessment model.

Conservation_Population	GMU	CP_Pop_ID	EXTANT	Ne_Est	Ne_Yr	Genetic status	RBT_Pres	Hyb_Risk	Com_Nnat	Supp	Barrier	Str_Lth	Networked	Fire_Risk	Flow_Cat	Disease_Risk	Temp_Risk
Osier Creek	RG Headwaters	13010005cp006	YES	361	2010	Unaltered (< 1%)	NO	None	NO	NO	Complete	5.9	Isolated	High	Low	Limited	LowRisk
Lake Fork Conejos River	RG Headwaters	13010005cp007	YES	71	2004	Unaltered (< 1%)	NO	None	NO	NO	Complete	1	Isolated	High	Low	Minimal	LowRisk
Lake Fork Conejos River	RG Headwaters	13010005cp008	YES	673	2012	Unaltered (< 1%)	NO	None	NO	NO	Complete	4	Isolated	High	Moderate	Moderate	LowRisk
Rio de los Pinos	RG Headwaters	13010005cp009	YES	42	2005	Unaltered (< 1%)	NO	None	NO	NO	Complete	0.9	Isolated	High	High	Limited	LowRisk
Cascade Creek	RG Headwaters	13010005cp010	YES	892	2000	Unaltered (< 1%)	NO	None	NO	NO	Complete	4.7	Isolated	High	Low	Limited	LowRisk
Costilla Creek, State Line Creek	Lower RG	13020101cp001	YES	1,122	2008	>1% and <=10%	NO	None	NO	NO	Complete	14.6	Weakly	High	Low	Limited	LowRisk
Costilla, Frey, Glacier, Patten Creeks	Lower RG	13020101cp002	YES	2,488	2010	Unaltered (< 1%)	NO	None	NO	NO	Complete	15.2	Moderately	High	Low	Limited	LowRisk
E. Unnamed Trib. #2 to Costilla Creek (Powderhouse)	Lower RG	13020101cp003	YES	378	2009	Unaltered (< 1%)	NO	None	NO	NO	Complete	6.2	Isolated	High	Low	Limited	LowRisk
E. Unnamed Trib. #2 to Costilla Creek (Powderhouse)	Lower RG	13020101cp004	YES	28	2004	NT-Sus_Hyb	NO	>10km	YES	NO	Complete	2.1	Isolated	High	Very Low	Limited	LowRisk
NW Unnamed Trib. to Costilla Creek (La Cueva)	Lower RG	13020101cp005	YES	109	2010	>1% and <=10%	NO	<10km	NO	NO	None	5.1	Isolated	High	Very Low	Limited	LowRisk
Comanche, Gold, Grassy, Holman, LaBelle Creeks	Lower RG	13020101cp006	YES	2,863	2012	Unaltered (< 1%)	NO	None	NO	NO	Complete	44.7	Moderately	High	Low	Limited	LowRisk
Chuck Wagon, Comanche, Fernandez Creeks	Lower RG	13020101cp007	YES	148	2010	>1% and <=10%	NO	None	NO	NO	Complete	8.6	Weakly	High	Low	Limited	LowRisk
Lower (Chuck Wagon, Comanche, Fernandez)	Lower RG	13020101cp007L	YES	237	2012	NT-Sus_Hyb	YES	Symp	NO	NO	None	5.5	Weakly	High	Low	Limited	Acute
Unnamed Trib. to Ute Creek	Lower RG	13020101cp008	YES	471	2005	Unaltered (< 1%)	NO	<10km	NO	NO	None	13.8	Isolated	High	Low	Limited	Chronic
Cabresto Creek	Lower RG	13020101cp009	YES	373	2013	Unaltered (< 1%)	NO	>10km	YES	NO	Partial	13.7	Isolated	High	Low	Minimal	LowRisk
Bitter Creek	Lower RG	13020101cp010	YES	204	2006	Unaltered (< 1%)	NO	>10km	NO	NO	Partial	2.9	Isolated	High	Very Low	Minimal	LowRisk
Columbine, Deer, PlacerFk, Willow Creeks	Lower RG	13020101cp011	YES	660	2010	Unaltered (< 1%)	NO	None	NO	NO	Complete	10.7	Moderately	High	Moderate	Limited	LowRisk
Lower (Columbine, Deer, Placer Fork, Willow)	Lower RG	13020101cp011L	YES	438	2010	Unaltered (< 1%)	NO	None	YES	NO	None	7.1	Moderately	High	Moderate	Limited	LowRisk
San Cristobal Creek	Lower RG	13020101cp012	YES	268	2006	Unaltered (< 1%)	NO	None	NO	NO	None	6.5	Isolated	High	Low	Limited	LowRisk
Yerba Creek	Lower RG	13020101cp013	YES	115	2005	Unaltered (< 1%)	NO	>10km	YES	YES	Partial	4.7	Isolated	High	Low	Limited	LowRisk
Italianos Creek	Lower RG	13020101cp015	YES	114	2005	NT-Sus_Un	NO	>10km	NO	NO	Complete	3.8	Isolated	High	Low	Limited	LowRisk
Gavilan Creek	Lower RG	13020101cp016	YES	133	2005	Unaltered (< 1%)	NO	>10km	YES	NO	None	3.4	Isolated	High	Low	Limited	LowRisk
South Fork Rio Hondo	Lower RG	13020101cp017	YES	142	2005	Unaltered (< 1%)	NO	>10km	YES	NO	None	6.3	Isolated	High	Moderate	Limited	LowRisk
Tienditas Creek	Lower RG	13020101cp018	YES	19	est	Unaltered (< 1%)	NO	>10km	YES	NO	None	3.2	Isolated	High	Low	Limited	LowRisk
Frijoles Creek	Lower RG	13020101cp019	YES	169	2008	Unaltered (< 1%)	NO	None	YES	NO	Partial	5	Isolated	High	Low	Limited	LowRisk
Palociento Creek	Lower RG	13020101cp020	YES	91	est	Unaltered (< 1%)	NO	None	YES	YES	Complete	3.9	Isolated	High	Low	Limited	LowRisk
Rio Grande del Rancho	Lower RG	13020101cp021	YES	182	calc	>1% and <=10%	NO	>10km	YES	NO	None	4.3	Isolated	High	Low	Limited	LowRisk
Rito la Presa	Lower RG	13020101cp022	YES	648	2008	Unaltered (< 1%)	NO	>10km	NO	NO	None	9.1	Isolated	High	Moderate	Minimal	LowRisk
Lower (Rito La Presa)	Lower RG	13020101cp022L	YES	135	est	Unaltered (< 1%)	NO	>10km	YES	NO	None	5.8	Isolated	High	Moderate	Minimal	LowRisk
Policarpio Creek	Lower RG	13020101cp023	YES	261	2005	Unaltered (< 1%)	NO	None	NO	NO	Complete	4.8	Isolated	High	Low	Limited	LowRisk
Unnamed Trib. to Rio Pueblo (Osha)	Lower RG	13020101cp024	YES	33	2005	>1% and <=10%	NO	None	NO	NO	Complete	8.8	Isolated	High	Low	Limited	LowRisk
Rito Angostura	Lower RG	13020101cp025	YES	573	2003	>1% and <=10%	NO	None	NO	NO	Complete	6.4	Isolated	High	Low	Limited	LowRisk
Alamitos Creek	Lower RG	13020101cp026	YES	438	2010	Unaltered (< 1%)	NO	>10km	NO	NO	Complete	4.1	Isolated	High	Low	Minimal	LowRisk
Lower (Alamitos)	Lower RG	13020101cp026L	**NO**	-			NO				None	7.3	Isolated	High	Moderate		LowRisk
Middle Fork Rio Santa Barbara	Lower RG	13020101cp027	YES	357	2003	Unaltered (< 1%)	NO	None	YES	NO	Complete	7	Isolated	High	High	Limited	LowRisk
East Fork Rio Santa Barbara	Lower RG	13020101cp028	YES	169	2009	Unaltered (< 1%)	NO	None	YES	NO	Partial	4.1	Isolated	High	Moderate	Limited	LowRisk
Rio Santa Barbara	Lower RG	13020101cp029	YES	463	2009	NT-Sus_Hyb	NO	<10km	YES	NO	None	14.5	Isolated	High	Moderate	Limited	LowRisk
Rio de las Trampas	Lower RG	13020101cp030	YES	548	calc	NT-Sus_Hyb	NO	<10km	NO	NO	None	8.2	Isolated	High	Moderate	Limited	LowRisk
Rio San Leonardo	Lower RG	13020101cp031	YES	299	calc	NT-Sus_Hyb	NO	<10km	NO	NO	Partial	5.8	Isolated	High	Low	Limited	LowRisk
Rio de la Cebolla, Truchas	Lower RG	13020101cp032	YES	545	2007	Unaltered (< 1%)	NO	<10km	NO	NO	Partial	17.2	Isolated	High	Moderate	Limited	LowRisk
Rio Quemado	Lower RG	13020101cp034	YES	1,223	2007	NT-Sus_Un	NO	None	NO	NO	None	16.8	Isolated	High	Moderate	Limited	LowRisk
Jicarita Creek	Lower RG	13020101cp035	YES	169	calc	Unaltered (< 1%)	NO	None	NO	NO	Partial	4.1	Isolated	High	Moderate	Limited	LowRisk
Unnamed Trib. to Rio Santa Barbara (Indian)	Lower RG	13020101cp036	YES	94	calc	NT-Sus_Hyb	NO	None	Unknown	NO	Complete	2.8	Isolated	High	Low	Limited	LowRisk
Rio Medio	Lower RG	13020101cp037	YES	1,285	calc	NT-Sus_Hyb	YES	Symp	YES	NO	None	13.1	Isolated	High	Low	Limited	LowRisk

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Conservation_Population	GMU	CP_Pop_ID	EXTANT	Ne_Est	Ne_Yr	Genetic status	RBT_Pres	Hyb_Risk	Com_Nnat	Supp	Barrier	Str_Lth	Networked	Fire_Risk	Flow_Cat	Disease_Risk	Temp_Risk
Rio Frijoles, Rio Jaroso	Lower RG	13020101cp038	YES	1,109	1992	NT-Sus_Hyb	YES	Symp	YES	NO	None	12.5	Isolated	High	Low	Limited	LowRisk
Rio Molino	Lower RG	13020101cp040	YES	273	2012	Unaltered (< 1%)	NO	None	NO	NO	Complete	5.6	Isolated	High	Low	Limited	LowRisk
Casias Creek	Lower RG	13020101cp041	YES	818	2013	Unaltered (< 1%)	NO	None	NO	NO	Complete	7.1	Moderately	High	Moderate	Limited	LowRisk
Nabor Creek	Lower RG	13020102cp001	YES	447	2006	Unaltered (< 1%)	NO	None	NO	NO	Complete	5.9	Isolated	High	Very Low	Limited	LowRisk
Little Willow Creek	Lower RG	13020102cp002	YES	279	2002	NT-Sus_Hyb	YES	Symp	NO	NO	Complete	3.7	Isolated	High	Low	Limited	LowRisk
Poso Creek	Lower RG	13020102cp003	YES	281	2005	NT-Sus_Hyb	NO	None	YES	YES	Complete	3.9	Isolated	High	Very Low	Limited	LowRisk
Jaroso Creek	Lower RG	13020102cp004	YES	240	2003	NT-Sus_Hyb	NO	None	NO	NO	None	8	Isolated	High	Very Low	Limited	LowRisk
Canjilon Creek	Lower RG	13020102cp005	YES	642	2004	>1% and <=10%	NO	<10km	NO	NO	None	8.1	Isolated	High	Low	Limited	LowRisk
El Rito	Lower RG	13020102cp006	YES	1,786	2008	Unaltered (< 1%)	NO	None	NO	NO	Complete	12.7	Isolated	High	Moderate	Limited	LowRisk
El Rito	Lower RG	13020102cp007	YES	1,450	2008	NT-Sus_Hyb	YES	Symp	YES	NO	Complete	5.3	Isolated	High	Moderate	Limited	Acute
Canones Creek	Lower RG	13020102cp008	YES	1,849	2010	Unaltered (< 1%)	NO	None	NO	NO	Complete	10.7	Isolated	High	Low	Limited	LowRisk
Polvadera Creek	Lower RG	13020102cp009	YES	0	2010	Unaltered (< 1%)	NO	None	NO	NO	Complete	13.1	Isolated	High	Very Low	Limited	LowRisk
Rio Del Oso, Rito De Abiquiu, Rito Del Oso	Lower RG	13020102cp010	**NO**	-	2012	NT-Sus_Un	NO	None	NO	NO	None	12.5	Isolated	High	Very Low	Limited	LowRisk
Wolf Creek	Lower RG	13020102cp011	YES	20	2010	>10% and <=20%	NO	None	YES	NO	Complete	0.6	Isolated	High	Low	Limited	LowRisk
East Fork Wolf Creek	Lower RG	13020102cp012	YES	513	2009	Unaltered (< 1%)	NO	None	NO	NO	Complete	3.7	Isolated	High	Very Low	Limited	LowRisk
Capulin Creek	Lower RG	13020201cp001	**NO**	-	2012	Unaltered (< 1%)	NO	None	NO	NO	None	12	Isolated	High	Low	Limited	LowRisk
Medio Dia Creek	Lower RG	13020201cp002	**NO**	-	2012	NT-Sus_Un	NO	None	NO	NO	None	0.7	Isolated	High	Very Low	Limited	LowRisk
Rio Cebolla	Lower RG	13020202cp001	YES	539	2012	NT-Sus_Un	NO	None	YES	YES	Complete	7.3	Isolated	High	Low	Moderate	LowRisk
Rito de las Palomas	Lower RG	13020202cp002	YES	404	calc	Unaltered (< 1%)	NO	None	YES	NO	None	6.9	Isolated	High	Very Low	Limited	LowRisk
Rio de Las Vacas, Anastacio, de las Perchas	Lower RG	13020202cp003	YES	662	na	>1% and <=10%	NO	None	NO	NO	Complete	4.5	Weakly	High	Low	Limited	LowRisk
Lower (Las Vacas, Anastacio, de las Perchas)	Lower RG	13020202cp003L	YES	2,266	na	>1% and <=10%	NO	None	YES	YES	None	15.4	Weakly	High	Low	Limited	LowRisk
La Jara Creek	Lower RG	13020204cp001	YES	26	est	>1% and <=10%	NO	None	NO	NO	None	4.4	Isolated	High	Low	Limited	LowRisk
Rito de los Pinos	Lower RG	13020204cp002	YES	54	est	NT-Sus_Un	NO	None	YES	NO	Complete	2.3	Isolated	High	Very Low	Limited	LowRisk
Rio Puerco	Lower RG	13020204cp003	YES	1,418	2009	>1% and <=10%	NO	None	NO	NO	None	14.4	Isolated	High	Low	Limited	LowRisk
Rio Mora	Pecos	13060001cp001	YES	75	calc	Unaltered (< 1%)	NO	None	Unknown	NO	None	2.4	Isolated	High	Low	Limited	LowRisk
Unnamed Trib. to Rio Mora	Pecos	13060001cp002	YES	115	calc	>1% and <=10%	NO	None	Unknown	NO	Partial	3.2	Isolated	High	Low	Limited	LowRisk
Rio Valdez	Pecos	13060001cp003	YES	135	2004	Unaltered (< 1%)	NO	None	NO	NO	Complete	3.7	Isolated	High	Moderate	Limited	LowRisk
Pecos River	Pecos	13060001cp004	YES	425	2003	>1% and <=10%	NO	None	NO	NO	Complete	6.3	Isolated	High	Low	Limited	LowRisk
Rito Del Padre, Rito Maestas	Pecos	13060001cp005	YES	668	1990	>1% and <=10%	NO	<10km	YES	NO	Complete	9.9	Isolated	High	Low	Limited	LowRisk
Rito los Esteros	Pecos	13060001cp006	YES	80	calc	Unaltered (< 1%)	NO	<10km	YES	NO	None	2.5	Isolated	High	Low	Limited	LowRisk
Jacks Creek	Pecos	13060001cp007	YES	561	2003	Unaltered (< 1%)	NO	<10km	NO	NO	Complete	11.3	Isolated	High	Low	Moderate	LowRisk
Cave Creek	Pecos	13060001cp008	YES	89	calc	>1% and <=10%	NO	None	Unknown	NO	Partial	2.7	Isolated	High	Low	Limited	LowRisk
Macho Creek	Pecos	13060001cp009	YES	153	2002	Unaltered (< 1%)	NO	<10km	NO	NO	Complete	3.4	Isolated	High	Very Low	Limited	LowRisk
Dalton Creek	Pecos	13060001cp010	YES	69	2006	Unaltered (< 1%)	NO	<10km	NO	NO	Complete	6.7	Isolated	High	Low	Moderate	LowRisk
Bear Creek	Pecos	13060001cp011	YES	282	calc	NT-Sus_Un	NO	None	NO	NO	Complete	5.6	Isolated	High	Low	Limited	LowRisk
Pinelodge Creek	Pecos	13060005cp001	YES	29	2010	Unaltered (< 1%)	NO	None	NO	NO	Complete	3.9	Isolated	High	Very Low	Limited	Acute

APPENDIX D

LITERATURE CITED

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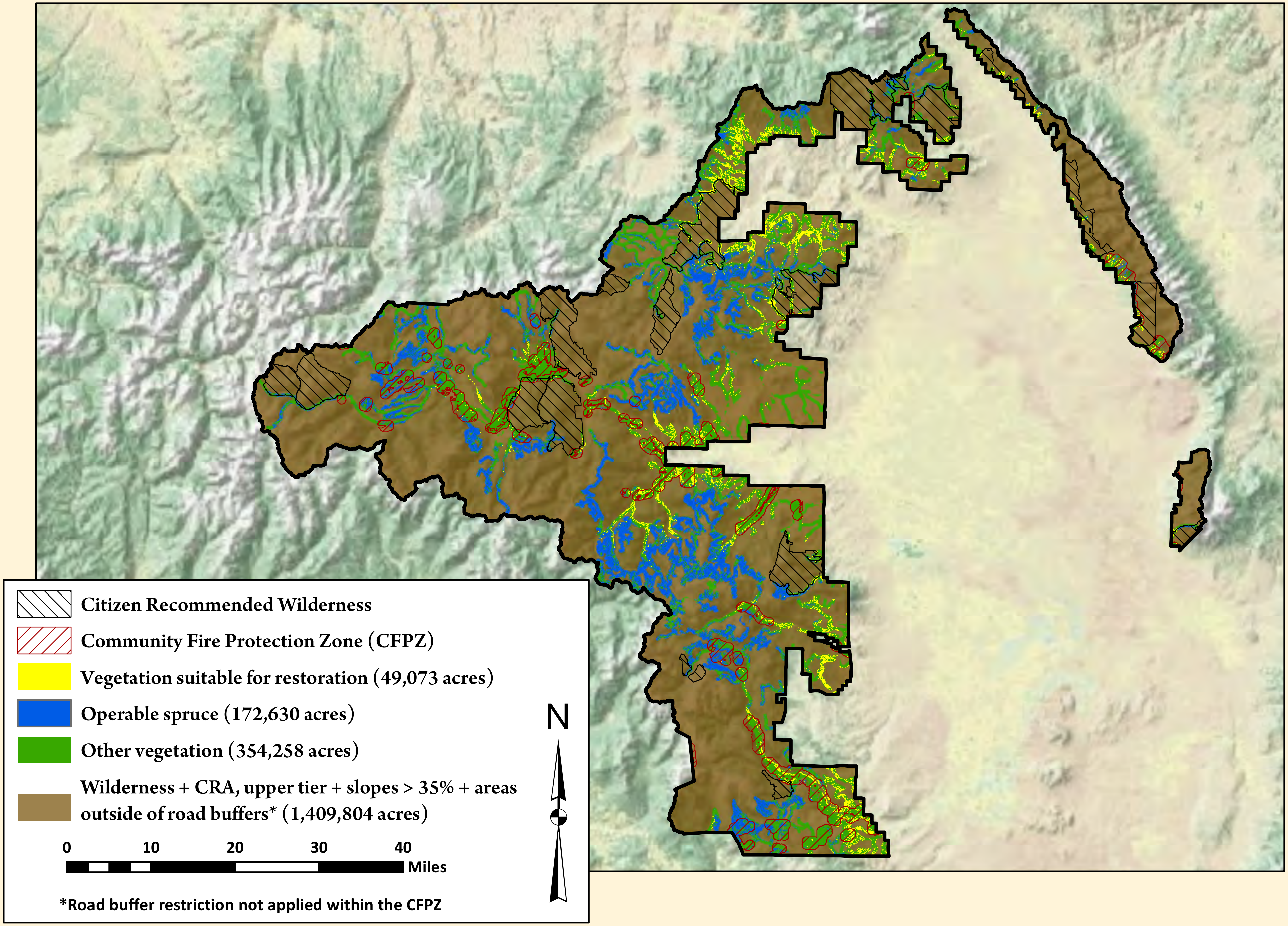
Opportunities for Mechanical Forest Restoration, Rio Grande National Forest



Landscape-Scale Fire Management on the Rio Grande National Forest