Friends of the Wild Swan
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December 13, 2023

U.S. Forest Service
Swan Lake Ranger District Attn: Christopher Dowling (Rumbling Owl)
200 Ranger Station Road
Bigfork, MT 59911
Via email to: https://www.fs.usda.gov/project/?project=64924

Mr. Dowling,

Please accept the following comments on the Rumbling Owl Project scoping proposed action on behalf of Friends of the Wild Swan. We incorporate by reference the comments submitted by Swan View Coalition.

The project area contains important habitat for many wildlife species due to its proximity to roadless areas and the Bob Marshall Wilderness. It also includes the Holland Lake bull trout critical habitat core area and lynx critical habitat. An Environmental Impact Statement is warranted to assess the impacts to threatened species such as grizzly bear, Canada lynx, wolverine and bull trout.

It appears that the project area has not been adequately surveyed to provide the accurate data necessary for the environmental analysis. For example: " Acquired lands in Section 33 hold many existing, undetermined roads." These roads must be identified and reclaimed.

• Old growth forest habitat is slated for commercial thinning and fuels reduction. Logging removes the habitat attributes that are necessary for old-growth associated wildlife and birds. The EIS must disclose (preferably with an aerial photograph map) where existing old-growth forest habitat is located, where recruitment old-growth is located and where the proposed cutting units are located. The EIS must analyze what the effects of logging will be on existing and recruitment old growth forest habitat both in terms of blowdown and other effects on the forest itself as well as on old-growth dependent wildlife.

How much old-growth forest habitat is there in the project area? Where is it? What is next to it? How connected is it? Where are mature stands that can be recruited as replacement old growth? What old-growth dependent wildlife are using it? Will this project log in old-growth forest habitat?

Is the old growth habitat fragmented? Does it have abrupt edges and have forested connections between patches been narrowed? (Very likely because this area was in a checkerboard of Plum Creek lands that were heavily logged and roaded). How big are the old growth patches? Are they sufficient to meet the needs of old growth dependent wildlife? We believe there should be an effort by the Flathead to connect rather than fragment old-growth forest habitat. Please explain and provide the science to justify how logging will increase the quantity, patch size and connectivity of old growth forests.

• It is unlikely that there are sufficient snags and down woody material. Please explain how logging will restore these attributes.

• The EIS needs to fully evaluate the effects to wildlife including old-growth associated wildlife (which has been missing in other environmental analysis on the Flathead but is essential to determine the impacts). Is the project area currently meeting the needs of old-growth associated species? Will the proposed action impact old-growth forests by either building roads in or adjacent to old-growth forest and/or placing cutting units adjacent to old-growth forest? Please provide the best available science to back up your contention that fuels reduction can be done in or adjacent to old growth and not impact the use of this scarce habitat by old-growth associated wildlife.

• For all wildlife the Flathead needs to quantify what current habitat availability, local population monitoring, and current status of the species indicate about current population health in this project landscape, or in other words, <u>is the current habitat enough?</u> If it is, how much more can you take and still not trigger significant population impacts? If there currently isn't enough habitat, how can you justify taking more?

• All the wildlife species in the project area require corridors to move for foraging, denning, nesting and seasonal habitats. The EIS must analyze and disclose: Where are these corridors? What is the habitat quality in them? What size are they? Are they wide enough to protect from edge effects and provide security? Are they fragmented by roads or past logging units? How much canopy cover, thermal cover or hiding cover is in them? How much down woody debris and snags are in them? What type of habitat is considered suitable?

Corridors of interior forest habitat between old growth habitat with a minimum width of >100 meters have been recommended by scientists. Does the Flathead have any actual width criteria you are using to define corridors in the project area? All corridor habitat in the project area should be mapped and both current and long-term objectives defined for maintaining these corridors over time.

• How is this project moving the Flathead towards or away from the goal to maintain and recruit old growth forests? How does this project sequester carbon from old trees?

• The EIS must analyze the blowdown effects to old-growth forests, riparian areas, wetlands or other forest habitats. It must also disclose whether blowdown will be salvage logged.

• Where is the current lynx foraging and denning habitat located? How will it be maintained, how will it be improved, how is it connected? How will it be impacted by this project? What are the effects to critical habitat for lynx? Will it be adversely modified? Lynx avoid clearcuts, are proposed seedtree units adjacent to other openings? Winter foraging habitat is limited – how much is there? Where is it?

• The main criteria for lynx foraging habitat is the presence of snowshoe hares. Where is the important hare habitat in this project area, and what is the estimated population density (low, medium, high)? Where is current hare habitat in the cumulative effects area? Where is current red squirrel habitat in the project and cumulative effects area? How will the foraging habitat be affected by this project as well as previous logging and roads?

• Will logging take place in mature multi-story habitat? If so, why?

• Section 33, previously a Plum Creek heavily logged and roaded section, was acquired from the Rocky Mountain Elk Foundation to maintain big game winter range. How will this project affect those habitat attributes such as thermal cover, hiding cover, security, etc? Are there many seeps and wetlands in the project area that are moist sites for elk? How will those areas be protected? What are the impacts of roads on those areas?

• How much big game thermal cover is there? Where is it? How is it connected? How much hiding cover is there? Where is it? How is it connected?

• Guidelines for elk security are a minimum of 250 acres for providing security under favorable conditions; under less favorable conditions the minimum must be >250 acres. Effective security areas may consist of several cover-types if the block is relatively unfragmented. Among security areas of the same size, one with the least amount of edge and the greatest width generally will be the most effective. Wallows, springs and saddles may require more cover than other habitats.

• Generally, elk security areas become more effective the farther they are from an open road. The minimum distance between a security area and an open road should be one half mile. The function of this ≥ one half mile "buffer" is to reduce and disperse hunting pressure and harvest that is concentrated along open roads. Failure to accomplish this function will reduce the effective size of the security area and may render it ineffective. When cover is poor and terrain is gentle, it may require more than one half mile from open roads before security is effective. (Hillis et al, 1991)

• Roads may be closed to motorized travel to provide elk security and a buffer between security areas and open roads. However, the minimum distance between open roads and security areas increases as closed-road densities increase within both the security area and buffer. (Id.)

• To be biologically meaningful, analysis unit boundaries should be defined by the elk herd home-range, and more specifically by the local herd home-range during hunting season. Elk vulnerability increases when less than 30% of analysis unit is comprised of security area. (Id.)

• These guidelines represent minimums and do not necessarily justify reducing elk security to meet these levels (i.e., if 50% of an analysis unit is security, do not assume that 20% of the unit is excess security). (Id.)

• What is the current total and open road density? How much grizzly bear core area is there in each subunit? Why are new roads being built? Why aren't more roads being decommissioned? How does this project favor the needs of the grizzly bear? How does this project maintain the 2011 on the ground baseline conditions for grizzly bears?

• Grizzly bear habitat requirements such as low road densities and security core protect a suite of other species such as elk, moose, mule deer, etc.

• How will this project maintain viability of sensitive species? How can that be measured when there are no conservation strategies or Forest Plan standards for sensitive species?

• Wolverine have been given Endangered Species Act protection, the Flathead must consult with the US Fish and Wildlife Service over the impacts to wolverine. Scientific studies are emerging about landscape effects from logging and other human activities so assumptions about habitat usage, prey availability and motorized use might change.

For example, Fisher, et al Wolverines (Gulo gulo luscus) on the Rocky Mountain slopes: natural heterogeneity and landscape alteration as predictors of distribution found: Wolverines were more abundant in rugged areas protected from anthropogenic development. Wolverines were less likely to occur at sites with oil and gas exploration, forest harvest, or burned areas, even after accounting for the effect of topography.

Wolverines elsewhere avoid human-disturbed areas (Carroll et al. 2001; Rowland et al. 2003; May et al. 2006) and recreational and industrial activity (Krebs et al. 2007). Human activities such as trapping, poaching, and road mortality have accounted for 46% (North America; Krebs et al. 2004) to 52% (Scandinavia; Persson et al. 2009) of known-cause wolverine mortalities across their range.

Wolverines avoid roads and other human development in British Columbia (Krebs et al. 2007), Norway (May et al. 2008), Idaho (Copeland et al. 2007), Montana (Carroll et al. 2001), and throughout the northwestern United States (Rowland et al. 2003).

Wolverine occurrence also increases with topographic ruggedness, where there is a combination of low- and high-elevation habitats. Bighorn sheep (Ovis canadensis Shaw, 1804) (Festa-Bianchet 1988), mule deer (Odocoileus hemionus (Rafinesque, 1817)) (D'Eon and Serrouya 2005), and other ungulates winter at lower elevations; in Scandinavia, wolverines showed significant selection for lower elevation habitats during winter months (Landa et al. 1998). It is possible that wolverines require lower elevations for foraging and higher elevations for predation refuge. Persistent spring snow cover has been hypothesized as important (Schwartz et al. 2009; Copeland et al. 2010) but is not a good predictor at this scale, since spring snow cover was sufficiently persistent across our study landscape to prevent modelling but wolverine occurrence still varied.

Southwest Crown of the Continent monitoring detected wolverines at elevations ranging from 3,346-7,567 feet.

How will this project impact wolverine habitat, foraging or displacement?

• Habitat fragmentation is generally defined as the process of subdividing a continuous habitat type into smaller patches, which results in the loss of original habitat, reduction in patch size, and increasing isolation of patches. (Heilman et al. 2002)

Habitat fragmentation is considered to be one of the single most important factors leading to loss of native species (especially in forested landscapes) and one of the primary causes of the present extinction crisis. **Although it is true that natural disturbances such as fire and disease fragment native forests, human activities are by far the most extensive agents of forest fragmentation.** For example, during a 20-year period in the Klamath–Siskiyou ecoregion, fire was responsible for 6% of forest loss, while clear-cut logging was responsible for 94% (emphasis added) (Id.)

Depending on the severity of the fragmentation process and sensitivity of the ecosystems affected, native plants, animals, and many natural ecosystem processes (e.g., nutrient cycling, pollination, predator–prey interactions, and natural disturbance regimes) are compromised or fundamentally altered. For many species, migration between suitable habitat patches becomes more difficult, leading to smaller population sizes, decreased gene flow, and possible local extinctions. (Id.)

As native forests become increasingly fragmented, ecosystem dynamics switch from being predominantly internally driven to being predominantly externally driven. Simultaneously, remnant patches become altered by changes within the patches themselves as the remnants become more and more isolated, thereby resulting in further ecological degradation across the landscape. Declines in forest species as a result of fragmentation have been documented for numerous taxa, including neotropical migrant songbirds, small mammals and invertebrates Forest fragmentation has also been associated with increased susceptibility to exotic invasion (Id.)

Among the common changes in forests over the past two centuries are loss of old forests, simplification of forest structure, decreasing size of forest patches, increasing isolation of patches, disruption of natural fire regimes, and increased road building, all of which have had negative effects on native biodiversity. These trends can be reversed, or at least slowed, through better management. (Noss 1999)

This project must reduce fragmentation and edge effects and increase patch size and core areas. Past management through even-aged silvicultural prescriptions have contributed to the fragmentation of forest habitat to the detriment of many bird and wildlife species. Large and small openings should be allowed to be created through natural processes rather than clearcut logging

• What monitoring will be done for wildlife? fish? old-growth dependent wildlife? sensitive plants? other? What past monitoring has been done to determine whether the proposed treatments actually achieve the desired results?

• How will logging in RMZs affect native fish? What is the current condition in the riparian areas? How will this project protect rather than adversely impact fish habitat and water quality? No logging or road building should be done in riparian areas.

• The 4 miles of temporary roads and 8 miles of new permanent roads will be constructed in/through RMZs with stream crossings (but they are not identified on the maps). There should not be any stream crossings which will negatively impact fish habitat and water quality. Roads should be decommissioned and removed, not upgraded and rebuilt.

• The scoping notice identified a culvert replacement for fish passage on Owl Creek Loop Road, will that allow for lake trout to move or expand?

• Hauer, et al. (1999) found that bull trout streams in wilderness habitats had consistent ratios of large to small and attached to unattached large woody debris. However, bull trout streams in watersheds with logging activity had substantial variation in these ratios. They identified logging as creating the most substantive change in stream habitats.

"The implications of this study for forest managers are twofold: (i) with riparian logging comes increased unpredictability in the frequency of size, attachment, and stability of the LWD and (ii) maintaining the appropriate ratios of size frequency, orientation, and bank attachment, as well as rate of delivery, storage, and transport of LWD to streams, is essential to maintaining historic LWD characteristics and dynamics. Our data suggest that exclusion of logging from riparian zones may be necessary to maintain natural stream morphology and habitat features. Likewise, careful upland management is also necessary to prevent cumulative effects that result in altered water flow regimes and sediment delivery regimes. While not specifically evaluated in this study, in general, it appears that patterns of upland logging space and time may have cumulative effects that could additionally alter the balance of LWD delivery, storage, and transport in fluvial systems. These issues will be critical for forest managers attempting to prevent future detrimental environmental change or setting restoration goals for degraded bull trout spawning streams."

Muhlfeld, et al. (2009) evaluated the association of local habitat features (width, gradient, and elevation), watershed characteristics (mean and maximum summer water temperatures, the number of road crossings, and road density), and biotic factors (the distance to the source of hybridization and trout density) with the spread of hybridization between native westslope cutthroat trout *Oncorhynchus clarkii lewisi* and introduced rainbow trout *O. mykiss* in the upper Flathead River system in Montana and British Columbia.

They found that hybridization was positively associated with mean summer water temperature and the number of upstream road crossings and negatively associated with the distance to the main source of hybridization. Their results suggest that hybridization is more likely to occur and spread in streams with warm water temperatures, increased land use disturbance, and proximity to the main source of hybridization.

The EIS must use the best available science to analyze how logging riparian habitat will impact native fish and water quality.

• The Holland Lake bull trout core area is critical habitat. It is threatened by non-native lake trout and mysis shrimp and a leaking sewage pond that is the Forest Service's responsibility.

The EIS must fully and completely analyze the impacts to bull trout critical habitat and westslope cutthroat trout habitat. There is no standard for sediment, temperature, pool frequency and bank stability in the Forest Plan. Sediment is one of the key factors impacting water quality and fish habitat. [*See* USFWS 2010]

> The introduction of sediment in excess of natural amounts can have multiple adverse effects on bull trout and their habitat (Rhodes et al. 1994, pp. 16-21; Berry, Rubinstein, Melzian, and Hill 2003, p. 7). The effect of sediment beyond natural background conditions can be fatal at high levels. Embryo survival and subsequent fry emergence success have been highly correlated to percentage of fine material within the streambed (Shepard et al. 1984, pp. 146, 152). Low levels of sediment may result in sublethal and behavioral effects such as increased activity, stress, and emigration rates; loss or reduction of foraging capability; reduced growth and resistance to disease; physical abrasion; clogging of gills; and interference with orientation in homing and migration (McLeay et al. 1987a, p. 671; Newcombe and MacDonald 1991, pp. 72, 76, 77; Barrett, Grossman, and Rosenfeld 1992, p. 437;Lake and Hinch 1999, p. 865; Bash et al. 2001n, p. 9; Watts et al. 2003, p. 551; Vondracek et al. 2003, p. 1005; Berry, Rubinstein, Melzian, and Hill 2003, p. 33). The effects of increased suspended sediments can cause changes in the abundance and/or type of food organisms, alterations in fish habitat, and long-term impacts to fish populations (Anderson et al. 1996, pp. 1, 9, 12, 14, 15; Reid and Anderson 1999, pp. 1, 7-15). No threshold has been determined in which fine-sediment addition to a stream is harmless (Suttle et al. 2004, p. 973). Even at low concentrations, fine-sediment deposition can decrease growth and survival of juvenile salmonids.
>
> Aquatic systems are complex interactive systems, and isolating the effects of sediment to fish is difficult (Castro and Reckendorf 1995d, pp. 2-3). The effects of sediment on receiving water ecosystems are complex and multi-dimensional, and further compounded by the fact that sediment flux is a natural and vital process for aquatic systems (Berry, Rubinstein, Melzian, and Hill 2003, p. 4). Environmental factors that affect the magnitude of sediment impacts on salmonids include duration of exposure, frequency of exposure, toxicity, temperature, life stage of fish, angularity and size of particle, severity/magnitude of pulse, time of occurrence, general condition of biota, and availability of and access to refugia (Bash et al. 2001m, p. 11). Potential impacts caused by excessive suspended sediments are varied and complex and are often masked by other concurrent activities (Newcombe 2003, p. 530). The difficulty in determining which environmental variables act as limiting factors has made it difficult to establish the specific effects of sediment impacts on fish (Chapman 1988, p. 2). For example, excess fines in spawning gravels may not lead to smaller populations of adults if the amount of juvenile winter habitat limits the number of juveniles that reach adulthood. Often there are multiple independent variables with complex inter-relationships that can influence population size.

> The ecological dominance of a given species is often determined by environmental variables. A chronic input of sediment could tip the ecological balance in favor of one species in mixed salmonid populations or in species communities composed of salmonids and nonsalmonids (Everest et al. 1987, p. 120). Bull trout have more spatially restrictive biological requirements at the individual and population levels than other salmonids (USFWS (U.S. Fish and Wildlife Service) 1998, p. 5). Therefore, they are especially vulnerable to environmental changes such as sediment deposition.

Aquatic Impacts

• Classify and analyze the level of impacts to bull trout and westslope cutthroat trout in streams, rivers and lakes from sediment and other habitat alterations:

Lethal: Direct mortality to any life stage, reduction in egg-to-fry survival, and loss of spawning or rearing habitat. These effects damage the capacity of the bull trout to produce fish and sustain populations.

Sublethal: Reduction in feeding and growth rates, decrease in habitat quality, reduced tolerance to disease and toxicants, respiratory impairment, and physiological stress. While not leading to immediate death, may produce mortalities and population decline over time.

Behavioral: Avoidance and distribution, homing and migration, and foraging and predation. Behavioral effects change the activity patterns or alter the kinds of activity usually associated with an unperturbed environment. Behavior effects may lead to immediate death or population decline or mortality over time.

Direct effects:

Gill Trauma - High levels of suspended sediment and turbidity can result in direct mortality of fish by damaging and clogging gills (Curry and MacNeill 2004, p. 140).

Spawning, redds, eggs - The effects of suspended sediment, deposited in a redd and potentially reducing water flow and smothering eggs or alevins or impeding fry emergence, are related to sediment particle sizes of the spawning habitat (Bjornn and Reiser 1991, p. 98).

Indirect effects:

Macroinvertebrates - Sedimentation can have an effect on bull trout and fish populations through impacts or alterations to the macroinvertebrate communities or populations (Anderson, Taylor, and Balch 1996, pp. 14-15).

Feeding behavior - Increased turbidity and suspended sediment can affect a number of factors related to feeding for salmonids, including feeding rates, reaction distance, prey selection, and prey abundance (Barrett, Grossman, and Rosenfeld 1992, pp. 437, 440; Henley, Patterson, Neves, and Lemly 2000, p. 133; Bash et al. 2001d, p. 21).

Habitat effects - All life history stages are associated with complex forms of cover including large woody debris, undercut banks, boulders, and pools. Other habitat characteristic important to bull trout include channel and hydrologic stability, substrate composition, temperature, and the presence of migration corridors (Rieman and McIntyre 1993, p. 5).

Physiological effects - Sublethal levels of suspended sediment may cause undue physiological stress on fish, which may reduce the ability of the fish to perform vital functions (Cederholm and Reid 1987, p. 388, 390).

Behavioral effects - These behavioral changes include avoidance of habitat, reduction in feeding, increased activity, redistribution and migration to other habitats and locations, disruption of territoriality, and altered homing (Anderson, Taylor, and Balch 1996, p. 6; Bash et al. 2001t, pp. 19-25; Suttle, Power, Levine, and McNeely 2004, p. 971).

• Native fish evolved with fire, they did not evolve with roads and logging.

> "Although wildfires may create important changes in watershed processes often considered harmful for fish or fish habitats, the spatial and temporal nature of disturbance is important. Fire and the associated hydrologic effects can be characterized as "pulsed" disturbances (*sensu* Yount and Niemi 1990) as opposed to the more chronic or "press" effects linked to permanent road networks. Species such as bull trout and redband trout appear to have been well adapted to such pulsed disturbance. The population characteristics that provide for resilience in the face of such events, however, likely depend on large, well-connected, and spatially complex habitats that can be lost through chronic effects of other management. Critical elements to resilience and persistence of many populations for these and similar species will be maintaining and restoring complex habitats across a network of streams and watersheds. Intensive land management could make that a difficult job." (Rieman and Clayton 1997)

• The project relies on BMPs to protect water quality and fish habitat. First, there is no evidence that application of BMPs actually protects fish habitat and water quality. Second, BMPs are only maintained on a small percentage of roads or when there is a logging project. What is the life expectancy for Best Management Practices? How often will they need to be re-applied and what is the expectation for securing funding to keep these roads maintained given the Forest Service's road budget?

BMPs fail to protect and improve water quality because of the allowance for "naturally occurring degradation." In Montana, "naturally-occurring degradation" is defined in ARM 16.20.603(11) as that which occurs after application of "all reasonable land, soil and water conservation practices have been applied." In other words, damage caused directly by sediment (and other pollution) is acceptable as long as BMPs are applied. The result is a never-ending, downward spiral for water quality and native fish.

Here's how it works:
• Timber sale #1 generates sediment damage to a bull trout stream, which is "acceptable" as long as BMPs are applied to project activities.
• "Natural" is then redefined as the stream condition after sediment damage caused by Timber Sale #1.
• Timber sale #2 – in the same watershed – sediment damage would be acceptable if BMPs are applied again – same as was done before.
• "Natural" is again redefined as the stream condition after sediment damage caused by Timber Sale #2.

The downward spiral continues with disastrous cumulative effects on bull trout, westslope cutthroat trout and most aquatic life.

BMPs are not “reasonable.” Clearly, beneficial uses are not being protected. In Montana, state water quality policy is not being followed. § 75-5-101 et seq. and ARM 16.20.701 et seq.

• The EIS must disclose the costs to continually apply BMPs to the already bloated road network as well as the 32.6 miles of new roads when the Flathead's entire road budget can only pay to maintain a fraction of the roads on whole forest.

• The EIS must evaluate and consider whether the proposed treatments to reduce fire and fuel hazard will actually have the desired effect during the timeframe before vegetation regrows.

The analysis must consider that in order to maintain this so called “fire proof” condition on the ground it entails repeated entries that have negative consequences for water quality, fish and wildlife. This must be analyzed as a cumulative effect. In addition, the combination of repeated entries is a programmatic vegetative management practice that represents a significant departure from the Goals and Objectives and Desired Future Condition envisioned and analyzed by the Forest Plan. We are facing “perpetual management” in an uncertain funding environment.

This issue is addressed in Rhodes and Baker’s Fire Probability, Fuel Treatment Effectiveness and Ecological Tradeoffs in Western U.S. Public Forests. Following are excerpts:

Using Equation 1, our results indicate that if treatments were repeated every 20 years across all USFS lands in the West, it would take about 720 years (36 cycles of treatments), on average, before it is expected that high-severity fire affects slightly more than 50% of treated areas while fuels are reduced. Treatments would have to be repeated at 20-year intervals for 340 years (17 cycles of treatments) before high-moderate severity fire is expected to encounter more than 50% of treated areas. Even after this duration of repeated treatments, it is likely that almost 50% of treated areas will be cumulatively affected by repeated treatments without compensatory benefits from reduced fire severity. These West-wide estimates provide perspective, but include forest types, such as subalpine forests, typified by low frequency, high-severity fire, where fuel treatments are unlikely to encounter fire [4]. Other forests, such as ponderosa pine, burn more often.

Even in ponderosa pine forests that burn relatively frequently, our regional analysis indicates that after 17 cycles of treatments, only slightly more than 50% of treated areas could potentially have fire severity reduced, on average. Our results indicate that high-severity fire is far from inevitable in areas left untreated and is, instead, expected to affect only a relatively small fraction of such areas at the broad scale of our analysis. Factoring in the probability of fire, using our framework, can significantly improve the assessments of the risks posed to aquatic systems by treating or not treating forest fuels. Where site-specific data on fire probabilities exist, the framework can be used to help locate treatments where they are most likely to encounter higher severity fire, increasing the likelihood of treatment benefits. In fact, our results indicate that such efforts are crucial.

There are several important factors that influence the aquatic tradeoffs among fuel treatments, fire, and aquatic systems that our framework does not address. Although the probability of outcomes is critical to assessing the expected value of options, the ecological costs of the outcomes of treatment vs non-treatment are also important in assessing the expected value of these options. With respect to the aquatic context, there is an ongoing need to fully evaluate tradeoffs such as the severity and persistence of the negative and positive impacts on watersheds and aquatic populations from fuel treatments and higher severity fire [8, 45]. An additional related issue is how effective treatments are when they encounter fire under a broad array of conditions affecting fire behavior [3]. While our analysis does not address these factors, it refines evaluation of net impacts of fuel treatment vs non-treatment by providing a framework for estimating the likelihood of fire occurrence in a given time frame.

At the scales of our analysis, results indicate that even if fuel treatments were very effective when encountering fire of any severity, treatments will rarely encounter fire, and thus are unlikely to substantially reduce effects of high-severity fire.

• In addition to Cohen's work on defensible space near structures, new science is validating the same approach in Syphard, et al., The role of defensible space for residential structure protection during wildfires which concludes:

"Structures were more likely to survive a fire with defensible space immediately adjacent to them. The most effective treatment distance varied between 5 and 20 m (16–58 ft) from the structure, but distances larger than 30 m (100 ft) did not provide additional protection, even for structures located on steep slopes. The most effective actions were reducing woody cover up to 40% immediately adjacent to structures and ensuring that vegetation does not overhang or touch the structure. Multiple-regression models showed landscape-scale factors, including low housing density and distances to major roads, were more important in explaining structure destruction. The best long-term solution will involve a suite of prevention measures that include defensible space as well as building design approach, community education and proactive land use planning that limits exposure to fire."

So, what is important for protecting homes and other structures is what the homeowners do on their property, not the thinning the Forest Service does miles away. What have homeowners adjacent to Forest Service land done to fire proof their homes and other structures on their property?

• How will the Flathead maintain the proposed new road system in the project area? What is the road budget?

• How will climate change impact your assumptions about this project?

• The Flathead must fully analyze the impacts of climate change. Published scientific reports indicate that climate change will be exacerbated by logging, that climate change will lead to increased wildfire severity (including drier and warmer conditions that may render obsolete the proposed effects of the project) and stream flows will be altered with reduced water in the summer and/or peak flows/flood events outside of historical norms. The Forest Service must candidly disclose, consider, and fully analyze the published scientific papers addressing climate change in these contexts. [See the Montana Climate Assessment at montanaclimate.org]

• Controlling weeds and preventing their spread is a huge issue that the Flathead does not have a grip on. Current methods are obviously not working, weeds spread on forest roads, in cutting units, landings, burn piles, and on to private property. The best way to prevent weeds from spreading out of control is not to disturb the native vegetation. Please do not attempt to dupe the public into believing that the same past failed mitigation measures to control weeds will somehow miraculously work in this project. This project will spread weeds, not reduce them adding another impact that will reduce forage for wildlife and increase competition with native plants.

We expect our comments be given full consideration. Please keep us informed.

/s/Arlene Montgomery
Program Director

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DEFINING ELK SECURITY: THE HILLIS PARADIGM

ABSTRACT

Elk vulnerability may be reduced, and hunter opportunity may be increased, by providing security areas for elk during the hunting season. We define security area requirements for land managers so that timber harvest decisions can reflect elk security needs.

To provide a reasonable level of bull survival, each security area must be a nonlinear block of hiding cover ≥ 250 acres in size and ≥ one-half mile from any open road. Collectively, these blocks must equal at least 30% of the analysis unit. Vegetation density, topography, road access, hunter-use patterns and elk movements are variables that must be considered when applying these guidelines. Examples are provided that illustrate how the security guidelines are applied in the field.

INTRODUCTION

Timber harvest affects elk vulnerability by changing the structure, size, juxtaposition and accessibility of security areas. Biologists have recently provided working definitions of "security," "security area," and "elk vulnerability" (Lyon and Christensen 1990). However, elk and timber managers still await research answers to current questions such as: "How large must a cover block be to provide effective security, how far must a security area be from a road, and how much of the area should provide security to meet elk vulnerability objectives?"

We developed guidelines for retaining elk security areas west of the Continental Divide in Montana. We suggest that the concepts presented here could assist managers in providing security areas elsewhere. We also hope this stimulates constructive criticism and research that improve the guidelines.

Special thanks go to S. D. Rose for helpful editorial comments. J. E. Firebaugh and R. E. Henderson provided technical reviews of the manuscript. We thank O. L. Daniels and C. W. Spoon for supporting development and application of these guidelines.

STUDY AREA

We devised guidelines applicable to the situations we know in the Clark Fork River drainage (excluding the Flathead River drainage). The area is characterized by steep slopes extensively forested by ponderosa pine, Douglas-fir, lodgepole pine, western larch and subalpine fir. Average elk populations and hunter numbers are at 30-year highs in the area, while the average bull/cow ratio observed by Montana Department of Fish, Wildlife and Parks (MDFWP) biologists in early spring has declined during the same period (MDFWP, Missoula, unpubl. data). The majority of elk habitat in the area is managed by the Lolo, Bitterroot and Deerlodge National Forests; although substantial portions are owned by Plum Creek Timber Company, Champion International Corporation, other private landowners, Montana Department of State Lands and USDI Bureau of Land Management.

MANAGEMENT OBJECTIVES

Lonner and Cada (1982) proposed that, "hunting recreational opportunities are good when hunting season lengths are relatively long, harvest rates are uniform, and rules and regulations few. The present 35-day general elk-hunting season in Montana permits a diversity of choice [for hunters] with regard to time, weather conditions, hunter density and area. A lengthy hunting season has little meaning if the majority of the harvest occurs in the first few days."

Nine years since Lonner and Cada's (1982) contribution, MDFWP and the three national forests within the study area formally share the following objectives: 1) maintain the current, relatively unregulated, 5-week hunting season; 2) distribute the bull harvest evenly over the entire hunting season; and 3) maintain a desired level of mature bulls in the post-hunting season population (For. Plan, Lolo Natl. For., 1986; For. Plan, Bitterroot Natl. For., 1987; For. Plan, Deerlodge Natl. For., 1987; Draft Elk Manage. Plan, MDFWP, Helena, 1991). We developed guidelines to meet these objectives.

The agencies have decided to maintain habitat security levels that allow desired numbers of bull elk to escape harvest, rather than impose more restrictions on hunters (e.g., shorter hunting seasons, antler-point restrictions, limited licenses). The recreational opportunities resulting from this type of management are becoming increasingly rare nationwide (Anon. 1988).

DOCUMENTATION

We developed guidelines from the following background of knowledge:

1. Elk behavior changes in response to the hunting season (Marcum 1975; Morgantini and Hudson 1979, 1985; Canfield 1988; Lyon and Canfield 1991).

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2. Elk avoid areas adjacent to roads with vehicular traffic, especially during the hunting season (Marcum 1975, Perry and Overly 1976, Lyon 1979, Irwin and Peek 1983, Lyon 1983, Lyon et al. 1985, Lyon and Canfield 1991).

3. Elk spend more time in dense cover during hunting season than they do before the hunting season (Marcum 1975, Irwin and Peek 1983, Canfield 1988). Large cover-blocks contribute to security more than small blocks (Canfield 1988, Lyon and Canfield 1991).

4. Elk movements generally are confined to habitats within a traditionally used home range (Edge et al. 1985, Lyon et al. 1985, Edge et al. 1986).

5. Road closures may either increase or decrease elk vulnerability depending upon the influences of cover, topography and hunting pressure, both within and adjacent to a security area (Basile and Lonner 1979, Lyon et al. 1985).

SECURITY-AREA GUIDELINES

How Large Must a Cover Block Be?

Larger is better—To meet the hunting opportunity objectives outlined here, managers should strive to retain, perpetuate, or replace the largest security areas possible. We assume that as security areas increase in size, elk become harder for hunters to find, and liberal hunting opportunities become less costly in terms of elk vulnerability.

Minimum size—In the lower Clark Fork drainage, conditions are favorable for elk to elude hunters: cover is dense, terrain is steep, and forest communities are largely unfragmented. Lyon and Canfield (1991) found that elk in this area selected for large, connected, vegetation communities (i.e., forest blocks of similar canopy structure). All other factors held constant, 236-acre unfragmented communities met minimum security requirements for 60% of the radioed elk. For the purposes of these guidelines, 250 acres appears to be the minimum-sized area for providing security under favorable conditions; under less favorable conditions, the minimum must be >250 acres.

Variables to consider—Effective security areas may consist of several different cover-types if the block is relatively unfragmented. For example, regenerated cutting units that provide reasonable cover might be found within an effective security area (Canfield et al. 1986). Among security areas of the same size, one with the least amount of edge and the greatest width generally will be the most effective. Rugged topography may increase security if it substantially decreases the accessibility of the area to hunters. Wallows, springs and saddles may require more cover than other habitats because both hunters and elk recognize and target these destinations.

How Far Must a Security Area Be from a Road?

Minimum distance—Generally, security areas become more effective the farther they are from an open road. Considering documented road-avoidance by elk (Lyon 1983, Lyon et al. 1985), the minimum distance between a security area and an open road should be one half mile. The function of this ≥ one half mile "buffer" is to reduce and disperse hunting pressure and harvest that is concentrated along open roads (Daneke 1980). Failure to accomplish this function will reduce the effective size of the security area and may render it ineffective.

Road design considerations—Road-design features may inadvertently turn designated security areas into hunter destinations. For example, trailheads, turnouts and/or parking areas in close proximity to security areas will concentrate hunting pressure in the vicinity and increase elk vulnerability. Similarly, open roads located both above and below a security area on a slope will encourage hunters to walk through the security area.

Cover and terrain—When cover is poor and terrain is gentle, it may require a distance >one half mile from open roads before security is effective. In such situations, hunters may identify the security area from the road, and the gentle terrain will deter few hunters from hiking. Conversely, if the security area is hidden or difficult to reach from a road, elk may find security in situations < one half mile from an open road.

Closed roads—Roads may be closed (to motorized travel) to provide security and a buffer between security areas and open roads. However, the minimum distance between open roads and security areas increases as closed-road densities increase within both the security area and buffer. Closed roads located within security areas may increase elk vulnerability by providing hunters with walking and shooting lanes. Use of horses and increasing use of mountain bikes by hunters on closed roads allows them better access and increases elk vulnerability, compared to unroaded habitats. Therefore, roads within security areas should be kept to an absolute minimum.

How Much of the Area Should Provide Security?

Analysis unit—First, a standardized "habitat analysis unit" (Lyon and Christensen 1990) must be described. To be biologically meaningful, analysis unit boundaries should be defined by the elk herd home-range (Edge et al. 1986), and more specifically by the local herd home-range during hunting season. Typically, the hunting season home-range includes the local herd transitional-range and at least the upper edge of winter range. These boundaries should be verified in advance by radio telemetry, particularly where elk vulnerability is at issue. Without telemetry data, biologists should test their home-range predictions against the experience of reliable local hunters and outfitters. Analysis units should not be adjusted for land ownership; instead, they should reflect the cumulative habitat conditions perceived by elk.

Minimum amount of security—Our collective experience suggests elk vulnerability increases when less than 30% of an analysis unit is comprised of security areas (Canfield 1991). Where bull survival objectives are high, it may be necessary to retain greater than 30% of the analysis unit in security.

Spatial arrangement—In conjunction with considering "how much security," it is critical to consider spatial arrangement of security areas across the landscape. The arrangement should provide for the habitat needs of elk through the 5-week hunting season (e.g., forage and water). Providing security only on dry, harsh, steep slopes may allow elk to avoid hunters early in hunting season; however, it is unlikely that elk will stay in harsh sites for extended periods (Marcum 1975). Further, security areas should cover a wide elevational range so they are available to elk under various weather conditions (e.g., security areas at high elevations may be unusable by elk during periods of deep snow).

A few large, or several minimum-sized, security areas may comprise the same combined proportion of an analysis unit. The best balance between security-area sizes and numbers for an analysis unit will result from creative thinking firmly based on knowledge of local elk-movement and hunting patterns.

APPLICATION OF THE MANAGEMENT GUIDELINES

We suggest that security areas should be ≥ 250 acres in size, ≥ one half mile from an open road, and should comprise ≥ 30% of a valid analysis unit. Unquestioning adherence to these guidelines may lead to serious misapplications and should be avoided. We believe the guidelines are properly applied when used to compare relative security levels in an analysis unit over time or to compare and evaluate the cumulative impacts of various timber-harvest alternatives on security. These guidelines represent minimums and do not necessarily justify reducing security to meet these levels (i.e., if 50% of an analysis unit is security, do not assume that 20% of the unit is excess security).

Inferences from detailed knowledge of a local elk herd—such as that typically obtained by radio telemetry—should override these management guidelines whenever discrepancies occur. For example, radioed elk have shown us site-specific exceptions where security is provided along highways or in small cover-blocks that hunters do not find. Similarly, traditional migration corridors and other elk concentration areas, if known, may deserve special considerations that are not covered by these guidelines (USDA 1991).

A comprehensive, sustained timber-management planning effort is required to obtain the greatest benefits from these guidelines. Radio-telemetry data should be collected ≥ 1 before year preparing alternative management strategies, and it may take ≥ 1 year to budget and prepare for a projected telemetry effort. Future timber harvest rotations, and recruitment of new security areas, should be projected to evaluate the best options for any proposed timber sale. Proposed timber harvests in remote and heavily forested analysis units should be carefully approached because the rare opportunity exists to retain elk security by design in these units, rather than by default as dictated by past logging practices.

In analyzing security requirements for a specific area, interpretation of the guidelines is needed to ensure that the result makes biological sense for local conditions. The point of designating elk security areas is not to meet some generalized guidelines, but to provide functional habitat.

We present examples of actual management problems we have addressed, to illustrate: I) guideline adjustments that made designated security areas reflect reality, and (2) provisions for meeting present and future security needs.

Example I

The Sequoia-Brewster area lies about 20 miles from Missoula. The area's entrance road ends at a gate on level terrain (Fig. 1). The ease of walking in the area and the concentration of hunters at the end of the road suggested to us that an area only one half mile from the parking area would not provide adequate security. Therefore, the buffer between the parking and security areas was increased to 1 mile.

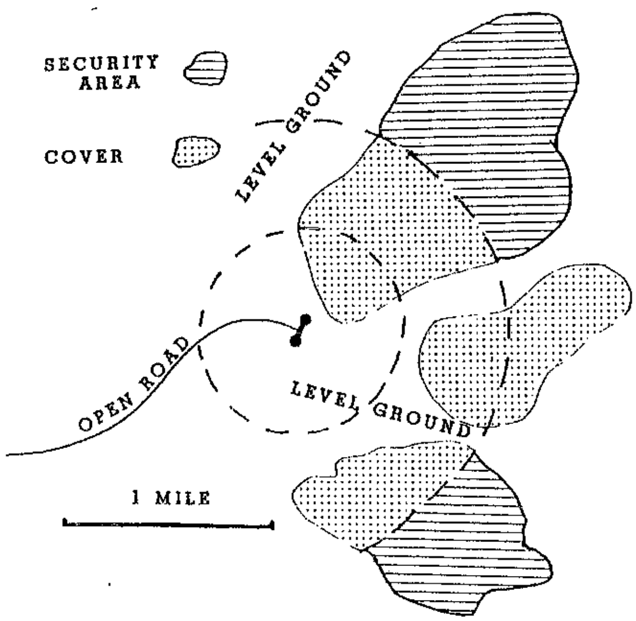

Figure 1. *Sequoia-Brewster area, showing the relationship of security to an open road, recognizing the trailhead, level ground, and concentrated use. The one half mile zone was increased to 1 mile.*

Example 2

The Tujo area includes a ridge between two parallel, creek-bottom roads; the two roads are connected by a road that switchbacks over the steep ridge (Fig. 2). This connecting road affects security in two ways: 1) its zigzag path accesses a large area and reduces the size of security areas A, B, and C (Fig. 2); and 2) it provides easy access to the ridgetop. Hunters can drive to the top and walk on a level, closed road into security area A. Likely hunting routes from the ridgetop to the creekside roads (and often to a second vehicle) are all downhill.

Keeping the connecting road open during the hunting season seriously compromises the protection we would expect from a large (2,400-acre) block such as security area A. To increase the area and quality of security, we proposed closing the connecting road during the hunting season. Since much of the area's popularity is due to the easy access provided by the open connecting road, public involvement on the issue is planned.

Example 3

The Sapphire Divide area (23,000 acres) lies 25 miles southeast of Missoula. The area is unusual because past logging was concentrated in the high-elevation basins along a major ridgetop-road. This fragmented the productive habitats that would have made good security areas. The 13% of the area that now provides security is all located in low- to mid-elevation sites that are low in productivity and characterized by steep, rocky ground.

Radio-tracking studies (Marcum 1975) in the 1970s showed that Sapphire Divide elk used the heavily logged, high elevations until hunting season. When hunters entered the area along the ridgetop road, elk immediately dropped down to the steep, dry slopes below. The elk adopted a weekly pattern of movement, returning to the productive, higher areas at mid-week and fleeing to the unproductive secure areas on weekends (when hunting pressure was highest). Thus the security areas were not only inadequate in acreage, they also were located in the

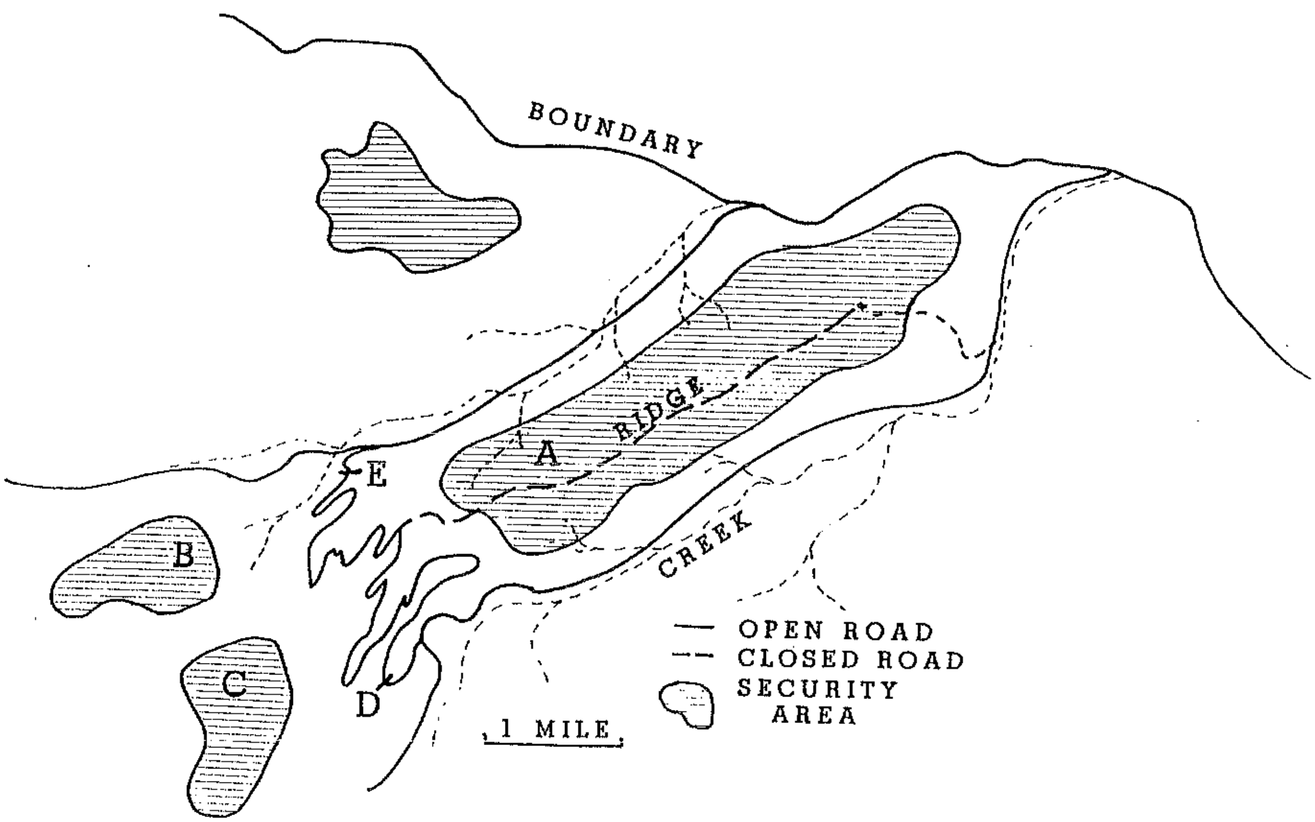

Figure 2. *Tujo area, showing how security area A is affected by the ridgetop closed road. The proposed solution is to close the connecting road at points D and E.*

wrong place. This made bulls especially vulnerable.

Predictably, the bull/cow ratio of this elk population is extremely low. After the 1989 hunting season, this ratio had declined to 3:100 (J.E. Firebaugh, MDFWP, Missoula, pers. commun.), suggesting that bull mortality due to hunting is very high.

To recover security in this analysis area, we first proposed to decrease hunter access to the high-elevation basins by closing entry roads near the points where they cross the divide from the west (Fig. 3). Second, to allow recovery of large cover-blocks in the productive, high-elevation basins, we developed a long-term strategy for the spatial arrangement of timber harvest: deferring timber harvest in designated large blocks (Fig. 3) to allow contiguous areas to regain cover at the same time, and reduce the area's fragmentation. Third, future timber harvests will be designed to minimize fragmentation by concentrating logging in small areas not currently providing security. The initial logging entries will revisit previously logged land, joining (in effect) the scattered, recovering units (Fig. 4). This will create a block of recovered cutting-units that will provide the next generation of security, totalling about 25% of the analysis unit by the year 2000.

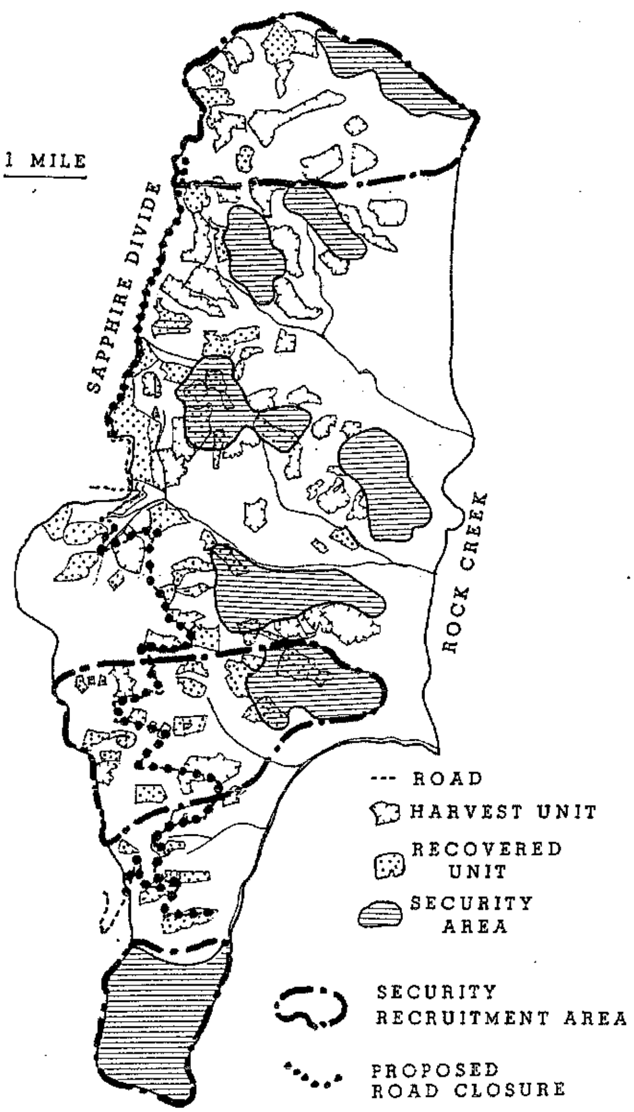

Figure 3. *Sapphire Divide area, showing designated security recruitment areas and roads proposed for closure. Note how the past harvest pattern has fragmented cover.*

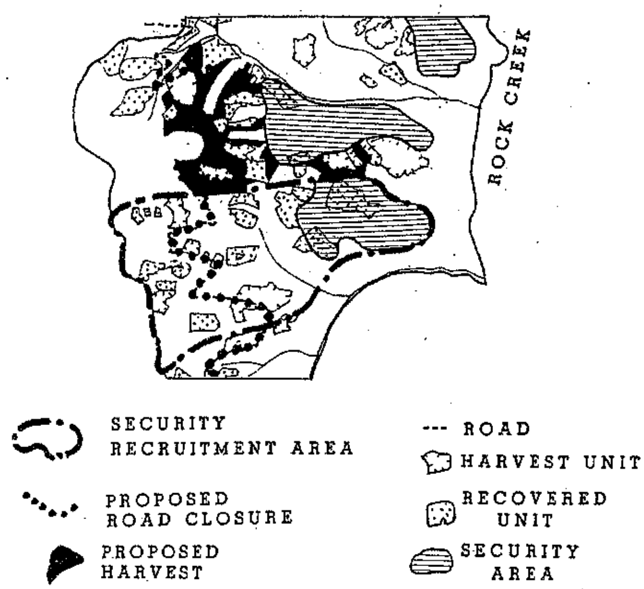

Figure 4. *South Sapphire Divide area, showing a clustered timber harvest strategy designed to create a large block of future security. Note how proposed cutting units are adjacent to recovered harvest units.*

CONCLUSIONS

During the last year, these guidelines were applied to nine elk herd-units involving 14 timber sales. Two disturbing trends were discovered. First, most herd units already had less than the minimum 30% security due to past timber harvest; in many of these cases, there were strong indications that bull survival was declining or at risk. Second, even in situations where security was substantially less than 30%, all remaining security stands were targeted for timber harvest. This indicates that timber harvest decisions made over the next few years will potentially severely impact remaining security and, ultimately, hunter opportunity.

Additional research is needed to test and refine these guidelines. However, based on the rapid, apparent decline of security, it is critical that we begin applying these guidelines immediately. Planning must not only address the quality and spatial arrangement of existing security areas, but also must provide for the regeneration of replacement security areas where a sustained timber harvest is desired.

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ARTICLE

Wolverines (*Gulo gulo luscus*) on the Rocky Mountain slopes: natural heterogeneity and landscape alteration as predictors of distribution

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Abstract: A species' occurrence can be influenced by natural and anthropogenic factors; disentangling these is a precursor to understanding the mechanisms of distribution. Anthropogenic factors may be especially important at contracting range edges. We test this premise for wolverines (*Gulo gulo luscus* L., 1758) at the edge of their Rocky Mountain range in Alberta, Canada, a mosaic of natural heterogeneity and extensive landscape development. As wolverines have a suspected negative response to human activity, we hypothesized their occurrence on the Rockies' slopes is predicted by a combination of natural and anthropogenic features. We surveyed wolverines at 120 sites along a natural and anthropogenic gradient using hair trapping and noninvasive genetic tagging. We used abundance estimation, generalized linear, and hierarchical models to determine whether abundance and occurrence was best predicted by natural land cover, topography, footprint, or a combination. Wolverines were more abundant in rugged areas protected from anthropogenic development. Wolverines were less likely to occur at sites with oil and gas exploration, forest harvest, or burned areas, even after accounting for the effect of topography. The relative paucity of wolverines in human-impacted portions of this range edge suggests that effective conservation requires managing landscape development, and research on the proximal mechanisms behind this relationship.

Key words: range edge, wolverine, *Gulo gulo luscus*, occupancy models, abundance estimation, habitat fragmentation, landscape scale.

Résumé : La présence d'une espèce en un lieu donné peut être influencée par des facteurs naturels et humains; la compréhension des mécanismes de répartition commence entre autres par la clarification des rôles de ces facteurs. Les facteurs humains peuvent s'avérer particulièrement importants aux bordures d'aires de répartition en contraction. Nous vérifions cette hypothèse pour le carcajou (*Gulo gulo luscus L.*, 1758) à la bordure de son aire de réparation dans les montagnes Rocheuses de l'Alberta (Canada), une mosaïque d'hétérogénéité naturelle et de secteurs aménagés. Comme il est soupçonné que le carcajou réagit négativement à l'activité humaine, nous avons postulé que sa présence sur les pentes des Rocheuses peut être prédite par une combinaison de caractéristiques naturelles et anthropiques. Nous avons étudié des carcajous en 120 sites le long d'un gradient naturel et anthropique en utilisant le prélèvement de poils à l'aide de pièges et le marquage génétique non invasif. Nous avons utilisé l'estimation de l'abondance et des modèles linéaires généralisés et hiérarchiques pour déterminer si le meilleur prédicteur de l'abondance et de la présence en un site était la couverture naturelle du sol, le relief, l'empreinte ou une combinaison de ces facteurs. Les carcajous étaient plus abondants dans les secteurs accidentés protégés de l'aménagement humain. Ils étaient moins susceptibles d'être présents dans des sites d'exploitation pétrolière et gazière et de coupe forestière ou dans des brûlis, et ce, même en tenant compte de l'effet du relief. La rareté relative des carcajous dans les portions de cette bordure d'aire de répartition touchées par des impacts d'origine humaine laisse croire que la conservation efficace nécessite la gestion de l'aménagement du paysage et de la recherche sur les mécanismes proximaux qui sous-tendent cette relation. [Traduit par la Rédaction]

Mots-clés : bordure d'aire de répartition, carcajou, *Gulo gulo luscus*, modèles d'occupation, estimation de l'abondance, fragmentation de l'habitat, échelle du paysage.

Introduction

Habitat loss, fragmentation, and alteration are a primary cause of many species' declines, and remain a pervasive anthropogenic phenomenon affecting ecological systems (Fahrig 1997, 2003). Determining the correlates of a species' spatial distribution across heterogeneous (and fragmented) landscapes is a key precursor to elucidating the ecological processes creating those patterns (e.g., Wiens et al. 1993). In particular, disentangling natural from anthropogenic correlates of distribution is a necessary requirement for effective conservation and management, and is often demanded when species conservation potentially conflicts with economically important landscape development. This task is further complicated because pattern and process can change markedly among landscapes as ecological and spatial contexts change, potentially preventing reliable inference from other landscapes (Fisher et al. 2005; Wheatley and Johnson 2009); this may be particularly true of circumboreal species distributed over highly varied landscapes, such as wolverines (*Gulo gulo* L., 1758).

Wolverines once inhabited boreal, tundra, and mountain habitats across North America and Eurasia (Pasitschniak-Arts and Larivière 1995) but their range has contracted, and populations declined,

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since European colonization (Weaver et al. 1996; Laliberte and Ripple 2004; Aubry et al. 2007). On the eastern edge of their Rocky Mountain range in the province of Alberta, wolverines are listed as "Data deficient", reflecting a lack of sufficient data for legal designation (Petersen 1997; Alberta Fish and Wildlife Division 2008). Historical trapping records suggest wolverines were distributed across Alberta's Rocky Mountains, adjacent foothills, and boreal forests (Petersen 1997; Poole and Mowat 2001; Alberta Fish and Wildlife Division 2008), but their current distribution remains unknown and wolverines' range here receives continued human perturbation.

This landscape is a topographically diverse conifer forest mosaic with oil and gas exploration, forest harvesting, coal mining, roads, and motorized recreational access. All of these impacts remove forest cover or increase human access, but of these oil and gas exploration is the most spatially extensive. It produces very narrow seismic lines—ca. 3 m wide linear corridors cut into forests—crisscrossing the landscape in densities sometimes exceeding 25 km/km² (see also Schneider et al. 2003). Seismic lines remove forest cover and increase access for industrial activities (heavy-truck haulage, well pads, and pipelines) and motorized recreation (snowmobiles and off-road vehicles). Extensive spatial linear features and accompanying human activity are known to affect the movement, distribution, and ecological interactions of other mammals in this region (Whittington et al. 2005; Muhly et al. 2011; Fisher et al. 2012; McKenzie et al. 2012).

This anthropogenic mosaic grades into rugged, high-elevation mountain landscapes largely protected from anthropogenic footprint. The current edge of wolverines' distribution is believed to straddle this gradient (Laliberte and Ripple 2004), but the landscape features contributing to range demarcation (and by inference, range contraction) remain unknown. Natural features likely have an effect; we suspected that habitat alteration has a significant added effect that has gone unnoticed, or has been absorbed into a shifting baseline (sensu Pauly 1995) of wolverine rarity. Wolverines elsewhere avoid human-disturbed areas (Carroll et al. 2001; Rowland et al. 2003; May et al. 2006) and recreational and industrial activity (Krebs et al. 2007). Human activities such as trapping, poaching, and road mortality have accounted for 46% (North America; Krebs et al. 2004) to 52% (Scandinavia; Persson et al. 2009) of known-cause wolverine mortalities across their range. These studies focussed on individual mortality and site selection via telemetry; none have systematically examined wolverine abundance and occurrence across a gradient of landscape development and natural heterogeneity to examine the relative contribution of each in demarcating wolverine distribution. This was our objective.

We hypothesized that wolverines would be more abundant in areas without landscape development and that the probability of wolverine occurrence varies along a spatial gradient as a function of (*i*) land cover, (*ii*) topography, and (*iii*) the degree of landscape alteration, measured as seismic-line density and the percentage of area regenerating from forest fire and timber harvest. We predicted that wolverine abundance and occurrence would increase with land cover and topographic heterogeneity and decrease with habitat alteration.

Materials and methods

To test these hypotheses, we used noninvasive genetic tagging (NGT) through hair trapping (Waits 2004; Kendall and McKelvey 2008) to survey spatial patterns of wolverine occurrence (e.g., Flagstad et al. 2004; Mulders et al. 2007; Hedmark and Ellegren 2007; Fisher et al. 2011; Magoun et al. 2011). For robust inference, we related these parameters to landscape composition using three approaches: abundance estimation models (Amstrup et al. 2010), species distribution models (Franklin and Miller 2009), and occupancy models (MacKenzie et al. 2002, 2006), ranked in an information–theoretic framework, to determine those factors that best explained wolverine occurrence.

Study area

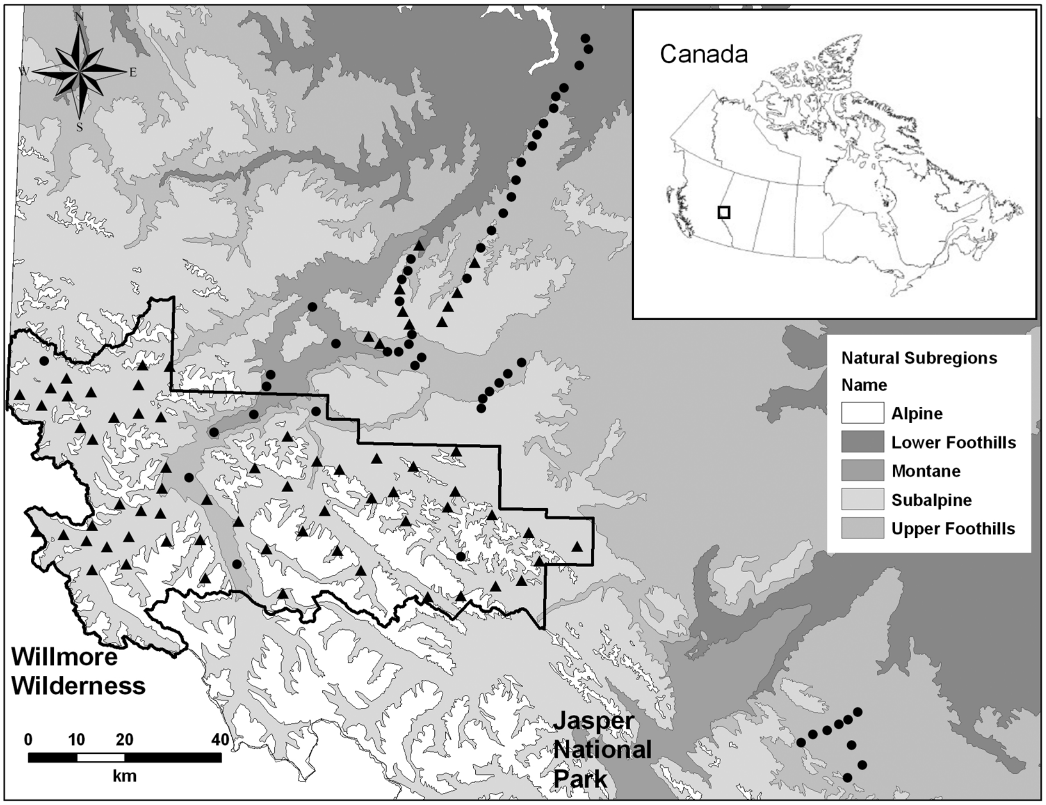
We sampled wolverine occurrence along an approximately east–west gradient (trending to northwest–southeast) spanning the Main Ranges, Front Ranges, and Upper Foothills of the Rocky Mountains in Alberta, Canada (Fig. 1). The area receives high precipitation and winter snow accumulation >2 m. The western end of the gradient is topographically rugged with peaks up to 3000 m, steep-sloped ridges, and wide valley bottoms. Slopes are forested by Engelmann spruce (*Picea engelmanni* Parry ex Engelm.) and subalpine fir (*Abies lasiocarpa* (Hook.) Nutt.). The mountains grade eastward into subalpine, upper foothills, and montane natural subregions (Downing and Pettapiece 2006), with elevations ranging to 1700 m. Forests are commonly mixed mature lodgepole pine (*Pinus contorta* Douglas ex Loudon) with white spruce (*Picea glauca* (Moench) Voss) or balsam fir (*Abies balsamea* (L.) Mill.). The west is protected from development within the Willmore Wilderness Area, a 4600 km² conservation area exempt from forest harvesting, mining, petroleum exploration, roads, and motorized transport, though with recreation, off-road trails, and large burns. From the Willmore, the landscape grades into an increasingly intensive network of roads and seismic lines for petroleum exploration (Fig. 2); conifer forests have been harvested since approximately 1955. This is a mosaic landscape of different forest stand ages, habitat alteration, motorized access, and industrial and recreational human activity. Fur trapping occurs across both landscapes with about <5 animals taken each year (Petersen 1997; Poole and Mowat 2001).

Experimental design

Methods and design mirror Fisher et al. (2011, 2012). Wolverine occurrence was sampled with noninvasive genetic tagging (NGT) via hair sampling at 120 survey sites (Fig. 3). Hair traps consisted of a tree loosely wrapped with Gaucho® barbed wire (Bekaert, Brussels, Belgium). We baited this tree with a large (ca. 15 kg) skinned beaver carcass and O'Gorman's LDC extra scent lure (O'Gorman's Co., Montana, USA). Sampling sites were deployed in early December and sampled monthly through the end of March—a period when food is scarce and bait is most effective in attracting mammals. We sampled within a systematic probabilistic design. Where no motorized access exists, we employed a systematic design constrained by helicopter access and avalanche risk. Sixty-six sites were placed 5727 ± 1574 m (mean ± SD) apart; 30 were sampled in 2006–2007 and 36 in 2007–2008, for a total area of ~4200 km² sampled. Where motorized access exists, this systematic design was constrained by road and trail access. Fifty-four foothills sites were deployed 4335 ± 5218 m (mean ± SD) apart. We sampled from early December through March 2004–2005, and again in 2005–2006; the first year's data were used in abundance estimation only.

Hair samples were collected monthly from the barbed-wire hair traps using sterile techniques. Species were identified from follicular DNA (Wildlife Genetics International, Nelson, British Columbia, Canada). DNA was extracted from hairs using QIAGEN®'s DNEasy™ Tissue Kits (QIAGEN, Hilden, Germany) and analysed to identify species using sequence-based analysis of the 16S rRNA gene of mitochondrial DNA (mtDNA) (sensu Johnson and O'Brien 1997), then compared with a DNA reference library of known mammal species. Samples identified as wolverine were assayed using microsatellite analysis to identify unique individuals using seven microsatellite markers, a number considered adequate for genetic capture–mark–recapture studies (Paetkau 2004). We summed wolverine presences across 3 months (Dec.–Jan., Jan.–Feb., Feb.–Mar.) to yield a 0–3 count of species occurrences at each site—the dependent data for species distribution models. Monthly

Fig. 1. Presence (triangles) and absence (circles) of wolverines (*Gulo gulo luscus*) at 120 hair-trapping stations in the Front Ranges, Main Ranges, and Foothills of the Rocky Mountains of west-central Alberta, Canada. This landscape is a mosaic of high-elevation alpine patches, mid-elevation subalpine forests, and montane and foothills forests. The western portion of the study area is protected from anthropogenic development within the Willmore Wilderness Area (black border).



occurrences by individuals informed capture histories for abundance estimation models.

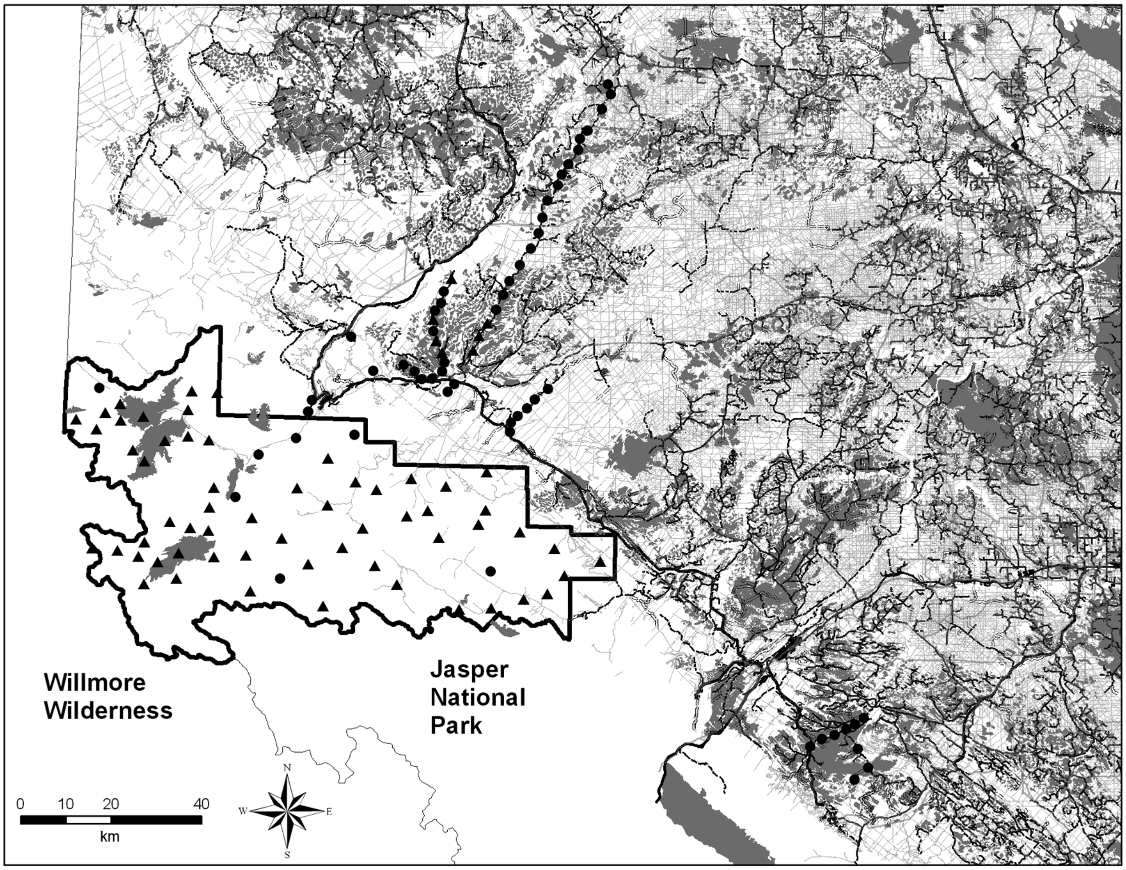
Abundance estimation

We used the Rcapture package (Baillargeon and Rivest 2007) in program R version 2.14.2 (R Development Core Team 2012) to estimate wolverine abundance. It is not feasible to relate abundance to the gradient of anthropogenic disturbance (since abundance is calculated for discrete areas, whereas the gradient is continuous). However, legislated landscape protection plays a role in the degree of disturbance (together with surface accessibility, existing land tenures, underlying geomorphology, and petroleum prices), so we asked whether wolverine abundance differed between the protected and the unprotected portions of the gradient. Models assumed a demographically closed population: mortality rates among a small population of large carnivores are expected to be near-zero over a 3-month period; our sampling period pre-dates mean kit emergence; dispersal occurs in this period (Inman et al. 2012), but there is no evidence that immigration differs from emigration. Rcapture calculates loglinear mark–recapture models (Cormack 1989) based on flexible assumptions of (*i*) no variation in hair-trap capture probability among individuals, M_0; (*ii*) variation in space, M_h; (*iii*) variation through time, M_t; (*iv*) variation in time and space, M_{th}; (*v*) behavioural variation resulting in a trap effect, M_b. Chao's (1987), Darroch et al.'s (1993), and Poisson (Rivest and Baillargeon 2007) model variants were also calculated. We selected the model with assumptions (heterogeneity, behaviour, temporal variability) that adequately fit the modelled data—a key requirement of abundance models (Baillargeon and Rivest 2007) that is reflected in low standard errors—balanced by model deviance and parsimony (Akaike's information criterion (AIC) score; Burnham and Anderson 2002). The foothills provided a sample size too small for mark–recapture analysis. Because wolverine detectability was the same in each study area (see Results), we could assume the ratio of detected animals inside and outside the Willmore approximated the ratio of total animals in these two areas and applied MacKenzie and Kendall's (2002) equation, which estimates relative abundances by adjusting for detection probabilities from occupancy models. In both cases, we divided the abundance estimate from this model by the estimated effective sampling area (e.g., Williams et al. 2002), calculated in GIS (ArcGIS version 9.3; ESRI, Inc., Redlands, California, USA) by buffering points in the sampling array with a 100 km² circle, approximating half a mean adult wolverine home range in Canadian mountain landscapes (Banci 1987, 1994).

Landscape quantification

Landscape composition (habitat availability) was quantified using a LandSat thematic-mapped GIS land-cover data set incorporating a digital elevation model, with a habitat-identification algorithm that classified 16 land-cover types (McDermid et al. 2009). Eight natural land-cover variables occurred sufficiently often in the study area to allow modelling: closed conifer forest, moderate conifer forest, open conifer forest, mixedwood forest, open wetland, upland shrubs, upland herbaceous habitats, and regenerating areas (for descriptions see McDermid et al. 2009). We calculated a topographic ruggedness index (TRI; Riley et al. 1999) based on a 25 m digital elevation model data from the Alberta Base Data set. Seismic line density (km/km²) obtained from government digital map inventory was used as a surrogate for anthropogenic habitat alteration and human activity. Seismic lines mark current and past oil and gas exploration, are correlated with current industrial activity (wellpads, drill sites, and pipelines), and provide recreational motorized access. They are also spatially

Fig. 2. Presence (triangles) and absence (circles) of wolverines (*Gulo gulo luscus*) at 120 hair-trapping stations in the Front Ranges, Main Ranges, and Foothills of the Rocky Mountains of west-central Alberta, Canada. The protected area of the Willmore Wilderness (black border) has two large burns (grey patches), whereas the landscape outside is a mosaic of trails and off-road motorized access, seismic lines for oil and gas development (thin lines), roads (thick lines), and forest harvesting (grey patches).



extensive, so lend themselves to modelling habitat alteration at large spatial scales. We used ArcGIS version 9.3 Spatial Analyst, spatial analysis routines, and the Regional Analysis function of Patch Analyst to calculate the percentage of each variable within a 5000 m radius buffer (78.5 km^2) around each sampling site. This area produces best-fit models for wolverines among a range of scales, and although some overlap among buffers exists, there is no evidence of inflation of type I error or biased estimates (Fisher et al. 2011).

Hierarchical occupancy modelling

Species detection is often imperfect and decreases with increasing rarity (MacKenzie et al. 2005, 2006). Species occupancy at a site (ψ) can be modelled in conjunction with its probability of detection (p): the probability of detecting that species if present (MacKenzie et al. 2006). If wolverine p differed between the design constraints (avalanche vs. trail), this might confound the habitat selection analysis. To ensure that data from across the entire study area could be reliably combined in generalized linear models for the habitat selection analysis, we tested whether p varied among design constraints, or through time, and whether significant landscape predictors of wolverine occupancy would mirror those from generalized linear models. We used custom single-season hierarchical occupancy models in software PRESENCE version 4.9 (Hines 2006). Detection histories comprised monthly wolverine detections and nondetections at each site, repeated across 3 months. Models assumed ψ was either constant, or varied with topographic ruggedness, seismic-line density, regenerating fire and cutblocks, or a combination of ruggedness and seismic-line density. Models further assumed that p was either constant, or differed among sampling constraints, or through time, or a combination of these. We ranked models by AIC weights and calculated evidence ratios (ERs) to weigh support for each covariate. From per-survey estimates of p, we calculated the probability of false absence (pfa) for a given survey duration as $(1 - p)^t$ (Long and Zielinski 2008), with $t = 3$ independent surveys.

Habitat selection

We used generalized linear models to test hypotheses about wolverines' relationship to landscape composition, since these are more flexible than occupancy models for this purpose. Because there were no differences in detectability among sites, the response variable was the frequency of monthly wolverine detections and nondetections at each site (0–3), across the study area. Habitat selection varies with habitat availability, and we had no a priori hypotheses about wolverines' relationships with natural landscape features in this landscape, with the exception of regenerating areas. To reduce the seven nondisturbance land-cover variables for model selection (Burnham and Anderson 2002), we used generalized linear models (Poisson errors, log link; *R* version 2.14.2) and the minimum adequate model approach (Crawley 2007) to identify which land-cover variables best explained wolverine occurrence data. The percentage of mixedwood forest was the only significant land-cover predictor. We additionally retained the "regenerating areas" variable—which included burned and harvested areas greater than ~10 years old—to test hypotheses about disturbed habitat. We then formulated 12 competing hypotheses about the importance of elevation, landscape ruggedness,

Fig. 3. Occurrence of wolverines (*Gulo gulo luscus*) was sampled with noninvasive genetic tagging via hair sampling in the Rocky Mountains of Alberta, Canada. Hair traps consisted of a tree loosely wrapped with barbed wire, baited with a large skinned beaver carcass and scent lure. Cameras placed on traps showed this method was effective at detecting wolverines.

mixedwood forest cover, seismic-line density, and regenerating areas in explaining wolverine occurrence (Table 1). We ranked models based on AIC scores and normalized AIC weights (which describe the weight of evidence in support of each model; Burnham and Anderson 2002). We summed AIC weights and calculated ERs (Anderson 2008) to summarize the overall importance of each variable in explaining wolverine occurrence; ER = 2 suggests there is twice the evidence for inclusion of an explanatory variable than its exclusion. We averaged the parameter estimates of the top models using *R* package MuMIn (Bartón 2012).

Results

Abundance

We identified 26 wolverines within the Willmore Wilderness Park (12 males, 14 females, at 66 sites), with overlapping space use (Fig. 4). The M_t model had low AIC score and low SE (1.3), estimating 27.2 wolverines. However, wolverine capture probability was heterogeneous and varied through time, thus fitting the M_{th} Chao model assumptions (Table 2), which estimated 28 wolverines (SE = 2.2) in this protected area. Other models' assumptions were unsupported by data, had higher AIC scores, or produced imprecise parameter estimates (Table 2). With 28 wolverines in an effective sampling area of 4140 km², we estimated density as 1 wolverine/148 km², or 6.8 wolverines/1000 km².

In the developed landscape to the east of the Willmore Wilderness, we detected five wolverines in year 1 (two males, three females, at 54 sites). Following pipeline installation through some sites, only three of these were detected in year 2 (Fig. 4). We estimated seven wolverines in this landscape in 2004–2005 and four wolverines in 2005–2006. With an effective sampling area of 2334 km² in 2004–2005, we estimated density as 1 wolverine/333 km², or 3 wolverines/1000 km². We sampled 2260 km² in 2005–2006 and estimated density as 1 wolverine/565 km², or 1.8 wolverines/1000 km².

Occupancy and probability of detection

Wolverine detectability did not vary between the two sampling design constraints. There was little evidence that *p* varied with sampling constraint alone (ER = 0.03) or with a combination of survey period and sampling constraint (ER = 0.37; Table 3). This evidence indicates that wolverines were equally detectable, when present, regardless of whether the systematic design was constrained by avalanche or road access. Equal detectability among sites justifies the use of combined data across the entire study area within species distribution models for habitat selection analysis. There was some evidence that the probability of wolverine detection increased January through March (ER = 1.22; Fig. 5). The best-supported model suggests that after 3 months of hair-trap surveys, there was an 87% probability of correctly assigning a site as occupied via hair-trapping (pfa = 0.13). Accounting for *p*, wolverines were more likely to occupy sites with increasingly rugged topography (ΣAIC weights = 0.70, ER = 2.31). There was weak evidence that occupancy varied with both ruggedness and seismic-line density (ΣAIC weights = 0.30, ER = 0.43). There was no evidence that wolverine occupancy varied with amount of regenerating area after timber harvest for fire (ER = 0.0).

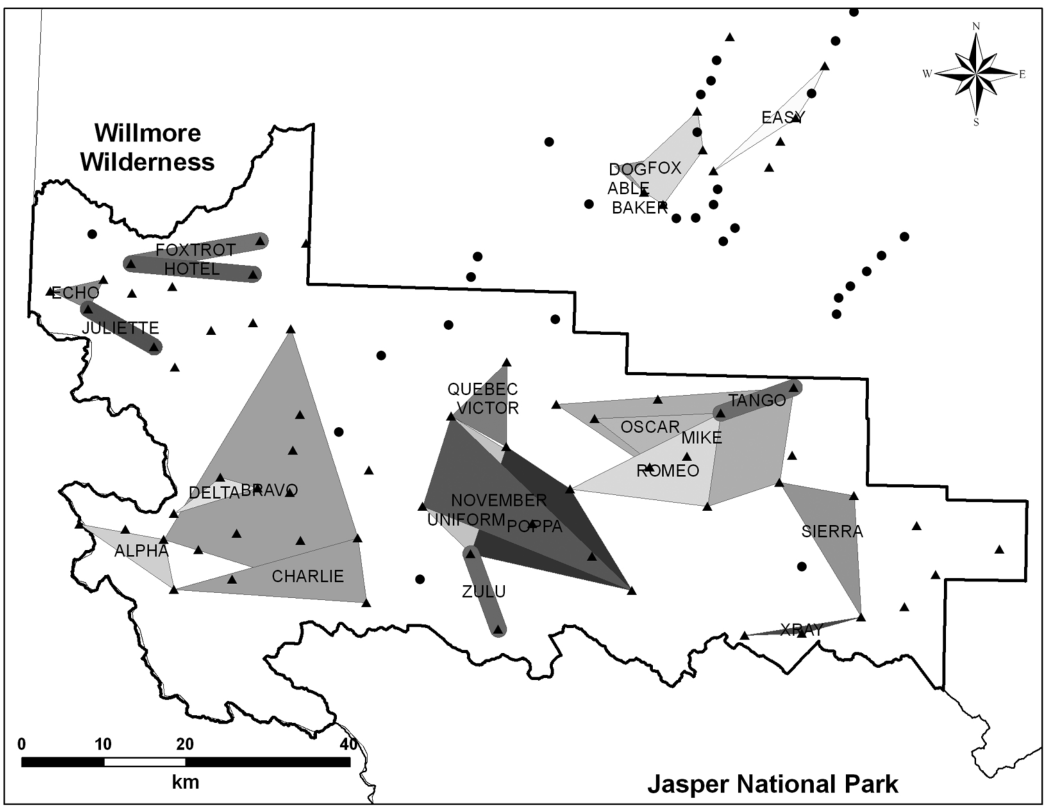
Habitat selection

Wolverines were more likely to occur in more topographically rugged terrain and areas where industrial activity and habitat

Table 1. Hypotheses about association of wolverines (*Gulo gulo luscus*) with features of the Alberta landscape and the corresponding models used to assess the explanatory variables.

Model	Hypothesis: wolverine occurrence is predicted by
1	Global model: proportion of mixedwood forest cover, proportion of regenerating areas, seismic-line density, landscape ruggedness, and sample-site elevation
2	Mixedwood forest cover, regenerating areas, seismic-line density, and landscape ruggedness
3	Mixedwood forest cover, regenerating areas, and seismic-line density
4	Mixedwood forest cover and regenerating areas
5	Regenerating areas only
6	Regenerating areas and seismic-line density
7	Regenerating areas, seismic-line density, and landscape ruggedness
8	Seismic-line density and landscape ruggedness
9	Landscape ruggedness only
10	Mixedwood forest and landscape ruggedness
11	Mixedwood forest and seismic-line density
12	Mixedwood forest, seismic-line density, and landscape ruggedness

Fig. 4. Minimum convex polygons (MCP) of "spatial detection ranges" of wolverine (*Gulo gulo luscus*) individuals (identified by names) detected at >1 site in the Main Ranges, Front Ranges, and Foothills of the Rocky Mountains of west-central Alberta, Canada. Twenty-six wolverines were detected within the Willmore Wilderness Park (black border); outside the Park, we detected 5 wolverines in 2004–2005 (shown) and only 3 of these again in 2005–2006. Wolverines were detected but not identified, or detected only once, at triangles outside MCPs and undetected at circles.



alteration was low. Wolverine occurrence was negatively related to seismic-line density (ER = 499) and was positively related to landscape ruggedness (ER = 61.5) (Table 4). Regenerating areas was related to wolverine occurrence (ER = 249), but this relationship is more difficult to decipher. The parameter estimate for REGEN was unstable in the multivariate model; it was negative in the single-variable model, but positive in the multi-variable model (Table 5), since regenerating areas and ruggedness were negatively correlated (see Caveats). Additional variables did not sufficiently improve explanatory power to warrant the penalty for an added parameter (Arnold 2010).

Discussion

Wolverine abundance differed between landscapes

The rugged, undeveloped end of the study area had 2–3 times the wolverine density of the less-rugged, developed end. By comparison, with 80% of the spatial effort (but twice the temporal effort) we identified only five wolverines outside the undeveloped Willmore Wilderness. Wolverine densities vary widely across western North America, ranging from 3 to 20 wolverines/1000 km², depending on location, trapping pressure, and habitat quality (Hornocker and Hash 1981; Banci and Harestad 1990; Lofroth and Krebs 2007; Golden et al. 2007; Inman et al. 2012). Many of these

Table 2. Estimated abundance of wolverines (*Gulo gulo luscus*) in the Rockies of west-central Alberta, based on Rcapture models with flexible assumptions of (*i*) no variation in hair-trap capture probability among individuals, M_0; (*ii*) variation among individuals only, M_h; (*iii*) variation through time, M_t; (*iv*) variation in time and individuals, M_{th}; (*v*) behavioural variation resulting in a trap effect, M_b; and Chao's (1987), Darroch et al.'s (1993), and Poisson (Rivest and Baillargeon 2007) model variants.

Model	Abundance estimate	SE	Model deviance	df	AIC score
M_0	27.8	1.7	17.09	5	39.72
M_t	27.2	1.3	6.58	3	33.21
M_b	51.7	38.0	4.36	4	28.99
M_{bh}	35.0	23.6	3.98	3	30.61
M_{th} Chao	28.2	2.2	4.75	2	33.38
M_{th} Darroch	33.2	10.9	4.75	2	33.38
M_{th} Poisson	30.0	4.7	4.75	2	33.38

Note: SE, standard error; df, degrees of freedom; AIC, Akaike's information criterion.

estimates are now 20–30 years old and none examines density estimates across landscapes with a marked gradient of habitat alteration. Our estimated 6.8 wolverines/1000 km² is similar to neighbouring British Columbia (6.2 wolverines/1000 km²; Lofroth and Krebs 2007) and Yukon (5.6 wolverines/1000 km²; Banci and Harestad 1990). The estimate of 2–3 wolverines/1000 km² is lower than most estimates from western North America, except for recent estimates from Montana (3.5 wolverines/1000 km²; Inman et al. 2012). The low density was unexpected, since wolverine populations have supported trapping throughout this region in past decades (Poole and Mowat 2001). Density differences inside and outside the protected area should be considered in the context of their close proximity (Fig. 3), which are <10–20 km apart in some places—much closer than wolverine home-range movements.

We used a standard method for estimating effective sampling area, but newly developed hierarchical models—which model encounter rates on spatial capture arrays as a basis for estimating effective sampling area—provide density estimates that sometimes differ from standard methods (Gardner et al. 2009). A hierarchical density estimator may have changed our conclusions if wolverine densities had differed only slightly inside and outside the park; however, the magnitude of the differences that we observed lends strong support to our conclusions.

Wolverines occupied rugged and undeveloped sites

Wolverines were more likely to occur at sites with rugged topography and low anthropogenic footprint. Similarly, May et al. (2006) found that Scandinavian wolverine home-range locations were better predicted by human infrastructure than by habitat. Wolverines avoid roads and other human development in British Columbia (Krebs et al. 2007), Norway (May et al. 2008), Idaho (Copeland et al. 2007), Montana (Carroll et al. 2001), and throughout the northwestern United States (Rowland et al. 2003). Inferences from range retractions coinciding with European colonization may also suggest wolverines are sensitive to human development at continental scales (Laliberte and Ripple 2004; Aubry et al. 2007).

We used seismic lines as an indicator of anthropogenic landscape alteration that causes habitat fragmentation and loss of forest canopy. Fragmentation is not synonymous with a barrier effect, as wolverines often cross these linear features (J.T. Fisher, unpublished snow-tracking data). Fragmentation can, however, alter ecological processes that indirectly affect species' distributions. We hypothesize that interspecific interactions play a role. Wolverines have a broad prey base (Hornocker and Hash 1981; Banci and Harestad 1990; Lofroth et al. 2007) including caribou neonates (Gustine et al. 2006), but reproductive rates are driven by winter availability of ungulate carcasses (Persson 2005). Anthropogenic activity may provide predation refuges for ungulates (e.g., Muhly et al. 2011) thereby reducing carcass availability. Alternatively, competition among carnivores may increase with fragmentation and human activity; seismic lines can alter movement by wolves (*Canis lupus* L., 1758), increasing encounter rates with other species and predation rates (James and Stuart-Smith 2000; Whittington et al. 2005; McKenzie et al. 2012), a factor implicated in the declines of Alberta woodland caribou (*Rangifer tarandus caribou* (Gmelin, 1788)) (e.g., Sorensen et al. 2008; Schneider et al. 2010). Seismic lines may therefore increase competition or intraguild predation for wolverines. In Scandinavia, wolves and Eurasian lynx (*Lynx lynx* (L., 1758)) are important influences on wolverine habitat selection (Mattisson et al. 2011*a*, 2011*b*; van Dijk et al. 2008*a*, 2008*b*). However, interspecific processes have never been examined in the markedly more predator-diverse North American landscape, where wolverines coexist with multiple ursid, canid, felid, and large mustelid species (Fisher et al. 2011); this remains a significant gap.

Habitat alteration and accompanying human activity may degrade habitat quality and depress naturally late-onset reproduction, low reproductive rates, juvenile survival, and population growth rates (Banci and Harestad 1988; Krebs et al. 2004; Persson et al. 2006). Low adult survival in harvested populations (Krebs et al. 2004) shows that anthropogenic mortality is typically additive, often leading to population declines (Lofroth and Ott 2007; Dalerum et al. 2008). Natural predation on wolverines is also higher in trapped than untrapped landscapes (Krebs et al. 2004). Human activity may therefore increase mortality through increased natural or human predation; alternatively denning and rearing areas may be abandoned owing to perceived risk. Whatever the mechanism, we show that the probability of wolverine occurrence decreases across a gradient of increasing anthropogenic landscape development.

Wolverine occurrence also increases with topographic ruggedness, where there is a combination of low- and high-elevation habitats. Bighorn sheep (*Ovis canadensis* Shaw, 1804) (Festa-Bianchet 1988), mule deer (*Odocoileus hemionus* (Rafinesque, 1817)) (D'Eon and Serrouya 2005), and other ungulates winter at lower elevations; in Scandinavia, wolverines showed significant selection for lower-elevation habitats during winter months (Landa et al. 1998). It is possible that wolverines require lower elevations for foraging and higher elevations for predation refuge. Persistent spring snow cover has been hypothesized as important (Schwartz et al. 2009; Copeland et al. 2010) but is not a good predictor at this scale, since spring snow cover was sufficiently persistent across our study landscape to prevent modelling but wolverine occurrence still varied. Finally, rugged areas may offer more den sites in steep, snow-covered slopes with large talus boulders (Magoun and Copeland 1998) and such den sites may be limiting factors for breeding females. However, wolverines also den in flatter landscapes in lower foothills, boreal forest, and arctic tundra.

Caveats

Wolverine detectability was imperfect and varied through time. For large mobile organisms, detectability is affected by movement in and out of sites that is assumed to be non-Markovian (Mackenzie et al. 2006). Variable wolverine detectability could result from changes in mobility owing to snow conditions or female denning (which occurs in this period). Understanding the relationship between mobility and detectability is an ongoing area of research. Notably, if wolverine detectability had differed among design constraints—avalanche risk vs. trail access—then estimates from species distribution models could be affected. In fact, all evidence from occupancy models showed that there was no effect of design constraint on wolverine detectability, indicating that the results observed from habitat selection models (which used the same data) were due to ecological signal, not an artefact of sampling, providing confidence in the conclusions.

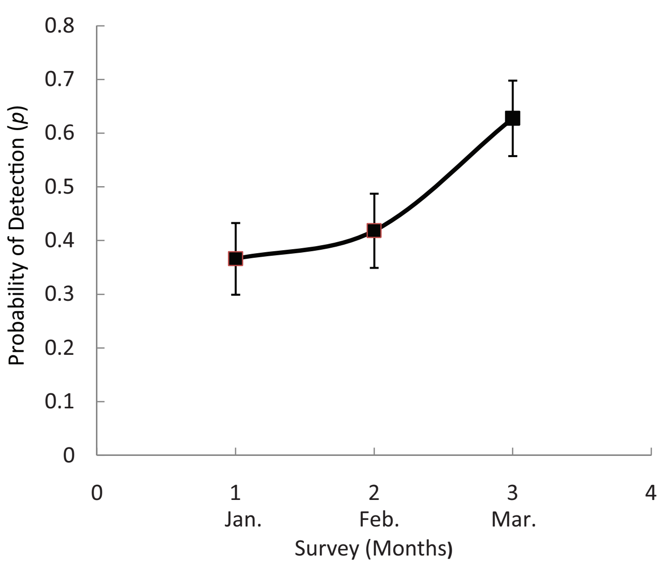
Table 3. Selection of wolverine (*Gulo gulo luscus*) occupancy models in west-central Alberta.

Model	AIC	ΔAIC	AIC weight	Model likelihood	No. of parameters*	−2(log likelihood)
ψ(RUGGED),p(SURVEY)	298.96	0.00	0.38	1.00	5.00	288.96
ψ(RUGGED),p(SURVEY+CONSTRAINT)	299.71	0.75	0.26	0.69	6.00	287.71
ψ(RUGGED+SEISMIC),p(SURVEY)	300.56	1.60	0.17	0.45	6.00	288.56
ψ(RUGGED+SEISMIC),p(SURVEY+CONSTRAINT)	301.50	2.54	0.11	0.28	7.00	287.50
ψ(RUGGED),p(.)	303.79	4.83	0.03	0.09	3.00	297.79
ψ(RUGGED),p(CONSTRAINT)	304.54	5.58	0.02	0.06	4.00	296.54
ψ(RUGGED+SEISMIC),p(.)	305.41	6.45	0.02	0.04	4.00	297.41
ψ(RUGGED+SEISMIC),p(CONSTRAINT)	306.36	7.40	0.01	0.02	5.00	296.36
ψ(SEISMIC),p(SURVEY+CONSTRAINT)	315.83	16.87	0.00	0.00	6.00	303.83
ψ(SEISMIC),p(SURVEY)	318.38	19.42	0.00	0.00	5.00	308.38
ψ(.),p(SURVEY+CONSTRAINT)	319.44	20.48	0.00	0.00	5.00	309.44
ψ(SEISMIC),p(CONSTRAINT)	320.31	21.35	0.00	0.00	4.00	312.31
ψ(REGEN),p(SURVEY+CONSTRAINT)	320.42	21.46	0.00	0.00	6.00	308.42
ψ(SEISMIC),p(.)	323.10	24.14	0.00	0.00	3.00	317.10
ψ(.),p(CONSTRAINT)	323.77	24.81	0.00	0.00	3.00	317.77
ψ(REGEN),p(CONSTRAINT)	324.80	25.84	0.00	0.00	4.00	316.80
ψ(REGEN),p(SURVEY)	352.11	53.15	0.00	0.00	5.00	342.11
ψ(.),p(SURVEY)	356.05	57.09	0.00	0.00	4.00	348.05
ψ(REGEN),p(.)	356.80	57.84	0.00	0.00	3.00	350.80
ψ(.),p(.)	360.67	61.71	0.00	0.00	2.00	356.67

Note: Occupancy (ψ) could be constant (.), vary with topographic RUGGEDness, SEISMIC line density, or REGENerating forest fires and cutblocks within a 5 km radius. Probability of detection (p) could differ by sampling design CONSTRAINTs or among SURVEYs.

*Number of estimated β parameters in the model.

Fig. 5. Wolverines (*Gulo gulo luscus*) were imperfectly detected via hair trapping in the mountain landscape of west-central Alberta, Canada. The probability of detecting wolverines, when present at a site, increased monthly from Dec. through Mar. After three surveys, the probability of false absence was reduced to ~13%. Bars represent standard errors.



After accounting for differences in land cover and topography, developed landscapes with human activity resulted in fewer wolverines across this natural and anthropogenic gradient spanning 30 individuals and an area in excess of 6000 km².

Topography and habitat alteration are unavoidably correlated on this edge of wolverines' distribution. Rugged areas are less likely to be developed, and topographic ruggedness was negatively correlated with both seismic-line density (Pearson's r = −0.765, $p < 0.0001$) and regenerating areas (Pearson's r = −0.503, $p < 0.0001$). Sampling design could not avoid this correlation, as no large tracts of undeveloped areas remain in subalpine and foothills landscapes (Fig. 2), and the alpine remains primarily undeveloped. This begs the question: is topography masking some signal from anthropogenic development, or vice versa? Generalized linear modelling provided strong evidence that this correlation does not obfuscate the signal that we detected, as the effects of seismic-line density and regenerating areas remained even after accounting for topographic ruggedness (model 9 vs. model 7, ΔAIC = 17.33; Table 5). If otherwise, model ΔAIC scores would be smaller, and relative support for either the habitat alteration or the ruggedness models weaker, as they share variance. Instead, ΔAIC and evidence ratios are high—strong support for including both seismic-line density and topography in the model. Hierarchical models provided similar evidence, though the effect of habitat alteration was weaker because some of the variance was attributed to temporal changes in detectability. The response of wolverines to regenerating areas requires more investigation, as multicollinearity among variables changed the direction of this relationship in our models.

Implications for wolverine landscape ecology

Wolverine occurrence decreases with increasing anthropogenic landscape development at this range margin, and wolverine density changes very abruptly. Alone, the 30 wolverines in the protected landscape would not likely persist long term (e.g., Reed et al. 2003; Traill et al. 2010), but Alberta wolverines' high genetic variability indicates that they are connected to, and exchanging DNA within, a larger population (Kyle and Strobeck 2001, 2002). However, connectivity may prove detrimental. To the west, wolverines are overharvested and in decline (Lofroth and Ott 2007) and are subject to anthropogenic habitat loss (Krebs et al. 2007). If in addition anthropogenic habitat alteration at the eastern range margin creates a population sink (sensu Pulliam 1988; Pulliam and Danielson 1991), together these may result in population decline. Moreover, though Rocky Mountain wolverine densities are (comparatively) high, density does not equal quality (Wheatley et al. 2002); Brøseth et al. (2010) suggest wolverine population growth rates can decrease as density increases.

We have shown a large-scale spatial correlation between wolverine occurrence and habitat fragmentation on this edge of their range. If fragmentation is altering ecological processes resulting in reduced wolverine distribution and wolverine declines, then identifying the mechanisms responsible should be the next target for investigation. As wolverines exist at very low densities, and

Table 4. Selection of wolverine (*Gulo gulo luscus*)–habitat models in west-central Alberta.

Model	Habitat variables	Residual deviance*	Residual df	AIC score	ΔAIC	AIC weight
7	REGEN+SEISMIC+RUGGED	90.4	116	220.23	0	0.539
2	MIXED+REGEN+SEISMIC+RUGGED	89.5	115	221.32	1.09	0.312
1	MIXED+REGEN+SEISMIC+RUGGED+ELEV	89.2	114	223.02	2.79	0.133
3	MIXED+REGEN+SEISMIC	98.4	116	228.19	7.96	0.010
6	REGEN+SEISMIC	103.8	117	231.67	11.44	0.002
12	MIXED+SEISMIC+RUGGED	102.0	116	231.79	11.56	0.002
8	SEISMIC+RUGGED	104.5	117	232.32	12.09	0.001
10	MIXED+RUGGED	105.1	117	232.90	12.67	0.001
11	MIXED+SEISMIC	108.2	117	236.05	15.82	0.000
9	RUGGED	111.7	118	237.56	17.33	0.000
4	MIXED+REGEN	119.7	117	247.49	25.52	0.000
5	REGEN	152.9	118	278.75	27.26	0.000

Note: AIC, Akaike's information criterion. MIXED is the proportion of area in mixedwood (co-dominant deciduous and coniferous); REGEN is the proportion of area regenerating (fires and cutblocks <20 years old); SEISMIC is the seismic-line density in kilometres of seismic line per square kilometre of area; RUGGED is the topographic ruggedness index; ELEV is the elevation of the sample site (metres above sea level). Wolverine occurrence counts were modeled against GIS habitat data measured at a 5000 m radius using generalized linear models. The best-supported model suggests regenerating areas, seismic-line density, and topographic ruggedness best explain wolverine occurrence.

*Null model deviance is 161.7 on 119 degrees of freedom (df).

Table 5. Estimated β parameters from wolverine (*Gulo gulo luscus*) species distribution models.

Model	Parameter	Estimate	SE	*p*	RVI*
7, 2 averaged	Intercept	−3.074	1.082	0.0049	
	REGEN	6.225	1.806	0.0006	1.00
	SEISMIC	−1.874	0.546	0.0007	1.00
	RUGGED	0.002	0.001	0.0013	1.00
	MIXED	−2.692	2.974	0.3704	0.35
5	Intercept	−0.231	0.115	0.0438	
	REGEN	−1.457	0.754	0.0534	

Note: Generalized linear models of wolverine occurrence in foothills and mountain landscapes suggest wolverines were positively predicted by topographic ruggedness and negatively predicted by seismic-line density, regenerating areas, and mixedwood forest. SE, standard error.

*Relative variable importance (RVI) is the sum of AIC weights over all models (Bartón 2012).

over vast areas, and across landscapes with markedly different ecological characteristics and disturbance regimes, multiple inferences from landscape-scale studies will be needed to derive the ecological mechanisms caused by human use of shared landscapes.

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Forest Fragmentation of the Conterminous United States: Assessing Forest Intactness through Road Density and Spatial Characteristics

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Over the past few centuries, widespread disturbance of native forests of the conterminous United States has dramatically altered the composition, structure, extent, and spatial pattern of forestlands (Curtis 1956, Whitney 1994). These forests have been either permanently replaced by other land uses or degraded to varying degrees by unsustainable forestry practices, forest fragmentation, exotic species introduction, or alteration of natural disturbance regimes.

Habitat fragmentation is generally defined as the process of subdividing a continuous habitat type into smaller patches, which results in the loss of original habitat, reduction in patch size, and increasing isolation of patches (Andrén 1994). Habitat fragmentation is considered to be one of the single most important factors leading to loss of native species (especially in forested landscapes) and one of the primary causes of the present extinction crisis (Wilcox and Murphy 1985). Although it is true that natural disturbances such as fire and disease fragment native forests, human activities are by far the most extensive agents of forest fragmentation (Burgess and Sharpe 1981). For example, during a 20-year period in the Klamath–Siskiyou ecoregion, fire was responsible for 6% of forest loss, while clear-cut logging was responsible for 94% (Staus et al. 2001). Depending on the severity of the fragmentation process and sensitivity of the ecosystems affected, native plants, animals, and many natural ecosystem processes (e.g., nutrient cycling, pollination, predator–prey interactions, and natural disturbance regimes) are compromised or fundamentally altered. For many species, migration between suitable habitat patches becomes more difficult, leading to smaller population sizes, decreased gene flow, and possible local extinctions (Wilcove 1987, Vermeulen 1993).

FOREST FRAGMENTATION CAN BE MEASURED AND MONITORED IN A POWERFUL NEW WAY BY COMBINING REMOTE SENSING, GEOGRAPHIC INFORMATION SYSTEMS, AND ANALYTICAL SOFTWARE

As native forests become increasingly fragmented, ecosystem dynamics switch from being predominantly internally driven to being predominantly externally driven (Saunders et al. 1991). Simultaneously, remnant patches become altered by changes within the patches themselves (Chen et al. 1995, Woodroffe and Ginsberg 1998) as the remnants become more and more isolated, thereby resulting in further ecological degradation across the landscape. Declines in forest species as a result of fragmentation have been documented for numerous taxa, including neotropical migrant songbirds

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(Whitcomb et al. 1981, Ambuel and Temple 1983), small mammals (Henderson et al. 1985, Verboom and Apeldoorn 1990), and invertebrates (Mader 1984). Forest fragmentation has also been associated with increased susceptibility to exotic invasion (Rejmánek 1989).

Concern over the widespread negative effects of fragmentation has led to calls for managing ecosystems at a regional scale (Franklin 1993), and it has led researchers to examine spatial patterns over large geographic extents (O'Neill et al. 1997, Jones et al. 1997, Riitters et al. 2000b). Quantitative methods have been developed to compare different landscapes, to identify landscape changes over time, and to correlate landscape pattern to ecological function (Turner 1989). Many indices can be calculated from the spatial patterning of land cover (Urban et al. 1987, Turner 1989, McGarigal and Marks 1995, Schumaker 1996), forming one of the major analytical pursuits of landscape ecology (Forman and Godron 1986).

Krummel and colleagues (1987) and O'Neill and colleagues (1988) examined landscape patterns based on high-altitude aerial photography and US Geological Survey (USGS) quadrangles (at 1:250,000 scale). The indices they chose to examine, which were found to be reasonably independent of one another, captured major features of landscape pattern. More recent assessments utilized a "sliding window" filter to reduce the complexity of the data and to draw out landscape patterns of interest (Jones et al. 1997, Riitters et al. 2000a, 2000b). Most landscape assessments have relied on land cover databases developed from coarse AVHRR (Advanced Very High Resolution Radiometer) satellite imagery (O'Neill et al. 1996, 1997, Loveland et al. 2000, Riitters et al. 2000a, 2000b). Although such assessments remain useful at continental scales, analysis of finer resolution imagery has been recommended when studying smaller geographic areas (O'Neill et al. 1997). Using classified Landsat Thematic Mapper (TM) imagery from National Land Cover Data (NLCD; Vogelmann et al. 1998), researchers have begun to examine spatial pattern at finer resolutions. Jones and colleagues (1997) examined numerous landscape indicators using the data set from NLCD for the mid-Atlantic states, with the primary research focus being water quality, and Riitters and colleagues (1997) employed multiple window sizes to examine landscape patterns of subwatersheds using the data set from NLCD for the Chesapeake Bay watershed.

Our objective was to build a forest fragmentation database for the conterminous United States by utilizing the high-resolution NLCD database, roads, and a series of fragmentation indices that quantify forest landscape patterns. Because of the numerous negative impacts that roads have on native forest ecosystems (Trombulak and Frissell 2000), roads data played a prominent role in the fragmentation assessment. We focused our analysis on forest ecoregions, as defined by the World Wildlife Fund (Ricketts et al. 1999), but we also summarized results at larger regional and national scales. Ecoregions can be defined as relatively large units of land containing a distinct assemblage of natural communities and species, with boundaries that approximate the original extent of natural communities prior to major land use change (Olson et al. 2001). Because of the scope of the project and the lack of complete, uniform data sets, we conducted the analysis without consideration for ownership, forest type, stand age, forest health, or type of disturbance. In this article, after describing the assessment of forest fragmentation, we review the methodology that created this database and some of its potential uses for conservation scientists, restoration scientists, land managers, policymakers, and others. We then offer a review of the strengths and limitations of the database and make recommendations for future modification and research.

Analyzing and mapping forest fragmentation

We used six basic geographic information systems (GIS) data layers from five separate sources: (1) national land cover data based on 30 meter (m) resolution Landsat 5 TM satellite imagery (Vogelmann et al. 1998), (2) USGS 1:100,000 scale roads, (3) US Census Bureau Topologically Integrated Geographic Encoding and Referencing (TIGER) 1:100,000 scale highways and US boundaries, (4) Bureau of Transportation Statistics (BTS) 1:100,000 scale boundaries for urbanized areas with a population of greater than 50,000, and (5) World Wildlife Fund ecoregions (Ricketts et al. 1999). We used the TIGER roads and BTS urban boundaries to define our units of analysis and the data set from NLCD and USGS roads data for the fragmentation analysis.

Choosing the unit of analysis. In general, the better the ecological subdivision of a region, the more sensitive and interpretable any landscape pattern index will be (O'Neill et al. 1996). Of the few ecological assessments that have analyzed large regions, most employed the watershed as the basic unit of study (Jones et al. 1997), which may be a reasonable subdivision for some ecological research questions, particularly regarding effects of land use on aquatic ecosystems. For regional assessments of forest spatial pattern and fragmentation, however, dissecting the landscape by watersheds can be considered to artificially sever intact forest patches and alter analytical results. For example, many forest organisms have no difficulty moving from one watershed to another within the same forest patch, in effect treating watershed boundaries as highly permeable. Roads, however, have been shown to be a significant barrier to movement for many forest organisms. Units of study should be defined according to a significant source of forest fragmentation, such as major roads and highways (Trombulak and Frissell 2000). For example, Anderson and colleagues (1999) used an analytical unit they termed an "ecoblock," which was defined by paved and unpaved roads, railroads, power lines, and bodies of water.

We defined our units of analysis, termed *land units,* using the TIGER highway data (US interstates, US routes, and state and county highways) and the borders of the conterminous United States. We used TIGER highway data instead of USGS highway data to delineate land units, because TIGER data on highways were more complete and up-to-date. Only those areas that were at least 2000 hectares (ha) were included as land units. We decided on 2000 ha after exploring a number of size limits, because this size reduced the amount of land units to a manageable number, yet was sufficiently small in comparison with the average land unit size. We used BTS data to identify and remove urban areas from the analysis, assuming that the amount of intact forest would be minimal in those areas. A final land units GIS data layer was created to which fragmentation analysis results could be linked.

Assessing fragmentation. For the purposes of calculating fragmentation statistics, we combined the 21 potential NLCD classes into two classes: forest (including woody wetlands) and nonforest (including water). Only portions of the largest interstates were delineated in the NLCD data set. Thus, to account for the fragmenting effect of roads, we superimposed a 30 m resolution raster version of the USGS roads data set onto the NLCD forest–nonforest data set. We used the USGS roads data, because this data set presented smaller roads in more detail than did the TIGER roads data set. All forest and nonforest patches smaller than 1 ha were reclassified to match the surrounding land cover type to decrease the number of very small patches and thus the time required for processing data. The resulting land units were at least 2000 ha, did not include urban areas, and contained both forest and nonforest patches that were at least 1 ha in size.

Because highways defined the land units, land unit boundaries did not match up directly with the ecoregion boundaries. In every case, the outermost land unit boundaries extended outside the ecoregion. For most ecoregions, the land unit area was a fairly close approximation of the ecoregion area (see figure 4b). For five ecoregions made up of smaller forest ecoregions surrounded by large nonforest ecoregions, we matched the land unit boundaries to the ecoregion boundaries to avoid skewing the fragmentation results by including large areas of nonforest habitat. These "island" ecoregions (figure 1) were the Great Basin montane forests, Wasatch and Uinta montane forests, Colorado Rockies forests, Arizona Mountains forests, and Madrean Sky Islands montane forests.

We conducted spatial analyses for the conterminous US portion of 39 forest ecoregions, as defined by the World Wildlife Fund (figure 1; Ricketts et al. 1999), 21 in the East and 18 in the West. To quantify landscape patterns, we calculated 33 class-level and 39 landscape-level metrics (or indices) using FRAGSTATS, a software program for analyzing spatial patterns (McGarigal and Marks 1995). Additionally, we calculated road density directly from the 1:100,000 scale USGS roads data set, which included all size classes of roads except for four-wheel drive roads. Results for the 72 indices were then spatially linked back to the land units GIS database. (See box 1 for a list of the attributes associated with each land unit.) Because of the lack of compatible, nationwide data sets for natural fragmentation, such as fire, windthrow, or flooding, we did not attempt to distinguish natural and anthropogenic fragmentation within the land units.

Interpretation of fragmentation results. This GIS data set was designed to help address a wide range of ecological inquiries pertaining to forest fragmentation. As an example, we provide one possible interpretation of the results by combining 5 of the 72 indices using an unweighted additive scoring method. The indices used included road density (kilometers per kilometers squared [km/km^2]); total core area index (percentage of all forest area within a land unit that is considered core area, based on a 90 m edge buffer distance); mean nearest neighbor (the average distance in meters from one forest patch to the nearest forest patch); class area (total amount of forest in hectares within each land unit); and percentage of landscape (percentage of a land unit that is composed of forest). We calculated these five indices for each land unit and aggregated the results by ecoregion using natural breaks. This method, natural breaks, uses the Jenks's optimization method, which identifies breakpoints that minimize the sum of variance within each class and maximize the variance between classes (Jenks and Caspall 1971). In this case, each land unit received a score for each of the five indices, ranging from 1 (highest fragmentation outcome) to 5 (lowest fragmentation outcome). The individual scores were then combined into one composite score for each land unit, ranging from 5 (highest possible level of fragmentation) to 25 (lowest possible level of fragmentation).

Forest fragmentation of the conterminous United States

A total of 19,953 land units (18,659 in the East and 1294 in the West) were delineated, which covered approximately 3.6 million km^2 (2.5 million km^2 in the East and 1.1 million km^2 in the West). The mean area of land units was 13,297 ha for eastern forest ecoregions and 86,851 ha for western forest ecoregions. The number of land units ranged from 9 in the North Cascade Forest (ecoregion 23) to 2777 in the Southern Great Lakes Forest (table 1; ecoregion 36). Slightly over 50% of the forest ecoregions were actually covered by forest, and approximately 33% of the ecoregions were covered by core (or interior) forest, with a 90 m edge buffer distance. The percentage of core area values ranged from 9.8 in the Southern Great Lakes Forest (ecoregion 36) to 68.1 in the Eastern Forest–Boreal Transition (ecoregion 14). The number of forest patches differed considerably between East and West, with nearly four times as many patches in the East as in the West. The mean forest patch size ranged from 21 ha in the Southern Great Lakes Forest (ecoregion 36) to 268 ha in the Central Pacific Coastal Forest (ecoregion 9). The mean forest patch size was approximately 92 ha in the West and 67 ha in the East.

The land unit database was constructed to give users a variety of quantified forest fragmentation results. Summaries could be made over a number of geographic extents, including country, region, biome, state, or ecoregion. For this study, we compiled results at the country (conterminous United States), region (East versus West), and ecoregion levels and included them as separate files in the database. Fragmentation

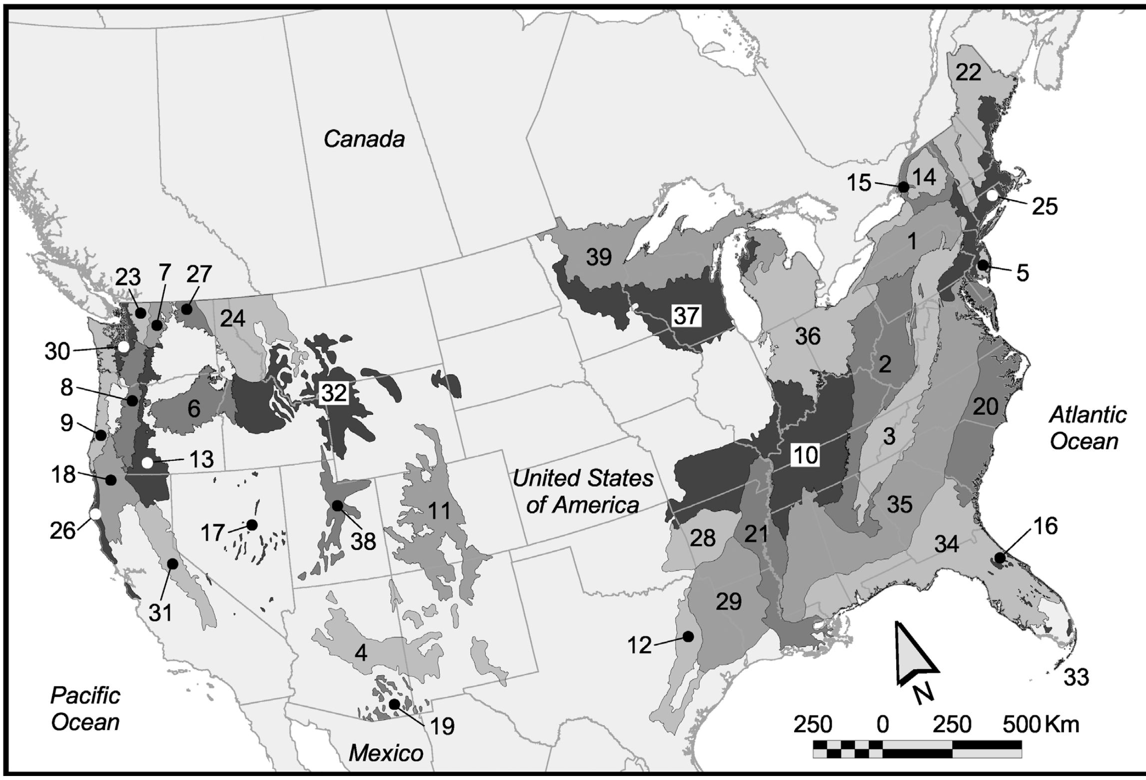

Figure 1. Forest fragmentation was analyzed for 39 forested ecoregions: (1) Allegheny Highland Forest, (2) Appalachian Mixed Mesophytic Forest, (3) Appalachian/Blue Ridge Forest, (4) Arizona Mountain Forest, (5) Atlantic Coastal Pine Barren, (6) Blue Mountain Forest, (7) Cascade Mountain Leeward Forest, (8) Central and Southern Cascade Forest, (9) Central Pacific Coastal Forest, (10) Central US Hardwood Forest, (11) Colorado Rockies Forest, (12) East Central Texas Forest, (13) Eastern Cascade Forest, (14) Eastern Forest/Boreal Transition, (15) Eastern Great Lakes Lowland Forest, (16) Florida Sand Pine Scrub, (17) Great Basin Montane Forest, (18) Klamath–Siskiyou Forest, (19) Madrean Sky Island Montane Forest, (20) Middle Atlantic Coastal Forest, (21) Mississippi Lowland Forest, (22) New England/Acadian Mixed Forest, (23) North Cascade Forest, (24) North Central Rockies Forest, (25) Northeastern Coastal Forest, (26) Northern California Coastal Forest, (27) Okanogan Forest, (28) Ozark Mountain Forest, (29) Piney Wood Forest, (30) Puget Lowland Forest, (31) Sierra Nevada Forest, (32) South Central Rockies Forest, (33) South Florida Rockland, (34) Southeastern Conifer Forest, (35) Southeastern Mixed Forest, (36) Southern Great Lakes Forest, (37) Upper Midwest Forest/Savanna Transition, (38) Wasatch and Uinta Montane Forest, and (39) Western Great Lakes Forest. (See Ricketts et al. 1999 for a discussion of ecoregion.)

metrics summarized for the country using ordinal scores for our five example indices show the national pattern of forest fragmentation (figure 2). In figure 2, it is easy to see the differences in land unit size between East and West, as well as regions in the country where forests appear more intact. Moving east to west, some of the larger, more intact areas include the Northwoods of Maine, Adirondack Park in New York, the Boundary Waters area of northern Minnesota, Glacier National Park and the Bob Marshall Wilderness area of Montana, the Selway–Bitterroot region of Idaho, the North Cascades and Olympic Mountains of Washington, and the Klamath–Siskiyou region of southwest Oregon and northwest California. Higher levels of forest fragmentation can be seen in southern New England; portions of the mid-Atlantic states; the Piedmont of the Southeast; and large sections of Ohio, Indiana, Michigan, Wisconsin, southern Florida, and the Mississippi Valley. All of these examples are located in the eastern United States, where the size of the land units is much smaller than in the West. Some land units in western Wyoming also received low scores, mostly in regions where naturally occurring nonforested lands intermix with forested areas.

Forest fragmentation at the ecoregion level

Although it is useful to consider forest fragmentation at the national level, the strength of the land unit database is realized best when focusing on smaller geographic extents. Examining forest fragmentation at the ecoregion level is particularly important for several reasons. General forest type, ecology, and disturbance histories are far more similar for land units within ecoregions than they are between them. This similarity helps considerably when trying to choose appropriate

Table 1. Summary of results for road density and selected fragmentation metrics for each ecoregion, the western and eastern portion of the study area, and the entire conterminous United States.

Region	Ecoregion area (ha)	Number of land units	Percent forest[a]	Percent core area[b]	Number of forest patches	Mean forest patch size (ha)
Allegheny Highland Forest	7,675,748	602	69.7	46.5	66,514	90
Appalachian Mixed Mesophytic Forest	17,854,294	1,602	76.7	52.8	125,894	123
Appalachian/Blue Ridge Forest	14,827,932	1,301	72.5	50.8	142,238	97
Arizona Mountain Forest	10,330,107	101	48.5	29.9	76,303	65
Atlantic Coastal Pine Barren	825,117	113	54.6	27.0	19,918	28
Blue Mountain Forest	5,898,031	47	48.3	28.6	71,800	83
Cascade Mountain Leeward Forest	1,456,954	17	62.3	39.9	14,914	142
Central and Southern Cascade Forest	4,090,056	65	68.0	46.5	31,894	163
Central Pacific Coastal Forest	3,745,165	88	84.0	62.4	15,401	268
Central US Hardwood Forest	27,580,236	1,886	50.3	28.3	327,957	49
Colorado Rockies Forest	12,283,430	134	58.9	38.5	69,452	105
East Central Texas Forest	5,119,185	445	29.5	10.9	98,364	22
Eastern Cascade Forest	5,045,576	103	37.8	22.9	76,509	75
Eastern Forest/Boreal Transition	2,659,400	78	82.3	68.1	11,236	242
Eastern Great Lakes Lowland Forest	2,374,371	224	66.5	47.4	32,640	93
Florida Sand Pine Scrub	386,176	90	34.0	16.5	21,452	33
Great Basin Montane Forest	534,324	27	48.3	24.5	3,942	63
Klamath–Siskiyou Forest	4,610,238	110	77.1	52.1	32,762	182
Madrean Sky Island Montane Forest	1,097,147	31	21.3	10.1	7,756	28
Middle Atlantic Coastal Forest	12,624,046	1,055	58.2	33.4	126,351	62
Mississippi Lowland Forest	10,690,623	675	25.4	14.5	86,938	42
New England/Acadian Mixed Forest	10,741,731	611	83.2	64.2	41,023	231
North Cascade Forest	1,304,363	9	72.1	49.8	14,775	188
North Central Rockies Forest	9,313,772	71	66.1	46.3	52,617	180
Northeastern Coastal Forest	8,217,277	1,185	69.4	46.5	101,060	77
Northern California Coastal Forest	1,223,314	85	75.5	50.5	13,690	160
Okanogan Forest	1,303,530	38	55.5	32.7	13,359	93
Ozark Mountain Forest	5,836,909	253	67.2	46.9	44,875	105
Piney Wood Forest	13,523,604	957	69.0	46.3	101,365	102
Puget Lowland Forest	1,496,320	163	71.7	48.2	31,865	136
Sierra Nevada Forest	4,889,313	166	46.4	25.4	55,452	71
South Central Rockies Forest	14,530,308	107	37.7	23.4	121,234	77
South Florida Rockland	219,994	17	34.1	12.3	12,573	29
Southeastern Conifer Forest	23,103,750	1,695	53.5	31.0	229,194	59
Southeastern Mixed Forest	32,933,256	2,606	68.4	42.3	268,860	92
Southern Great Lakes Forest	20,178,698	2,777	25.4	9.8	255,440	21
Upper Midwest Forest/Savanna Transition	15,150,620	1,602	31.5	14.4	204,508	28
Wasatch and Uinta Montane Forest	3,817,489	79	54.5	27.8	33,129	62
Western Great Lakes Forest	18,232,102	861	72.0	49.6	106,790	135
Western Conterminous United States	86,969,605	1,294	50.9	31.9	595,252	92
Eastern Conterminous United States	250,755,098	18,659	56.5	35.3	2,100,742	67
Entire Conterminous United States	337,724,703	19,953	54.8	34.3	2,695,994	72

Note: Number of land units for the western, eastern, and entire conterminous United States is less than the sum of land units for each ecoregion because some land units are shared by two or more ecoregions.

a. *Percent forest* is the amount of the entire land unit area that is composed of forest.

b. *Percent core area* is the amount of forest cover composed of core forest area using a 90 m edge effects distance.

fragmentation indices and interpret them in an ecologically meaningful fashion. For example, comparing a deciduous forest type in the eastern United States, which is more likely to be naturally contiguous but heavily disturbed by humans, with a dry conifer forest type in the western United States, which may be naturally patchy and minimally disturbed by humans, can cause serious problems in the interpretation of the calculated results.

Forest fragmentation profiles can be created and compared for each ecoregion. For example, using ordinal scores for our five indices, we generated individual ecoregion fragmentation profiles (figure 3). These histograms were calculated by using the amount of land represented in each cumulative ordinal score class as a percentage of the total land unit area for each ecoregion. Starting from the eastern seaboard (ecoregion 20) and heading west to the final forest ecoregion before the Plains states (ecoregion 10), forest fragmentation profiles show different conditions. Among these five ecoregions, forest fragmentation is high along the coast (ecoregion 20) and in the Piedmont region (ecoregion 35),

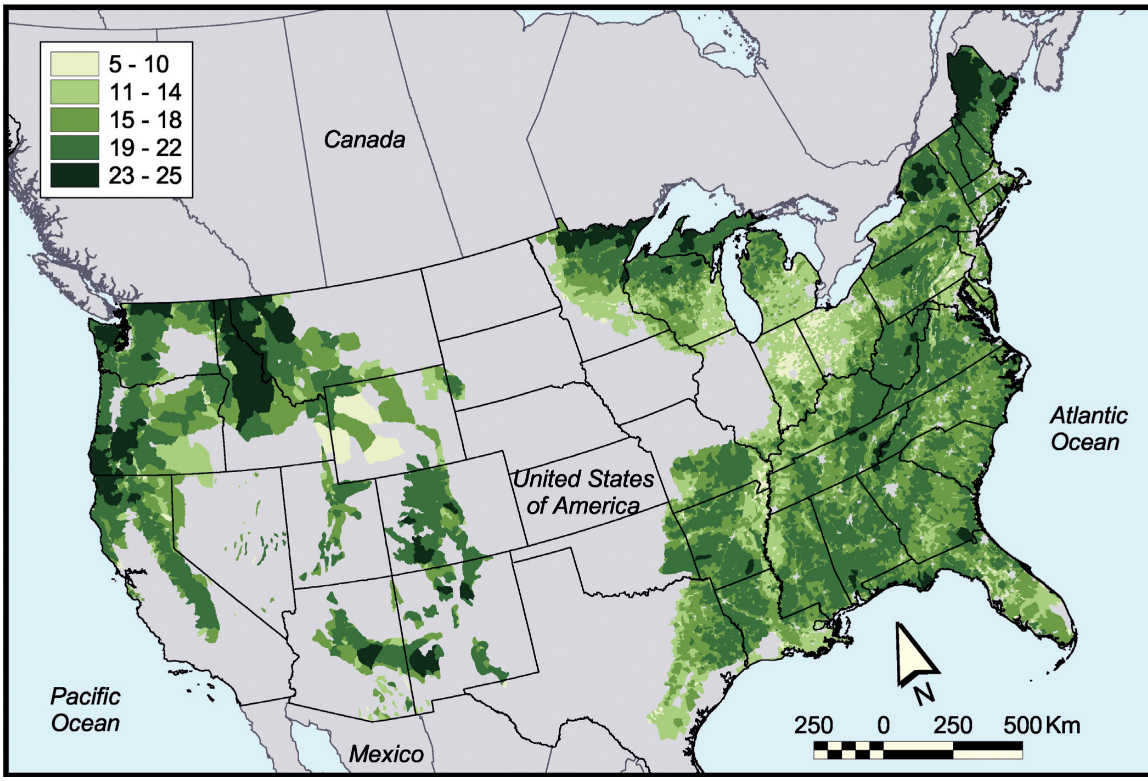

Figure 2. Map of cumulative ordinal scores results for all land units in the conterminous United States. Note that ordinal score ranges were determined using fragmentation results for all land units. Higher scores (darker areas) denote less fragmented forest landunits.

decreases in the Appalachian region (ecoregions 2 and 3), and increases again in ecoregion 10. These are obviously general results; more specific attributes could be tracked within ecoregions over time. One of the strengths of this database and methodology, however, is that it can be replicated cost-effectively as a tool for monitoring forest fragmentation. For example, forest fragmentation is one of nine indicators included in the conservation of biological diversity criteria for the Montréal Process (Montréal Process 1996). The Montréal Process was convened to develop and implement internationally agreed criteria and indicators for the conservation and sustainable management of temperate and boreal forests. Numerous technical challenges regarding the assessment, reporting, and monitoring of identified criteria and indicators still exist. For example, land cover and road data sets are often unavailable, lack appropriate detail, or are outdated. For forest fragmentation, the methodology outlined in this article, or a modified version of it, might serve as a foundation for ongoing monitoring for member nations, particularly where roads are numerous across the landscape. As a parallel process, periodic updates of the underlying data sets would be required to produce a more accurate assessment.

Looking more closely at just one ecoregion (figure 4), the Middle Atlantic Coastal Forest, further observations can be made and the potential utility of the land unit database explored. Figure 4a shows the forest–nonforest land cover upon

Table 2. Data ranges used to determine ordinal ranking for each selected fragmentation metric for ecoregion 20 (Middle Atlantic Coastal Forest).

Fragmentation metric	Ordinal score data range 1	2	3	4	5
Road density (km/km^2)	3.583 – 6.418	2.318 – 3.582	1.740 – 2.317	1.301 – 1.739	0.208 – 1.300
Class area (ha)	153 – 5099	5099 – 11855	11855 – 22977	22977 – 42416	42416 – 77981
Percentage of landscape	7.37 – 31.66	31.67 – 46.41	46.42 – 58.78	58.79 – 71.51	71.52 – 92.78
Total core area index (%)	7.13 – 31.13	31.14 – 43.93	43.94 – 54.08	54.09 – 64.28	64.29 – 86.66
Mean nearest neighbor (m)	145.57 – 285.55	89.46 – 145.56	63.00 – 89.45	45.76 – 62.99	30.00 – 45.75

Note: Ranges were determined using natural breaks classification, which is based on Jenks's optimization method (Jenks and Caspall 1971).

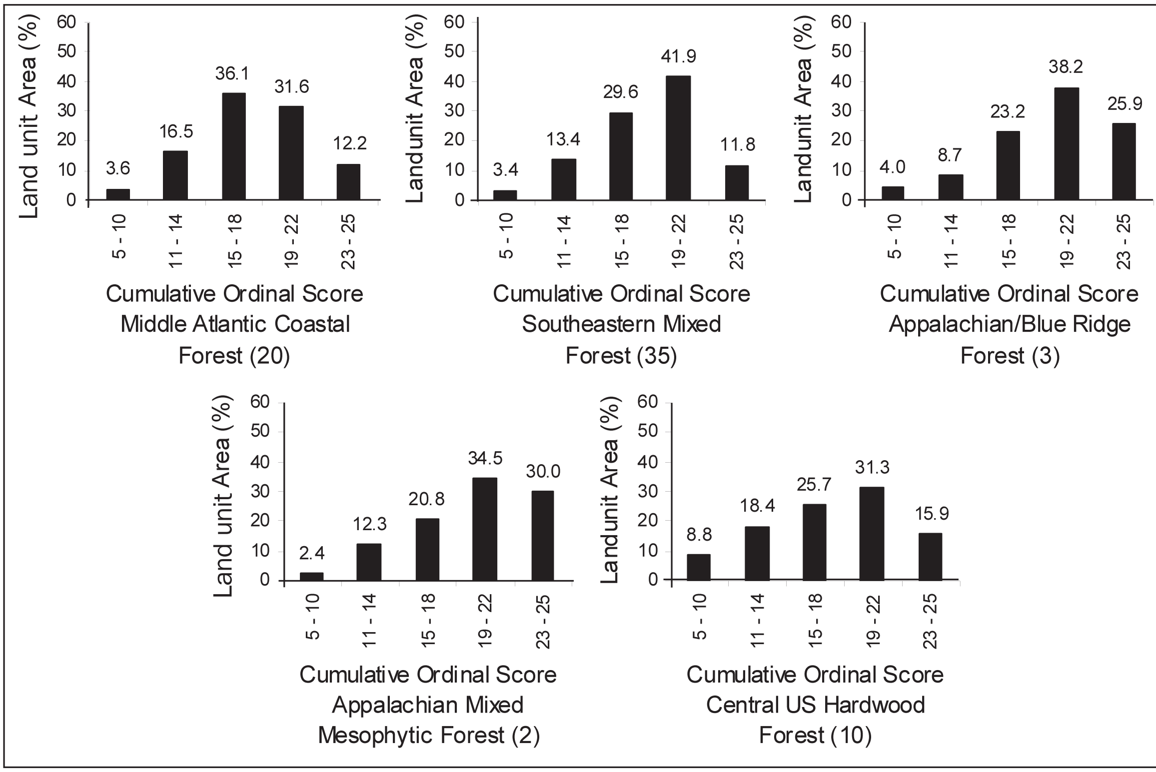

Figure 3. Amount of land represented in each cumulative ordinal score class as a percentage of the total land unit area for each of five eastern US ecoregions. (Please refer to figure 1 for ecoregion locations.)

which fragmentation indices, except road density, were calculated. Figure 4b shows the cumulative ordinal score results for this ecoregion using our five indices. Data ranges for each of the five indices used to determine ordinal ranking are presented in table 2. Note that the range in ordinal scores in figure 2 and figure 4b is identical, but the mapped results of each figure appear very different. This difference is due to differences in scoring within each figure: Figure 2 scores are based on all 19,953 land units in the conterminous United States; the results in figure 4B were generated by scoring only the 1055 land units that made up that particular ecoregion.

Other important features in figure 4b differ from those of figure 2. First, the irregular size and shape of land units is evident. Second, the spatial distribution of the cumulative results provides important information. Most of the higher scoring land units are located along the coast, while lower scoring land units reside in the western half and northernmost portions of the ecoregion. Connected land units of similar score are evident as are isolated, high-scoring land units surrounded by lower scoring land units. It is important to remember that this initial analysis does not distinguish among various forest quality attributes such as native versus plantation or late seral versus early seral forests.

Figure 4c demonstrates an extended utility of the database. This figure shows the cumulative ordinal score results along with existing protected areas taken from a protected areas database (DellaSala et al. 2001). GAP status codes pertain to the USGS GAP Analysis Program, in which "GAP" refers to a geographic approach to planning for diversity (Scott et al. 1994). GAP status 1 and status 2 lands (in blue) are essentially protected from conversion to nonnatural land cover, with GAP 1 lands emphasizing more management to promote native biodiversity and GAP 2 lands emphasizing less. GAP 3 lands (in orange) are also protected from conversion to nonnatural land cover, but they are subject to various extractive uses.

Many of the GAP 1, 2, and 3 protected areas correspond to some of the highest-scoring land units in this ecoregion; however, other high-scoring land units remain outside these existing protected areas. With this information, conservation planners can focus on areas that have more intact forests from which they can design and prioritize conservation activity. For example, the area with high forest intactness between Hofmann State Forest, Bladen Lakes State Forest, and Green Swamp could receive a higher priority for protection as a link between existing protected areas. Planners can gain a perspective on regional forest loss and fragmentation, and possibly forecast future problem areas, once a time-series analysis is completed. By repeating the assessment periodically, changes in forest condition at the regional scale could be tracked with empirical data routinely reported and ongoing management actions updated to reflect current information.

Ecological thresholds

In developing the forest fragmentation data presented in this article, we made no attempt to include known ecological thresholds in the scores. Thus, all scoring was intentionally un-

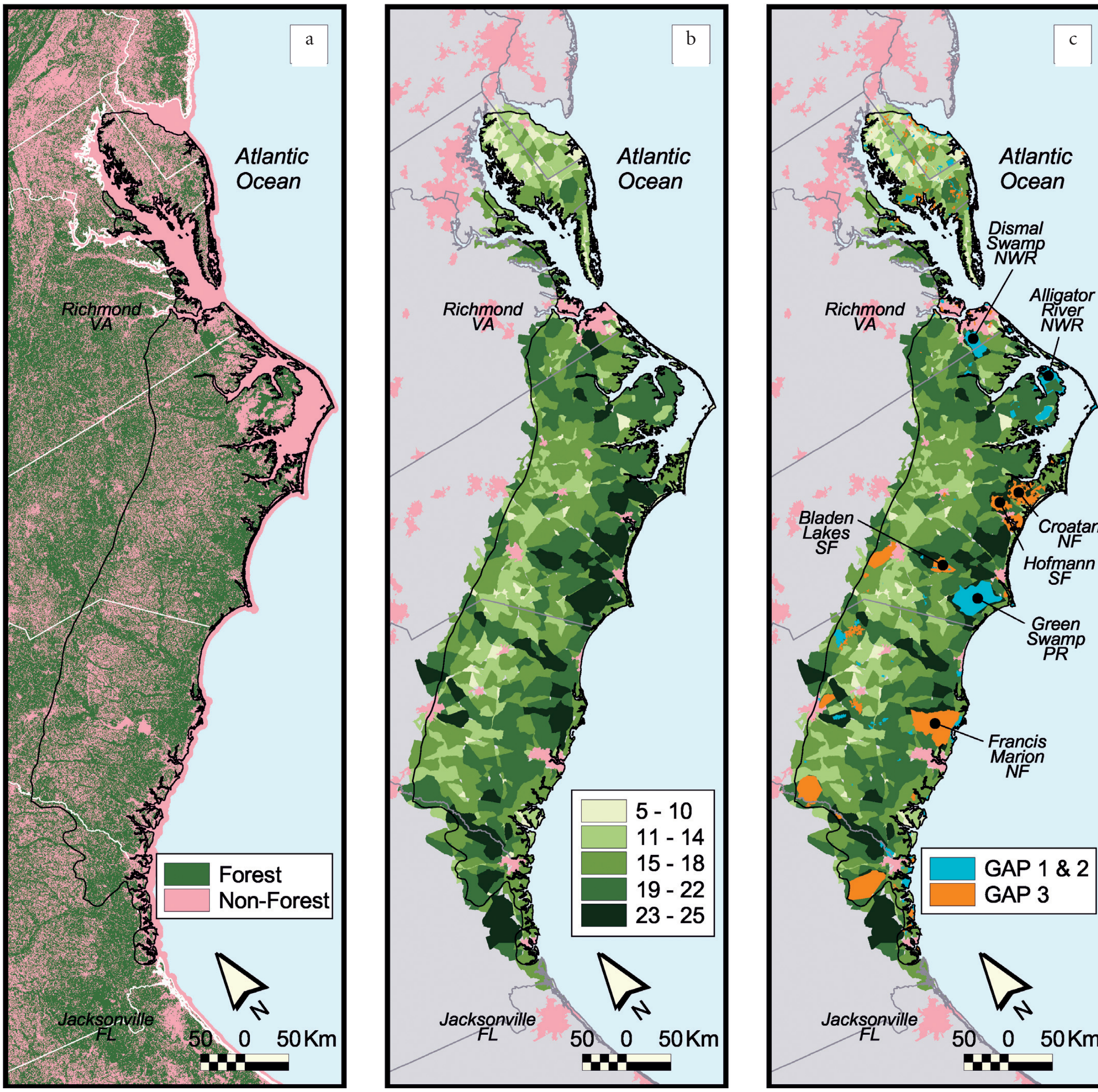

Figure 4. (a) National Land Cover Data reclassified as either forest or nonforest for ecoregion 20 (Middle Atlantic Coastal Forest), with a 30 meter resolution raster version of USGS 1:100,000 scale roads added as nonforest. Forest included coniferous forest, deciduous forest, mixed forest, and forested wetland classes. (b) Cumulative ordinal score results for all land units in ecoregion 20. Please refer to table 2 for the data ranges used to determine ordinal ranks for selected fragmentation metrics. (c) Protected areas for ecoregion 20 overlaying cumulative ordinal score results. GAP status 1 and 2 are lands protected from conversion to nonnatural land cover with greater emphasis on conserving native biodiversity for GAP 1. GAP 3 lands are also protected from conversion to nonnatural land cover, but subject to various extractive uses. For (b) and (c), pale red areas denote cities with a population of at least 50,000 people.

weighted and relative. We did not try to include ecological thresholds because of the general lack of reliable threshold data. However, that does not preclude use of the land units database to address specific conservation issues where ecological thresholds are better understood. For the Middle Atlantic Coastal Forest ecoregion, for example, we offer two different representations of the data (figures 5a, 5b). Figure 5a shows road density scores for each land unit using three natural breaks in the data. The best range for road density was 0.0–1.8 km/km^2 and included the majority of the ecoregion. In comparison, conservation planners in charge of the eastern red wolf *(Canis rufus)* recovery effort, which is centered in and around the Alligator River National Wildlife Refuge (figure 4b), could be concerned about the impact of roads on recovery efforts. Although there has been some variability based on species and geographic location, the scientific literature reports an approximate road density threshold of 0.5 km/km^2 for long-term persistence of wolves (Thiel 1985, Mladenoff et al. 1995). Reviewing road density results for the Middle Atlantic Coastal Forest ecoregion with this ecological threshold tells a very different story than that presented by natural breaks. There are very few places where road density in this area is below the threshold that is required for successful long-term existence for large carnivore populations in the Middle Atlantic Coastal Forest ecoregion, although those areas that do exist are near the wolf recovery area (figure 5b).

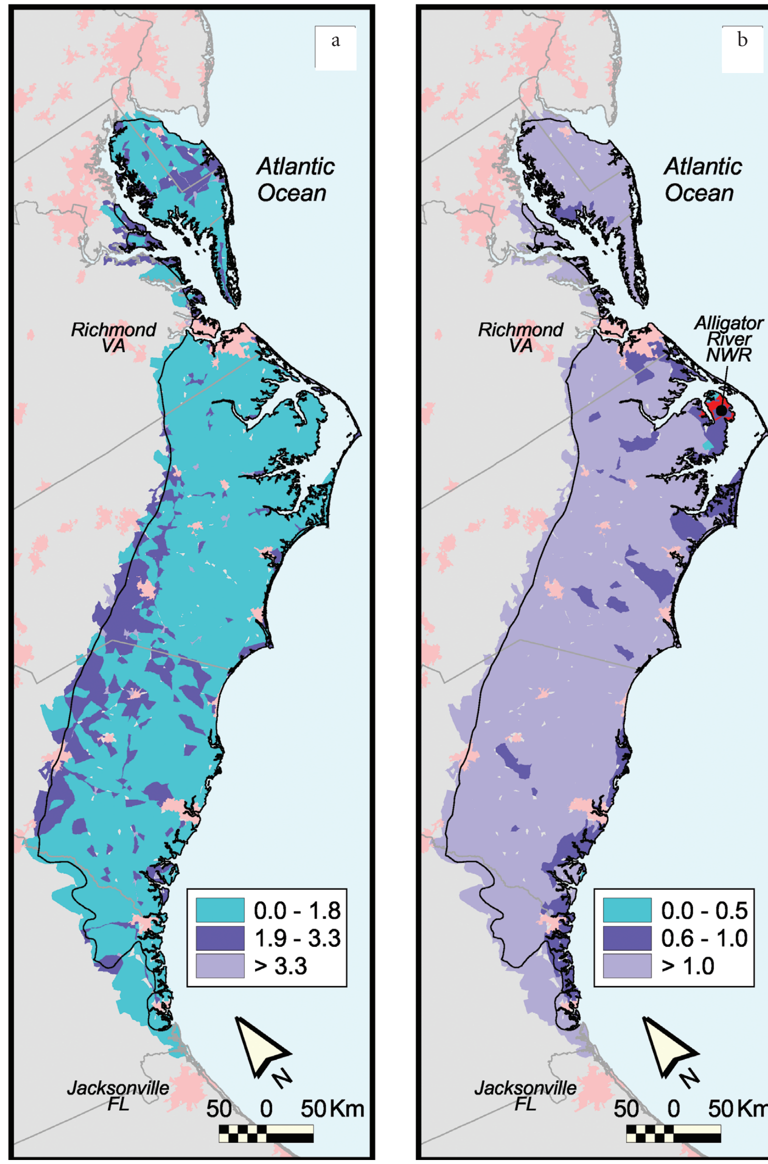

Figure 5. (a) Road density results (km/km^2) for ecoregion 20 (Middle Atlantic Coastal Forest) using natural breaks classification, based on Jenks's optimization method. (b) Road density results (km/km^2) for ecoregion 20 using biologically based classification ranges. Note that for both (a) and (b), the displayed results are actual road density values per land unit and not ordinal score results. For both panels, pale red areas denote cities with a population of at least 50,000 people.

Critical assessment and research recommendations

Roads. The emphasis on roads in the establishment of an analytical unit and as an index for fragmentation is unusual for a forest fragmentation analysis of this scope. Roads have been included in other studies (Jones et al. 1997) but have rarely been so prominent in the research design. In fact, some research efforts have found roads too problematic and have elected to avoid them altogether (Heinz Center 1999). We believe our use of roads is an important contribution and fully warranted by the overwhelming body of scientific literature describing the negative impacts that roads have on natural systems (Trombulak and Frissell 2000). There are other ways to examine roads, but roads are too important to just ignore. There is also an issue of scale, particularly as it applies to roads. The map scale of the roads data used in generating the forest fragmentation database (1:100,000) is reasonable as a first approximation, especially when analyzing such a large geographic extent, but incorporating finer scales (e.g., 1:24,000) is more desirable. We are currently applying the same basic approach described in this article for various subregions around the country using 1:24,000 scale roads data and including additional forest quality information. At this scale, the total length of roads increases roughly 40% for these areas. Furthermore, while there is a fair amount of agreement between scales in terms of roads distribution and concentration, there are examples where the 1:100,000 roads data contained very few roads, but the 1:24,000 scale roads data showed an extensive network.

By using highways, we offer a different approach to dissecting landscapes into ecologically meaningful analytical units. This technique worked particularly well in much of the eastern United States, where the highway network is extensive, by dissecting the landscape into smaller units of analysis. In regions where the road network is less dense, use of highway-defined land units resulted in units of analysis that encompassed areas substantially different than the ecoregion being studied. Addressing this problem in the future may call for using different criteria to define land units, depending on the type of ecoregion being analyzed.

Natural versus anthropogenic disturbance. Another difficult analytical issue pertains to natural forest patchiness. Fragmentation is not always an ecological negative. Natural patchiness is important to many forest types, whether disturbance is caused by large-scale fires or localized wind-

Box 1. General items, ordinal score items, and fragmentation indices for the land units database.

Item	Level	Brief description
AREA	n.a.	Area in square meters
PERIMETER	n.a.	Perimeter length in meters
LANDUNITS#	n.a.	Internal identification number
LANDUNITS-ID	n.a.	User assigned unique identification number
CBILABEL	n.a.	Textual identification
CBICODE	n.a.	Identification (1 = land unit, 2 = non–land unit)
ROAD-DENS-S1	n.a.	Original road density ordinal score
CA-S1	n.a.	Original class area ordinal score
PCT-LAND-S1	n.a.	Original percentage of landscape ordinal score
TCAI-S1	n.a.	Original total core area index ordinal score
MNN-S1	n.a.	Original mean nearest neighbor ordinal score
SUM-S1	n.a.	Sum of all used original ordinal scores
ROAD-DENS-S2	n.a.	Expanded road density ordinal score
CA-S2	n.a.	Expanded class area ordinal score
PCT-LAND-S2	n.a.	Expanded percentage of landscape ordinal score
TCAI-S2	n.a.	Expanded total core area index ordinal score
MNN-S2	n.a.	Expanded mean nearest neighbor ordinal score
SUM-S2	n.a.	Sum of all used expanded ordinal scores
ROAD-LENGTH	n.a.	Total USGS road length in meters
ROAD-LENGTH-KM	n.a.	Total USGS road length in kilometers
TOTAL-SQKM	n.a.	Total land unit area in square kilometers
ROAD-DENS	n.a.	Land unit road density in km/km^2
TYPE	Class	Patch type
CA	Class	Class area
TA	Class	Total landscape area
PCT-LAND	Class	Percentage of landscape
LPI	Class	Largest patch index
NP	Class	Number of patches
PD	Class	Patch density
MPS	Class	Mean patch size
PSSD	Class	Patch size standard deviation
PSCV	Class	Patch size coefficient of variation
TE	Class	Total edge
ED	Class	Edge density
LSI	Class	Landscape shape index
MSI	Class	Mean shape index
AWMSI	Class	Area weighted mean shape index
DLFD	Class	Double log fractal dimension
MPFD	Class	Mean patch fractal dimension
AWMPFD	Class	Area weighted mean patch fractal dimension
C-PCT-LAND	Class	Core area percentage of landscape
TCA	Class	Total core area
NCA	Class	Number of core areas
CAD	Class	Core area density
MCA1	Class	Mean core area per patch
CASD1	Class	Patch core area standard deviation
CACV1	Class	Patch core area coefficient of variation
MCA2	Class	Mean area per disjunct core
CASD2	Class	Disjunct core area standard deviation
CACV2	Class	Disjunct core area coefficient of variation
TCAI	Class	Total core area index
MCAI	Class	Mean core area index
MNN	Class	Mean nearest neighbor distance
NNSD	Class	Nearest neighbor standard deviation
NNCV	Class	Nearest neighbor coefficient of variation
L-TA	Landscape	Total area
L-LPI	Landscape	Largest patch index
L-NP	Landscape	Number of patches
L-PD	Landscape	Patch density
L-MPS	Landscape	Mean patch size
L-PSSD	Landscape	Patch size standard deviation
L-PSCV	Landscape	Patch size coefficient of variation
L-TE	Landscape	Total edge
L-ED	Landscape	Edge density
L-LSI	Landscape	Landscape shape index
L-MSI	Landscape	Mean shape index
L-AWMSI	Landscape	Area weighted mean shape index

throw. In some forest types, such as ponderosa pine *(Pinus ponderosa)*, natural fragmentation is a sign of higher ecological integrity. Intensively managed ponderosa pine forests often display greater tree densities than unmanaged, native stands.

Because of the limits of the input data, it was not possible to differentiate in this study between natural and anthropogenic disturbance. For many forest types, the combination of 30 m resolution satellite imagery and a minimum mapping unit of 1 ha eliminated the majority of smaller natural openings. With regard to natural patchiness, we intentionally avoided the most problematic ecoregions, such as those characterized by open forest or savannas. This problem, however, could not be avoided entirely. For example, Jeffrey pine *(Pinus jeffreyi)* forests, which are naturally patchy forests that grow in very harsh serpentine soils on a small percentage of the Klamath–Siskiyou ecoregion, showed up in the land cover database as quite patchy. Differentiating between Jeffrey pine natural openings and neighboring clearcut blocks was not possible without exhaustive effort. Expanding this effort for the other open forest types scattered throughout the country was untenable. This problem would have been far more serious had the data scale been more detailed, thereby resulting in the delineation of small openings. More detailed investigations will need to address this problem by using disturbance data.

Fragmentation index redundancy and applicability. It has been stated that many fragmentation indices are redundant over a range of spatial and attribute scales, making it important to choose the most relevant indicators (Cain et al. 1997). In addition, indices should be carefully chosen and interpreted to provide ecologically relevant information specific to each research question. We included all of the class- and landscape-level fragmentation results in the land units database to allow for the widest possible utility. We believe that a national forest fragmentation database should be as inclusive as possible, because we are still in the early stages of interpreting spatial pattern. It is still unknown which index (or suite of indices) tells us the most about forest fragmentation, and until we learn more about the mechanism and impact of forest fragmentation, we believe it is better to provide too much

Box 1, continued

L-DFLD	Landscape	Double log fractal dimension
L-MPFD	Landscape	Mean patch fractal dimension
L-AWMPFD	Landscape	Area weighted mean patch fractal dimension
L-TCA	Landscape	Total core area
L-NCA	Landscape	Number of core areas
L-CAD	Landscape	Core area density
L-MCA1	Landscape	Mean core area per patch
L-CASD1	Landscape	Patch core area standard deviation
L-CACV1	Landscape	Patch core area coefficient of variation
L-MCA2	Landscape	Mean area per disjunct core
L-CASD2	Landscape	Disjunct core area standard deviation
L-CACV2	Landscape	Disjunct core area coefficient of variation
L-TCAI	Landscape	Total core area index
L-MCAI	Landscape	Mean core area index
L-MNN	Landscape	Mean nearest neighbor distance
L-NNSD	Landscape	Nearest neighbor standard deviation
L-NNCV	Landscape	Nearest neighbor coefficient of variation
L-SHDI	Landscape	Shannon's diversity index
L-SIDI	Landscape	Simpson's diversity index
L-MSIDI	Landscape	Modified Simpson's diversity index
L-PR	Landscape	Patch richness
L-PRD	Landscape	Patch richness density
L-RPR	Landscape	Relative patch richness
L-SHEI	Landscape	Shannon's evenness index
L-SIEI	Landscape	Simpson's evenness index
L-MSIEI	Landscape	Modified Simpson's evenness index
L-CONTAG	Landscape	Contagion index

n.a. Not applicable.

data rather than not enough. With this database, it may be useful to employ principal component-based factor analysis (Johnston 1980), a multivariate procedure designed to identify the most important factors driving variability, as demonstrated by Cain and colleagues (1997). It would also be advantageous to incorporate promising new indices, such as patch cohesion (Shumaker 1996).

Spatial filtering techniques using discrete units, such as watersheds (Riitters et al. 1997), have been used to analyze and map regional spatial patterns. This technique has been applied using multiple window sizes (9 x 9 pixels, 27 x 27 pixels, and 81 x 81 pixels) that sense the landscape at different scales to model habitat suitability for species. Hybridizing our approach with spatial filtering algorithms may prove very fruitful.

Conclusions

Land cover data derived from satellite imagery offers outstanding potential for analyzing forest fragmentation (Riitters et al. 2000b). In this article we outline a methodology for assessing forest fragmentation and offer a comprehensive data set for further investigation by researchers. Repeated use of our methodology could become part of a national forest monitoring protocol. Emerging spatial analysis techniques, along with computer mapping advances, have the potential to promote meaningful planning for biodiversity conservation at multiple spatial and temporal scales. Although we are making advances in planning at multiple spatial scales (Poiani et al. 2000), we are still at the early experimental stages of handling the topic analytically in the GIS environment. Despite the numerous technical advances, we see little value in computer mapping technologies unless they can work in close concert with field biology. Without a strong commitment to field surveys and evaluations, we will lose a tremendous opportunity to effectively address the many conservation issues of our time. In the meantime, it is premature to conclude that any region's forests have recovered (Moffat 1998) until one of the most important measures of biodiversity decline, habitat fragmentation, is properly assessed.

Acknowledgments

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Large woody debris in bull trout (*Salvelinus confluentus*) spawning streams of logged and wilderness watersheds in northwest Montana

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Abstract: We measured large woody debris (LWD) in 20 known bull trout (*Salvelinus confluentus*) spawning stream reaches from logged and wilderness watersheds in northwestern Montana. Mean bankfull width of stream reaches was 14.1 m ranging from 3.9 to 36.7 m. Streams were large enough to move LWD and form aggregates. We determined the characteristics of individual pieces of LWD that were interactive with the stream channel. Large, short pieces of LWD attached to the stream bank were the most likely to be positioned perpendicular to stream flow, while large, long pieces either tended to be parallel to the flow or, when attached, were most apt to extend across the channel thalweg. Observations indicated that the majority of pools were formed as scour pools by either very large LWD pieces that were perpendicular to the stream or multipiece LWD aggregates. Among reaches in wilderness watersheds, ratios of large to small LWD, attached to unattached LWD, and with and without rootwads were relatively consistent. However, among reaches with logging in the watershed, these ratios varied substantially. These results suggest that logging can alter the complex balance of delivery, storage, and transport of LWD in northern Rocky Mountain streams, and therefore, the likely substantive change in stream habitats.

Résumé : Nous avons mesuré les gros débris ligneux (GDL) dans 20 tronçons de cours d'eau servant de frayères à l'omble à tête plate dans des bassins exploités par l'industrie forestière et des bassins sauvages du nord-ouest du Montana. La largeur moyenne des tronçons à pleins bords était de 14,1 m, avec une fourchette de 3,9 m à 36,7 m. Les cours d'eau étaient assez larges pour que les GDL se déplacent et forment des agrégats. Nous avons déterminé les caractéristiques des morceaux de GDL qui interagissaient avec le chenal. Les morceaux gros et courts attachés à la berge étaient les plus susceptibles de se positionner perpendiculairement au courant, tandis que les morceaux gros et longs se plaçaient parallèlement au courant ou, s'ils étaient attachés, étaient les plus susceptibles de se placer en travers du thalweg du chenal. Les observations ont montré que la majorité des fosses sont le résultat de l'affouillement causé soit par de très gros morceaux de GDL perpendiculaires au courant, soit par des agrégats composés de plusieurs morceaux de GDL. Parmi les tronçons des bassins sauvages, les rapports des gros aux petits GDL, des GDL attachés aux GDL non attachés, des GDL avec et sans attaches racinaires, étaient relativement constants, alors qu'ils variaient considérablement parmi les tronçons des bassins soumis à l'exploitation forestière. Ces résultats permettent de penser que l'exploitation forestière peut altérer l'équilibre complexe de l'apport, de l'installation et du transport des GDL dans les cours d'eau du nord des Montagnes Rocheuses, et donc occasionner des modifications potentiellement importantes des habitats lotiques.

[Traduit par la Rédaction]

Introduction

Large woody debris (LWD) plays numerous roles in the structure and function of stream ecosystems (Gregory et al. 1991). Riparian forests contribute LWD to a channel network, directly affecting both large- and small-scale stream morphology, hydrologic processes, and stream biota (Abbe and Montgomery 1996; Bisson and Montgomery 1996). Large wood accumulations influence the dissipation of stream energy and thus the ability of the stream to transport material. For example, LWD has been associated with channel avulsion, floodplain formation, and island development (Abbe and Montgomery 1996; Nanson and Knighton 1996). LWD also plays an important role in localized modification of streambed morphology (Bisson et al. 1987; Ralph et al. 1994) and pool frequency and channel geometry (Beschta and Platts 1986; Fausch and Northcote 1992; Richmond and Fausch 1995). The orientation and position of LWD in streams affect storage of organic and inorganic matter (Bilby and Ward 1989; Nakamura and Swanson 1993). Likewise, wood serves as trophic support of stream biota by providing organic matter for stream invertebrates and substratum for attachment and growth (Angermeier and Karr 1984; Benke et al. 1985; Hauer and Benke 1991).

The factors that directly affect introduction, stability, or character of stream LWD have a potentially significant influence on native fish populations that utilize streams for spawning, rearing, or growth and completion of life histories

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(Andrus et al. 1988). Stream characteristics affected by LWD and its implications on salmonid populations have been the focus of numerous studies (e.g., Marcus et al. 1990; Ralph et al. 1994; Riley and Fausch 1995). A species of particular concern is the bull trout (*Salvelinus confluentus*), which has been in decline throughout the Pacific Northwest and was recently (1998) listed as a threatened species under the U.S Endangered Species Act. Numerous explanations for its decline have been offered, including habitat degradation (Fraley and Shepard 1989), overharvest (Rieman and McIntyre 1996), and displacement by exotic species (Leary et al. 1993).

Despite a generally ubiquitous trend of decline, the bull trout populations of the Flathead Basin in northwest Montana were considered relatively healthy, until recently. Strong spawning populations from Flathead Lake, Swan Lake, and Hungry Horse Reservoir have been an important part of the native fish fauna and an important sport fishery. In the past several years, however, the frequency of bull trout spawning in tributaries of the North and Middle forks of the Flathead River (i.e., the Flathead Lake population) has seriously declined (Rieman and Myers 1997). Overfishing, competitive interactions, predation of juveniles, food web alterations in Flathead Lake, and loss of habitat for spawning and rearing have all been suggested as causes for this decline. It is likely, however, that no single factor can be isolated as the overriding ecological bottleneck. Rather, all these factors influence the Flathead Lake population. For example, inundation of spawning gravels with fine sediments or changes in channel form and complexity may be major factors affecting the decline in bull trout spawning in the tributary drainages of the North and Middle forks of the Flathead River (Weaver and Fraley 1991). Low frequencies of spawning in some of the tributaries of the Swan River have been associated with the presence of logging roads (Baxter et al. 1999). Although the mechanisms that may be leading to the observed decline in bull trout are unclear, either on the landscape or in specific streams, hydrologic and vegetative changes associated with land use clearly play an important role. We suggest that a significant part of that role may be the result of change in the frequency, character, and distribution of in-stream LWD.

Although LWD plays an important role among streams in forested watersheds of the Pacific Northwest Coastal and Cascade Mountains (e.g., Nakamura and Swanson 1993; Ralph et al. 1994) and in the central Rocky Mountains (e.g., Fausch and Northcote 1992; Richmond and Fausch 1995), little information is available regarding the character or function of LWD in forested streams of the northern Rocky Mountains. The processes that have been documented among Washington and Oregon streams or streams in Colorado might not be seamlessly applicable in western Montana. Differences in climatic regime, landscape geomorphology, hydrologic regime, and the size, density, and longevity of dominant riparian species among these regions will have direct bearing on the interactive relationship between stream structure and function and LWD.

Regardless of the potential causes of bull trout population declines or the current cumulative effects impinging on the health and long-term viability of bull trout populations in western Montana, the maintenance of productive spawning and rearing habitat will be critical to the long-term sustainability or recovery of bull trout (see Fausch and Northcote 1992). As the recent changes in the food web of Flathead Lake come to some new quasi-equilibrium, with its cascading effects on higher trophic levels (Spencer et al. 1991), population restoration for bull trout will be strongly affected by reproductive success and juvenile survivorship. LWD may play a critical role in maintaining appropriate stream habitat and thus affect the long-term sustainability of bull trout populations in the Flathead Basin.

The purpose of this study is to describe the characteristics and selected functions of LWD among an array of historical bull trout spawning streams in the Flathead Basin. Although in several instances, redds occurred within a study reach, it was not our intention to specifically locate bull trout redds or covariation of redds and LWD. We selected streams from each of the four major tributaries in the drainage: the North and Middle forks of the Flathead River (Flathead Lake bull trout population), the South Fork of the Flathead River (Hungry Horse Reservoir population), and the Swan River drainage (Swan Lake population). We also chose streams that represented different types and levels of land use. Streams in the North Fork and Swan River include tributary drainages with extensive logging and riparian clearcuts. Streams in the Middle and South forks were within Glacier National Park or designated wilderness, respectively. The primary objectives of the research were to (*i*) characterize LWD in known bull trout spawning streams of the Flathead Basin, (*ii*) examine relationships of LWD size, position, and orientation across an array of stream sizes, (*iii*) examine the role of LWD in affecting local-scale bedform and stream morphology, and (*iv*) examine the potential effect of land use and (or) riparian logging on the size frequency structure, orientation, and decay relationships of LWD.

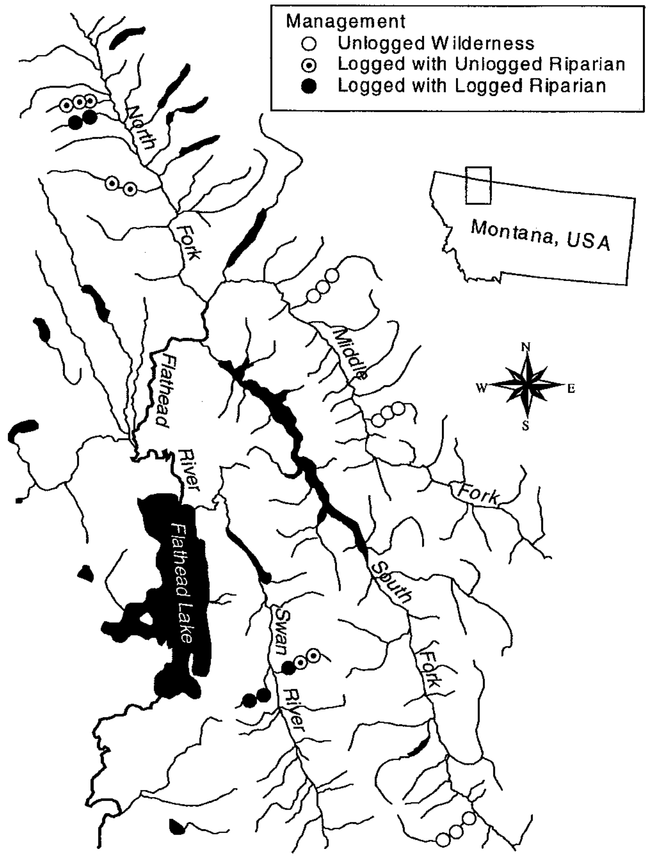
Materials and methods

Study area

This study was conducted in the Flathead Basin, a 22 241-km^2 drainage in northwestern Montana and southeastern British Columbia, along the west slope of the Continental Divide and within the belt series of the northern Rocky Mountains (Fig. 1). Sedimentary bedrock from the late Paleogene to the Proterozoic underlies the region and has been affected by low-grade metamorphosis. These mountain ranges are part of the Rocky Mountain Belt Supergroup and consist of argillites, siltites, and carbonates with a maximum stratigraphic thickness of 5200 m (Whipple et al. 1984). Colluvium and glacial till mantle the heavily forested valleys. During the height of the last major glaciation, about 20 000 years ago, the Flathead Basin was covered by glacial ice. The main glacial advance flowed from the cordilleran ice sheet down the Rocky Mountain Trench. Smaller valley glaciers flowed from the Livingston, Whitefish, Swan, Flathead, and Mission ranges to merge along the valley floors, forming trunk glaciers as much as 1000 m thick. Alluvial valley segments of tributary drainages formed with faulting and local accumulations of valley fill from alluvial and glacial sources.

Twenty stream reaches were selected from a stratified random design for study from eight streams distributed around the basin (Fig. 1). All reaches were in third- or fourth-order segments. We consulted the Montana Department of Fish, Wildlife, and Parks

Fig. 1. Map of the Flathead Basin and northern Continental Divide region of northwest Montana showing the names of major river drainages and study stream reaches in unlogged wilderness watersheds (open circles), logged watersheds with unlogged riparian (encircled dots), and logged watersheds with logged riparian (solid circles).



and selected study reaches within known bull trout spawning tributaries. We selected study streams within watersheds that had a land use history of either logging or wilderness management. Additionally, we selected specific study reaches within streams based on prevalent streamside management within the watershed. In some cases, as in Red Meadow Creek, the selected reaches flowed through riparian clearcuts, which occur commonly along that stream's length. Among other streams, such as Ole Creek in Glacier National Park, the riparian zone along the study reach was in an unaltered condition.

The substratum of each study reach was similar, generally composed of gravel and cobble, although occasionally, larger boulders (50–100 cm) were also present. The drainage area above the study reaches varied from 23 290 ha on Young's Creek to 1610 ha on Red Meadow Creek (Table 1).

Stream channel

Stream cross-sectional profiles, sinuosity, and gradient were measured at each stream reach using an Abney level, a Sonin® electronic distance measurer, and a leveling rod. Eleven transects (A–K) were taken across each stream reach at 10-m intervals covering a total reach length of 100 m. Each transect consisted of channel profiles measured perpendicular to the stream thalweg and to the top of the bankfull channel on both sides of the stream. Typically, 8–12 measures were taken to develop the cross-sectional profile at each transect. The profile data included all major breaks in elevation, the wetted channel width, water depth at the thalweg (at the time of measure), and height of the average bankfull channel. The change in bed height and water depth between each transect profile was measured using the Abney level, electronic distance measurer, and leveling rod. The 10-m intervals between each transect were identified as a stream section and referenced to the downstream transect.

LWD measurements

Measures of LWD were made within each 10-m stream section between each transect. LWD was defined as logs ≥10 cm in diameter and ≥1 m in length. Although there are no standard criteria established as to the minimum size that constitutes LWD, the criteria used here are the same as used in research at other locations (Andrus et al. 1988; Fausch and Northcote 1992; Richmond and Fausch 1995). Each piece of wood meeting the LWD criteria was measured if any part occurred within or was suspended above the bankfull stream channel. The diameter was measured at each end of the LWD piece with a 1-m caliper. The length of each piece was measured with the electronic distance measurer if the length was >2 m or with the caliper for shorter pieces. Piece volume was calculated as a tapered cylinder (Lienkaemper and Swanson 1987). All large rootwads were considered LWD regardless of length. Stumped rootwads with a length <1 m were common among streams with logged riparian areas. Volume of rootwads was estimated by measuring the diameter of the root structure across the dominant mass as one end of the cylinder, the bole of the tree stump above the root structure as the other end of the cylinder, and the distance between these measurements as the tapered cylinder length. The position and orientation of LWD to the channel were determined for each LWD piece. Piece position was recorded as one of three possibilities: (*i*) no contact with either bank, (*ii*) contacting either the left or right bank, or (*iii*) contacting both banks. In addition to simply contacting a bank, many pieces were strongly "attached" to one or, rarely, both banks. We classified an LWD piece as being attached if either or both ends were anchored into the stream bank.

Orientation of LWD is known to affect stream flow and bed morphology (Robison and Beschta 1990). Likewise, stream power affects piece orientation because hydraulic forces move unattached ends in a downstream direction (Nakamura and Swanson 1994). Piece orientation was divided into three categories: (*i*) at an about 0° angle (parallel) to the channel, (*ii*) at an about 45° angle to the channel, and (*iii*) at an about 90° angle (perpendicular) to the channel. We also noted whether an LWD piece had a rootwad attached to the bole, since this plays an important function in the attachment, orientation, and distribution dynamics of the piece.

The relative age of each piece was assessed using a modification of the Grette (1985) decay classification procedure, which divided LWD into four decay classes: (1) bark and branches attached, (2) bark and branches missing; wood solid with evidence of decay restricted to the outer perimeter, (3) wood showing significant signs of decay to at least depths of 5–10 cm, and (4) wood soft and decayed nearly or completely to the center of the piece. We later combined categories 3 and 4 for our analysis because of the infrequency of observing type 4 decay class LWD. We believe that the scarcity of decay class 4 wood is due to the rapidity of final decay and disappearance once a piece undergoes a transition from decay class 3 to class 4.

Data analyses

We conducted a variety of statistical analyses including χ^2, correlation analysis, ANOVA, and MANOVA using the statistical analysis software SPSS for Windows by SPSS, Inc. We considered test results to be significant at $\alpha = 0.95$.

Table 1. River drainage, stream name, watershed area above study reaches, number of pieces in aggregates or not in aggregates, and other characteristics of LWD.

Flathead River drainage	Stream Name	Drainage area (ha)	Reach and logging condition	LWD aggregate (no.)	LWD nonaggregate (no.)	LWD volume (m^3)	LWD attachment	
							Attached	Unattached
Middle Fork	Ole Creek	10 295	A[a]	0	13	8.8	5	8
			B[a]	18	11	7.4	17	12
			C[a]	0	32	30.8	7	25
	Nyack Creek	22 005	A[a]	0	7	0.4	1	6
			B[a]	0	5	3.0	2	3
North Fork	Red Meadow Creek	1 612	A[c]	10	41	35.9	35	16
			B[c]	53	70	73.3	95	28
	Whale Creek	9 836	A[b]	0	56	14.3	26	30
			B[b]	0	87	19.4	37	50
			C[b]	39	52	58.9	59	32
	Coal Creek	12 113	A[b]	51	72	33.3	57	66
			B[b]	11	25	19.8	19	17
South Fork	Young's Creek	23 289	A[a]	145	39	73.2	90	94
			B[a]	49	28	18.6	49	28
			C[a]	0	37	10.1	17	20
Swan River	Jim Creek	3 705	A[c]	71	46	65.2	72	45
			B[c]	0	46	10.9	35	11
	Goat Creek	5 602	A[b]	16	45	18.1	31	30
			B[b]	35	35	20.4	49	21
			C[c]	27	48	56.7	54	21

Note: Diameter class: 1 = 10–19 cm; 2 = 20–29 cm; 3 = 30–39 cm; 4 = 40 cm.
[a]Reach in unlogged wilderness watershed.
[b]Reach in logged watershed with unlogged riparian.
[c]Reach in logged watershed with logged riparian.

Results and discussion

General characteristics of study reaches

Mean (±SD) bankfull widths among all stream reaches combined were 14.2 ± 6.6 m with a range of 3.9–36.7 m across all transects. Study reaches were variable, both between and within streams. Stream gradients among all reaches were moderate (mean 1.0%, maximum 2.6%) but again highly variable. The thalweg bed elevation of some downstream transects was higher than that of upstream transects, clearly illustrating streambed complexity.

Five of the 20 study reaches had one or more side channels. In cases where side channels were present, there was always one dominant channel. The side channels were always small with only minor flow. Side channels in two stream reaches contained a high density of LWD and likely were abandoned main channels.

Thalweg stream depths across all transects ranged from a minimum of 0.15 m in Ole Creek reach A to a maximum of 1.33 m in Young's Creek reach C. An examination of the relationship between drainage area and stream depth showed a significantly, positive correlation (p = 0.016) with stream maximum depths; however, neither mean nor minimum stream depth was significantly correlated with basin size (Fig. 2).

General characterization of LWD

A total of 1320 pieces of LWD were counted and measured among all study reaches. The number of pieces and volume of LWD across all reaches were highly variable (Table 1). For example, reach A on Young's Creek had 184 pieces, while, in contrast, reach B on Nyack Creek had only five pieces of LWD.

Across all stream reaches, the size of the LWD was also extremely variable. About 70% of all LWD was in the smaller two diameter classes (10–19 cm: >35%; 20–29 cm: >30%) (Fig. 3A) and >50% of the LWD was between 1 and 4 m in length (Fig. 3B). Size frequency of diameter measures and tree length measures demonstrated a decreasing exponential curve with increasing piece size. Together, <50% of the LWD across all stream reaches consisted of pieces >30 cm in diameter and >4 m in length. However, as in other studies (e.g., Abbe and Montgomery 1996), we found that the larger LWD pieces played the primary role in streambed configuration and the formation of aggregates (see data analysis below). LWD pieces that had been moved by the stream into large debris jams, or aggregates, often spanned the stream channel and were extremely stable, owing to their mass and configuration with the stream banks and the LWD pieces within the jam.

Other studies have found that LWD attachment to one or both banks and (or) the presence of the tree's rootwad are important factors influencing the stability (i.e., the resistance to being moved during flood) and orientation of the LWD piece (Beschta and Platts 1986; Richmond and Fausch 1995). Among nonaggregated LWD, we found attachment to one or both banks and the number of pieces that had rootwads to be highly variable, commensurate with the high variation in LWD occurrence between stream reaches (Table 1). However, we did find statistically significant relation-

0° by diameter class				45° by diameter class				90° by diameter class				Rootwad	
1	2	3	4	1	2	3	4	1	2	3	4	With	Without
1	0	1	0	2	3	1	4	1	0	0	0	6	7
4	3	0	1	4	3	2	3	4	2	1	2	2	27
4	6	3	4	3	4	1	1	1	1	0	4	7	25
2	0	0	0	3	1	0	0	0	0	0	0	0	7
2	1	1	0	0	0	0	0	0	0	0	1	1	4
2	4	5	4	3	3	2	4	3	10	3	8	9	42
3	7	1	4	5	18	9	12	8	15	13	28	8	115
13	19	3	1	10	2	2	2	4	0	0	0	1	55
19	14	6	3	18	13	6	1	2	3	2	0	0	87
13	14	14	5	11	13	3	1	5	7	0	5	17	74
33	20	3	4	17	17	6	6	10	6	1	0	4	119
7	2	2	3	5	2	0	1	5	4	3	2	2	34
16	16	8	5	43	29	20	9	12	10	5	9	27	157
11	7	2	1	20	9	3	10	4	4	2	3	15	62
6	4	0	4	9	4	4	0	3	2	0	1	5	32
9	8	2	3	19	19	8	8	7	14	9	11	18	99
7	3	0	2	13	7	1	2	5	0	3	2	2	44
6	7	1	2	6	2	2	1	9	13	4	8	8	53
15	5	3	3	9	7	1	4	7	9	1	6	6	64
4	5	2	4	7	3	4	8	9	9	4	16	20	55

ships between the orientation of the LWD piece, the attachment of the piece to the bank, and both the volume and the length of the LWD piece. Using χ^2 analysis, we found that LWD tends to be significantly shorter among those pieces that are perpendicular to the stream flow (orientation 90°) than among those that are parallel (orientation 0°) to stream flow (Table 2). This is likely due to longer pieces being subject to rotation around an anchor point, such as a bank attachment, during flooding when stream power and floatation of the LWD are at their highest (Nakamura and Swanson 1994). We found a similar significant relationship of increased LWD piece diameter associated with pieces that were perpendicular to the channel compared with those parallel to the channel (Table 2). We also observed that bank attachment often extended a considerable distance onto the bank and back into the riparian vegetation. These pieces were often the most stable and demonstrated resistance to change in orientation.

In addition to girth and length influencing stream LWD position, we also examined the affect of LWD attachment on orientation. A χ^2 analysis revealed that LWD pieces that were perpendicular to the current had a significantly higher frequency of bank attachment (Table 3). We found these features to be particularly important, since perpendicular pieces were the most interactive with the stream channel in that they were often most responsible for change in streambed morphology and complexity (see LWD influence on streambed morphology below). We also examined LWD wood decay (Table 1). We found that among all stream reaches, most LWD was in decay class 2, i.e., most pieces had been stripped of their bark and branches and showed only the earliest signs of rotting at the surface. This finding has the following implications: (*i*) most of the wood has been in the stream for at least several years, long enough to loose the outer bark and limbs, but not so long as to enter advanced decay stages and (*ii*) the paucity of decay classes 3 and 4 suggests that once an LWD piece enters the latter stages of decay, decomposition processes occur rapidly. The latter stages of decay may be strongly enhanced during spring runoff as increased stream power causes decomposing logs to break apart. We did not conduct tests to directly determine the rate of LWD decomposition; however, based on our knowledge of aggregate accumulations at sites that we have visited regularly since the mid-1970's, we know that LWD can remain >20 years with no signs of surface decomposition. Thus, it appears that LWD probably remains in these streams for periods exceeding 50 years (sensu Andrus et al. 1988). To summarize, those pieces that were perpendicular to the flow tended to be attached to the stream bank, large in diameter, and short. This finding corroborates Nakamura and Swanson (1994).

LWD influence on streambed morphology

An important feature of stream habitat structure is the development and stability of streambed morphology. Streams that alternate between riffles, pools, and runs provide complex habitats that support high biodiversity, biomass, and secondary production of aquatic insects and fish. Complex

Fig. 2. Maximum, mean, and minimum thalweg depths among all transects for each stream reach regressed against the drainage area of each watershed above the study area. The maximum thalweg depth to basin area correlation coefficient (r = 0.51) is significant at the 0.05 level. Correlations between mean and minimum thalweg depths and basin area were nonsignificant.

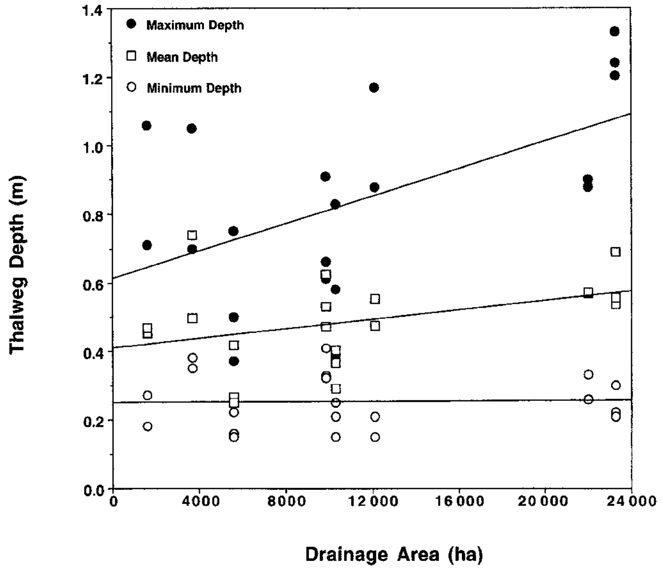
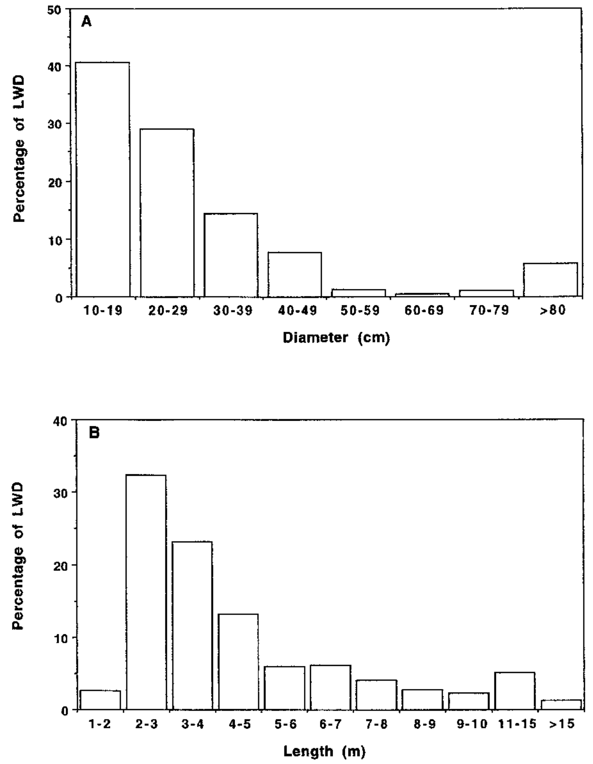


Fig. 3. Percentage of LWD in each of (A) eight bole diameter classes and (B) 11 length classes.



variation in stream habitat and streambed morphology is frequently required for different species to coexist (sensu Connell 1980).

Using correlation analysis, we examined the role of LWD and its influence on streambed morphology. We found that as the number of pieces of LWD and the volume of LWD increased within a stream section, there was a corresponding increase in the bedslope of the section immediately downstream (Fig. 4). The steepest bedslopes were all associated with upstream aggregates, large snags with rootwads, or large-diameter LWD oriented perpendicular to the stream thalweg. Each of these LWD structures performs important bed-forming functions, e.g., the retention of gravel on the upstream side of the structure and (or) the focus of stream flow and thus stream power and scour on the downstream bed material forming pools. Both of these factors lead to the aggradation of upstream gravel and cobble and the downstream degradation of bed material. These correlations between increased LWD piece frequency and volume and bedslope underscore the importance of LWD aggregates in stabilizing bedload, capturing gravel, and promoting pool formation.

Land use influence on LWD

Three of the eight watersheds (eight of 20 study reaches) examined were located within Glacier Park or the Bob Marshall Wilderness. These three watersheds are managed as wilderness and have never been logged or roaded. The other five watersheds are in the North Fork or the Swan River drainages and flow through lands managed for multiple use, but primarily for timber harvest.

We found a tight correlation (r = 0.99) between the frequency of large LWD (≥30 cm in diameter) and the frequency of small LWD (<30 cm in diameter) among the reaches draining wilderness areas (Fig. 5). In contrast, among reaches in watersheds with upstream logging, the large to small LWD relationship was poorly correlated (r = 0.18). These data suggest that even though variation in number of pieces of LWD among stream reaches may be high, there is a consistent and highly predictable relationship between the frequency of large-diameter trees and small-diameter trees in the LWD pool in wilderness watersheds that was not present in logged watersheds. In addition to the cross-watershed comparison of LWD size ratios, we further compared streams flowing through logged riparian zones with those flowing through unlogged riparian zones, but in logged watersheds. Among the logged watersheds, we found that streams flowing through logged riparian zones tended to have a higher large LWD to small LWD ratio than streams flowing from wilderness areas, while streams flowing from logged watersheds, but with unlogged riparian zones, usually had smaller ratios (Fig. 5).

We also examined the relationship of attachment of LWD to the stream bank. Again, we found that among wilderness watersheds, there was a relatively tight correlation (r = 0.94) between the frequency of attached LWD and the frequency of unattached LWD and a poorly correlated relationship (r = 0.30) among logged watersheds (Fig. 6). Additionally, we examined the relationship between the frequency of LWD pieces with rootwads and the frequency of those without rootwads. Again, among reaches in wilderness watersheds, there was a high correlation (r = 0.96) between LWD pieces with rootwads and those without rootwads, but among logged watersheds, this relationship was poorly correlated (r = 0.13) (Fig. 7).

Table 2. Chi square analysis of length and bole diameter versus attachment and orientation characteristics of nonaggregate LWD among all stream study reaches.

	Attachment characteristic			Orientation characteristic			
LWD characteristic	Attached	Unattached	Pearson χ^2	0°	45°	90°	Pearson χ^2
Length <4 m	219	155	0.05 (p = 0.82)	108	150	116	38.0 (p < 0.00)
Length ≥4 m	184	135		153	121	45	
Diameter <30 cm	236	232	35.4 (p < 0.00)	190	191	87	17.7 (p < 0.00)
Diameter ≥30 cm	167	58		71	80	74	

Table 3. Chi square analysis of attachment and orientation characteristics of nonaggregate LWD among all stream study reaches.

Attachment	Orientation characteristic			
characteristic	0°	45°	90°	Pearson χ^2
Attached	127	156	120	27.5 (p < 0.00)
Unattached	134	115	41	

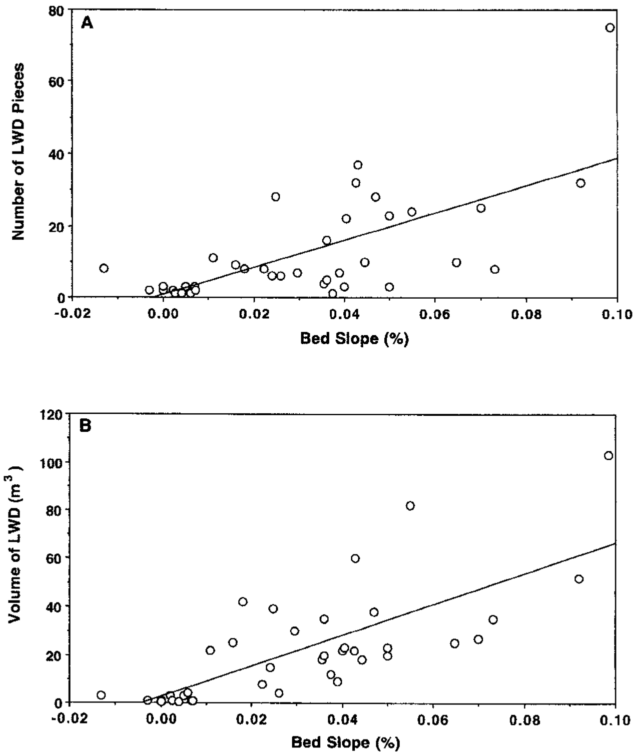
Among all watersheds, there was high variance in the frequency of LWD. Furthermore, LWD is not evenly distributed at the stream reach (100 m) spatial scale. However, the ratio of large to small LWD, the ratio of attached to unattached LWD, and the ratio of LWD with and without rootwads were relatively consistent across stream reaches in wilderness areas. However, among stream reaches in logged watersheds, these relationships were highly variable. These data suggest that logging or associated land use activities within a watershed may result in an alteration in the balance of delivery, storage, and transport of stream LWD, which in turn would have strong implications regarding effects on material transport and stream habitats.

It would require additional, focused study to determine the cause and effect relationship between size, attachment, and rootwad frequencies of stream LWD and specific logging practices. However, regardless of whether increased variance in size frequency is the result of direct actions that alter LWD input to the stream (e.g., cutting of large-diameter riparian trees) or indirect forces (e.g., postlogging blowdown), they may contribute to LWD characteristics that depart from relationships among streams in unlogged watersheds. Such departures may result in substantive habitat alteration and adverse effects on species dependent on habitats affected by stream LWD structure and function, such as pool size and frequency, surface/ground water exchange, and complex channel morphology.

Implications for watershed and streamside management

We found a close association between LWD and streambed morphology. For example, the steepest bedslopes were all associated with upstream aggregates of LWD. Among nonaggregated LWD, we found large pieces attached to stream banks and oriented perpendicular to the thalweg to be more closely associated with pool formation than parallel or unattached pieces. Among LWD pieces that were oriented perpendicular to stream flow and crossed the stream thalweg, there was a statistically significant higher frequency of long pieces (i.e., ≥4 m in length), a nearly significant proportion of large pieces (i.e., ≥30 cm in diameter), and a statistically significant higher frequency of LWD pieces attached to the stream bank (Table 4). These field observations suggest that LWD pieces that are perpendicular to the stream channel and engage the stream thalweg are the primary influence promoting pool formation. Field measurements further indicate that of the perpendicular LWD pieces, large-diameter pieces that are long and attached to the stream bank are the most stable (e.g., best able to resist reorientation and movement once in the channel). Grette (1985) and Richmond and Fausch (1995) also reported a significant positive relationship between LWD and the abundance of pools, as well as the importance of relatively few, stable LWD pieces that accounted for most of the pool formation.

Fig. 4. (A) Number of LWD pieces and (B) volume of LWD in the upstream 10-m stream section and the corresponding downstream bedslope.



Wohl et al. (1993) reported that stream depth, gradient, stream power, and the resistance of bed and bank materials to erosion were important determinants of pool size. We observed a similar relationship in which the streams of the larger drainages had the deepest pools, even though drainage size had no significant effect on mean or minimum thalweg

Fig. 5. Frequency of large-diameter (≤30 cm) versus small-diameter (<30 cm) LWD among study stream reaches in wilderness watersheds (solid circles) and in logged watersheds (open circles). Arrows denote study stream reaches with logged riparian areas. The trend line is for wilderness watersheds. Correlation coefficients are given in the text.

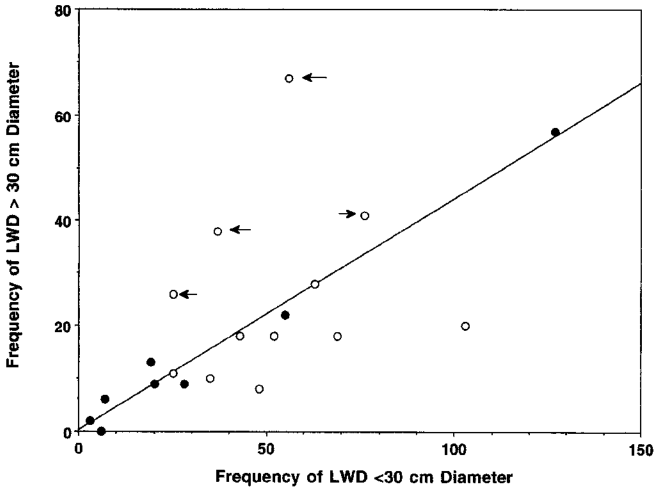
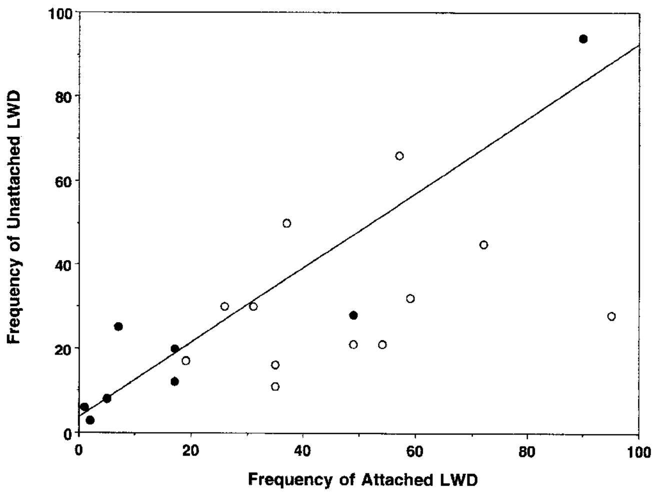


Fig. 6. Frequency of unattached versus attached LWD among study stream reaches in wilderness watersheds (solid circles) and in logged watersheds (open circles). The trend line is for wilderness watersheds. Correlation coefficients are given in the text.



depth. Other researchers have suggested that for large streams and rivers, LWD generally has a limited effect on gradient, stream power, maximum stream width, and maximum bankfull depth (Andrus et al. 1988; Evans et al. 1993). Our results support this suggestion, since within the largest watershed streams, such as Young's Creek, pool-forming LWD pieces occurred almost exclusively as aggregates (Table 1). In other words, as a stream gets larger, exemplified by Young's Creek in our study, stream power becomes sufficient to move virtually all wood that enters the channel. However, this does not mean that as stream size increases, LWD becomes of little consequence but rather that the role of LWD may change as aggregates interact with the dynam-

Fig. 7. Frequency of LWD with rootwad versus without rootwad among study stream reaches in wilderness watersheds (solid circles) and in logged watersheds (open circles). The trend line is for wilderness watersheds. Correlation coefficients are given in the text.

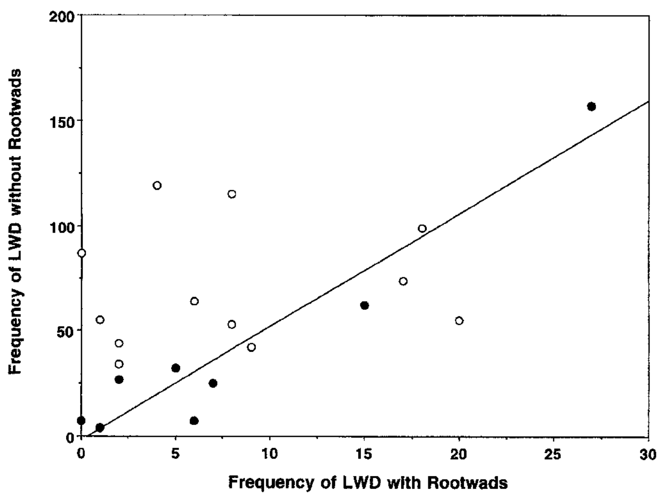


Table 4. Chi square analysis of length, bole diameter, and attachment characteristics of nonaggregate LWD crossing the stream channel and engaging the stream thalweg versus LWD within the bankfull channel but not engaging the stream thalweg.

	Crossing characteristic		
LWD characteristic	Across thalweg	Not across thalweg	Pearson χ^2
Length <4 m	40	321	6.64 (p = 0.01)
Length ≥4 m	56	254	
Diameter <30 cm	57	396	3.38 (p = 0.07)
Diameter ≥30 cm	39	179	
Attached	66	324	5.20 (p = 0.02)
Unattached	30	251	

ics of flood waters affecting anabranching and (or) avulsion behavior (e.g., Nanson and Knighton 1996).

Robison and Beschta (1990) and Richmond and Fausch (1995) showed that changing relationships between LWD and stream flow influenced pool types and that the majority of pools were formed by LWD spanning the channel perpendicular to flow. Richmond and Fausch (1995) found plunge and dammed pools to be the most prevalent pool type in the small subalpine streams of Colorado. Bilby and Ward (1989) found a similar pool type in smaller streams (<7 m wide) in southwestern Washington, but mainly scour pools in large streams (Bilby and Ward 1991). We observed similar situations among streams in the Flathead Basin where pools in the smaller streams (e.g., Goat Creek, Red Meadow Creek) were primarily associated with plunging or dammed water around LWD and with scour pools around aggregates in the largest streams (e.g., Young's Creek, Coal Creek).

In forested watersheds, LWD is an essential component in the formation of stream morphology and provides habitat for aquatic insects and fish. However, the relationship between stream size and power and the position and role played by LWD in the modification of bedform and channel development is a changing one. It is apparent from this study that large-diameter, shorter pieces of LWD attached to the stream bank have a higher frequency of perpendicular orientation and that larger, longer pieces attached to the bank tend to interact with the channel, as represented by those logs that cross the thalweg. Thus, a greater degree of pool-forming interaction with stream flows is represented by large, long pieces of LWD attached to the stream bank. Likewise, as the stream size increases, a concurrent increase in the size of LWD comprising an aggregate is needed to remain stable and interactive with the channel. Thus, the interaction of stream power, bed characteristics, and LWD piece diameter, length, and position largely determines the structure and function of stream LWD. The distribution of LWD among size classes, and attachment and orientation categories appears relatively consistent across streams in unlogged watersheds but becomes less predictable in streams that have been influenced by logging. A detailed investigation into the specifics of various logging histories would be necessary to determine how specific site prescriptions affect the outcome of LWD relationships associated with forest streams.

The implications of this study for forest managers are twofold: (*i*) with riparian logging comes increased unpredictability in the frequency of size, attachment, and stability of the LWD and (*ii*) maintaining the appropriate ratios of size frequency, orientation, and bank attachment, as well as rate of delivery, storage, and transport of LWD to streams, is essential to maintaining historic LWD characteristics and dynamics. Our data suggest that exclusion of logging from riparian zones may be necessary to maintain natural stream morphology and habitat features. Likewise, careful upland management is also necessary to prevent cumulative effects that result in altered water flow regimes and sediment delivery regimes. While not specifically evaluated in this study, in general, it appears that patterns of upland logging over space *and* time may have cumulative effects that could additionally alter the balance of LWD delivery, storage, and transport in fluvial systems. These issues will be critical for forest managers attempting to prevent future detrimental environmental change or setting restoration goals for degraded bull trout spawning streams (cf. Reeves et al. 1991).

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[Article]

Local Habitat, Watershed, and Biotic Factors Influencing the Spread of Hybridization between Native Westslope Cutthroat Trout and Introduced Rainbow Trout

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Abstract.—The invasion of nonnative fishes in freshwater systems is often facilitated by the interaction of biotic and abiotic factors operating at multiple spatial and temporal scales. We evaluated the association of local habitat features (width, gradient, and elevation), watershed characteristics (mean and maximum summer water temperatures, the number of road crossings, and road density), and biotic factors (the distance to the source of hybridization and trout density) with the spread of hybridization between native westslope cutthroat trout *Oncorhynchus clarkii lewisi* and introduced rainbow trout *O. mykiss* in the upper Flathead River system in Montana and British Columbia. The presence of hybridization and the proportion of rainbow trout admixture were estimated using seven diagnostic microsatellite loci. We defined logistic and linear regression models including various combinations of spatial and environmental factors and used an information-theoretic approach to evaluate the relative plausibility of these models. Models combining measures of water temperature, disturbance, and source connectivity were the best-approximating ones for the presence of hybridization. Hybridization was positively associated with mean summer water temperature and the number of upstream road crossings and negatively associated with the distance to the main source of hybridization. The best-approximating models associated with the level of introgression among hybridized sites included measures of temperature, source connectivity, and the density of trout. The proportion of rainbow trout admixture was negatively related to the distance to the source and positively related to mean summer water temperature and density. Our results suggest that hybridization is more likely to occur and spread in streams with warm water temperatures, increased land use disturbance, and proximity to the main source of hybridization. However, habitat features alone may not limit the spread of hybridization; populations with high proportions of admixture and high densities may have to be reduced or eliminated.

Exotic species are one of the greatest threats to global biodiversity and are a major concern in the conservation of freshwater ecosystems (Mack et al. 2000; Rahel 2000). Human disturbances of the landscape, such as intentional and accidental species translocations and habitat alterations, often create secondary contact between previously isolated species (Allendorf et al. 2001). In many cases, nonnative species are implicated in the decline and extinction of native biota through competition, predation, the spread of disease and parasites, and hybridization and introgression (Pimm 1989; Rahel 2000).

The invasion success of introduced species is often influenced by the interaction of abiotic and biotic factors operating at multiple spatial and temporal scales. In freshwaters, the major factors associated with the invasion and establishment of nonnative fishes include habitat conditions (local and watershed), connectivity, biotic resistance, and evolutionary history (Dunham et al. 2002; Benjamin et al. 2007; Fausch 2008). Water temperature plays a major role in determining the distribution and abundance of

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stream-dwelling salmonid species and has been correlated with invasion success in freshwater systems (Paul and Post 2001; Dunham et al. 2003; McMahon et al. 2007). Human-mediated habitat disturbances that increase stream temperatures and degrade riparian and stream habitats have also been correlated with the invasions of nonnative species (Thurow et al. 1997). Furthermore, theoretical models and empirical evidence suggest that the invasion and spread of nonnative species is freshwaters is strongly related to stream connectivity and the proximity of native populations to nonnative sources. However, little information is available as to the interactive role of these factors in determining the invasion of nonnative salmonids in freshwater systems.

Hybridization can be a major consequence of species introductions, especially in circumstances in which nonnative species hybridize with rare or endangered taxa and thus threaten the persistence of those taxa. Introgressive hybridization is more common in fish than in any other vertebrate taxa. This is particularly true for salmonids, for which widespread introgression among nonnative and native taxa has often created hybrid swarms over extensive geographical areas (Allendorf and Leary 1988; Leary et al. 1995; Allendorf et al. 2001). Additionally, interspecific hybridization may cause outbreeding depression as a result of the break-up of coadapted gene complexes and the disruption of local adaptations (Templeton 1986; Barton and Hewitt 1989; Rhymer and Simberloff 1996). Thus, hybridization is considered a leading cause of the decline and extinction of many freshwater fishes throughout North America (Miller et al. 1989).

Hybridization and introgression with introduced rainbow trout *Oncorhynchus mykiss* are considered the greatest threats facing many native populations of cutthroat trout *O. clarkii* in western North America (Behnke 1992; Leary et al. 1995). Introgressive hybridization with introduced rainbow trout has been especially detrimental to native westslope cutthroat trout *O. clarkii lewisi*, threatening this highly divergent subspecies with genomic extinction (Allendorf and Leary 1988; Allendorf et al. 2001). Nonhybridized populations of westslope cutthroat trout persist in less than 10% of their historical range in the United States (Shepard et al. 2005) and less than 20% of their range in Canada (COSEWIC 2006). Consequently, many remaining populations are restricted to small, fragmented headwater habitats, where the long-term sustainability of these populations is uncertain (Hilderbrand and Kershner 2000).

The upper Flathead River system is considered a regional and rangewide stronghold for nonhybridized westslope cutthroat trout. Hybridization with introduced, nonnative rainbow trout, however, has led to a rapid spread of introgression (Hitt et al. 2003; Boyer et al. 2008), threatening the genetic and ecological characteristics of the migratory and resident populations that have persisted in the basin since the last glacial period (~14,000 years ago). This study was intended to examine the local habitat features, watershed characteristics, and biotic factors associated with the occurrence (presence or absence) and amount of hybridization (proportion of rainbow trout admixture) between native westslope cutthroat trout and nonnative rainbow trout in the upper Flathead River drainage from the headwaters of the North Fork Flathead River in Canada downstream to the main-stem Flathead River upstream of Flathead Lake. We hypothesized that hybridization would be more likely in warmer, low-elevation streams in close proximity to hybridized populations with high proportions of rainbow trout admixture. Alternatively, we predicted that westslope cutthroat trout would be more common in headwater streams characterized by colder water temperatures, less land disturbance, and greater distances from hybridized source populations. Finally, we hypothesized that the proportion of rainbow trout admixture in hybridized populations would be associated with water temperature, the density of trout *Oncorhynchus* spp., and source connectivity. Our objectives were to examine the occurrence and extent of rainbow trout introgression in relation to these abiotic and biotic factors. Understanding the factors influencing the distribution and spread of hybridization will enable fisheries managers to focus conservation and management programs for westslope cutthroat trout and other salmonids threatened with the loss of genetic integrity.

Methods

Study area.—The study area included the tributaries to the North Fork and main-stem Flathead rivers in northwestern Montana and southeastern British Columbia. The North Fork Flathead River originates in the Rocky Mountains of southeastern British Columbia and flows into northwestern Montana, where it forms the western border of Glacier National Park before joining the main-stem Flathead River, which flows into Flathead Lake (Figure 1). This interconnected drainage contains migratory and resident populations of westslope cutthroat trout, a species of special concern in Montana and a blue-listed species at risk in British Columbia. Adfluvial and fluvial populations migrate from Flathead Lake and the Flathead River, respectively, to spawn in streams within the North Fork and Middle Fork drainages (Muhlfeld et al. 2009b).

Recent studies in the Flathead River drainage have

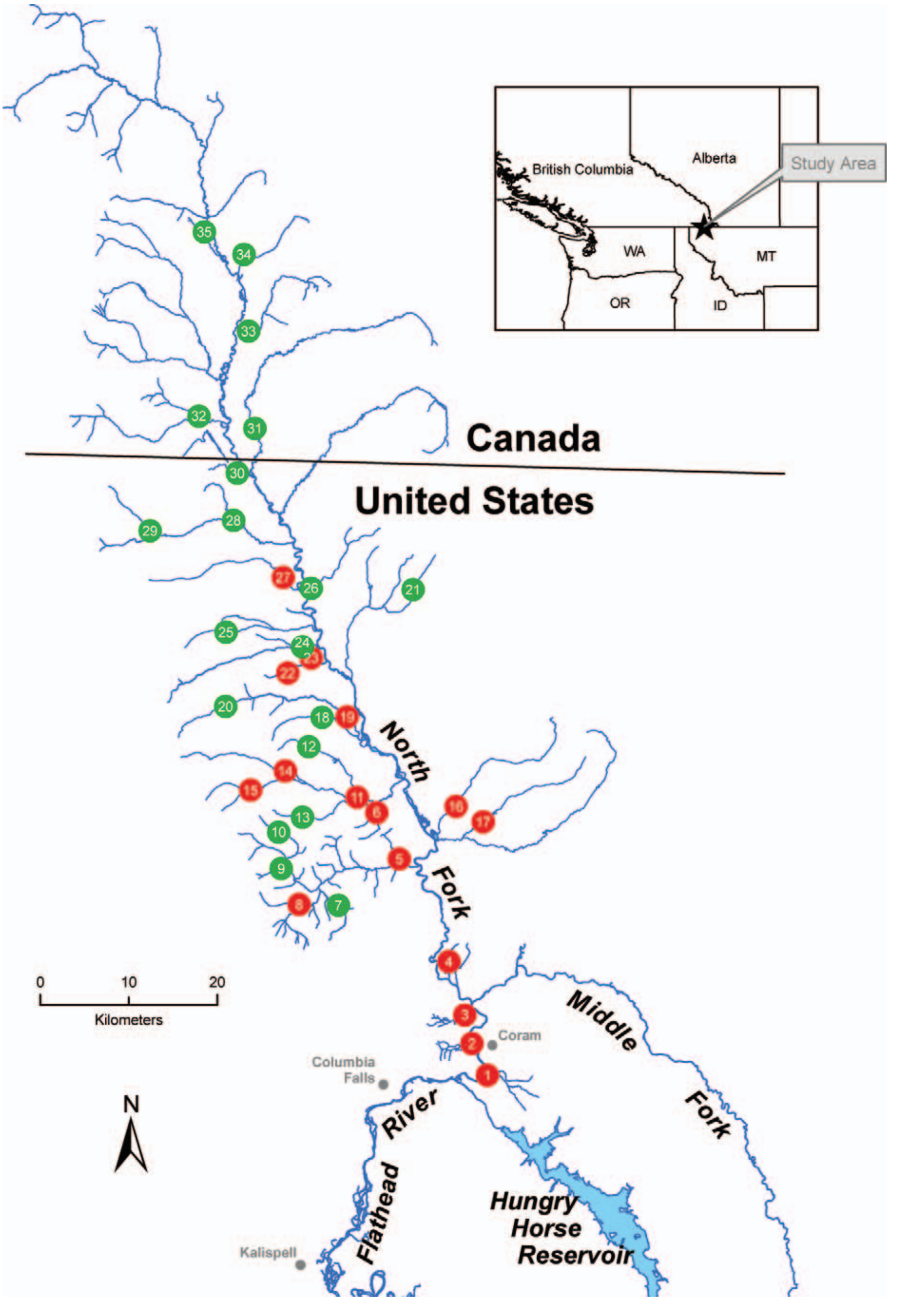

FIGURE 1.—Study area and sampling sites with hybridized (red) and nonhybridized (green) populations. The sampling site codes correspond to those in Table 1.

shown that hybridization is spreading upstream from source populations with high levels of rainbow trout ancestry. Hitt et al. (2003) found evidence of rainbow trout introgression in 7 of 11 populations that were determined to be nonhybridized in 1984, suggesting that hybridization has recently spread upstream in this system. In addition, these authors showed that the presence of hybridization was more strongly associated with neighborhood characteristics (i.e., distance and spatial attributes) than with environmental gradients.

However, their study did not assess how environmental and biotic factors influence the amount of nonnative rainbow trout introgression, nor did it include samples collected in the headwaters in Canada. Furthermore, recent genetics data (Boyer et al. 2008) and radiotelemetry studies (Muhlfeld et al., in press) indicate that the major source of hybridization in the system is Abbot Creek, a tributary to the main stem that contains a hybrid swarm with a high proportion (0.92) of rainbow trout admixture (Boyer et al. 2008). This stream is also located about 5 km downriver of a former private rainbow trout hatchery (Sekokini Springs), and anecdotal evidence suggests that approximately 70,000 rainbow trout were illegally released in 1997 when operations ceased (B. Marotz, Montana Fish, Wildlife and Parks, personal communication). Boyer et al. (2008) found that the amount of admixture tended to decrease with distance upstream from Abbot Creek, but no other abiotic or biotic factors were considered in the analysis. In this study, we expand on this research by using recent microsatellite DNA data to understand the relative importance of abiotic and biotic factors influencing both the presence/absence (occurrence) and degree of hybridization (proportion of rainbow trout admixture) throughout the interconnected river system.

Study design and data collection.—Fish population and habitat data were collected at 35 sites in the upper Flathead River system in Montana and British Columbia (Table 1; Figure 1). Streams were sampled during the low-flow period (July–September) from 2004 through 2007, and genetic samples were collected in 2003 and 2004 (Boyer et al. 2008). All sample sites were located downstream of physical barriers to fish migration. Migratory cutthroat and rainbow trout, therefore, could have theoretically accessed each site within the interconnected study area. Sampling occurred throughout the system and represented the full range of environmental and geographic variation within it (Figure 1).

Dependent variables.—We used the microsatellite DNA data reported by Boyer et al. (2008) to determine the occurrence of hybridization and the proportion of rainbow trout admixture for each site using seven diagnostic microsatellite loci. Fish were captured by electrofishing in stream reaches ranging from 250 m to 1 km in length to minimize the sampling of related individuals. Total lengths were recorded, and a portion of the fish tissue was excised and stored in a 95% solution of ethanol. The vast majority of sampled trout were less than 200 mm in length (i.e., age 1 and age 2). Population admixture was calculated as the proportion of nonnative rainbow trout alleles found among individuals within a population. Hybridization was declared present in a tributary if rainbow trout alleles were detected in the sample at one or more loci. A sample was considered to consist of nonhybridized westslope cutthroat trout if no rainbow trout alleles were detected; the power to detect rainbow trout genetic contributions as small as 1% in a hybrid swarm was at least 0.94 with our techniques (Boecklen and Howard 1997).

Biotic variables.—We examined the influence of two biotic metrics, trout density and distance to the source of hybridization, on the occurrence and degree of hybridization. We considered the distance to Abbot Creek as a measure of stream connectivity to the source of hybridization in the system and trout density as a measure of the influence of demographic support in facilitating or reducing the likelihood of hybridization at each site. The stream distance from the mouth of Abbot Creek to each sample site was measured in ArcGIS 9.2 (ESRI, Redlands, California).

Trout densities were estimated in the same sections 1–2 years after the genetics sampling. Abundance estimates were conducted in 150-m sections using the multiple-pass depletion method (Zippin 1958). A hydrologic break (e.g., a riffle or vertical drop) was selected for the upper boundary, and a block net (12.7-mm mesh) was placed across the channel at the lower boundary before sampling. A minimum of three passes were completed in each section with one or two backpack electrofishing units (Smith-Root Model 15–D) working from the upstream boundary downstream to the block net. The total lengths (mm) of all captured trout were recorded. Based on length-at-age data for the upper Flathead River system (C. Muhlfeld, unpublished data), individuals less than 75 mm were considered young-of-the-year fish. These individuals were not included in the abundance estimates owing to poor sampling efficiency and variable emergence times across streams. Ten wetted widths were systematically taken every 15 m through the sample section and were used to calculate the wetted stream surface area. Population estimates were calculated using the depletion model in the MICROFISH 3.0 computer program (Van Deventer and Platts 1985), which estimates abundance from the counts and capture probabilities derived from the multi-pass sampling. Although the removal method typically produces biased and variably underestimated population density or abundance estimates, we accounted for this by maintaining similar capture probabilities across sites. Trout density (fish/m^2) was calculated by dividing the estimate of fish population by the wetted stream surface area. At 11 of the 35 sites, abundance was estimated in more than one year. In these situations, we averaged the densities across years. Georeferenced locations were obtained at

TABLE 1.—Summary of the local habitat features, watershed characteristics, and biotic factors in each study site in the upper Flathead River drainage.

Site		Local habitat features			Watershed characteristics			
Name	Number	Gradient (%)	Elevation (m)	Stream width (m)	Maximum temperature (°C)	Mean summer temperature (°C)	Road density	Road crossings
Abbott	1	0.01	950	2.64	20.2	15.00	0.70	24
Ivy	2	0.07	977	2.03	12.6	10.60	0.50	4
Rabe	3	0.04	996	3.42	16.2	12.40	1.20	11
Third	4	0.03	962	2.18	13.3	10.70	0.07	0
Langford	5	0.02	1,130	2.56	10.8	9.40	0.43	8
Meadow	6	0.03	1,134	2.15	19.4	14.00	0.24	4
Skookoleel	7	0.08	1,200	6.10	11.8	8.70	0.49	6
Nicola	8	0.07	1,280	3.90	9.7	7.60	1.32	9
Werner	9	0.06	1,303	5.98	11.1	8.30	1.22	9
Kletomus	10	0.09	1,390	4.20	12.5	9.10	0.45	0
Cyclone, lower	11	0.02	1,260	3.78	18.6	13.10	0.83	14
Cyclone, upper	12	0.07	1,430	11.67			0.44	3
Deadhorse	13	0.04	1,260	3.60	13.7	9.80	0.42	6
North Fork Coal	14	0.03	1,259	2.30	13.9	10.00	0.71	33
South Fork Coal	15	0.05	1,340	6.50	14.1	10.20	0.45	6
Anaconda	16	0.05	1,110	5.05	16.3	12.00	0.02	1
Dutch	17	0.02	1,110	4.71	16.8	12.60	0.03	1
Moran	18	0.05	1,230	3.70	13.2	9.60	0.65	9
Hay Creek, lower	19	0.02	1,090	6.70	13.0	10.10	0.42	14
Hay Creek, upper	20	0.04	1,430	6.70	11.2	8.50	0.32	5
Akokala	21	0.03	1,340	6.30	14.8	10.90	0.03	0
South Fork Red Meadow	22	0.03	1,240	2.40	12.3	9.20	0.31	2
Red Meadow	23	0.03	1,150	7.40	15.5	11.90	0.59	31
Hawk	24	0.02	1,176	1.30	14.6	10.10	1.56	5
Moose	25	0.02	1,130	4.00	10.1	7.60	0.52	11
Ford	26	0.03	1,154	4.03	15.9	11.20	0.00	0
Tepee	27	0.03	1,210	3.80	17.7	11.80	1.05	17
Ketchikan	28	0.02	1,278	3.15	13.8	10.10	0.04	0
Tuchuck	29	0.03	1,536	5.70	12.1	9.10	0.04	1
Colts	30	0.06	1,239	3.78	12.6	9.60	0.24	0
Sage	31	0.00	1,280	13.60	12.7	10.60	0.29	29
Burnham	32	0.05	1,273	3.19	16.5	11.40	0.59	14
Commerce	33	0.02	1,334	5.92	14.3	11.20	0.29	4
Middlepass	34	0.05	1,405	5.04	11.2	9.20	0.16	8
Parker	35	0.04	1,395	4.54	8.4	6.60	0.82	1

the upstream limit of each sample section using a Global Positioning System unit (TSC1 Asset Surveyor; Trimble Navigation, Sunnyvale, California).

Local habitat and watershed variables.—Local habitat features included measures of stream size, gradient, and elevation. Site gradient (measured at the reach scale) and elevation were derived from 1:25,000 U.S. Geological Survey maps using ArcGIS. Mean stream width was calculated as the average of the ten wetted-width measurements collected during the population estimate.

Watershed variables included measures of stream temperature and land disturbance. Thermographs were deployed at each site to record water temperatures hourly during the year in which the abundance estimates were made. The water temperature metrics used were the mean and maximum summer temperatures. The mean summer temperature was calculated as the mean of the daily averages from 1 July to 30 September. The maximum water temperature at each site was the highest recorded temperature during the sampling period. Temperature data were unavailable for one site (upper Cyclone Creek).

Road density metrics were used as indicators of land use disturbance. Roads can alter the hydrologic and geomorphic regimes in downstream areas (Trombulak and Frissell 2000), and measures of road density and stream crossings have been correlated with the spatial extent of timber harvest activity in the Flathead River system (Hauer and Blum 1991). Therefore, we estimated road density and the number of road–stream intersections upstream of each site (Baxter et al. 1999) from the U.S. Forest Service's Flathead National Forest Infrastructure Application (INFRA) database in ArcGIS.

Data analysis.—We first tested for differences ($P <$ 0.10) between the hybridized and nonhybridized sites for each independent variable using Mann–Whitney *U*-tests. We used logistic and linear regression analyses to evaluate the associations between the nine independent

TABLE 1.—Extended.

Site		Biotic factors			
Name	Number	Distance to source (km)	Hybridization present	Trout density (fish/m^2)	% Rainbow trout admixture
Abbott	1	0.0	Yes	0.16	91.6
Ivy	2	6.4	Yes	0.08	49.3
Rabe	3	13.9	Yes	0.22	49.1
Third	4	16.9	Yes	0.19	65.8
Langford	5	40.3	Yes	0.12	33.1
Meadow	6	58.3	Yes	0.06	3.5
Skookoleel	7	54.2	No	0.04	0.0
Nicola	8	55.1	Yes	0.07	1.8
Werner	9	56.0	No	0.08	0.0
Kletomus	10	62.7	No	0.10	0.0
Cyclone, lower	11	59.7	Yes	0.07	11.6
Cyclone, upper	12	59.7	No	0.05	0.0
Deadhorse	13	67.9	No	0.14	0.0
North Fork Coal	14	67.9	Yes	0.23	7.3
South Fork Coal	15	74.6	Yes	0.02	0.6
Anaconda	16	48.3	Yes	0.07	20.6
Dutch	17	49.3	Yes	0.04	13.0
Moran	18	64.4	No	0.06	0.0
Hay Creek, lower	19	64.7	Yes	0.05	1.4
Hay Creek, upper	20	81.0	No	0.07	0.0
Akokala	21	86.8	No	0.01	0.0
South Fork Red Meadow	22	77.2	Yes	0.07	0.3
Red Meadow	23	75.0	Yes	0.15	2.2
Hawk	24	74.1	No	0.10	0.0
Moose	25	89.6	No	0.12	0.0
Ford	26	84.7	No	0.08	0.0
Tepee	27	87.7	Yes	0.05	1.3
Ketchikan	28	103.3	No	0.22	0.0
Tuchuck	29	108.4	No	0.11	0.0
Colts	30	107.0	No	0.08	0.0
Sage	31	114.1	No	0.01	0.0
Burnham	32	116.4	No	0.03	0.0
Commerce	33	130.7	No	0.05	0.0
Middlepass	34	139.5	No	0.05	0.0
Parker	35	143.7	No	0.05	0.0

variables and the occurrence (presence/absence) of hybridization among all study sites and the proportion of rainbow trout admixture among hybridized sites, respectively. First, we attempted to reduce the number of independent variables in the final variable sets to avoid potential model selection biases caused by such large candidate model sets (Ramsey and Schafer 2002; Taper 2004; Kutner et al. 2004). Therefore, for the logistic and linear regression analyses we included all nine variables in both forward and backward stepwise regression processes and included all of the variables selected in the first model selection as the final variable set, regardless of whether or not they were retained in the final model. For the logistic regression analysis, the stepwise model included stream width, mean summer water temperature, the number of road crossings, and the distance to the source of hybridization, whereas in the linear regression analysis mean summer water temperature, stream width, the distance to the source, and trout density were included.

Next, we used all subsets of the logistic and linear regression models (a priori) representing all possible combinations of the four remaining variables in each analysis and employed an information-theoretic approach to evaluate the relative plausibility of the competing models (Burnham and Anderson 2002). A Hosmer–Lemeshow goodness-of-fit test of the global model (including all factors) indicated that the logistic model provided a good fit to the presence/absence data. Therefore, we used Akaike's information criterion (Akaike 1973) with adjustment for small sample size (AIC_c; Hurvich and Tsai 1989) to rank the competing models relative to the one with the lowest score. Models were considered equally plausible if their AIC_c scores were within 2.0 of that of the best model (Burnham and Anderson 2002). The classification cutoff was 0.5 for each logistic model, and all models included a constant and an error term. For the linear regression analysis, it was necessary to perform a logit transformation on the proportion of rainbow trout

TABLE 2.—Model selection results for candidate logistic regression models with various combinations of local habitat features (stream width), watershed characteristics (mean summer water temperature and number of upstream road crossings), and biotic factors (distance to the source of hybridization) in relation to the occurrence of hybridization between native westslope cutthroat trout and nonnative rainbow trout at 35 sites in the upper Flathead River drainage. The number of parameters (k) includes intercept and error terms. Models were ranked according to their corrected Akaike information criterion values (AIC_c).

Model	k	AIC_c	ΔAIC_c
Mean temperature, distance to source, number of road crossings	5	28.98	0.00
Mean temperature, distance to source	4	29.02	0.04
Mean temperature, distance to source, width	5	32.02	3.04
Mean temperature, distance to source, number of road crossings, width	6	32.12	3.14
Distance to source, number of road crossings	4	32.67	3.69
Distance to source, number of road crossings, width	5	33.48	4.50
Distance to source	3	35.32	6.34
Distance to source, width	4	36.73	7.75
Mean temperature	3	42.05	13.07
Mean temperature, number of road crossings, width	5	42.46	13.48
Mean temperature, width	4	42.70	13.72
Mean temperature, number of road crossings	4	43.46	14.48
Number of road crossings, width	4	45.35	16.37
Width	3	48.69	19.71
Number of road crossings	3	49.28	20.30

admixture in order to meet the assumptions of normality and homogeneity of variance and account for the correct variation behavior of the proportional data (i.e., using multiple alleles across all fish in each sample). The final variable selection and model development followed the same procedures as for the logistic regression analysis.

Results

A total of 971 individuals were collected from 35 locations in 33 streams (mean per stream, 28; SD, 7). Nineteen of the 35 locations (54%) showed no evidence of rainbow trout introgression (Table 1; Figure 1). Streams with hybrid populations were smaller (mean width, 3.9 m; range, 2.0–7.4 m) and lower in elevation (mean, 1,137 m; range, 950–1,280 m) than streams with nonhybridized westslope cutthroat trout (mean width, 5.4 m; range, 1.3–13.6 m [Mann-Whitney $U = 97.5$, $P = 0.07$]; mean elevation, 1,304 m; range, 1,130–1,536 m [$U = 47.0$, $P < 0.01$]). The mean and maximum summer temperatures were significantly higher in streams with hybrids. The mean water temperature was 11.5°C (range, 7.6–15.0°C) in streams containing hybrid populations and 9.6°C (range, 6.6–11.4°C) in streams with nonhybridized westslope cutthroat trout populations ($U = 63.5$, $P < 0.01$); the maximum temperature averaged 20.2°C among hybridized populations, versus 16.5°C among nonhybridized populations ($U = 82.5$, $P = 0.03$). Hybrid populations occurred in streams with significantly ($U = 99.5$; $P = 0.08$) more upstream road intersections (mean, 11; range, 0–33) than those containing westslope cutthroat trout (mean, 6; range, 0–29), but no differences were detected for road density ($U = 123.5$; $P = 0.35$). No differences in gradient were found between streams occupied by hybrid trout (mean, 0.04; range, 0.01–0.07) and nonhybridized westslope cutthroat trout (mean, 0.04; range, 0.01–0.09; $U = 110.5$, $P = 0.17$). The same was true for the density of trout (nonhybridized sites: mean = 0.076 fish/m^2; hybridized sites: mean = 0.103 fish/m^2) ($U = 116$, $P = 0.24$).

The best-approximating logistic regression model contained the watershed variables mean summer water temperature and number of upstream road crossings in combination with the biotic variable distance to the source of hybridization (Table 2). However, one other model (with the variables mean summer water temperature and distance to the source of hybridization) was equally plausible. Both models had overall classification accuracies greater than 85%. The occurrence of hybridized trout was positively associated with mean summer water temperature and the number of

TABLE 3.—Coefficients (B) and standard errors (SEs) for the two most plausible logistic regression models of the occurrence of hybridization between native westslope cutthroat trout and nonnative rainbow trout in the upper Flathead River drainage (see Table 2).

Variable	B	SE
Model 1		
Mean temperature	0.955	0.56
Distance to source	−0.103	0.043
Number of road crossings	0.128	0.086
Constant	−3.532	4.64
Model 2		
Mean temperature	1.104	0.584
Distance to source	−0.099	0.047
Constant	−4.131	4.518

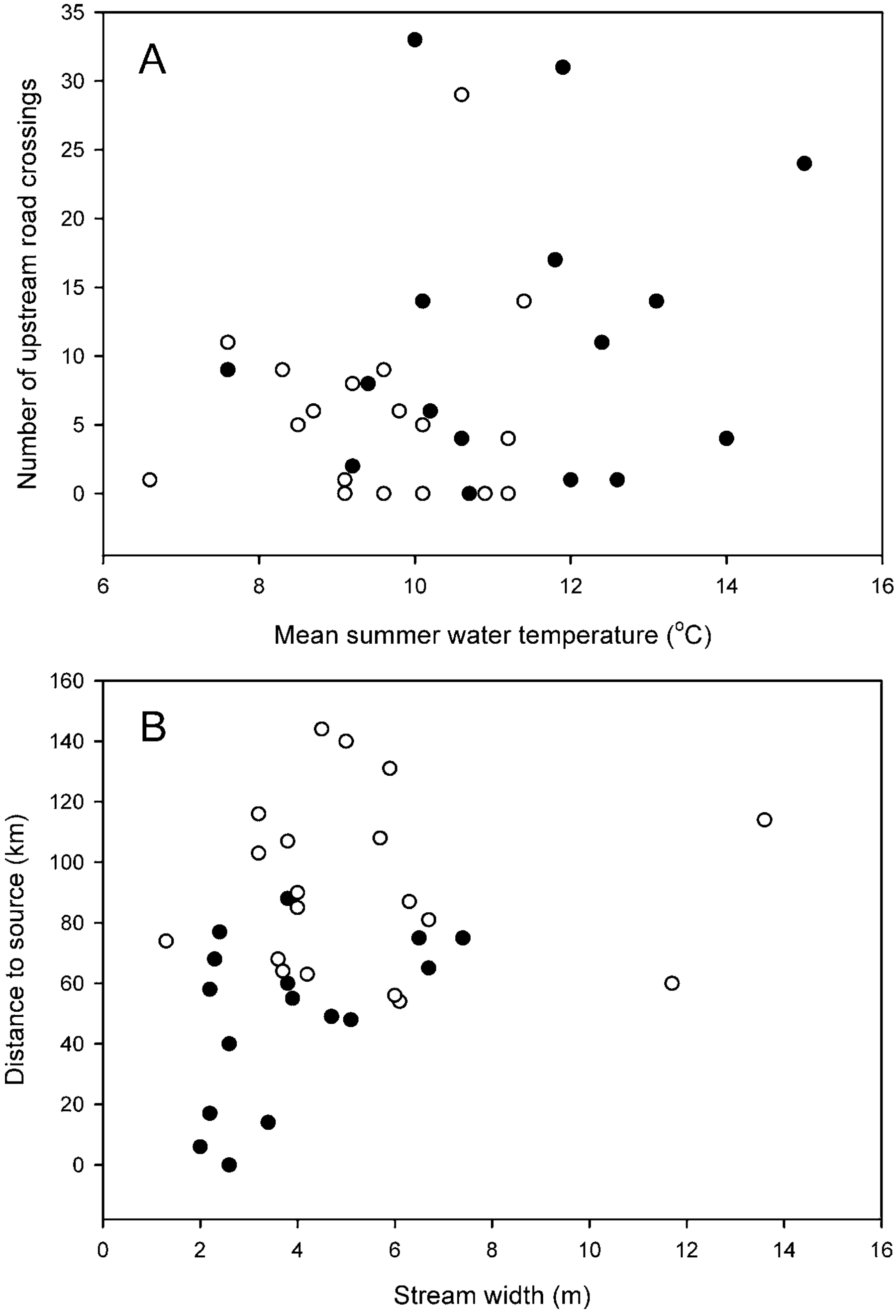

FIGURE 2.—Presence (closed circles) and absence (open circles) of hybridization between native westslope cutthroat trout and nonnative rainbow trout in relation to **(A)** mean summer water temperature and the number of upstream road crossings and **(B)** stream width and the distance from the source of hybridized individuals.

upstream road crossings and negatively associated with the distance to the source of hybridization (Table 3; Figure 2). The best-approximating linear regression models associated with the level of introgression among hybridized sites included summer water temperature, distance to the source, and trout density (Table 4). The proportion of rainbow trout admixture (logit transformed) was negatively related to the

TABLE 4.—Model selection results for candidate linear regression models of the proportion of nonnative genetic admixture between native westslope cutthroat trout and nonnative rainbow trout in 16 hybridized streams in the upper Flathead River drainage. Density refers to the density of both trout species at the sampling site; other variables are explained in Table 2.

Model	k	AIC_c	ΔAIC_c
Mean temperature, distance to source	4	48.871	0
Mean temperature, distance to source, density	5	49.169	0.298
Distance to source	3	49.518	0.647
Distance to source, density	4	50.494	1.623
Mean temperature, distance to source, width	5	52.958	4.087
Distance to source, width	4	53.093	4.222
Mean temperature, distance to source, density, width	6	54.455	5.584
Distance to source, density, width	5	54.856	5.985
Mean temperature, density	4	72.573	23.702
Density	3	73.205	24.334
Mean temperature, density, width	5	74.563	25.692
Mean temperature, width	4	74.619	25.748
Density, width	4	74.965	26.094
Width	3	75.264	26.393
Mean temperature	3	75.284	26.413

distance to the source and positively related to mean temperature and density (Table 5). The occurrence and amount of introgression was negatively related to stream width in both regression analyses (Figure 3), although it was not included in the top models.

Discussion

Conservation of aquatic biodiversity requires an understanding of the invasion patterns and factors promoting extinction by hybridization. We evaluated the influence of local habitat features, large-scale watershed characteristics, and biotic factors associated with the spread of hybridization between an introduced, nonnative species and a native species of conservation concern. Our results provide evidence supporting the hypothesis that hybridization is more likely to occur and spread in streams with warm water temperatures, increased land use disturbance, and proximity to the main source of hybridization. Our findings provide fisheries managers with a better understanding of the factors that promote the success of invasions and the loss of biodiversity through extinction by hybridization.

TABLE 5.—Model coefficients (B) and standard errors (SEs) for the four most plausible linear regression models of the proportion of rainbow trout admixture in the upper Flathead River drainage (see Table 4).

Variable	B	SE
Model 1		
Mean temperature	0.242	0.121
Distance to source	−0.072	0.009
Constant	−1.39	1.559
Model 2		
Mean temperature	0.251	0.111
Distance to source	−0.065	0.009
Density	6.376	3.422
Constant	−2.512	1.551
Model 3		
Distance to source	−0.077	0.009
Constant	1.586	0.506
Model 4		
Distance to source	−0.071	0.01
Density	6.017	3.923
Constant	0.639	0.784

Our results are concordant with those of other studies in Europe and North America that have found that invasion success is often facilitated by a complex interaction of many abiotic and biotic factors, such as local habitat conditions, temperature, connectivity, human influences, and biotic resistance (Paul and Post 2001; Dunham et al. 2002; Rich et al. 2003; Kitano 2004; Carveth et al. 2006; Jeschke and Strayer 2006). However, the relative influences of these factors vary among geographical areas and hybrid zones of native cutthroat trout and nonnative rainbow trout. For example, Rubidge and Taylor (2005) showed that the level of hybridization decreased with increasing distance from Koocanusa Reservoir in British Columbia (indicating that the reservoir acts as a source of rainbow trout) but found no evidence that stream order, magnitude, or elevation influenced the extent of hybridization among localities. Conversely, Weigel et al. (2003) found evidence of ecological barriers (e.g., water temperature) restricting the spread of hybridization between westslope cutthroat trout and nonnative rainbow trout in the Clearwater River system of Idaho. These authors found that many tributaries located close to the original stocking locations did not contain hybridized populations and that the degree of hybridization showed negative associations with site elevation and positive associations with stream width. In contrast, Gunnell et al. (2008) found that the primary factor influencing the geographic distribution of rainbow trout introgression with native Yellowstone cutthroat trout (*O. clarkii bouvieri*) was fluvial distance from the stocking locations and, to a lesser extent, stream elevation. In our study, the independent variables site elevation, distance to the source, and water temperature were correlated, making it difficult to ascertain the relative effects of each variable on the geographic distribution of hybridization in the system.

Source proximity strongly influenced the occurrence and proportion of rainbow trout admixture across the large, interconnected drainage. Other studies have also shown that invasion success and hybridization are largely governed by the spatial arrangement of nonnative source populations. For instance, in the upper Kootenay River drainage in British Columbia, Rubidge and Taylor (2005) showed clustering among hybridized locations and decreasing hybridization with increasing distance from Koocanusa Reservoir. Likewise, Gunnell et al. (2008) found that rainbow trout introgression with native Yellowstone cutthroat trout declined with distance from a known rainbow trout stocking source. We found that the distance to the source of hybridization appears to be the leading factor associated with the presence of hybridization and the amount of rainbow trout admixture among hybridized sites. However, the spread of hybridization was also influenced by the additive effects of temperature and a measure of land use degradation, suggesting that hybridization is facilitated by a complex combination of spatial and environmental factors in this westslope cutthroat trout–rainbow trout hybrid zone.

The observed genotypic gradient of decreasing hybrid occurrence and decreasing levels of introgression from main-stem sites to upper-elevation tributaries is consistent with the results of several studies in other rainbow trout–cutthroat trout hybrid zones (Kruse et al. 2000; Weigel et al. 2003; Rubidge and Taylor 2005; Gunnell et al. 2008). Elevation was significantly related to the occurrence and level of hybridization, but we expected this to be the case since site elevation increased with increasing distance from the main source of hybridization (Abbot Creek) and elevation is strongly correlated with water temperature. Similarly, Weigel et al. (2003) found that the level of introgression between introduced rainbow trout and native westslope cutthroat trout was negatively related to elevation. In contrast, Rubidge and Taylor (2005) found no evidence that stream size and elevation influence the extent of hybridization among sites in the Kootenay River drainage, although their study streams did not include the wide range of site elevations included in ours.

Hybridization was more likely at warmer sites and the level of rainbow trout introgression was positively related to mean summer water temperature, suggesting that temperature is also a noteworthy factor promoting invasion by nonnative rainbow trout and hybridization with native cutthroat trout. Water temperature plays an important role in determining the distribution of many stream-dwelling salmonid species owing to its direct effects on physiology, behavior, and ecological interactions (Paul and Post 2001; Dunham et al. 2003; Carveth et al. 2006; McMahon et al. 2007). Indeed, water temperature has an important influence on the distribution and abundance of westslope cutthroat trout throughout their current range (Shepard et al. 2005). For example, Sloat et al. (2005) found that westslope cutthroat trout resided in streams with cool water temperatures (maximum daily temperature, $\leq$16.5°C) in the Madison River drainage in Montana, which is nearly identical to our findings in the Flathead River system (mean maximum temperature, 16.5°C). Although westslope cutthroat trout and rainbow trout have nearly identical optimum growth temperatures, rainbow trout have higher upper tolerance limits and grow over a wider range of temperatures than westslope cutthroat trout, according to a laboratory study (Bear et al. 2007). Also, rainbow trout are native to lower-elevation systems along the Pacific coast (MacCrimmon 1971). These temperature relationships, therefore, may account for the displacement of native cutthroat trout by nonnative rainbow trout in montane systems.

We detected a general pattern of nonhybridized populations persisting in colder, headwater streams at higher elevations than those occupied by hybridized populations. These results are consistent with those of other studies that have examined the genetic distribution of hybridization in situations in which previously allopatric populations of nonnative rainbow trout and native westslope cutthroat trout have come into artificial secondary contact (Weigel et al. 2003; Rubidge and Taylor 2005). At first glance, the data appear to support the elevation refugia hypothesis. That is, cold temperatures in headwater streams impart a competitive advantage to native salmonids and thus account for the greater resistance to invasion and the displacement of nonnatives (Paul and Post 2001; McMahon et al. 2007). However, the overlap in temperature regimes and local habitat conditions among sites with and without hybrids and the significant association between the amount of admixture and the distance to the source of hybridization suggest that headwater streams will not provide a refuge from hybridization if the sources of hybridization persist and spread in the system. Furthermore, because westslope cutthroat trout and rainbow trout

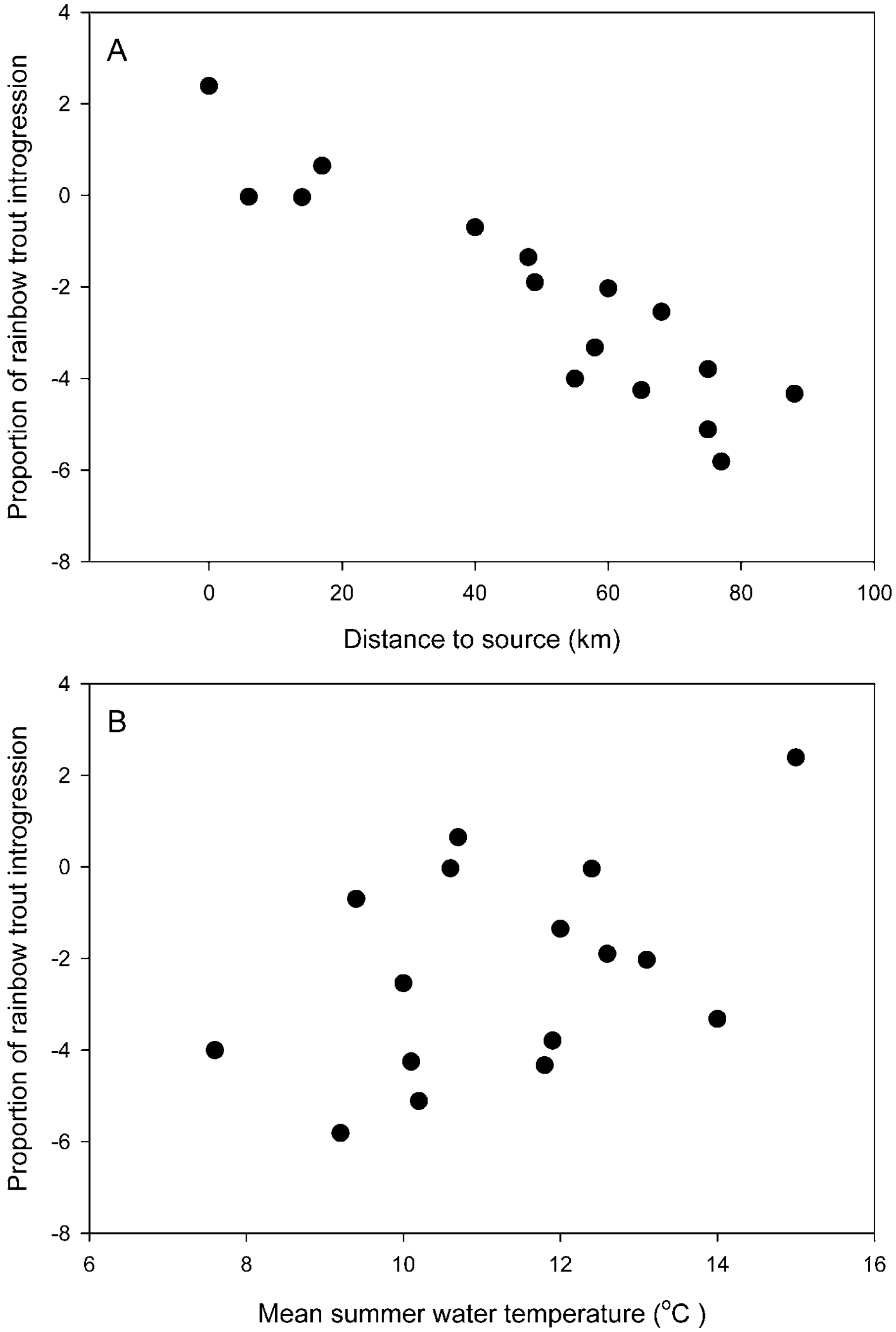

FIGURE 3.—Proportion of rainbow trout introgression (logit transformed) versus **(A)** the distance from the source of hybridized individuals, **(B)** mean summer water temperature, **(C)** trout density, and **(D)** stream width.

have virtually identical optimum growth temperatures in the laboratory (13.68°C and 13.18°C, respectively; Bear et al. 2007), temperature alone may not prevent or slow the spread of hybridization. Additional research is needed to compare the thermal preferences of hybrids with those of both parental species in natural environments.

The association between the presence of hybridiza-

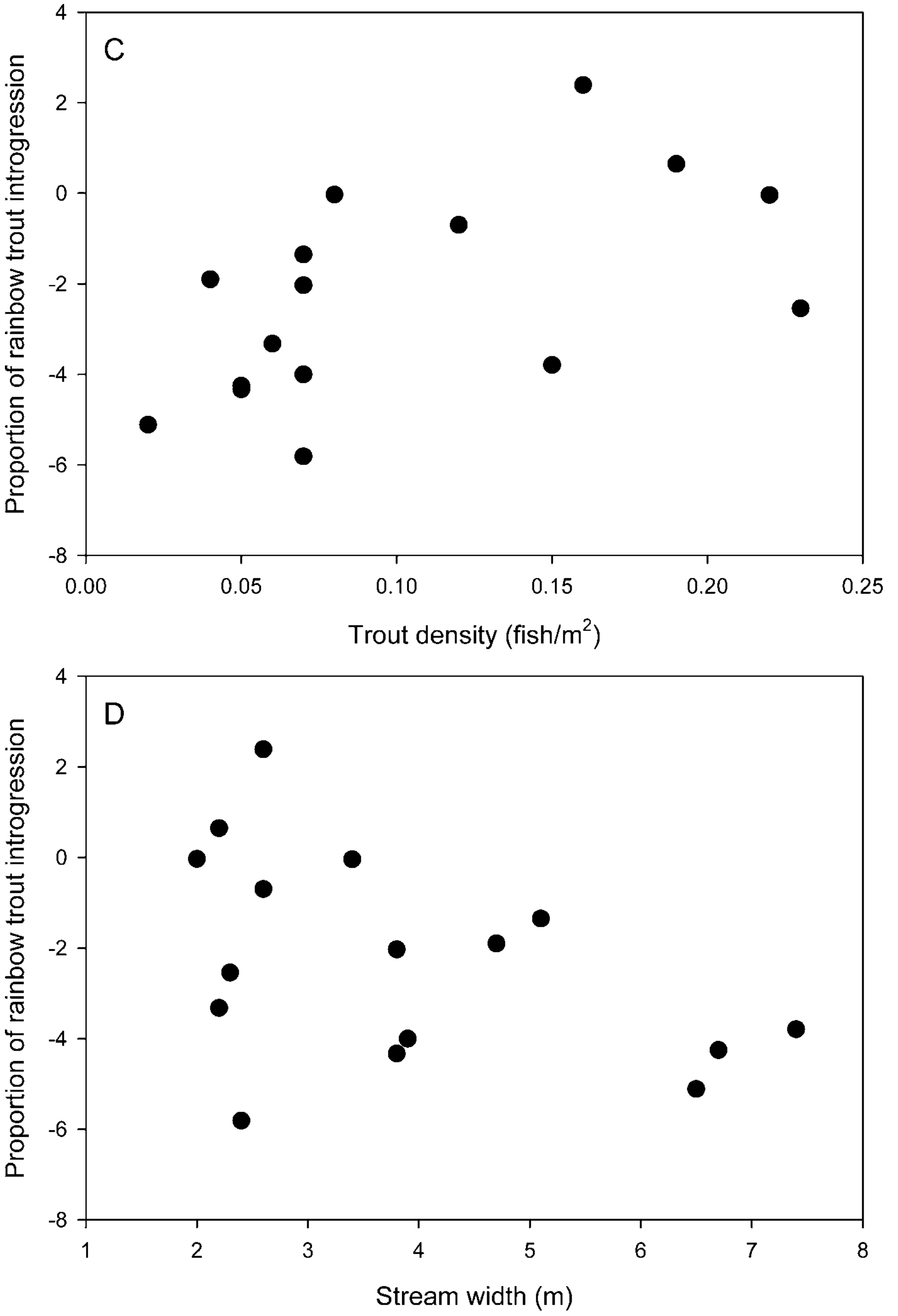

FIGURE 3.—Continued.

tion and the number of upstream road crossings suggests that hybridization is also more likely in streams with increased disturbance. Similarly, Shepard (2004) found that invasive brook trout *Salvelinus fontinalis* displaced native westslope cutthroat trout in a southwestern Montana stream with higher water temperatures, lower frequencies of debris and pools, and greater erosion and deposition of fine sediments than two adjacent, undisturbed streams. Land use disturbances can make systems more prone to the successful invasion of nonnative competitors by changing the availability and quality of habitats, which may result in the displacement or complete replacement of native taxa (Allendorf et al. 2001; Jeschke and Strayer 2006). This has been observed for a variety of salmonid species (Taylor et al. 1984; Fausch et al. 2001; Paul and Post 2001; Shepard 2004) as well as other vertebrate species (Haig et al. 2004; Schwartz et al. 2004) and many plant species (Arnold 1997). Road metrics are often used as a surrogate for habitat disturbance because they may negatively impact salmonid populations by increasing stream sediment loads; obstructing fish movements; degrading spawning, rearing, and reproductive habitats; and providing vectors for fishing pressure and the stocking of nonnative species (Meehan 1991; Trombulak and Frissell 2000; McCaffery et al. 2007). Other disturbances unaccounted for in this study, such as drought, wildfire, and flooding, may also affect the invasion success of rainbow trout in novel environments. Fausch et al. (2001) concluded that flood disturbance regimes strongly influenced the invasion success of rainbow trout in five Holarctic regions with similar temperature regimes. These authors showed that invasive rainbow trout are more successful when fry emerge in areas and periods of low flood probability as opposed to areas with summer and fall floods that wash them away.

Hybrid zones are areas of contact between genetically distinct populations where hybridization occurs and are formed and maintained by selection and dispersal (Barton and Hewitt 1989). Our results suggest that the dispersal of hybridized individuals from hybrid source populations is a significant factor in the spread of hybridization in the upper Flathead River system; these results are corroborated by a recent telemetry study (Muhlfeld et al. 2009b) and fine-scale genetic analyses (Boyer et al. 2008). However, Muhlfeld et al. (2009a) found that hybridization rapidly reduces fitness in later-generation westslope cutthroat trout–rainbow trout hybrids in a stream in the North Fork Flathead River (Langford Creek; site 5 in this study). Despite the apparent occurrence of outbreeding depression, the authors concluded that hybridization may still spread because of (1) the relatively high reproductive success of first-generation hybrids and a few males with high levels of admixture; (2) higher straying rates in rainbow trout (Boyer et al. 2008); and (3) and the fact that all of the progeny of hybrids are hybrids (Epifanio and Philipp 2001). Thus, source connectivity and dispersal barriers probably explain the distribution patterns of hybridization in this system.

We found that the proportion of rainbow trout admixture among hybridized sites was positively related to the abundance of trout. Although biotic resistance from native species may limit the extent to which competitors become established (Pimm 1989), we hypothesized that sites with higher densities would be more susceptible to hybridization because rainbow trout and westslope cutthroat trout have similar mating behaviors that would facilitate interbreeding and the formation of hybrid swarms. Alternatively, our results suggest that introgressed populations have a fitness advantage. However, we do not believe that this is the case because fish densities were not significantly different among hybridized and pure sites and our recent work indicates that hybridization rapidly reduces fitness in later-generation hybrids (Muhlfeld et al. 2009a). Additional research is needed to understand the demographic and ecological consequences of hybridization in old and new hybrid swarms in a variety of stream environments.

Covariation among spatial and environmental variables and stocking history precludes us from making definitive conclusions regarding the relative influences of factors that limit or promote the spread of hybridization. This is a problem in many invasive species studies owing to the increase in the prevalence of unauthorized introductions of nonnative species (Rahel 2000). The sites where disturbance was more common and temperatures were warmer were also closer to the source of hybridization than the colder, less disturbed sites in the headwaters. Additionally, the purported illegal release of an estimated 70,000 rainbow trout in 1997 from a private hatchery in the lower portion of the drainage probably played a significant role in the recent proliferation and current distribution of hybridized trout in the system, and many studies have shown that propagule pressure (the number and frequency of introduced individuals) plays an important role in the establishment and spread of exotic species (see Lockwood et al. 2006 for a review). The observed spatial distribution of hybridization may not be entirely the result of stocking history, however, as rainbow trout tend to establish population strongholds at low-elevation sites in the drainages into which they are introduced. For example, Weigel et al. (2003) found a pattern of elevational separation between native westslope cutthroat trout and nonnative rainbow trout despite the fact that the rainbow trout had access to all of the sampling locations in their study area. Similarly, Paul and Post (2001) showed that rainbow trout stocked extensively over a wide range of elevations on the eastern slopes of the Rocky Mountains generally moved downstream and hybridized with and displaced native cutthroat trout populations.

Management Implications and Conclusions

Our data suggest that westslope cutthroat trout are particularly susceptible to hybridization with nonnative rainbow trout in situations in which anthropogenic habitat disturbances increase water temperature and degrade stream habitats. Habitat degradation and fragmentation have been identified as leading factors in the decline and extirpation of westslope cutthroat trout populations throughout their range (Shepard et al. 2005). Currently, the headwaters of the North Fork Flathead River in British Columbia have been targeted for coal bed methane development and open-pit coal mining. Our research shows that this area supports the majority of the remaining nonhybridized westslope cutthroat trout populations in the transboundary system (Boyer et al. 2008; Muhlfeld et al., in press; this study). Thus, protection of pure migratory populations and their critical spawning and rearing habitats in the headwater portion of the drainage are critical to the long-term persistence of nonhybridized westslope cutthroat trout populations and migratory life history forms in the Flathead River and similar freshwater systems.

The petition to list the westslope cutthroat trout as a threatened species under the Endangered Species Act was recently denied because the subspecies is widely distributed, numerous nonintrogressed westslope cutthroat trout populations are distributed in secure habitats throughout the subspecies' historic range, and numerous westslope cutthroat trout are nonintrogressed or nearly so. (USFWS 2003).

Although headwater streams currently contain nonhybridized westslope cutthroat trout populations, our data suggest that habitat conditions alone are not sufficient to maintain "secure" habitats in open systems and that headwater streams will not provide refuge from hybridization if the sources of hybridization persist in this open system. Indeed, many studies have found that pure cutthroat trout populations only persist above upstream migration barriers in situations in which nonnative rainbow trout have been introduced (Sloat et al. 2005; Ostberg and Rodriguez 2006; Metcalf et al. 2008).

Our results indicate (1) that hybridization is likely to spread further, causing additional westslope cutthroat trout populations to be lost, unless populations with high amounts of rainbow trout admixture are suppressed or eliminated and (2) that the protection of hybridized populations facilitates the expansion of hybridization. To preserve nonhybridized westslope cutthroat trout populations, managers should consider eradicating hybridized populations with high levels of rainbow trout admixture and restoring streams characterized by warm temperatures and high levels of disturbance.

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BIOLOGICAL EFFECTS OF SEDIMENT ON BULL TROUT AND THEIR HABITAT –

GUIDANCE FOR EVALUATING EFFECTS

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BIOLOGICAL EFFECTS OF SEDIMENT ON BULL TROUT AND THEIR HABITAT

Anthropogenic sediment input into water bodies can have a variety of impacts to fish species from behavioral effects such as avoidance or abandonment of cover to lethal effects. The Washington Fish and Wildlife Office reviews numerous projects where sediment is generated during construction. A scientific approach was needed to determine the concentration and duration of sediment input where adverse effects of project-related sediment would occur.

The following document addresses the biological effects of sediment on bull trout and their habitat. The document is divided into two sections:

1. A literature review on the biological effect of sediment on fish (Page 3).
2. Effects analysis for project related sediment input (Page 23).

The literature review addresses the different types of sediment and the biological effects on bull trout. Direct effects include gill trauma and impacts to spawning, redds, eggs, and alevins. Indirect effects include impacts to macroinvertebrates, feeding efficiency, habitat, physiological stress, and behavioral changes.

The effects analysis section provides a step-by-step process to determine the concentration and duration of sediment input to a stream where adverse affects occur. Newcombe and Jensen (1996) and Anderson et al (1996) provide the basis for the analyzing sediment effects to bull trout and their habitat.

Introduction

As a stream or river flows downslope, it transports sediment and dissolved matter (Skinner and Porter 2000, p. 252). A stream has a natural amount of sediment that is transported through the system that varies throughout the year in response to natural hydrological changes (Galbraith et al. 2006, p. 2488). The amount of sediment that a stream can transport annually is based on numerous factors: precipitation, surface water transport, erosion, topography, geology, streamflow, riparian vegetation, stream geomorphologic characteristic, human disturbance, atmospheric deposition, etc. (Bash et al. 2001o, p. 7;Berry et al. 2003, p. 7). Therefore, different watersheds will have different levels or concentrations of turbidity and suspended sediment. A glaciated stream will have higher sediment levels than a spring fed stream (Uehlinger et al. 2002, p. 1;Ahearn 2002, p. 2).

Many watersheds are subject to anthropogenic disturbances that can produce substantial inputs of sediments into streams (Barrett et al. 1992, p. 437). Turbidity, suspended solids, sediment, and siltation have been consistently listed as impairments in the U.S. Environmental Protection Agency's (EPA) 305(b) water quality reports in rivers and streams, lakes, reservoirs, ponds, wetlands, and oceans shoreline waters (Berry, Rubinstein, Melzian, and Hill 2003, p. 4). The EPA's 305(b) list provides the U.S. Congress and the public a means of determining or assessing the current condition of water quality within each individual state. Excessive sedimentation, natural and anthropogenic, has been estimated to occur in 46 percent of all streams and rivers in the U.S. and is considered the most important factor limiting fish habitat and causing water quality impairment (Judy et al. 1984 as cited in Henley et al. 2000, p. 126;Berry, Rubinstein, Melzian, and Hill 2003, pp. 4, 7). One of the most pervasive influences of land-use activities on stream ecosystems is an increase in sediment yield resulting from point source discharges associated with in-stream activities (Suren and Jowett 2001, p. 725).

Aquatic organisms have adapted to the natural variation in sediment load that occurs seasonally within the stream (ACMRR/IABO Working Party on Ecological Indices of Stress to Fishery Resources 1976, pp. 13, 15;Birtwell 1999, p. 7). Field experiments have found a thirty-fold increase in salmonids' (coho salmon) tolerance to suspended solids between August and November when naturally occurring concentrations are expected to be high (Cederholm and Reid 1987, p. 388).

The introduction of sediment in excess of natural amounts can have multiple adverse effects on bull trout and their habitat (Rhodes et al. 1994, pp. 16-21;Berry, Rubinstein, Melzian, and Hill 2003, p. 7). The effect of sediment beyond natural background conditions can be fatal at high levels. Embryo survival and subsequent fry emergence success have been highly correlated to percentage of fine material within the streambed (Shepard et al. 1984, pp. 146, 152). Low levels of sediment may result in sublethal and behavioral effects such as increased activity, stress, and emigration rates; loss or reduction of foraging capability; reduced growth and resistance to disease; physical abrasion; clogging of gills; and interference with orientation in homing and migration (McLeay et al. 1987a, p. 671;Newcombe and MacDonald 1991, pp. 72, 76, 77;Barrett, Grossman, and Rosenfeld 1992, p. 437;Lake and Hinch 1999, p. 865;Bash et al. 2001n, p. 9;Watts et al. 2003, p. 551;Vondracek et al. 2003, p. 1005;Berry, Rubinstein, Melzian, and Hill

2003, p. 33). The effects of increased suspended sediments can cause changes in the abundance and/or type of food organisms, alterations in fish habitat, and long-term impacts to fish populations (Anderson et al. 1996, pp. 1, 9, 12, 14, 15;Reid and Anderson 1999, pp. 1, 7-15). No threshold has been determined in which fine-sediment addition to a stream is harmless (Suttle et al. 2004, p. 973). Even at low concentrations, fine-sediment deposition can decrease growth and survival of juvenile salmonids.

Aquatic systems are complex interactive systems, and isolating the effects of sediment to fish is difficult (Castro and Reckendorf 1995d, pp. 2-3). The effects of sediment on receiving water ecosystems are complex and multi-dimensional, and further compounded by the fact that sediment flux is a natural and vital process for aquatic systems (Berry, Rubinstein, Melzian, and Hill 2003, p. 4). Environmental factors that affect the magnitude of sediment impacts on salmonids include duration of exposure, frequency of exposure, toxicity, temperature, life stage of fish, angularity and size of particle, severity/magnitude of pulse, time of occurrence, general condition of biota, and availability of and access to refugia (Bash et al. 2001m, p. 11). Potential impacts caused by excessive suspended sediments are varied and complex and are often masked by other concurrent activities (Newcombe 2003, p. 530). The difficulty in determining which environmental variables act as limiting factors has made it difficult to establish the specific effects of sediment impacts on fish (Chapman 1988, p. 2). For example, excess fines in spawning gravels may not lead to smaller populations of adults if the amount of juvenile winter habitat limits the number of juveniles that reach adulthood. Often there are multiple independent variables with complex inter-relationships that can influence population size.

The ecological dominance of a given species is often determined by environmental variables. A chronic input of sediment could tip the ecological balance in favor of one species in mixed salmonid populations or in species communities composed of salmonids and nonsalmonids (Everest et al. 1987, p. 120). Bull trout have more spatially restrictive biological requirements at the individual and population levels than other salmonids (USFWS (U.S. Fish and Wildlife Service) 1998, p. 5). Therefore, they are especially vulnerable to environmental changes such as sediment deposition.

Bull trout are apex predators that prey on a variety of species including terrestrial and aquatic insects and fish (Rieman and McIntyre 1993, p. 3). Fish are common in the diet of individual bull trout that are over 110 millimeters or longer. Large bull trout may feed almost exclusively on fish. Therefore, when analyzing impacts of sediment on bull trout, it is very important to consider other fish species that are part of their prey base. While sediment may not directly impact bull trout, the increased sediment input may affect the spawning and population levels of Chinook and coho salmon, cutthroat trout, and steelhead, or other species that are potential prey for bull trout. The following effects of sediment are not specific to bull trout alone. All salmonids can be affected similarly.

This document identifies the biological effects of sediment on fish and their habitat including the different life stage(s) affected by sediment input. It also provides an analysis to determine the level of sediment concentrations and duration that results in adverse effects to bull trout (and all salmonids) and their habitat.

Sediment Classifications and Definitions

Sediment within a stream can be classified into a variety of categories: turbidity, suspended sediment, bedload, deposited sediment, and wash load (Waters 1995, pp. 13-14;Bash et al. 2001l, pp. 3-4). Sediment category definitions include:

- Turbidity - Optical property of water which results from the suspended and dissolved materials in the water. This causes light to be scattered rather than transmitted in straight lines. Turbidity is measured in nephelometric turbidity units (NTUs). Measurements of turbidity can quickly estimate the amount of sediment within a sample of water.
- Suspended sediment - Represents the actual measure of mineral and organic particles transported in the water column. Suspended sediment is measured in mg/L and is an important measure of erosion, and is linked to the transport of nutrients, metals, and industrial and agricultural chemicals through the river system.
- Bedload - Consists of larger particles on the stream bottom that move by sliding, rolling, or saltating along the substrate surface. Bedload is measured in tons/day, or tons/year.
- Deposited sediment - The intermediate sized sediment particles that settle out of the water column in slack or slower moving water. Based on water velocity and turbulence, these intermediate size particles may be suspended sediment or bedload.
- Wash load - Finest particles in the suspended load that are continuously maintained in suspension by the flow turbulence. Therefore significant quantities are not deposited in the bed.

Suspended sediment, turbidity, and deposited sediment are not associated with specific particle sizes, as there will be considerable overlap depending on velocity, turbulence, and gradient (MacDonald et al. 1991, p. 98;Waters 1995, p. 14). Turbidity cannot always be correlated with suspended solid concentrations due to the effects of size, shape and refractive index of particles (Bash et al. 2001k, p. 5). Turbidity and suspended sediment affect the light available for photosynthesis, visual capability of aquatic animals, gill abrasion, and physiology of fish. Suspended and deposited sediment affect the habitat available for macroinvertebrates, the quality of gravel for fish spawning, and the amount of habitat for fish rearing (Waters 1995, p. 14).

The size of particles within the stream is also important. The quantity of “fines” within a stream ecosystem is usually associated with the degree of fish population declines (Castro and Reckendorf 1995c, p. 2). Particle diameters less than 6.4 mm are generally defined as “fines” (Bjornn et al. 1977c, p. 1;Shepard, Leathe, Waver, and Enk 1984, p. 148;Hillman et al. 1987, p. 185;Chapman 1988, p. 14;Bjornn and Reiser 1991, p. 103;Rieman and McIntyre 1993, p. 6;Castro and Reckendorf 1995b, p. 2;MBTSG (The Montana Bull Trout Scientific Group) 1998a, p. 8).

Biological Effects of Sediment on Bull Trout

<u>Classification of Sediment Effects</u>

In the absence of detailed local information on population dynamics and habitat use, any increase in the proportion of fines in substrates should be considered a risk to the productivity of an environment and to the persistence of associated bull trout populations (Rieman and McIntyre 1993, p. 6). Specific effects of sediment on fish and their habitat can be put into three classes that include (Newcombe and MacDonald 1991, pp. 72-73;Waters 1995, pp. 81-82;Bash et al. 2001j, p. 10):

Lethal: Direct mortality to any life stage, reduction in egg-to-fry survival, and loss of spawning or rearing habitat. These effects damage the capacity of the bull trout to produce fish and sustain populations.

Sublethal: Reduction in feeding and growth rates, decrease in habitat quality, reduced tolerance to disease and toxicants, respiratory impairment, and physiological stress. While not leading to immediate death, may produce mortalities and population decline over time.

Behavioral: Avoidance and distribution, homing and migration, and foraging and predation. Behavioral effects change the activity patterns or alter the kinds of activity usually associated with an unperturbed environment. Behavior effects may lead to immediate death or population decline or mortality over time.

<u>Direct Effects</u>

Gill trauma

High levels of suspended sediment and turbidity can result in direct mortality of fish by damaging and clogging gills (Curry and MacNeill 2004, p. 140). Fish gills are delicate and easily damaged by abrasive silt particles (Bash et al. 2001i, p. 15). As sediment begins to accumulate in the gill filaments, fish excessively open and close their gills to expunge the silt. If irritation continues, mucus is produced to protect the gill surface, which may impede the circulation of water over the gills and interfere with fish respiration (Bash et al. 2001h, p. 15). Gill flaring or coughing abruptly changes buccal cavity pressure and is a means of clearing the buccal cavity of sediment. Gill sediment accumulation may result when fish become too fatigued to continue clearing particles via the cough reflex (Servizi and Martens 1991a, p. 495).

Fish are more susceptible to increased suspended sediment concentrations at different times of the year or in watersheds with naturally high sediment such as glaciated streams. Fish secrete protective mucous to clean the gills (Erman and Ligon 1985, p. 18). In glaciated systems or during winter and spring high flow conditions when sediment concentrations are naturally high, the secretion of mucous can keep gills clean of sediment. Protective mucous secretions are inadequate during the summer months, when natural sediment levels are low in a stream system. Consequently, sediment introduction at this time may increase the vulnerability of fish to stress and disease (Bash et al. 2001g, p. 12).

Spawning, redds, eggs, and alevins

The effects of suspended sediment, deposited in a redd and potentially reducing water flow and smothering eggs or alevins or impeding fry emergence, are related to sediment particle sizes of the spawning habitat (Bjornn and Reiser 1991, p. 98). Sediment particle size determines the pore openings in the redd gravel. With small pore openings, more suspended sediments are deposited and water flow is reduced compared to large pore openings.

Survival of eggs is dependent on a continuous supply of well oxygenated water through the streambed gravels (Cederholm and Reid 1987, p. 384;Anderson, Taylor, and Balch 1996, p. 13). Eggs and alevins are generally more susceptible to stress by suspended solids than are adults. Accelerated sedimentation can reduce the flow of water and, therefore, oxygen to eggs and alevins. This can decrease egg survival, decrease fry emergence rates (Cederholm and Reid 1987, p. 384;Chapman 1988, pp. 12-16;Bash et al. 2001f, pp. 17-18), delay development of alevins (Everest, Beschta, Scrivener, Koski, Sedell, and Cederholm 1987, p. 113), reduce growth and cause premature hatching and emergence (Birtwell 1999, p. 19). Fry delayed in their emergence are also less able to compete for environmental resources than fish that have undergone normal development and emergence (intra- or interspecific competition) (Everest, Beschta, Scrivener, Koski, Sedell, and Cederholm 1987, p. 113). Sedimentation fills the interstitial spaces and can prevent alevins from emerging from the gravel (Anderson, Taylor, and Balch 1996, p. 13;Suttle, Power, Levine, and McNeely 2004, pp. 971-972).

Several studies have documented that fine sediment can reduce the reproductive success of salmonids. Natural egg-to-fry survival of coho salmon, sockeye and kokanee has been measured at 23 percent, 23 percent and 12 percent, respectively (Slaney et al. 1977, p. 33). Substrates containing 20 percent fines can reduce emergence success by 30-40 percent (MacDonald, Smart, and Wissmar 1991, p. 99). A decreases of 30 percent in mean egg-to-fry survival can be expected to reduce salmonid fry production to extremely low levels (Slaney, Halsey, and Tautz 1977, p. 33).

<u>Indirect Effects</u>

Macroinvertebrates

Sedimentation can have an effect on bull trout and fish populations through impacts or alterations to the macroinvertebrate communities or populations (Anderson, Taylor, and Balch 1996, pp. 14-15). Increased turbidity and suspended sediment can reduce primary productivity by decreasing light intensity and periphytic (attached) algal and other plant communities (Anderson, Taylor, and Balch 1996, p. 14;Henley, Patterson, Neves, and Lemly 2000, p. 129;Suren and Jowett 2001, p. 726). This results in decreased macroinvertebrates that graze on the periphyton.

Sedimentation also alters the habitat for macroinvertebrates, changing the species density, diversity and structure of the area (Waters 1995, pp. 61-78;Anderson, Taylor, and Balch 1996, pp. 14-15;Reid and Anderson 1999, pp. 10-12;Shaw and Richardson 2001, p. 2220). Certain groups of macroinvertebrates are favored by salmonids as food items. These include mayflies, caddisflies, and stoneflies. These species prefer large substrate particles in riffles and are negatively affected by fine sediment (Everest, Beschta, Scrivener, Koski, Sedell, and Cederholm

1987, p. 115;Waters 1995, p. 63). Increased sediment can affect macroinvertebrate habitat by filling of interstitial space and rendering attachment sites unsuitable. This may cause invertebrates to seek more favorable habitat (Rosenberg and Snow 1975, p. 70). With increasing fine sediment, invertebrate composition and density changes from available, preferred species (i.e., mayflies, caddisflies, and stoneflies) to non-preferred, more unavailable species (i.e., aquatic worms and other burrowing species) (Reid and Anderson 1999, p. 10;Henley, Patterson, Neves, and Lemly 2000, pp. 126, 130;Shaw and Richardson 2001, p. 2219;Suren and Jowett 2001, p. 726;Suttle, Power, Levine, and McNeely 2004, p. 971). The degree to which substrate particles are surrounded by fine material was found to have a strong correlation with macroinvertebrate abundance and composition (Birtwell 1999, p. 23). At an embeddedness of one-third, insect abundance can decline by about 50 percent, especially for riffle-inhabiting taxa (Waters 1995, p. 66).

Increased turbidity and suspended solids can affect macroinvertebrates in multiple ways through increased invertebrate drift, feeding impacts, and respiratory problems (Cederholm and Reid 1987, p. 384;Shaw and Richardson 2001, p. 2218;Berry, Rubinstein, Melzian, and Hill 2003, pp. 8, 11). The effect of turbidity on light transmission has been well documented and results in increased invertebrate drift (Waters 1995, p. 58;Birtwell 1999, pp. 21, 22). This may be a behavioral response associated with the night-active diel drift patterns of macroinvertebrates. While increased turbidity results in increased macroinvertebrate drift, it is thought that the overall invertebrate populations would not fall below the point of severe depletion (Waters 1995, p. 59). Invertebrate drift is also an important mechanism in the repopulation, recolonization, or recovery of a macroinvertebrate community after a localized disturbance (Anderson, Taylor, and Balch 1996, p. 15;Reid and Anderson 1999, pp. 11-12).

Increased suspended sediment can affect macroinvertebrates by abrasion of respiratory surface and interference with food uptake for filter-feeders (Anderson, Taylor, and Balch 1996, p. 14;Birtwell 1999, p. 21;Shaw and Richardson 2001, p. 2213;Suren and Jowett 2001, pp. 725-726;Berry, Rubinstein, Melzian, and Hill 2003, p. 11). Increased suspended sediment levels tend to clog feeding structures and reduce feeding efficiencies, which results in reduced growth rates, increased stress, or death of the invertebrates (Newcombe and MacDonald 1991, p. 73). Invertebrates living in the substrate are also subject to scouring or abrasion which can damage respiratory organs (Bash et al. 2001e, p. 25).

Feeding Efficiency

Increased turbidity and suspended sediment can affect a number of factors related to feeding for salmonids, including feeding rates, reaction distance, prey selection, and prey abundance (Barrett, Grossman, and Rosenfeld 1992, pp. 437, 440;Henley, Patterson, Neves, and Lemly 2000, p. 133;Bash et al. 2001d, p. 21). Changes in feeding behavior are primarily related to the reduction in visibility that occurs in turbid water. Effects on feeding ability are important as salmonids must meet energy demands to compete with other fishes for resources and to avoid predators. Reduced feeding efficiency would result in lower growth and fitness of bull trout and other salmonids (Barrett, Grossman, and Rosenfeld 1992, p. 442;Sweka and Hartman 2001, p. 138).

Distance of prey capture and prey capture success both were found to decrease significantly when turbidity was increased (Berg and Northcote 1985, pp. 1414-1415;Sweka and Hartman 2001, p. 141;Zamor and Grossman 2007, pp. 168, 170, 174). Waters (1995, p. 83) states that loss of visual capability, leading to reduced feeding, is one of the major sublethal effects of high suspended sediment. Increases in turbidity were reported to decrease reactive distance and the percentage of prey captured (Sweka and Hartman 2001, p. 141;Bash et al. 2001c, pp. 21-23;Klein 2003, pp. 1, 21). At 0 NTUs, 100 percent of the prey items were consumed; at 10 NTUs, fish frequently were unable to capture prey species; at 60 NTUs, only 35 percent of the prey items were captured. At 20 to 60 NTUs, significant delay in the response of fish to prey was observed (Bash et al. 2001b, p. 22). Loss of visual capability and capture of prey leads to depressed growth and reproductive capability.

To compensate for reduced encounter rates with prey under turbid conditions, prey density must increase substantially or salmonids must increase their active searches for prey (Sweka and Hartman 2001, p. 144). Such an increase in activity and feeding rates under turbid conditions reduces net energy gain from each prey item consumed (Sweka and Hartman 2001, p. 144).

Sigler et al. (1984, p. 150) found that a reduction in growth occurred in steelhead and coho salmon when turbidity was as little as 25 NTUs. The slower growth was presumed to be from a reduced ability to feed; however, more complex mechanisms such as the quality of light may also affect feeding success rates. Redding et al. (1987, p. 742) found that suspended sediment may inhibit normal feeding activity, as a result of a loss of visual ability or as an indirect consequence of increased stress.

Habitat Effects

Compared to other salmonids, bull trout have more specific habitat requirements that appear to influence their distribution and abundance (Rieman and McIntyre 1993, p. 7). All life history stages are associated with complex forms of cover including large woody debris, undercut banks, boulders, and pools. Other habitat characteristic important to bull trout include channel and hydrologic stability, substrate composition, temperature, and the presence of migration corridors (Rieman and McIntyre 1993, p. 5).

Increases in sediment can alter fish habitat or the utilization of habitats by fish (Anderson, Taylor, and Balch 1996, p. 12). The physical implications of sediment in streams include changes in water quality, degradation of spawning and rearing habitat, simplification and damage to habitat structure and complexity, loss of habitat, and decreased connectivity between habitat (Anderson, Taylor, and Balch 1996, pp. 11-15;Bash et al. 2001a, pp. 1, 12, 18, 30). Biological implications of this habitat damage include underutilization of stream habitat, abandonment of traditional spawning habitat, displacement of fish from their preferred habitat, and avoidance of habitat (Newcombe and Jensen 1996, p. 695).

As sediment enters a stream it is transported downstream under normal fluvial processes and deposited in areas of low shear stress (MacDonald and Ritland 1989, p. 21). These areas are usually behind obstructions, near banks (shallow water) or within interstitial spaces. This episodic filling of successive storage compartments continues in a cascading fashion downstream until the flow drops below the threshold required for movement or all pools have reached their storage capacities (MacDonald and Ritland 1989, p. 21). As sediment load increases, the stream compensates by geomorphologic changes in increased slope, increased channel width, decreased depths, and decreased flows (Castro and Reckendorf 1995a, p. 21). These processes contribute to increased erosion and sediment deposition that further degrade salmonid habitat.

Loss of acceptable habitat and refugia, as well as decreased connectivity between habitats, reduces the carrying capacity of streams for salmonids (Bash et al. 2001p, p. 30). This loss of habitat or exclusion of fish from their habitat, if timed inappropriately, could impact a fish population if the habitat within the affected stream reach is critical to the population during the period of the sediment release (Anderson, Taylor, and Balch 1996, p. 12;Reid and Anderson 1999, p. 13). For example, if summer pool habitat used by adults as holding habitat prior to spawning is a limiting factor within a stream, increased sediment and reduced pool habitat during the summer can decrease the carrying capacity of the stream reach and decrease the fish population. In systems lacking adequate connectivity of habitats, fish may travel longer distances or use less desirable habitats, increasing biological demands and reducing their fitness.

The addition of fine sediment (less than 6.4 mm) to natural streams during summer decreased abundance of juvenile Chinook salmon in almost direct proportion to the amount of pool volume lost to fine sediment (Bjornn et al. 1977b, p. 31). Similarly, the inverse relationship between fine sediment and densities of rearing Chinook salmon indicates the importance of winter habitat and high sediment loads (Bjornn et al. 1977a, pp. 26, 38, 40). As fine sediments fill the interstitial spaces between the cobble substrate, juvenile Chinook salmon were forced to leave preferred habitat and to utilize cover that may be more susceptible to ice scouring, predation, and decreased food availability (Hillman, Griffith, and Platts 1987, p. 194). Deposition of sediment on substrate may lower winter carrying capacity for bull trout (Shepard, Leathe, Waver, and Enk 1984, p. 153). Food production in the form of aquatic invertebrates may also be reduced.

Juvenile bull trout densities are highly influenced by substrate composition (Shepard, Leathe, Waver, and Enk 1984, p. 153;Rieman and McIntyre 1993, p. 6;MBTSG (The Montana Bull Trout Scientific Group) 1998b, p. 9). During the summer, juvenile bull trout hold positions close to the stream bottom and often seek cover within the substrate itself. When streambed substrate contains more than 30 percent fine materials, juvenile bull trout densities drop off sharply (Shepard, Leathe, Waver, and Enk 1984, p. 152). Any loss of interstitial space or streambed complexity through the deposition of sediment would result in a loss of summer and winter habitats (MBTSG (The Montana Bull Trout Scientific Group) 1998c, p. 9). The reduction of rearing habitat will ultimately reduce the potential number of recruited juveniles and therefore reducing population numbers (Shepard, Leathe, Waver, and Enk 1984, pp. 153-154). In fact, Johnston et al. (2007, p. 125) found that density-dependent survival during the earliest of the juvenile stages (between egg and age-1) regulated recruitment of adult bull trout in the population.

Although an avoidance response by fish to increased sediment may be an initial adaptive survival strategy, displacement from cover could be detrimental. It is possible that the consequences of fish moving from preferred habitat, to avoid increasing levels of suspended sediment, may not be beneficial if displacement is to sub-optimal habitat, because they may be stressed and more vulnerable to predation (Birtwell 1999, p. 12).

In addition to altering stream bed composition, anthropogenic input of sediment into a stream can change channel hydrology and geometry (Owens et al. 2005, pp. 694-695). Sediment release can reduce the depth of pools and riffle areas (Anderson, Taylor, and Balch 1996, p. 12). This can reduce available fish habitat, decrease fish holding capacity, and decrease fish populations (Anderson, Taylor, and Balch 1996, pp. 12, 14).

Physiological Effects

Sublethal levels of suspended sediment may cause undue physiological stress on fish, which may reduce the ability of the fish to perform vital functions (Cederholm and Reid 1987, p. 388, 390). Stress is defined as a condition perceived by an organism which threatens a biological function of the organism, and a set of physiological and behavioral responses is mounted to counteract the condition (Overli 2001, p. 7). A stressor is any anthropogenic or natural environmental change severe enough to require a physiological response on the part of a fish, population, or ecosystem (Anderson, Taylor, and Balch 1996, pp. 5-6;EPA (U.S. Environmental Protection Agency) 2001a, pp. 1-2;Jacobson et al. 2003, p. 2). At the individual level, stress may affect physiological systems, reduce growth, increase disease, and reduce the individual's ability to tolerate additional stress (Anderson, Taylor, and Balch 1996, p. 7;Bash et al. 2001q, p. 17). At the population level, the effects of stress may include reduced spawning success, increased larval mortality, reduced recruitment to succeeding life stages and, therefore, overall population declines (Bash et al. 2001r, p. 17).

Upon encountering a stressor, the fish responds through a series of chemical releases in its body. These primary chemical and hormonal releases include catecholamine (e.g. epinephrine, norepinehprine) in the circulatory system, corticosteroids (e.g. cortisol) from the interregnal tissue, and hypothalamic activation of the pituitary gland (Gregory and Wood 1999, p. 286;Schreck et al. 2001, p. 5;Barton 2002, p. 517;Davis 2006, p. 116). Primary chemical releases result in secondary releases or changes in plasma, glucose, tissue ion, metabolite levels, and hematological features. These secondary responses relate to physiological adjustments in metabolism, respiration, immune and cellular function (Mazeaud et al. 1977, p. 201;Barton 2002, p. 517;Haukenes and Buck 2006, p. 385). After secondary responses, continued stress results in tertiary stress responses which affect whole-animal performance such as changes in growth, condition, resistance to disease, metabolic scope for activity, behavior, and ultimately survival (Pickering et al. 1982, p. 229;Barton 2002, p. 517;Portz et al. 2006, pp. 126-127).

Stress in a fish occurs when the homeostatic or stabilizing process in the organism exceed the capability of the organism to compensate for the biotic or abiotic challenge (Anderson, Taylor, and Balch 1996, p. 5). The response to a stressor is an adaptive mechanism that allows the fish to cope with the real or perceived stressor in order to maintain its normal or homeostatic state (Barton 2002, p. 517). Acclimation to a stressor can occur if compensatory physiological responses by the fish are able to re-establish a satisfactory relationship between the changed environment and the organism (Anderson, Taylor, and Balch 1996, p. 5). The ability of an individual fish to acclimate or tolerate the stress will depend on the severity of the stress and the physiological limits of the organism (Anderson, Taylor, and Balch 1996, p. 5). In a natural system, fish are exposed to multiple chemical and physical stressors which can combine to cause adverse effects (Berry, Rubinstein, Melzian, and Hill 2003, p. 4). The chemical releases from each stressor results in a cumulative or additive response (Barton et al. 1986, pp. 245, 247;EPA (U.S. Environmental Protection Agency) 2001b, pp. 3, 25;Cobleigh 2003, pp. 16, 39, 55;Milston et al. 2006, p. 1172).

Stress in fish results in extra cost and energy demands. Elevated oxygen consumption and increased metabolic rate result from the reallocation of energy to cope with the stress (Barton and Schreck 1987, pp. 259-260;Contreras-Sanchez et al. 1998, pp. 439, 444;McCormick et al. 1998, pp. 222, 231). An approximate 25 percent increase in metabolic cost, over standard metabolism requirements, is needed to compensate for a perceived stress (Barton and Schreck 1987, p. 260;Davis 2006, p. 116). Stressed fish would thus have less energy available for other life functions such as seawater adaptation, disease resistance, reproduction, or swimming stamina (Barton and Schreck 1987, p. 261;Contreras-Sanchez, Schreck, Fitzpatrick, and Pereira 1998, p. 444).

Tolerance to suspended sediment may be the net result of a combination of physical and physiological factors related to oxygen availability and uptake by fish (Servizi and Martens 1991b, p. 497). The energy needed to perform repeated coughing (see Gill trauma section) increases metabolic oxygen demand. Metabolic oxygen demand is related to water temperature. As temperatures increase, so does metabolic oxygen demand, but concentrations of oxygen available in the water decreases. Therefore, a fish's tolerance to suspended sediment may be primarily related to the capacity of the fish to perform work associated with the cough reflex. However, as sediment increases, fish have less capability to do work, and therefore less tolerance for suspended sediment (Servizi and Martens 1991c, p. 497).

Once exposed to a stressor, the primary chemical releases can take one-half to twenty-four hours to peak (Schreck 1981, p. 298;Barton 2002, p. 520;Quigley and Hinch 2006, p. 437). Recovery or return of the primary chemical release to normal or resting levels can take two hours to two weeks (Mazeaud, Mazeaud, and Donaldson 1977, pp. 205-206;Schreck 1981, p. 313). In a study of handling stress, chemical release of cortisol peaked at two hours and returned to normal in four hours. However, complete recovery took 2 weeks (Pickering, Pottinger, and Christie 1982, pp. 236, 241). Fish exposed to two or more stresses require longer recovery times than fish exposed only to one stressor indicating the cumulative effects of stress (Sigismondi and Weber 1988, pp. 198-199).

Redding el al.(1987, pp. 740-741) observed higher mortality in young steelhead trout exposed to a combination of suspended sediment (2500 mg/L) and a bacteria pathogen, than when exposed to the bacteria alone. Physiological stress in fishes may decrease immunological competence, growth, and reproductive success (Bash et al. 2001s, p. 16).

Behavioral effects

Increased turbidity and suspended sediment may result in behavior changes in salmonids. These changes are the first effects evoked from increased levels of turbidity and suspended sediment

(Anderson, Taylor, and Balch 1996, p. 6). These behavioral changes include avoidance of habitat, reduction in feeding, increased activity, redistribution and migration to other habitats and locations, disruption of territoriality, and altered homing (Anderson, Taylor, and Balch 1996, p. 6;Bash et al. 2001t, pp. 19-25;Suttle, Power, Levine, and McNeely 2004, p. 971). Many behavioral effects result from changes in stream habitat (see Habitat effects section). As suspended sediment concentration increases, habitat may be lost which results in abandonment and avoidance of preferred habitat. Stream reach emigration is a bioenergetic demand that may affect the growth or reproductive success of the individual fish (Bash et al. 2001u, p. 12). Pulses of sediment result in downstream migration of fish, which disrupts social structures, causes downstream displacement of other fish and increases intraspecific aggression (McLeay et al. 1987b, pp. 670-671;Bash et al. 2001v, pp. 12, 20;Suttle, Power, Levine, and McNeely 2004, p. 971). Loss of territoriality and the breakdown of social structure can lead to secondary effects of decreased growth and feeding rates, which may lead to mortality (Berg and Northcote 1985, p. 1416;Bash et al. 2001w, p. 20).

Downstream migration by bull trout provides access to more prey, better protection from avian and terrestrial predators, and alleviates potential intraspecific competition or cannibalism in rearing areas (MBTSG (The Montana Bull Trout Scientific Group) 1998d, p. 13). Benefits of migration from tributary rearing areas to larger rivers or estuaries may be increased growth potential. Increased sedimentation may result in premature or early migration of both juveniles and adults or avoidance of habitat and migration of nonmigratory resident bull trout.

High turbidity may delay migration back to spawning sites, although turbidity alone does not seem to affect homing. Delays in spawning migration and associated energy expenditure may reduce spawning success and therefore population size (Bash et al. 2001x, p. 29).

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DETERMINING EFFECTS FOR SECTION 7 CONSULTATIONS

There are numerous factors that can influence project-specific sediment effects on bull trout and other salmonids. These factors include the concentration and duration of sediment input, existing sediment conditions, stream conditions (velocity, depth, etc.) during construction, weather or climate conditions (precipitation, wind, etc.), fish presence or absence (bull trout plus prey species), and best management practice effectiveness. Many of these factors are unknown.

Newcombe and Jensen (1996) and Anderson et al. (1996) provide the basis for analyzing sediment effects to bull trout and other salmonids and their habitat. Newcombe and Jensen (1996) conducted a literature review of pertinent documents on sediment effects to salmonids and nonsalmonids. They developed a model that calculated the severity of ill effect (SEV) to fish based on the suspended sediment dose (exposure) and concentration. No data on bull trout were used in this analysis. Anderson et al. (1996), using the methods used by Newcombe and Jensen (1996), developed a model to estimate sediment impacts to salmonid habitat.

A 15-point scale was developed by Newcombe and Jensen (1996, p. 694) to qualitatively rank the effects of sediment on fish (Table 1). Using a similar 15-point scale, Anderson et al. (1996)

Table 1 – Scale of the severity (SEV) of ill effects associated with excess suspended sediment on salmonids.

SEV	Description of Effect
	Nil effect
0	No behavioral effects
	Behavioral effects
1	Alarm reaction
2	Abandonment of cover
3	Avoidance response
	Sublethal effects
4	Short-term reduction in feeding rates; short-term reduction in feeding success
5	Minor physiological stress; increase in rate of coughing; increased respiration rate
6	Moderate physiological stress
7	Moderate habitat degradation; impaired homing
8	Indications of major physiological stress; long-term reduction in feeding rate; long-term reduction in feeding success; poor condition
	Lethal and paralethal effects
9	Reduced growth rate; delayed hatching; reduced fish density
10	0-20% mortality; increased predation; moderate to severe habitat degradation
11	> 20 – 40% mortality
12	> 40 – 60% mortality
13	> 60 – 80% mortality
14	> 80 – 100% mortality

ranked the effects of sediment on fish habitat (Table 2).

We analyzed the effects on different bull trout life history stages to determine when adverse effects of project-related sediment would occur. Table 3 shows the different ESA effect calls for bull trout based on severity of ill effect.

The effect determination for a proposed action should consider all SEV values resulting from the action because sediment affects individual fish differently depending on life history stage and site-specific factors. For juvenile bull trout, an SEV of 5 is likely to warrant a "likely to adversely affect" (LAA) determination. However, abandonment of cover (SEV 2), or an avoidance response (SEV 3), may result in increased predation risk and mortality if habitat features are limiting in the project's stream reach. Therefore, a LAA determination may be warranted at an SEV 2 or 3 level in certain situations. For subadult and adult bull trout, however, abandonment of cover and avoidance may not be as important. A higher SEV score is more appropriate for adverse effects to subadult and adult bull trout. In all situations, we assume that SEV scores associated with adverse effects are also sufficient to represent a likelihood of harm or harass[1].

Table 2 – Scale of the severity (SEV) of ill effects associated with excess suspended sediment on salmonid habitat.

SEV	Description of Effect
3	Measured change in habitat preference
7	Moderate habitat degradation – measured by a change in invertebrate community
10	Moderately severe habitat degradation – defined by measurable reduction in the productivity of habitat for extended period (months) or over a large area (square kilometers).
12	Severe habitat degradation – measured by long-term (years) alterations in the ability of existing habitats to support fish or invertebrates.
14	Catastrophic or total destruction of habitat in the receiving environment.

When evaluating impacts to habitat as a surrogate for species effects, adverse effects may be anticipated when there is a notable reduction in abundance of aquatic invertebrates, and an alteration in their community structure. These effects represent a reduction in food for bull trout and other salmonids, and correspond to an SEV of 7 – moderate habitat degradation.

Newcombe and Jensen (1996) used six data groups to conduct their analysis. These groups were 1) juvenile and adult salmonids (Figure 1), 2) adult salmonids (Figure 2), 3) juvenile salmonids (Figure 3), 4) eggs and larvae of salmonids and non-salmonids (Figure 4), 5) adult estuarine nonsalmonids (no figure provided), and 6) adult freshwater nonsalmonids (no figure provided). No explanation was provided for why juvenile and adult salmonids were combined

[1] Harm and harass in this context refers to the FWS's regulatory definition at 50 CFR 17.3. E.g., Harm means "an act which actually kills or injures wildlife. Such an act may include significant habitat modification or degradation where it actually kills or injures wildlife by significantly impairing essential behavior patterns, including breeding, feeding, or sheltering."

for group 1. As juveniles are more adapted to turbid water (Newcombe 1994, p. 5), their SEV levels are generally lower than for adult salmonids given the same concentration and duration of sediment (Figures 1-3).

Table 3 – ESA Effect calls for different bull trout life stages in relation to the duration of effect and severity of ill effect. Effect calls for habitat, specifically, are provided to assist with analysis of effects to individual bull trout.

	SEV	ESA Effect Call
Egg/alevin	1 to 4	Not applicable - alevins are still in gravel and are not feeding.
	5 to 14	LAA - any stress to egg/alevin reduces survival
Juvenile	1 to 4	NLAA
	5 to 14	LAA
Subadult and Adult	1 to 5	NLAA
	6 to 14	LAA
Habitat	1 to 6	NLAA
	7 to 14	LAA due to indirect effects to bull trout

The figures of Newcombe and Jensen (1996) have been modified in this document. In each figure, values (in mg/L) are provided for each duration to determine when adverse effects would occur. Specific values are also given for when harm would be likely to occur. For example:

> Figure 1 – This figure is for both juveniles and adults. From Table 2, bull trout are "likely to be adversely affected" given an SEV of 5. On Figure 1, a sediment concentration of 99 mg/L for one hour is anticipated to be the maximum concentration for an SEV of 4. At 100 mg/L, an SEV of 5 occurs. In addition, one hour of exposure to 5,760 mg/L is the maximum for an SEV of 7. Exposure to 5,761 mg/L for one hour would warrant an SEV of 8. This would be the threshold between harassment and harm. An SEV of 7 would be harassment, and an SEV of 8 would be considered harm.

The following provides some guidance on use of the figures.

Definitions from Newcombe and Jensen (1996, p. 696). These definitions are provided for consultations that may have impacts to bull trout prey such as Chinook and coho salmon.

> Eggs and larvae – eggs, and recently hatched fish, including yolk-sac fry, that have not passed through final metamorphosis.
>
> Juveniles – fry, parr, and smolts that have passed through larval metamorphosis but are sexually immature.

Adults – mature fish.

Bull trout use:

Newcombe and Jensen (1996) conducted their analysis for freshwater, therefore the use of the figures within this document in marine waters should be used with caution.

Figure 1 – Juvenile and Adult Salmonids. This figure should be used in foraging, migration and overwintering (FMO) areas. In FMO areas, downstream of local populations, both subadult and adult bull trout may be found.

Figure 2 – Adult Salmonids. This figure will not be used very often for bull trout. There may be circumstances, downstream of local population spawning areas that may have just adults, but usually this would not be the case. Justification for use of this figure should be stated in your consultation.

Figure 3 – Juvenile Salmonids. This figure should be used in local population spawning and rearing areas outside of the spawning period. During this time, only juveniles and sub-adults should be found in the area. Adults would migrate to larger stream systems or to marine water. If the construction of the project would occur during spawning, then Figure 1 should be used.

Figure 4 – Eggs and Alevins. This figure should be used if eggs or alevins are expected to be in the project area during construction.

Figure 5 – Habitat. This figure should be used for all projects to determine whether alterations to the habitat may occur from the project.

Background and Environmental Baseline

In determining the overall impact of a project on bull trout, and to specifically understand whether increased sediment may adversely affect bull trout, a thorough review of the environmental baseline and limiting factors in the stream and watershed is needed. The following websites and documents will help provide this information.

1. Washington State Conservation Commission's Limiting Factors Analysis. A limiting factors analysis has been conducted on watersheds within the State of Washington. Limiting factors are defined as "conditions that limit the ability of habitat to fully sustain populations of salmon, including all species of the family Salmonidae." These documents will provide information on the current condition of the individual watersheds within the State of Washington. The limiting factors website is http://salmon.scc.wa.gov. Copies of the limiting factors analysis can be found at the Western Washington Fish and Wildlife Library.

2. Washington Department of Fish and Wildlife's (1998) Salmonid Stock Inventory (SaSI). The Washington Department of Fish and Wildlife (WDFW) inventoried bull

trout and Dolly Varden (*S. malma*) stock status throughout the State. The intent of the inventory is to help identify available information and to guide future restoration planning and implementation. SaSI defines the stock within the watershed, life history forms, status and factors affecting production. Spawning distribution and timing for different life stages are provided (migration, spawning, etc.), if known. SaSi documents can be found at http://wdfw.wa.gov/fish/sasi/index.htm.

3. U.S. Fish and Wildlife Service's (USFWS 1998a) Matrix of Diagnostics/Pathways and Indicators (MPI). The MPI was designed to facilitate and standardize determination of project effects on bull trout. The MPI provides a consistent, logical line of reasoning to aid in determining when and where adverse affects occur and why they occur. The MPI provides levels or values for different habitat indicators to assist the biologist in determining the level of effects or impacts to bull trout from a project and how these impacts may cumulatively change habitat within the watershed.

4. Individual Watershed Resources. Other resources may be available within a watershed that will provide information on habitat, fish species, and recovery and restoration activities being conducted. The action agency may cite a publication or identify a local watershed group within the Biological Assessment or Biological Evaluation. These local groups provide valuable information specific to the watershed.

5. Washington State Department of Ecology (WDOE) - The WDOE has long- and short-term water quality data for different streams within the State. Data can be found at http://www.ecy.wa.gov/programs/eap/fw_riv/rv_main.html. Clicking on a stream or entering a stream name will provide information on current and past water quality data (when you get to this website, scroll down to the Washington map). This information will be useful for determining the specific turbidity/suspended sediment relationship for that stream (more information below).

6. Washington State Department of Ecology (WDOE) - The WDOE has also been collecting benthic macroinvertebrates and physical habitat data to describe conditions under natural and anthropogenic disturbed areas. Data can be found at http://www.ecy.wa.gov/programs/eap/fw_benth/index.htm. You can access monitoring sites at the bottom of the website.

7. U.S. Forest Service, Watershed Analysis Documents - The U.S. Forest Service (USFS) is required by the Record of Decision for Amendments to the USFS and Bureau of Land Management Planning Documents within the Range of the Northern Spotted Owl to conduct a watershed analysis for watersheds located on FS lands. The watershed analysis determines the existing condition of the watershed and makes recommendations for future projects that move the landscape towards desired conditions. Watershed analysis documents are available from individual National Forests or from the Forest Plan Division.

8. U.S. Fish and Wildlife Service - Bull Trout Recovery Plans and Critical Habitat Designations. The draft Bull Trout Recovery Plan for the Columbia River Distinct

Population Segment (DPS) (also the Jarbidge River and the St. Mary-Belly River DPS) and the proposed and final critical habitat designations provide current species status, habitat requirements, and limiting factors for bull trout within specific individual recovery units. These documents are available from the Endangered Species Division as well as the Service's web page (www.fws.gov).

These documents and websites provide baseline and background information on stream and watershed conditions. This information is critical to determining project-specific sediment impacts to the aquatic system. The baseline or background levels need to be analyzed with respect to the limiting factors within the watershed.

Consultation Sediment Analysis

The analysis in this section only applies to construction-related physiological and behavioral impacts, and the direct effects of fine sediment on current habitat conditions. Longer-term effects to habitat from project-induced channel adjustments, post-construction inputs of coarse sediment, and secondary fine sediment effects due to re-mobilization of sediment during the following runoff season, are not included in the quantitative part of this effects determination. Those aspects are only considered qualitatively.

The background or baseline sediment conditions within the project area or watershed will help to determine whether the project will have an adverse effect on bull trout. The following method should be followed to assist in reviewing effects determinations and quantifying take in biological opinions.

1) Determine what life stage(s) of bull trout will be affected by sedimentation from the project. Life history stages include eggs and alevins, juveniles, and sub-adults and adults. If projects adhere to approved work timing windows, very few should be constructed during periods when eggs and alevins are in the gravels. However, streambed or bank adjustments may occur later in time and result in increased sedimentation during the time of the year when eggs and alevins may be in the gravels and thus affected by the project.

2) Table 4 (Page 45) provides concentrations, durations, and SEV levels for different projects. This table will help in analyzing similar projects and to determine sediment level impacts associated with that type of project. Based on what life history stage is in the project area and what SEV levels may result from the project, a determination may be made on effects to bull trout.

3) Once a "likely to adversely affect" determination has been made for a project, the figures in Newcombe and Jensen (1996) or Anderson et al. (1996) are used to determine the concentration (mg/L) at which adverse effects[2] and "take" will occur (see Figures 1-5). For example, if a project is located in FMO habitat, Figure 1 would be used to determine the concentrations at which adverse effects will occur. Since Figure 1 is used for both adults and juveniles, an SEV of 5 (for juveniles) is used (see Table 2). For (a.) the level

[2] For the remainder of the document, references to "adverse effects" also refer to harm and harass under 50 CFR 17.3.

when instantaneous adverse effects occur, find the SEV level of 5 in the one hour column. The corresponding concentration is the instantaneous value where adverse affects occur. In this example, it is 148 mg/L. For (b), (c), and (d), adverse effects will occur when sediment concentrations exceed SEV 4 levels. The exact concentrations for this have been provided. For each category, find the SEV 4 levels and the corresponding concentration levels are the values used.

For impacts to individual bull trout, adverse effects would be anticipated in the following situations:

a. Any time sediment concentrations exceed 148 mg/L over background.
b. When sediment concentrations exceed 99 mg/L over background for more than one hour continuously.
c. When sediment concentrations exceed 40 mg/L over background for more than three hours cumulatively.
d. When sediment concentrations exceeded 20 mg/L over background for over seven hours cumulatively.

For habitat effects, use Figure 5 and the same procedure as above for individual bull trout. For example, adverse effects would be expected to occur in the following situations:

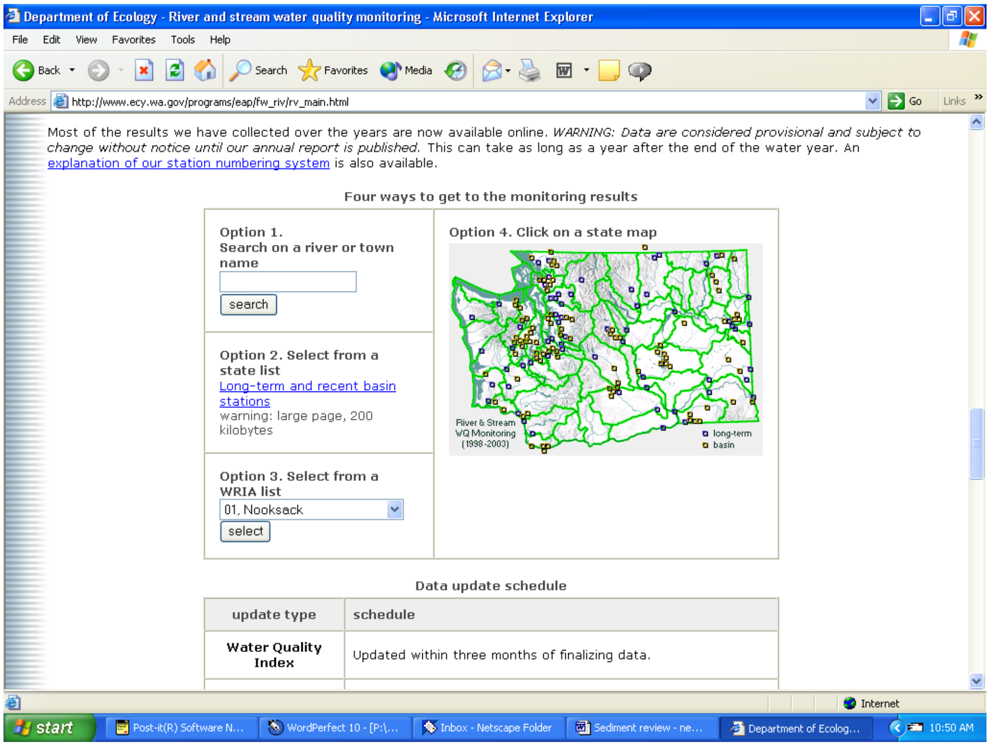
a. Any time sediment concentrations exceed 1,097 mg/L over background.
b. When sediment concentrations exceed 885 mg/L over background for more than one hour continuously.
c. When sediment concentrations exceed 345 mg/L over background for more than three hours cumulatively.
d. When sediment concentrations exceeded 167 mg/L over background for over seven hours cumulatively.

4) Because sediment sampling for concentration (mg/L) is labor intensive, many applicants prefer to monitor turbidity as a surrogate. To do this, the sediment concentration at which adverse effects to the species and/or habitat occurs is converted to NTUs. Two methods, regression analysis and turbidity to suspended solid ratio, are available for this conversion. The regression analysis method should be used first. If not enough data are available then the turbidity to suspended solid ratio method should be used.

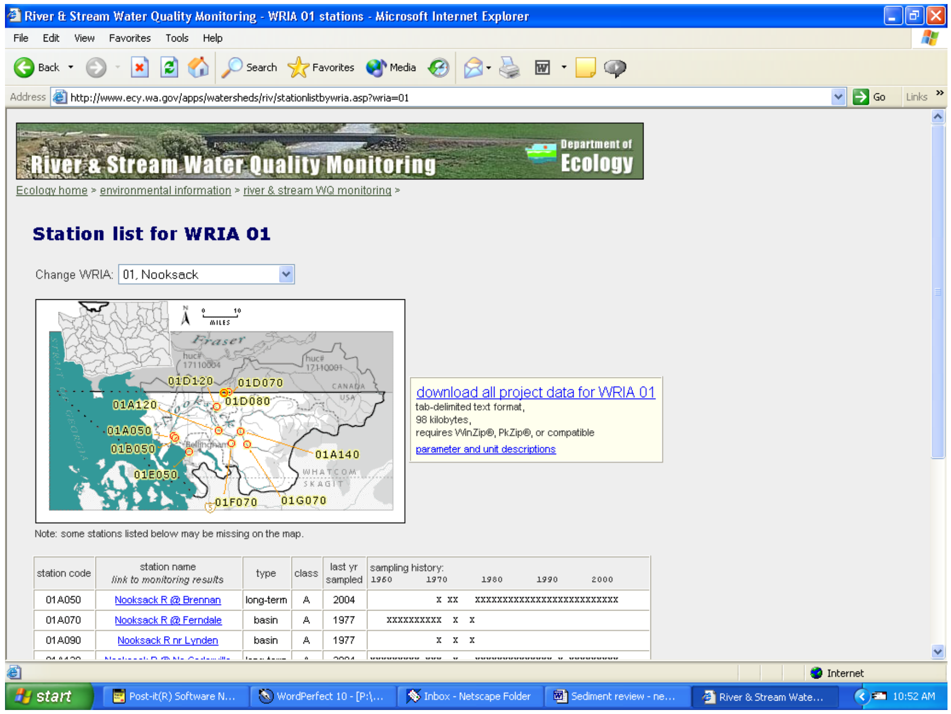
   a. Data – as described above in Background and Environmental Baseline, an attempt should be made to find turbidity and suspended solid information from the project area, action area, or the stream in which the project is being constructed. This information may be available from the Tribes, watershed monitoring groups, etc. Try to obtain information for the months in-water construction will occur, which is usually during the fish timing window (in most cases, July through September). If you are unable to find any data for the action area, use the WDOE water quality monitoring data. The following are the steps you need to go through to locate the information on the web and how to download the data:

i. Go to the WDOE webpage (http://www.ecy.wa.gov/programs/eap/fw_riv/rv_main.html).

ii. When you get to the website, the page will state "River and Stream Water Quality Monitoring." If you scroll down the page, you will see the following text and map.



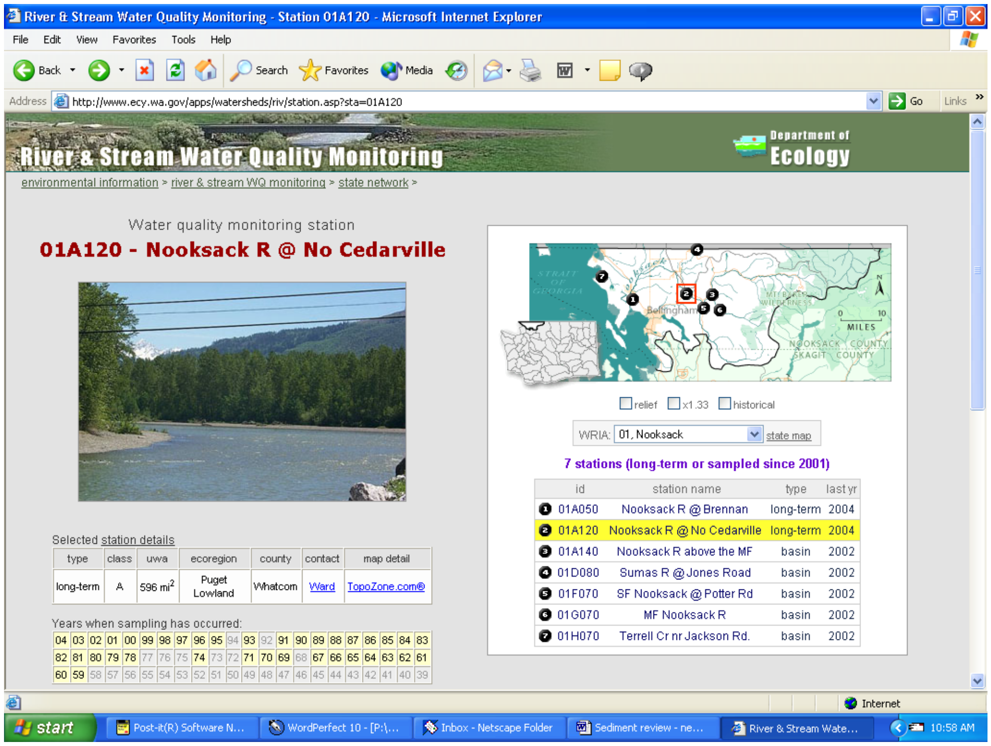
iii. The map shows all the water quality monitoring stations in Washington. You can click on a watershed, or go to Option 3, click on the down arrow and find your watershed. You will then get the following webpage. This is an example for the Nooksack River.



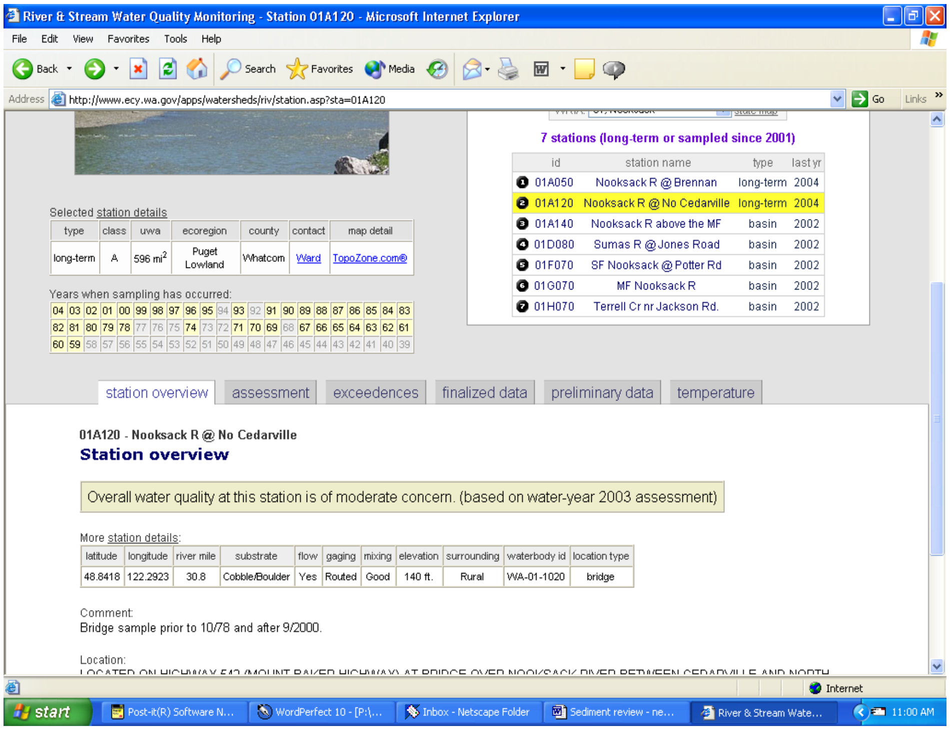
iv. This webpage shows you all the monitoring stations in this watershed. Scrolling down a little on the webpage, you get a list of the monitoring stations and the years that data were collected. The more years in which data were collected the better; however, you want to pick the monitoring station closest to the project site. If a project is located on a tributary, do not use data from the main river in the watershed. Find a monitoring station on a tributary and use that data. **Justification for the use of the data needs to be made in the BO.** The following language was used in the Anthracite Creek Bridge Scour BO. Changes to this paragraph to represent regression analysis are not italicized.

"*The guidance of Newcombe and Jensen (1996) requires a measurement of the existing suspended sediment concentration levels (mg/L) and duration of time that sediment impacts would occur. The Service used data available on the Washington Department of Ecology (WDOE) website to determine a ratio of turbidity (NTU) to suspended solids (mg/L)*(website to find the correlation between turbidity and suspended solids) *in Anthracite Creek. No water quality data was available for Anthracite Creek, so the Service used water quality monitoring data from a different tributary within the Snohomish River watershed. Patterson Creek, which is a tributary to the Snoqualmie River, was used to determine the ratio of turbidity to suspended solids* (correlation between turbidity and suspended solids). *The Service believes that Patterson Creek would have very comparable water quality data as Anthracite Creek. The turbidity to suspended solid ratio for Patterson Creek is 1:2.4 during the proposed months of construction (July through September).*" Delete the last sentence for regression analysis or put in the equation used for analysis and the R^2.

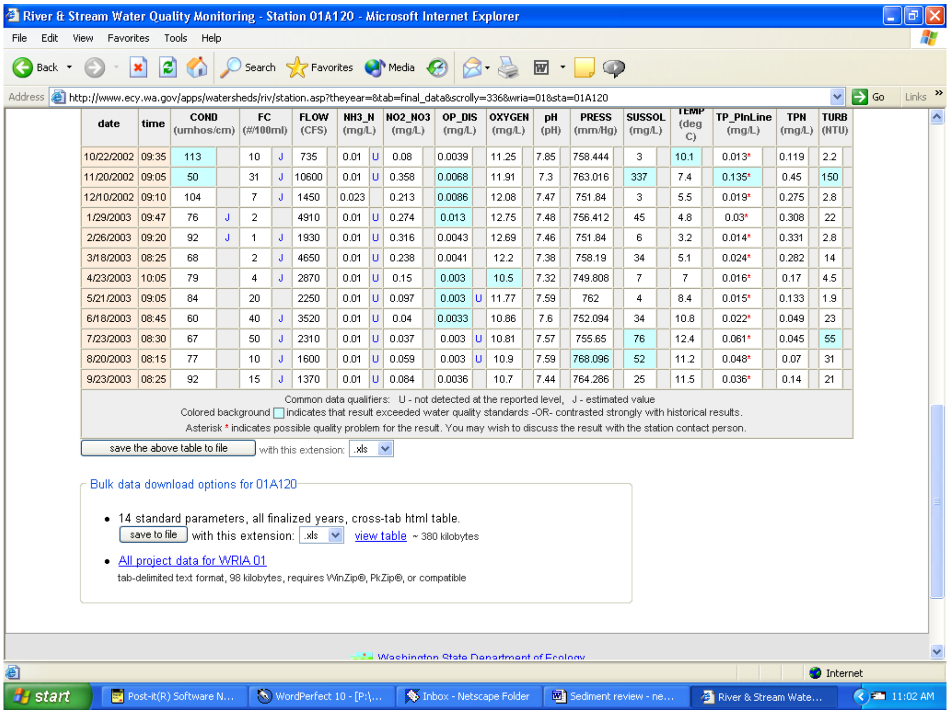
v. When you select the monitoring station, the following webpage appears. This monitoring station is on the Nooksack River at North Cedarville.



vi. Moving down the webpage, you find the following. The page shows the years data were collected and 4 to 6 tabs that provide different information. Click on the finalized data tab.



vii. Selecting the finalized data, a new page comes up; scrolling down that page you see the following. The top part of the page shows the finalized data for the most recent year data were collected. Below the data is a box that says "Bulk data download options..." Click on the "save to file" button for the 14 standardized data parameters. Follow the instructions to save this file. This saves all the data from that monitoring station so the regression analysis can be conducted.



River & Stream Water Quality Monitoring - Station 01A120 - Microsoft Internet Explorer

http://www.ecy.wa.gov/apps/watersheds/riv/station.asp?theyear=&tab=final_data&scrolly=336&wria=01&sta=01A120

date	time	COND (umhos/cm)	FC (#/100ml)	FLOW (CFS)	NH3_N (mg/L)	NO2_NO3 (mg/L)	OP_DIS (mg/L)	OXYGEN (mg/L)	pH (pH)	PRESS (mm/Hg)	SUSSOL (mg/L)	TEMP (deg C)	TP_PlnLine (mg/L)	TPN (mg/L)	TURB (NTU)
10/22/2002	09:35	113	10 J	735	0.01 U	0.08	0.0039	11.25	7.85	758.444	3	10.1	0.013*	0.119	2.2
11/20/2002	09:05	50	31 J	10600	0.01 U	0.358	0.0068	11.91	7.3	763.016	337	7.4	0.135*	0.45	150
12/10/2002	09:10	104	7 J	1450	0.023	0.213	0.0086	12.08	7.47	751.84	3	5.5	0.019*	0.275	2.8
1/29/2003	09:47	76 J	2	4910	0.01 U	0.274	0.013	12.75	7.48	756.412	45	4.8	0.03*	0.308	22
2/26/2003	09:20	92 J	1 J	1930	0.01 U	0.316	0.0043	12.69	7.46	751.84	6	3.2	0.014*	0.331	2.8
3/18/2003	08:25	68	2 J	4650	0.01 U	0.238	0.0041	12.2	7.38	758.19	34	5.1	0.024*	0.282	14
4/23/2003	10:05	79	4 J	2870	0.01 U	0.15	0.003	10.5	7.32	749.808	7	7	0.016*	0.17	4.5
5/21/2003	09:05	84	20	2250	0.01 U	0.097	0.003 U	11.77	7.59	762	4	8.4	0.015*	0.133	1.9
6/18/2003	08:45	60	40 J	3520	0.01 U	0.04	0.0033	10.86	7.6	752.094	34	10.8	0.022*	0.049	23
7/23/2003	08:30	67	50 J	2310	0.01 U	0.037	0.003 U	10.81	7.57	755.65	76	12.4	0.061*	0.045	55
8/20/2003	08:15	77	10 J	1600	0.01 U	0.059	0.003 U	10.9	7.59	768.096	52	11.2	0.048*	0.07	31
9/23/2003	08:25	92	15 J	1370	0.01 U	0.084	0.0036	10.7	7.44	764.286	25	11.5	0.036*	0.14	21

Common data qualifiers: U - not detected at the reported level, J - estimated value
Colored background indicates that result exceeded water quality standards -OR- contrasted strongly with historical results.
Asterisk * indicates possible quality problem for the result. You may wish to discuss the result with the station contact person.

save the above table to file with this extension: .xls

Bulk data download options for 01A120

- 14 standard parameters, all finalized years, cross-tab html table. save to file with this extension: .xls view table ~ 380 kilobytes
- All project data for WRIA 01 tab-delimited text format, 98 kilobytes, requires WinZip®, PkZip®, or compatible

viii. Open Excel and open the file that was just downloaded. Verify that all data appear to be available. After you have worked with these files, you will get an idea if something appears wrong. If the data looks like something is wrong, verify it by comparing the data to the finalized data on the webpage (look at each year's finalized data). After the file is open, delete all columns except the date, sussol (mg/L) and turb (NTU).

ix. Next delete the rows that do not need to be included. Only save the months in which the project will be constructed. For example, if work will be conducted during the work timing window of July 15 through August 31, delete all rows except those that contain data for July and August. The data consist of one data collection point each month. In addition, delete any values that have a "U" or "J" in the column to the right of the NTU value. This data may not be accurate; data may not be detectable at reported level or is an estimated value. The blue cells indicate the value exceeds water quality standards or contrasted strongly with historical results.

x. After deleting the unnecessary columns and rows, your data should contain 5 columns. You can now delete the columns to the right of the values. This will give you 3 columns. The first being the date, the second column contains the suspended solid data (mg/L) and the third column the turbidity (NTU) data.

b. Regression analysis. Once you have the data reduced to the months construction will occur, you can determine the relationship between turbidity and suspended solids using regression. The following steps will provide the regression equation using the data obtained above. These steps are for Excel 2007.

i. With your mouse, highlight both columns of data (suspended solid and turbidity), but do not include the heading information.

ii. Then click on "Insert", "Scatter" and then the graph that does not have any lines on it (should be the upper left graph).

iii. The graph is placed on your Excel sheet, so move it over so you can see all the data and the graph.

iv. Now add the trendline to the graph. This is done by clicking (left button) once on any of the points on the graph. Then right click. A window pops open and click on "Add Trendline." A "Format Trendline" window appears. Make sure Linear is checked, and down on the bottom, check Display Equation on chart and Display R-squared value on chart. Click on close.

   1. The X and Y data are opposite of what you want so you need to swap the values. This is done by left clicking once anywhere on the graph and then right click and click on "select data." A window pops open and you want to click on Edit. An Edit Series window appears and you want to click on the little red arrow next to Series X values. This allows you to select the data in the table. Upon clicking the red arrow, you will see the column under sussol (mg/L) being selected by a moving line around the cells. Select the data under Turb (NTU) by left clicking and holding the button down and drag all the way down to the last cell in that column. The whole column should have the moving line around all the cells. Click on the little red arrow in the Edit Series window. That will expand out the window and you will do the same for the Series Y values. Click on the red arrow next to that, then left click and hold and select all the cells in the column under Sussol (mg/L), and then click on the red arrow again. When the Edit Series window expands, click on OK, and then click on OK.

v. The equation that you want to use for your conversion from NTUs to suspended solids is now on the graph. Hopefully, your R-squared value is also high. This gives you an indication of how well your data fits the line. A one (1) is perfect. If this number is low (and a ballpark figure is less than 0.60) then you may want to consider using the ratio method to determine your conversion from NTUs to suspended solids.

   1. Outliers – sometimes there will be data that will be far outside the norm. These values can be deleted and that will help increase your R-squared value. If you are good at statistics there are ways of

determining outliers. If not, you will probably just use the data as is, unless you think something is really not right, then you may want to delete those data points.

vi. Using the equation for the regression analysis, convert the sediment concentrations found for when adverse affects occur to bull trout and their habitat (number 3 above) to NTUs. For our example, let's say our NTU to suspended solid equation is: $y = 1.6632x - 0.5789$. Adverse effects would then occur at (solve for x):

For impacts to the species adverse effect would occur in the following situations:

a. Any time sediment concentrations exceed 89 NTU over background.
b. When sediment concentrations exceed 60 NTU over background for more than one hour continuously.
c. When sediment concentrations exceed 24 NTU over background for more than three hours cumulatively.
d. When sediment concentrations exceeded 12 NTU over background for over seven hours cumulatively.

For impacts to habitat

a. Any time sediment concentrations exceed 660 NTU over background.
b. When sediment concentrations exceed 532 NTU over background for more than one hour continuously.
c. When sediment concentrations exceed 208 NTU over background for more than three hours cumulatively.
d. When sediment concentrations exceeded 101 NTU over background for over seven hours cumulatively.

c. Turbidity:suspended solid ratio: To calculate the turbidity to suspended solid ratio you need to download the same data off the Ecology website as described above. Sometimes the monitoring stations have limited amount of data and by running the regression analysis it is possible to get a negative slope (an increase in turbidity results in a decrease in suspended solids). This is very unlikely to occur in a stream. Other times you have so few data points that the R^2 value shows that the correlation between suspended solid and turbidity is not very good. When R^2 values are below 0.60, determine the turbidity to suspended solid ratio. The following are the steps needed to calculate the turbidity to suspended solid ratio.

i. After you deleted all the columns and rows of data you do not need, you should have 3 columns of data. The first being the date, the second column contains the suspended solid data (mg/L) and the third column the turbidity (NTU) data.

ii. Calculate the average turbidity and suspended solid value for all data. Average the turbidity column and average the suspended solid column.

iii. Calculate the turbidity to suspended solid value for the average turbidity and average suspended solid value obtained in ii. Divide the average suspended solid value by the average turbidity value.

iv. If any outliers are identified, they should be deleted. Recalculate the turbidity:suspended solid ratio if outliers have been removed (should automatically be done when values are deleted).

vii. Using the turbidity to suspended solid ratio, convert the sediment concentrations found for when adverse effects occur to bull trout and their habitat (number 3 above) to NTUs. For our example, let's say our NTU to suspended solid ratio is 2.1. Adverse effects to the species would then occur in the following situations:

a. Any time sediment concentrations exceed 70 NTU over background.
b. When sediment concentrations exceed 47 NTU over background for more than one hour continuously.
c. When sediment concentrations exceed 19 NTU over background for more than three hours cumulatively.
d. When sediment concentrations exceeded 10 NTU over background for over seven hours cumulatively.

Adverse effects to the species through habitat impacts would occur in the following situations:

a. Any time sediment concentrations exceed 522 NTU over background.
b. When sediment concentrations exceed 421 NTU over background for more than one hour continuously.
c. When sediment concentrations exceed 164 NTU over background for more than three hours cumulatively.
a. When sediment concentrations exceeded 80 NTU over background for over seven hours cumulatively.

5) Determine how far downstream adverse effects and take will occur. There is no easy answer for determining this. Table 4 provides some sediment monitoring data for a variety of projects. These data can be used to determine the downstream extent of sediment impacts for a project. Note that in Table 4 there is not a single downstream point that can always be used because sediment conveyance and mixing characteristics are different for each stream. **An explanation of how the distance downstream was determined needs to be included in each BO.**

Figure 1 – Severity of ill effect scores for juvenile and adult salmonids. The individual boxes provide the maximum concentration for that SEV. The concentration between 4 and 5 represents the threshold for harassment, and the concentration between 7 and 8 represents the threshold for harm.

Juvenile and Adult Salmonids
Average severity of ill effect scores

Concentration (mg/L)											
162755	10	11	11	12	12	13	14	14	-	-	-
59874	9	10	10	11	12	12	13	13	14	-	-
22026	8	9	10	10	11	11	12	13	13	14	-
8103	8	8	9	10	10	11	11	12	13	13	14
2981	$\boxed{5760}$ 7	8	8	9	9	10	11	11	12	12	13
1097	6	$\boxed{2335}$ 7	$\boxed{1164}$ 7	8	9	9	10	10	11	12	12
403	5	6	7	$\boxed{491}$ 7	8	9	9	10	10	11	12
148	5	5	6	7	$\boxed{214}$ 7	8	8	9	10	10	11
55	$\boxed{99}$ 4	5	5	6	6	$\boxed{95}$ 7	8	8	9	9	10
20	3	$\boxed{40}$ 4	$\boxed{20}$ 4	5	6	6	$\boxed{42}$ 7	8	8	9	9
7	3	3	4	$\boxed{8}$ 4	5	6	6	$\boxed{18}$ 7	$\boxed{8}$ 7	8	9
3	2	2	3	4	$\boxed{4}$ 4	5	5	6	7	$\boxed{4}$ 7	8
1	1	2	2	3	3	$\boxed{2}$ 4	5	5	6	7	$\boxed{2}$ 7
	1	3	7	1	2	6	2	7	4	11	30
	Hours			Days			Weeks		Months		

Figure 2 - Severity of ill effect scores for adult salmonids. The individual boxes provide the maximum concentration for that SEV. The concentration between 5 and 6 represents the threshold for harassment, and the concentration between 7 and 8 represents the threshold for harm.

Adult Salmonids
Average severity of ill effect scores

Concentration (mg/L)	Hours			Days			Weeks		Months		
	1	3	7	1	2	6	2	7	4	11	30
162755	11	11	12	12	13	13	14	14	-	-	-
59874	10	10	11	11	12	12	13	13	14	14	-
22026	9	10	10	11	11	12	12	13	13	14	14
8103	8	9	9	10	10	11	11	12	12	13	13
2981	8	8	9	9	10	10	11	11	12	12	13
1097	$\boxed{2190}$ 7	8	8	8	9	9	10	10	11	11	12
403	6	$\boxed{1095}$ 7	$\boxed{642}$ 7	8	8	9	9	10	10	11	11
148	$\boxed{156}$ 5	6	6	$\boxed{331}$ 7	$\boxed{175}$ 7	8	8	9	9	10	10
55	5	$\boxed{78}$ 5	6	6	7	$\boxed{94}$ 7	8	8	9	9	9
20	4	4	$\boxed{46}$ 5	$\boxed{24}$ 5	6	6	$\boxed{50}$ 7	$\boxed{27}$ 7	8	8	9
7	3	4	4	5	$\boxed{12}$ 5	6	6	7	$\boxed{14}$ 7	$\boxed{8}$ 7	8
3	2	3	3	4	4	$\boxed{7}$ 5	$\boxed{4}$ 5	6	6	7	$\boxed{4}$ 7
1	2	2	3	3	4	4	5	$\boxed{2}$ 5	$\boxed{1}$ 5	6	6

Figure 3 - Severity of ill effect scores for juvenile salmonids. The individual boxes provide the maximum concentration for that SEV. The concentration between 4 and 5 represents the threshold for harassment, and the concentration between 7 and 8 represents the threshold for harm.

Juvenile Salmonids
Average severity of ill effect scores

Concentration (mg/L)	1	3	7	1	2	6	2	7	4	11	30
162755	9	10	11	11	12	13	14	14	-	-	-
59874	9	9	10	11	11	12	13	14	14	-	-
22026	8	9	9	10	11	11	12	13	13	14	-
8103	$\boxed{13119}$ 7	8	9	9	10	11	11	12	13	13	14
2981	6	$\boxed{4448}$ 7	8	9	9	10	11	11	12	13	13
1097	6	6	$\boxed{1931}$ 7	8	9	9	10	11	11	12	13
403	5	6	6	$\boxed{687}$ 7	8	9	9	10	11	11	12
148	$\boxed{197}$ 4	5	6	6	$\boxed{254}$ 7	8	9	9	10	11	11
55	4	$\boxed{67}$ 4	5	6	6	$\boxed{96}$ 7	8	8	9	10	11
20	3	4	$\boxed{29}$ 4	5	6	6	$\boxed{36}$ 7	8	8	9	10
7	2	3	4	$\boxed{10}$ 4	5	6	6	$\boxed{13}$ 7	8	8	9
3	1	2	3	4	$\boxed{4}$ 4	5	6	6	$\boxed{5}$ 7	8	8
1	1	1	2	3	4	$\boxed{1}$ 4	5	6	6	8	8
	Hours			Days			Weeks		Months		

Figure 4 - Severity of ill effect scores for eggs and alevins of salmonids. The individual boxes provide the maximum concentration for that SEV. The concentration between 4 and 5 represents the threshold for both harassment and harm to eggs and alevins.

Eggs and Alevins of Salmonids
Average severity of ill effect scores

Concentration (mg/L)											
162755	7	9	10	11	12	13	14	-	-	-	-
59874	7	8	9	10	12	13	14	-	-	-	-
22026	7	8	9	10	11	12	13	-	-	-	-
8103	7	8	9	10	11	12	13	14	-	-	-
2981	6	7	8	10	11	12	13	14	-	-	-
1097	6	7	8	9	10	11	12	14	-	-	-
403	6	7	8	9	10	11	12	13	14	-	-
148	5	6	7	9	10	11	12	13	14	-	-
55	5	6	7	8	9	10	12	13	14	-	-
20	5	6	7	8	9	10	11	12	13	-	-
7	11 4	5	7	8	9	10	11	12	13	14	-
3	4	5	6	7	8	10	11	12	13	14	-
1	4	5	6	7	8	9	10	11	13	14	-
	1	3	7	1	2	6	2	7	4	11	30
	Hours			Days			Weeks		Months		

Figure 5 - Severity of ill effect scores for salmonid habitat. The individual boxes provide the maximum concentration for that SEV. The concentration between 6 and 7 represents the threshold for anticipating adverse effects to bull trout through habitat modifications.

Salmonid Habitat
Average severity of ill effect scores

Concentration (mg/L)											
162755	11	12	12	13	14	-	-	-	-	-	-
59874	10	11	12	12	13	14	-	-	-	-	-
22026	9	10	11	11	12	13	14	14	-	-	-
8103	8	9	10	11	11	12	13	14	14	-	-
2981	8	8	9	10	11	11	12	13	13	14	-
1097	7	7	8	9	10	10	11	12	13	13	14
	$\boxed{885}$										
403	6	7	7	8	9	10	10	11	12	12	13
		$\boxed{345}$	$\boxed{167}$								
148	5	6	6	7	8	9	9	10	11	12	12
				$\boxed{68}$							
55	4	5	6	6	7	8	9	9	10	11	11
					$\boxed{29}$						
20	3	4	5	5	6	7	8	8	9	10	11
						$\boxed{12}$					
7	2	3	4	5	5	6	7	7	8	9	10
							$\boxed{5}$				
3	2	2	3	4	5	5	6	7	8	8	9
								$\boxed{2}$			
1	1	1	2	3	4	4	5	6	7	7	8
	1	3	7	1	2	6	2	7	4	11	30
	Hours			Days			Weeks		Months		

Reference List

1. Anderson, P. G., B. R. Taylor, and G. C. Balch. 1996. Quantifying the effects of sediment release on fish and their habitats.Canadian Manuscript Report of Fisheries and Aquatic Sciences 2346.

2. Newcombe, C. P. and J. O. T. Jensen. 1996. Channel suspended sediment and fisheries: synthesis for quantitative assessment of risk and impact. North American Journal of Fisheries Management 16(4): 693-727.

3. Newcombe, C. P. 1994. Suspended sediment in aquatic ecosystems: ill effects as a function of concentration and duration of exposure.Victoria, British Columbia.

ESA Consultations:

While reviewing a project for sediment related impacts, there are a couple things to think about.

1. Time frame – how does sediment affect feeding, breeding, and sheltering. This is important when thinking about the likelihood of harm (significant impairment of essential behavior…) and/or harassment (significantly disrupt normal behavior…). During ESA consultations this must always be in the back of your mind.

2. Individual fish – Throughout this document, the term bull trout and their habitat are used. Please remember to think about risks to individual bull trout. The ESA is designed to protect individuals as well as populations, but effect determination and analysis or take are both about effects to individuals. For example, on page 4 of the Sediment Template (literature review), under Biological Effects of Sediment on bull trout, the last sentence in the first paragraph states “Specific effects of sediment on fish and their habitat can be put into three classes that include:” The document then defines lethal, sublethal, and behavioral effects. These effects can be to an individual or to multiple individuals within a reach.

3. Habitat – similarly, sediment input into a stream can alter habitat, and this can impact an individual bull trout as well as multiple bull trout within a reach. The preceding discussion addresses fish habitat in general and not necessarily critical habitat or PCE’s. An attempt was made to clarify this in the document. It was not possible to relate sediment input to the critical habitat PCE’s. The information needed to address sediment input and impacts to the PCEs can be found within the Sediment Template document.

Table 4 - Water quality monitoring data received by the Washington Fish and Wildlife Office. Calculated Values are exact SEV values for juvenile and adult salmonids (Figure 1) based on Newcombe and Jensen (1996), and for habitat (Figure 5) by Anderson et al. (1996).

Project and Watershed	Stream Characteristics at Project Location	Monitoring Locations	Original Sediment Data – how sediment data was provided in monitoring report.	Concentration (mg/L) used for determining SEV level. From original sediment data, concentration was either directly used, or was calculated using ratio or regression as stated in comments column.	Duration of elevated sediment concentration levels during project construction.	SEV (Juvenile and Adult Salmonids) Calculated SEV value for impacts to salmonids based on Newcombe and Jensen (1996)	SEV Habitat Calculated SEV value for habitat based on Anderson et al. (1996)	Comments
Culvert Removal or Removal and Replacement								
Siegel Creek Culvert Removal, Siegel Creek – Clark Fork River Watershed (Montana) Culvert removal Channel stabilization Bank reshaping	Lolo National Forest Bankfull width: 12 ft Average discharge: 2.8 CFS Slope: 6.7% Drainage area: 9,245 acres	Grab samples No distance Provided. Assume 150 ft. Automatic sampling - 150 ft downstream	Sediment load Ave: 0.07 tons/day Peak: 0.4 tons/day Sediment load Ave: 0.04 tons/day Peak: 0.3 tons/day	9.4 (average)* 53.7 (peak)* 5.4 (average)* 40.3 (peak)*	24 hrs* > 3 to 7 hrs* 24 hrs* > 3 to 7 hrs*	5 5 at 3 hrs 5 at 7 hrs 4 4 at 3 hrs 5 at 7 hrs	5 5 at 3 hrs 6 at 7 hrs 4 5 at 3 hrs 5 at 7 hrs	Creek dewatered during work. All sediment sampling was in mg/L. Concentration reached baseline at 1.5 miles downstream. Most of sediment appeared to settle within several hundred feet.
Sheep Creek Culvert Replacement Sheep Creek – Selway River Watershed (Idaho) Culvert replacement	Bitterroot National Forest Discharge: 1.5-2.0 CFS baseflow Channel width: 5 feet Slope: 8.9% Rosgen B4 channel	Approximately 100 ft. Distance not given, stated right below work area where water was put back in stream.	Baseline 1.69 mg/L 4.5 mg/L – 25 min 7.5 mg/L – 2 min 7.5 mg/L – 30 min 34.37 mg/L – 30 min 164.19 mg/L – 11 min 15,588.6 mg/L – 30 min 677 mg/L – 30 min 105.31 mg/L – 30 min 29.17 mg/L – 30 min 17.6 mg/L – 30 min 19.74 mg/L – 30 min 15,588.6 mg/L – 30 min	11..8 162.5 2,737.9 (average) 15,586.9 (peak)	1.5 hrs (building diversion dam and diverting stream) 15 min (diversion failure) 6.5 hrs (diversion removed and stream stabilizing, exact duration unknown, stopped monitoring before sediment conc. returned to background. 30 min (peak during diversion removal)	3 4 8 8	3 4 9 8	Creek dewatered during work. All sediment sampling in mg/L.

Project and Watershed	Stream Characteristics at Project Location	Monitoring Locations	Original Sediment Data	Concentration (mg/L) used for determining SEV level.	Duration of elevated sediment concentration.	SEV (Juvenile and Adult Salmonids)	SEV Habitat	Comments
Culvert Removal or Removal and Replacement, continued								
Graves Creek Road Repair Graves Creek – Quinault River Watershed (Washington) Road widening Culvert installation	Olympic National Park Project located 1.5 and 1.7 miles upstream of Upper Quinault Bridge Discharge: 3,200 – 3,700 cfs Slope: 0.4%	Distance from project site on tributary to the confluence with the Quinault was not provided. Road runs along Quinault River, so assume distance was less then 50 feet. Monitoring data is at confluence.	Baseline: 1.5 NTUs Confluence: 39 NTUs Below new culvert: 5.5 NTUs	52.5	2 hrs Monitoring report stated that construction was limited to less than two hours.	4	5	No diversion Culvert was installed on small trib. to Quinault River. Data indicates concentration and duration of sediment at trib. confluence with Quinault. Data analysis: Used Quinault River data downstream of Quinault Lake. No data available upstream. One year of data available – used July through October (4 months) NTU:SS ratio = 1:1.4 Regression: Negative slope Used ratio in analysis
Sulpher Creek State Route 241 Yakima County Culvert replacement	Project located approximately 1.5 miles of I-82 on SR141, near airport. Slope 3.5%	100 and 200 ft	Data provided in NTUs	100 ft 137.1 36.8 77.6 436.3 94.6 118.7 200 ft 33.8 50.0 55.5 213.0 147.2 141.0	 6 hr# 1 hr# 1 hr# 6 hr# 1 hr# 1 hr# 1 hr# 1 hr# 1 hr# 6 hr# 1 hr# 1 hr#	 6 4 4 7 4 5 4 4 4 6 5 5	 6 4 4 7 5 5 4 4 4 7 5 5	Dewatered stream Data analysis: Sulpher Creek has 2 monitoring stations, each a half mile apart. Both stations only have one year of data. Using individually, there would only be 2 points. Combined data for regression analysis. Used regression Regression: SS = 2.6561*NTU + 14.362 Ratios: Lower site ratio of 1:3.7 upper site has 1:3.3. Combined data 1:3.4.
Everett Vicinity Bridge 2/5N Seismic Retrofit Snohomish River and unnamed side channel Removal of 2 culverts of an existing temporary access road	Culverts removed in side channel Project located at Highway 2 over Snohomish River. Slope: In tidally influenced section of Snohomish River Construction occurred during low tide and channel had very little water running.	Work conducted in side channel of Snohomish River, sample taken 10 ft below confluence with river	Reading of 825 NTUs found, no background on that day, background next day was 15.6 NTUs.	713.4	2.5 hrs	6	7	Side channel not dewatered. Data analysis: Used Snohomish River data at Snohomish. 27 years of data on the lower Shohomish River. Used regression NTU:SS ratio = 1:2.1 Regression: SS = 0.878*NTU + 2.7839

Project and Watershed	Stream Characteristics at Project Location	Monitoring Locations	Original Sediment Data	Concentration (mg/L) used for determining SEV level.	Duration of elevated sediment concentration.	SEV (Juvenile and Adult Salmonids)	SEV Habitat	Comments
Culvert Removal or Removal and Replacement, continued								
Judd Creek Vashon Island Culvert replacement stream dewatered during construction. Water quality monitoring data for other Judd Creek project said "another stream simulation culvert replacement"	Judd Creek enters in NW corner of Quartermaster Harbor of Vashon Island. Monitoring report did not state where project was located. Drainage area: 3,292 acres. Discharge: 2.2 cfs Slope: 1.5% - used lower reach	100, 500, 1800 ft.	Data provided in graph format (NTUs). All values were estimated from graph	100 20 379.1 172 18.5 500 11.3 41.4 72.7 16.3 1800 19 41.4 9.2	 6 hrs 7 hrs 5 hrs 13 hrs 6 hrs 7 hrs 6 hrs 14 hrs 4 hrs 7 hrs 12 hrs	 4 7 6 5 4 5 5 5 4 5 4	 5 7 6 5 4 5 6 5 4 5 4	Stream was dewatered. Ecology does not monitor water quality in streams on Vashon Island. No stream water quality monitoring data available. Used 1:2 as an estimated average ratio.
Judd Creek Vashon Island Culvert Replacement stream dewatered during construction.	Judd Creek enters in NW corner of Quartermaster Harbor of Vashon Island. Drainage area: 3,292 acres. Discharge: 2.2 cfs Slope: 2.0%	100, 500, 1600 ft.	Data provided in graph format (NTUs). All values were estimated from graph	100 ft 9.6 49.7 20.6 500 ft 12 20.9 22.2 1,600 ft 10 22.5 11	 3 hrs 4 hrs 5.5 hrs 1.5 hrs 6 hrs 3.5 hrs 1 hr 2.5 hrs 2	 3 5 4 3 4 4 3 4 3	 3 5 5 3 5 4 3 4 3	Stream was dewatered. Ecology does not monitor water quality in streams on Vashon Island. No stream water quality monitoring data available. Used 1:2 as an estimated average ratio.
Harris Creek Snoqualmie River Culvert Replacement	Harris Cr. located approx. 2 miles north of Carnation, WA. Project in upper reaches of creek. Drainage area: 8,626 acres. Slope: 3.9% Discharge: 1.3 cfs (King County data)	Not provided	Document stated all water quality criteria were met except for one exceedance, 24 NTUs above background.	48	1 hr#	4	4	Stream was dewatered. Ecology does not monitor water quality in Harris Creek. No stream water quality monitoring data available. Used 1:2 as an estimated average ratio.

Project and Watershed	Stream Characteristics at Project Location	Monitoring Locations	Original Sediment Data	Concentration (mg/L) used for determining SEV level.	Duration of elevated sediment concentration.	SEV (Juvenile and Adult Salmonids)	SEV Habitat	Comments
Bank Stabilization								
Swede Heaven Bank Stabilization N.F. Stillaguamish River Project: 300 feet long, placing rock groins, LWD, and plantings	Project located approx. 5.5 miles west of Darrington, WA. Drainage area: 685 sq. miles. Discharge: 1,892 cfs Slope: 0.3% Bankfull width: 210 ft.	300, 600, and 1,200 ft downstream	Data provided in NTUs.	300 ft. 56.7 103.8 191.5 28.4 27.5 16.1 22.8 35.7 42.4 20.0 600 ft. 33.6 38.5 31.6 17.7 24.5 20.4 1,200 ft 47.6	 1 hrs** 3 hrs** 3 hrs** 30 min. 1.5 hrs 30 min 30 min 1.5 hrs 30 min 1 hrs[#] 2 hrs** 2 hrs** 3 hrs** 1 hrs# 30 min 30 min 1 hrs**	 4 5 6 3 4 3 3 4 3 3 4 4 4 3 3 3 4	 4 5 6 3 4 3 3 4 3 3 4 4 4 3 3 3 4	Construction area was diverted. Streambank was isolated. Data analysis 9 years of data available for the N.F. Stillaguamish River at Darrington, used July and August months when construction occurred. NTU:SS ratio = 1:3.5 Regression: Negative slope Used ratio in analysis
MP 9.2 Oil City Road Hoh River Riprap (170 ft) and LWD placement	No project location given, Oil City Road runs along the north bank of the lower Hoh River. Discharge: 2,541 cfs Drainage area: 253 sq. miles Slope: 0.3%	300 and 600 ft downstream	Monitoring data was only for LWD placement and not riprap installation Data provided in NTUs.	300 ft. 8.4 7.7 9.4 600 ft 7.5	 10 min 10 min 10 min 20 min	 2 1 2 2	 1 1 1 2	No information on how project constructed, dewatered. Project became influenced by WSDOT diversion dam release 5-6 miles upstream. 13 Years of data available for the Hoh River at the DNR Campground near the Hwy 101 Bridge. NTU:SS ratio = 1:1.2 Regression SS = 0.3874*NTU + 5.5385 Used regression analysis
SR 20 – debris jam Skagit River tributary	Project located at milepost 90 on SR20. No exact location, so used tributary just east of Concrete WA. Slope: 8.1%	Data stated sampling points located upstream and downstream of project area on the Skagit River. Two additional points located on two Skagit tributaries that are culverted under SR20.	Turbidity readings taken once a week in absence of any major rainfall and more frequently during a runoff producing rain event.	Met water quality standards.	Met water quality standards.			High turbidity was sampled, but this was due to runoff from rain events and not project. Channel was dewatered during construction.
Emergency Bank Protection Hoh River Rock placed in stream	No information on location of project. Work conducted in December.	Samples drawn 150 - 200 ft downstream of project.	Turbidity readings taken usually after large deposit of rock was placed in the river.	Met water quality standards. NTUs were provided for project, but levels were same as background.				NTU's read between 10.7 and 17.2. For emergency work, this seems very clear water.

Project and Watershed	Stream Characteristics at Project Location	Monitoring Locations	Original Sediment Data	Concentration (mg/L) used for determining SEV level.	Duration of elevated sediment concentration.	SEV (Juvenile and Adult Salmonids)	SEV Habitat	Comments
Bank Stabilization, continued								
Rivershore Lane Emergency Watershed Project South Fork Stilliguamish River Reconstructed 1,000 ft of riverbank and stabilized the bank with rock vanes, logs, and rootwad structures.	Project located 0.5 miles SE of Robe WA. Discharge: 461 cfs Slope: 0.4%	300, 600 ft, and 3.3 miles		600 ft 130.3 14.2 20.9 12.5 98.1 120.7 3.3 miles 50.1 32.8	 6 hrs 2.5 hrs 2 hrs 1 hr 1 hr 10.5 hrs 4 hrs 4.5 hrs**	 6 4 4 3 4 6 5 5	 6 4 4 3 5 7 5 5	Work area was dewatered by construction of a bypass channel. 9 years of data available for the N.F. Stillaguamish River at Darrington, used July and August months when construction occurred. NTU:SS ratio = 1:3.5 Regression had negative slope, used ratio. No 300 ft readings were taken, data logger not operating correctly.
Boulder Creek Bank Stabilization Montana	No project location was given. Unable to determine any stream characteristics information.	350 and 4,300 ft	Data estimated off of graph of monitoring data – in mg/L	350 ft 77.4 334.5 4,300 ft 13.25 155.6	 3.5 hrs 12.5 hrs 3.5 hrs 12.25 hrs	 5 7 4 6	 5 8 4 7	Project area was dewatered by constructing diversion channel.
Saxon Bank Stabilization Project South Fork Nooksack River Construct tree revetment and 3 rock vanes. Protecting 1,400 ft. of bank.	Project located at town of Saxon, WA. Slope: 0.7% Drainage area: 129 sq. miles Discharge: 748 cfs	300 ft	Summary of data provided in email which gave NTU levels when monitoring was above 5 NTU's, WA water quality standard.	43.0	4 hrs#	5	5	Had constructed an in-channel deflector to move the bulk of the river flow away from construction site. Data analysis. Two years of data for the S.F. Nooksack River at Potter Road. Used July through September data. NTU:SS ratio = 1:1.9 Regression: SS = 1.7249*NTU + 0.5206 Used regression
Lower Hutchinson Creek Project South Fork Nooksack River Installation of ELJs and levee setback	Project located at confluence of Hutchinson Creek and S.F. Nooksack River near Acme, WA. LEJs installed on S.F. Nooksack and Hutchinson Creek. S.F. Nooksack Slope: 0.7% Drainage area: 129 sq. miles Discharge: 748 cfs Hutchinson Creek Slope: 1.1%	300, 1200, 3000 ft.	Daily monitoring was provided in NTU's. Most work occurred either in dewatered section of Hutchinson Creek or outside wetted channel.	300 ft. 14 12	 1 hr 0.5 hr	 3 2	 3 2	Hutchinson Creek was diverted. Unable to tell from data where samples were taken, used estimated average ratio of 1:2.0 from S.F. Nooksack River (see previous entry for Saxon Bank project) NTU:SS ratio = 1:2.0 Project had low turbidity, no monitoring was done at 1200 and 3000 ft.

Project and Watershed	Stream Characteristics at Project Location	Monitoring Locations	Original Sediment Data	Concentration (mg/L) used for determining SEV level.	Duration of elevated sediment concentration.	SEV (Juvenile and Adult Salmonids)	SEV Habitat	Comments
Bank Stabilization, continued								
Green River Fish Restoration Project Green River Installation of in-stream gravel nourishment and construction of 2 ELJs	Project located at RM 60 on the Green River. 2 miles east of Palmer WA. Drainage area: 231 sq. miles Discharge 958 cfs Slope: 0.8%	300, 600, 1200, 2500 ft	Data provided in NTUS. No background values provided, so used first couple readings of the day as background.	300				Data analysis; 29 years of data for the Green River at Kanaskat. Data collected at Cumberland-Palmer Road bridge. Used July and August data. Ratio: 1:1.7 Regression: S = 0.0983*NTU + 1.9326 Used ratio, regression data not correlated.
				19.0	3.25	4	4	
				20.5	11.75#	5	5	
				39.9	9.5**	5	5	
				45.5	5.25	5	5	
				16.6	5.0	4	4	
				63.5	11.25**	6	6	
				74.6	10.5#	6	6	
				112.3	2.75**	5	5	
				27.0	7.75**	5	5	
				9.0	9.5**	4	4	
				87.1	11**	6	6	
				118.4	8.5#	6	6	
				600				
				11.1	3.25	4	4	
				121.9	0.75	4	5	
				28.8	11.75#	5	5	
				31.3	9.5**	5	5	
				35.7	9.0#	5	5	
				9.9	5.0	4	4	
				58.6	11.25**	6	6	
				67.3	10.5#	6	6	
				10.7	2.75**	3	3	
				23.5	7.75**	5	5	
				9.9	9.5**	4	4	
				121.8	11**	6	7	
				100.6	8.5#	6	6	
				1200				
				22.4	4.75	4	4	
				36.7	11.75#	5	6	
				20.6	9**	5	5	
				23.5	11.5#	5	5	
				20.2	2.25**	4	4	
				48.3	11.25**	5	6	
				130.3	6.75#	6	6	
				19.7	7.75**	5	5	
				18.8	11.75#	5	5	
				143.1	11**	6	7	
				75.6	9.0#	6	6	
				2500				
				11.4	4.75	4	4	
				19.1	3.0	4	4	
				13.4	10.0**	4	5	
				26.9	9.5	5	5	
				12.5	2.25**	3	3	
				33.4	11.25**	5	5	
				67.7	2.25#	5	5	
				48.8	4.5	5	5	
				20.9	7.75**	5	5	
				12.7	9.5**	4	4	
				104.1	11**	6	6	
				63.4	10.0#	6	6	

Project and Watershed	Stream Characteristics at Project Location	Monitoring Locations	Original Sediment Data	Concentration (mg/L) used for determining SEV level.	Duration of elevated sediment concentration.	SEV (Juvenile and Adult Salmonids)	SEV Habitat	Comments
Bank Stabilization, continued								
Maple Creek Channel Reconstruction Thornton Creek 2 culvert removals, 2 bridge installations, channel reconstruction with habitat enhancement, boulder clusters, porous weirs, logjams, etc.	Project located on the S.F. Thornton Creek, just upstream of Hale School, above 30th St. NE bridge. S.F. Thornton Creek Drainage area: 12.1 sq. miles Discharge: 8 cfs Slope: 0.3% Bankful: 8 ft	200, 600, and 1660 ft downstream	Data provided in NTUs in graph. Estimated values from graph. Project site was dewatered, data collected during rewatering site.	200 ft 131.8 600 ft 48.1 1660 ft 40.5	 1.75 hrs 3 hrs 1.5 hrs	 5 5 4	 5 5 4	Site was dewatered and had excessive flows that overtopped diversion dams and flushed system prior to monitoring. Data analysis King County water quality data was used. 30 years of data for Thornton Creek collected at mouth. Used July and August data. Ratio: 1:2.5 Regression: SS = 3.2973*NTU - 3.6295 Used regression.
Bridge Construction and/or Repair								
SR 90 – Wilson Creek Bridge Widening Project Wilson Creek tributary to Yakima River	Project located on Wilson Creek at I-90 Bridge at Ellensburg WA. Slope: 0.6% Drainage area: 13 sq, miles	100 and 200 ft downstream		100 ft. 55.2 21.4 20.6 200 ft. 202.3 28.2 22.5	 1 hr# 6 hrs 1 hr 2 hrs 4.5 hrs 1 hr	 4 4 3 5 4 3	 4 5 3 6 5 3	Data analysis 3 years of data for Wilson Creek at Highway 821. Used July through September data. NTU:SS ratio = 1:3.2 Regression SS = 2.4425NTU + 6.2212 Used regression
SR – 12 Black River Bridge Scour Protection Black River – Tributary to Chehalis River. Placement of riprap to protect bridge column, placement of filter blanket and streambed gravel, installation of temporary work platform.	Project located on Black River, approximately 2 miles SE of Oakville, WA Slope: 0.2% Drainage area: 144 sq. miles Discharge: 162 cfs	300, 500 and 600 ft	Data provided in NTUs.	300 ft 10.6 8.8 9.6 18.8 500 ft 12.0 8.1 19.1 600 ft 12.5 6.4 12.8	 0.5 hr 5 hr 5 hr 1 hr# 4.5 hr 4.5 hr 1 hr# 2.5 hr 4.5 hr 1 hr#	 2 4 4 3 4 4 3 3 3 3	 2 4 4 3 4 4 3 3 3 3	Inwater silt curtain used. Data analysis: Ecology monitoring site at project location did not have turbidity and SS data. Used the data from the Black River at Moon Road Bridge monitoring station approximately 2 miles upstream. Six years of data available, July through September. NTU:SS ratio = 1:1.5 Regression had negative slope. Used ratio.

Project and Watershed	Stream Characteristics at Project Location	Monitoring Locations	Original Sediment Data	Concentration (mg/L) used for determining SEV level.	Duration of elevated sediment concentration.	SEV (Juvenile and Adult Salmonids)	SEV Habitat	Comments
Bridge Construction and/or Repair, continued								
Monroe Trestle Bridge Skykomish River Removal of railroad trestle	Project location is unknown. Project near City of Monroe WA. Discharge: 3,946 cfs Drainage area: 842 sq. miles Slope: 0.2%	300 ft (three locations across stream)	Turbidity was only high on one side of stream, that data is analyzed.	Site 1 6.9	32 hrs	5	5	Used sediment curtain around project. Data analysis 26 years of data for Skykomish River at Monroe. Used July through September data. NTU:SS ratio = 1:1.9 Regression: SS = 0.8453*NTU + 1.9163 Used regression
Humptulips River Bridge Scour Repair Humptulips River Project involved repair and augment riprap and placement of LWD	Project located on Humptulips River at US 101 Bridge. Slope 0.4% Drainage area: 276 sq. miles, 132 Sq. miles at project location Discharge: 1,340 cfs Bankfull at project location: 80-220 ft.	300 ft.	Measurements were recorded throughout the day, 5 to 7 times. Data provided in NTUs. Because time between monitoring sampling was anywhere from one to two hours during sediment generating activities, the peak turbidity values may not have been captured.	7.6 11.0	6.5 hrs** 7 hrs#	4 4	4 4	No stream dewatering occurred. Data analysis. 25 years of data for the Humptulips near Humptulips at the Highway 101 Bridge. Used July through September data. NTU:SS ratio = 1:1.6 SS = 0.6514*NTU + 1.1202 Used regression
Humptulips River Bridge Scour Repair Humptulips River Project involved installation of rock barbs and LWD in stream.	Project located on Humptulips River at US 101 Bridge. Slope 0.4% Drainage area: 276 sq. miles, 132 Sq. miles at project location Discharge: 1,340 cfs Bankfull at project location: 80-220 ft.	300 ft.	Met water quality standards.					

Project and Watershed	Stream Charact. at Project Location	Monitoring Locations	Original Sediment Data	Concentration (mg/L) used for determining SEV level.	Duration of elevated sediment concent.	SEV (Juvenile and Adult Salmonids)	SEV Habitat	Comments
Open Trench or Dredging of Stream								
Williams Pipeline, Mt. Vernon Loop North Fork Stillaguamish River Project involved installing a pipeline under the NF. Stillaguamish River	Project located on the NF Stillaguamish approximately 1 mile north of Arlington WA. Drainage area: 262 sq miles Discharge: 1,896 cfs Slope: 0.3%	100, 600, 2000 ft and 1 mile	Monitoring conducted throughout project. Project also took samples for analysis in lab. Regression equation determined from lab analysis: SS = 2.3237*NTU + 3.6702 Equation provides higher total suspended solids then Ecology data.	100 ft				Stream is diverted and dewatered during trenching. Open trench is exposed to river when one side of river is trenched and dredging occurred on opposite side. Data analysis. Used regression from project monitoring determined in lab for both SS and NTUs. Regression SS = 2.3237*NTU + 3.6702
				185.7	1 hr	5	5	
				220.2	1 hr	5	5	
				83.4	4 hr	5	5	
				113.8	9 hrs	6	6	
				95.5	1 hr	4	5	
				312.5	20 hrs	7	8	
				338.9	20 hrs	7	8	
				76.2	4 hrs	5	5	
				145.3	12 hrs	6	7	
				1070.5	29 hrs	8	9	
				676.6	6 hrs	7	8	
				132.0	9.5 hrs	6	7	
				93.5	5 hrs	5	6	
				600 ft				
				25.9	1 hr	3	3	
				16.7	0.5 hr	3	3	
				25.4	8.5 hrs	5	5	
				13.0	3 hrs	4	4	
				37.4	8.5 hrs	5	5	
				73.0	21 hrs	6	7	
				19.8	0.5 hr	3	3	
				135.3	20.5 hrs	7	7	
				23.7	0.5 hr	3	3	
				59.8	1.5 hr	4	4	
				50.7	9.5 hrs	5	6	
				293.1	31.5 hrs	7	8	
				41.7	5.5 hrs	5	5	
				122.4	10 hrs	6	6	
				12.7	9.5 hrs	4	4	
				12.7	9 hrs	4	4	
				2000 ft				
				12.6	3 hrs	4	4	
				25.9	1.5 hrs	4	4	
				14.1	4 hrs	4	4	
				34.7	9 hrs	5	5	
				45.3	2 hrs	4	4	
				212.8	18 hrs	7	7	
				25.3	5 hrs	4	5	
				30.4	10.5 hrs	5	5	
				18.2	4 hrs	4	4	
				185.7	14.5 hrs	7	7	
				22.8	7.5	5	5	
				75.7	5.5 hrs	5	6	
				75.4	9.5 hrs	6	6	
				32.0	1.5 hrs	4	4	
				22.9	1 hrs	3	3	
				1 mile				
				20.5	1.5 hrs	4	4	
				16.5	1.5 hrs	3	3	
				45.5	2.5 hrs	4	5	
				23.1	3.5 hrs	4	4	
				394.6	0.5 hr	5	5	
				232.4	17 hrs	7	7	
				22.3	4.5 hrs	4	4	
				46.6	5.5 hrs	5	5	
				25.3	3.5 hrs	4	4	
				123.2	9.5 hrs	6	6	
				30.5	6.5 hrs	5	5	
				22.9	3.5 hrs	4	4	
				45.4	9 hrs	5	6	

Project and Watershed	Stream Characteristics at Project Location	Monitoring Locations	Original Sediment Data	Concentration (mg/L) used for determining SEV level.	Duration of elevated sediment concentration.	SEV (Juvenile and Adult Salmonids)	SEV Habitat	Comments
Open Trench or Dredging of Stream, continued								
Williams Pipeline, Mt. Vernon Loop. Pilchuck River Project involved installing a pipeline under the Pilchuck River. Used open trench method.	Exact project location unknown, used location where pipeline crosses the Pilchuck on topo map. Located SW of Machias, WA. Slope: 0.4% Drainage area: 127 sq. miles Discharge: 744 cfs	100, 400, and 1000 ft	Measurements taken every hour throughout construction.	100 ft. 54.9 400 ft. 38.5 1000 ft. 34.8	62 hrs 57 hrs 51 hrs	7 6 6	7 7 7	River was not dewatered or diverted. Open water trenching. Data analysis. 14 years of data for the Pilchuck River at Snohomish at the Highway 2 Bridge. Used July through September data. NTU:SS ratio = 1:2.3 Regression SS = 1.4319*NTU + 2.5223 Used regression
Williams Pipeline – Sumas Loop		Construction method:						
Smith Creek	Trib to mainstem Nooksack River by Lawrence WA Slope: 0.8%	Dam and pump	Met water quality standards.					
Saar Creek (two locations where crossed creeks)	Trib to Frasier River, creek enters Canada, located near Sumas, WA Slope: 0.6%	#1: Open cut #2: Dam and pump	Met water quality standards. Met water quality standards.					
Kenny Creek	Unable to locate creek	Open cut	Met water quality standards.					
Unnamed trib to Sumas River	Located 2 miles SE of Nooksack, WA. Slope: 2.3%	Dam and pump	Met water quality standards.					
Breakenridge Cr.	Trib to Sumas River, located 2 miles east of Nooksack, WA Slope: 1.9%	Dam and pump	Met water quality standards.					
Williams Pipeline – Mt. Vernon Loop		Construction method:						
Armstrong Creek	Trib to mainstem Stillaguamish at Arlington, WA Slope: 0.5%	Dam and pump	Met water quality standards.					
Trib to SF Stillaguamish River	Unable to locate creek	Dam and pump	Met water quality standards.					

Project and Watershed	Stream Characteristics at Project Location	Monitoring Locations	Original Sediment Data	Concentration (mg/L) used for determining SEV level.	Duration of elevated sediment concentration.	SEV (Juvenile and Adult Salmonids)	SEV Habitat	Comments
Open Trench or Dredging of Stream, continued								
Williams Pipeline – Snohomish Loop		Construction method:						
Sternoff Crossing	Unable to locate creek	Flume	Met water quality standards.					
Seidel Creek – had Siedel Creek on monitoring form	Trib to Bear Creek, 1.4 miles NE of Avondale, WA, which enters Sammamish River. Slope: 1.0%	Dam and pump	Met water quality standards.					
Struve Creek	Trib to Bear Creek, 1.1 miles SE of Cottage Lake, WA, which enters Sammamish River. Slope: 3.0%	Dam and pump	Met water quality standards.					
Williams Pipeline – Ft. Lewis Loop		Construction method:						
Muck Creek	Trib to the Nisqually River. Site located on Ft. Lewis, 2.7 miles W of Rocky Ridge.	Open cut	Met water quality standards.					
South Fork Creek	Trib to the Nisqually River. Site located on Ft. Lewis, 2.7 miles W of Rocky Ridge. Just South of Muck Creek crossing.	Open cut	Met water quality standards.					
Williams Pipeline Ft. Lewis Loop Nisqually River Project involved installing a pipeline under the Nisqually River Used open trench method.	Project located 0.8 miles SW if McKenna, WA Drainage area: 517 sq. miles Discharge: 1,500 cfs Slope: 0.1%	600, 1250, 2500, 5200 ft, 2 miles, and 4 miles	Samples taken approximately every hour. Samples at 2 miles was only taken once, two samples were taken at 4 miles (4.5 hours apart). These samples were used to determine downstream extent of plume. Data provided in NTUs.	600 ft. 35.1 1,250 ft. 24.4 2500 ft. 16.2 5200 ft. 12.8 2 miles 15.5 4 miles 9.5	22 hrs 22 hrs 22 hrs 22 hrs 4.5** Used 4 miles time 4.5**	6 5 5 5 4 4	6 6 5 5 4 4	Open cut, no diversion or dewatering occurred. Data analysis. 3 years of data for the Nisqually River at McKenna. Used July through September data. NTU:SS ratio = 1:0.8 Regression SS = 0.7159*NTU + 0.5214 Used regression

Project and Watershed	Stream Characteristics at Project Location	Monitoring Locations	Original Sediment Data	Concentration (mg/L) used for determining SEV level.	Duration of elevated sediment concentration.	SEV (Juvenile and Adult Salmonids)	SEV Habitat	Comments
Open Trench or Dredging of Stream, continued								
Maintenance Dredging and Disposal, Lower Snohomish River Snohomish River Clamshell and hydraulic dredging were used on the Upper and Lower Sediment Basins and the Navigational Channel. Disposal location was at Elliott Bay for clamshell dredging and Port of Everett's Riverside Business Park Disposal Site for the hydraulic dredging.	Downstream settling basin is located immediately west of the Everett Marina. Upstream settling basin is located southeast of the I-5 Bridge.	Background monitoring occurred 300 feet upstream of dredging. Clamshell dredging: samples taken at 600 ft. Three samples taken, surface (2 foot depth), mid, and bottom (2 feet above bottom). Hydraulic dredging: 300 ft for dredging activities – surface, mid and bottom readings, 600 ft for disposal activities. Samples taken twice daily, once during slack tide, once during strong ebb or flood tide. --------- Ebb tide sampling at 300, 600, 1500, 2250, and 2480 ft.		Clamshell dredging Mid and bottom reading: 58.3 Additional samples taken during ebb tide, which exceeded background levels. Not enough information provided to determine concentration and duration. Hydraulic dredging All within water quality standards.	1 hr	4	4	High turbidity readings were in mid to lower samples which may have been in higher salinity waters, not freshwater from river. Sediment analysis: Project location is in tidally influenced area. No sediment monitoring at this time location. Used lowest Snohomish River data, near City of Snohomish. 25 years of data, December through February. NTU:SS ratio = 1:1.9. Regression SS = 1.2748*NTU + 4.8946 Used regression -------- Dredging stopped during strong ebb tides to reduce sediment impacts.
Grays Harbor Dredging.	Exact location with Grays Harbor was not provided. Project was in tidal area	Samples taken at 300 and 600 feet from dredging operation. Samples taken at surface, midwater, and bottom.	Data provided in NTUs	Met water quality standards. Midwater and bottom samples highly variable. When samples were above water quality, resampling both background and at monitoring location, showed in compliance.				
Miscellaneous Activities								
Mount Vernon Wastewater Treatment Plant Outfall Project Skagit River Project involved extending the outfall from the river bank out into the thalwag of the river.	Project located in City of Mount Vernon. Drainage area: 3,093 sq. miles Discharege: 14,000 cfs Slope: 0.1%	Monitoring occurred 100 feet upstream of project and 300 feet downstream	Data provided in NTUs	Met water quality standards for sheet pile driving (cofferdam) and dewatering, no information provided on putting water back into site and removing sheet piles.				

Project and Watershed	Stream Characteristics at Project Location	Monitoring Locations	Original Sediment Data	Concentration (mg/L) used for determining SEV level.	Duration of elevated sediment concentration.	SEV (Juvenile and Adult Salmonids)	SEV Habitat	Comments
Miscellaneous Activities, continued								
Silver Creek Dam Removal Tributary to the White River. Project involved removal of 10-year-old log stringer dam about 5 ft high.	Project located approximately 1120 ft upstream of the confluence with the White River, near Silver Springs Campground. Approximately 3.3 miles SE of Snoquera, WA on Highway 410. Drainage area: 8.0 sq. miles Slope: 8.4% Discharege: 8.3 cfs	159, 559, and 1118 ft downstream	Data provided in NTUs in graph. Estimated values from graph. Project site was not dewatered, logs pulled out of stream and sediment released.	159 ft 114.5 559 ft 157.0 1118 ft. 55.2	1 hr 0.75 0.75	5 5 4	5 5 4	No BMPs or conservation measures used to minimize sedimentation. Sediment analysis. No gage located on creek. Paul Bakke monitored project and determined NTU to suspended sediment ratio of 1:1.9789 Used ratio: 1:2

* Values calculated from monitoring report. Concentration calculated using equation tons/day = 0.0027* cfs* mg/L (USACE (U.S. Army Corps of Engineers) 1995). Background concentration 1.5 mg/L (average). Stream velocity 2.76 cfs. Duration: monitoring report stated sediment concentration levels decreased to near pre-removal levels in about 24 hours (used for average values), peak values based on 8 to 10 hour work day.

** Exact duration is unknown as monitoring stopped when work day was over. Unable to determine when concentrations returned to baseline.

Exact duration is unknown as monitoring did not provide start or stop times to be able to make accurate determination.

Wildfire and Native Fish: Issues of Forest He of Sensitive Species

By Bruce Rieman and Jim Clayton

ABSTRACT

Issues related to forest health and the threat of larger, more destructive wildfires have led to major new initiatives to restructure and recompose forest communities in the western United States. Proposed solutions will depend, in part, on silvicultural treatments and prescribed burning. Large fires can produce dramatic changes in aquatic systems, including altered sediment and flow regimes, fish mortality, and even local extinctions. Responses of salmonid populations to large disturbances such as fire indicate that complexity and spatial diversity of habitats are important to the resilience and persistence of populations. Some populations retain the ecological diversity necessary to persist in the face of large fires, and natural events such as wildfire have been important in creating and maintaining habitat diversity. Although timber harvest and fire can precipitate similar changes in watershed processes, we do not necessarily expect the physical and ecological consequences of large fires and timber harvest to be the same. We agree that healthy forests are fundamental to healthy aquatic ecosystems. In their haste to restore unhealthy forests, however, managers must take care to avoid simplistic solutions that compound problems already present in the management of aquatic ecosystems and native fishes. Management to restore ecological structure, composition, and process is largely experimental and potentially risky. We propose that the mosaic of conditions in both terrestrial and aquatic systems provides an opportunity to learn and adapt new management without placing key remnant aquatic habitats and populations at risk.

Some concerned scientists and managers have suggested that the increasing risk of uncharacteristically large or damaging wildfires in forests across the western United States may threaten the integrity of whole landscapes. In response, major initiatives to correct such problems have been proposed, some of which rely largely on silvicultural activities such as increased timber harvest, thinning, and prescribed fire. Do these and associated activities represent higher or lower risks for native fishes and aquatic environments?

Wildfires and forest health have dominated much of the discussion and interest regarding management of forested lands in the West in recent years. It is increasingly evident that forested landscapes throughout the region have been dramatically altered by past land management activities. Selective and extensive timber harvest, silviculture, fire suppression, and grazing practices have substantially changed the structure and composition of forest communities (Franklin 1992; Lehmkuhl et al. 1994; Hessburg et al. 1997). The general decline in abundance of large, old trees and old forest (Henjum et al. 1994; Hessburg et al. 1997) is a familiar issue. But ecologists are now recognizing that changes in structure and composition are symptomatic of more fundamental changes in patterns of disturbance and other ecological processes. Forests that were once mosaics of species, ages, and patterns in crown cover have been simplified. Many are now dominated by higher-density, middle-aged stands that are more vulnerable to pest infestations and fire (Huff et al. 1995; Hessburg et al. 1997; Hann et al. 1997). The more homogeneous patterns in vegetation and the increased fuel loadings are thought to increase landscape vulnerability to larger stand-replacement fires (Agee 1988, 1994; Huff et al. 1995). Large fires burning throughout the West in recent years may have been the result of these changes (e.g., Christensen et al. 1989; Barbouletos and Morelan 1995), although climatic patterns may be important as well (e.g., Johnson et al. 1995, 1996).

The possibility of uncharacteristically large, damaging fires and declining forest health holds important implications for land managers. Larger fires mean lost timber values, increased threats to private property on an expanding urban-wild-land interface, and higher costs associated with fire suppression. Uncharacteristic fires may also threaten properties of ecosystems. Direct effects of intense fires and subsequent impacts on hydrologic regimes, erosion, debris flows, woody debris, and riparian cover can strongly influence the structure and function of aquatic systems (Swanson 1981; Megahan 1991; Bozek and Young 1994). Intense fires and the associated environmental responses (e.g., elevated stream temperatures, large sediment pulses, and debris flows) may

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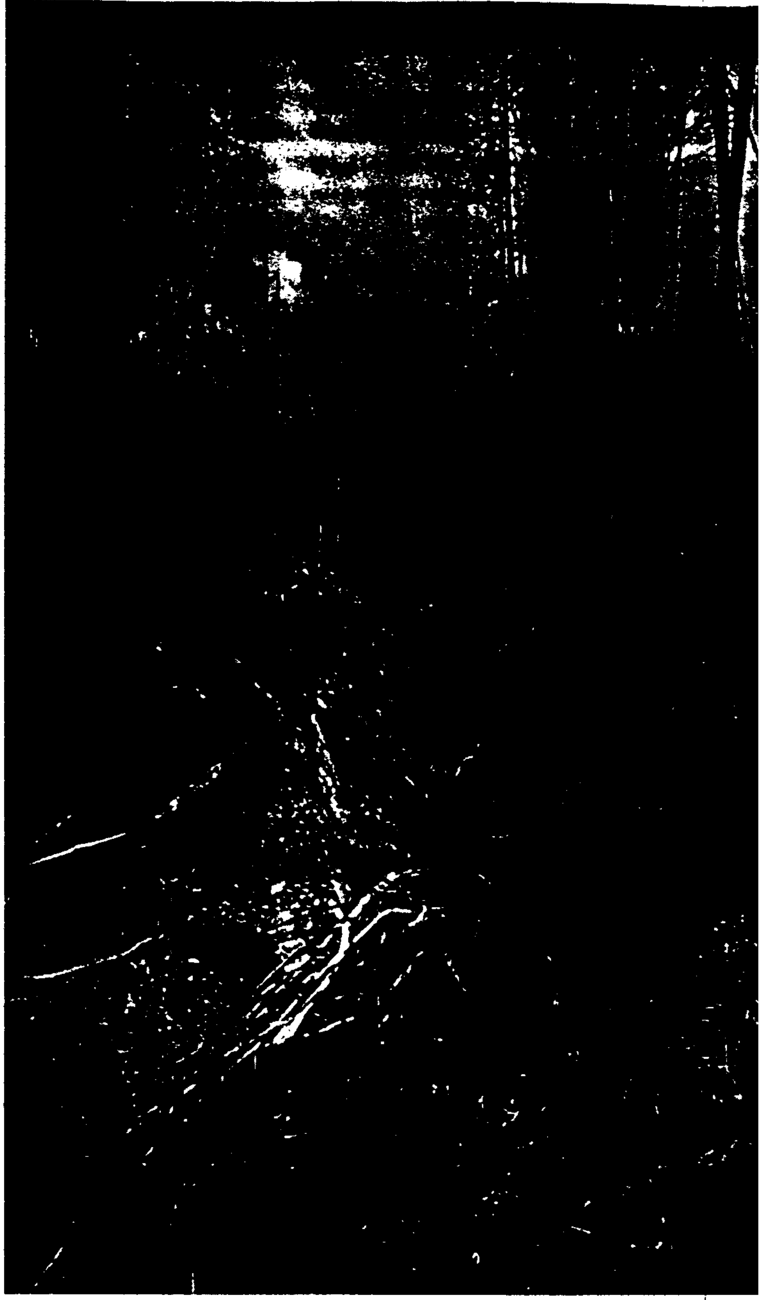
Alan Barta



Wildfires may trigger major debris flows, floods, and erosion that can significantly harm or even destroy small fish populations and other aquatic biota.

result in direct mortality of fish (Minshall and Brock 1991; Bozek and Young 1994; Rinne 1996) and other aquatic organisms (Rinne 1996), and even extinction of local populations (Propst et al. 1992; Rinne 1996).

Concern about such severe possibilities has focused interest on active vegetation management, galvanizing efforts to actively restructure or recompose forest communities (Barbouletos and Morelan 1995; see other papers in Eskew 1995). Managers have justified these initiatives by citing the spectrum of potential social, economic, and ecological costs. For example, some managers have argued that changing fire patterns represent one of the most important threats to the persistence of native, threatened, endangered, or sensitive species such as bull trout (*Salvelinus confluentus*), chinook salmon (*Oncorhynchus tshawytscha*), and redband/rainbow trout (*O. mykiss*). Given the general decline of many native fishes throughout the West (Williams et al. 1989; Nehlsen et al. 1991; Lee et al. 1997) and the seemingly pervasive nature of forest health problems, it may be prudent to proceed quickly with active vegetation management in all "unhealthy" forests. But such management carries costs and risks that also must be considered.

Forest health and ecosystem management initiatives seek to recreate a structure and composition in forest communities that is more consistent with natural disturbance regimes. Although prescribed and natural fires can play an important role (Carlson et al. 1995; Huff et al. 1995), silvicultural activities that include thinning and harvest will be desirable where feasible. Although wildfire and timber harvest both can result in changes to watershed processes that conceptually are similar (e.g., increased surface erosion and water yield), they also may differ fundamentally. Disturbance by fire affects ecosystems in complex ways, but activities that reduce the risk of fire, insect, or pathogen outbreaks by changing vegetation structure and composition, and fuel management also lead to complicated effects. Mimicking natural disturbance may not be as easy as hoped.

We believe that managers often hold overly simplistic views of the forest health problem. In our experience, some believe that forest health can be improved simply by managing vegetation through silvicultural treatments that manipulate structure and composition. While this viewpoint is attractive because it provides some clear direction for managers, key elements of large ecosystems such as soil and watershed processes can be discounted. Land management activities influence many ecosystem components, and clearly more than the forests have changed. In other cases managers recognize that forest attributes are part of a larger ecosystem, but still seem to imply that treating symptoms (terrestrial vegetation) will lead to a cascade of solutions. Put simply, restoring terrestrial vegetation to a more natural condition should lead to improvements in overall ecosystem health. Pathways to forest health decline were largely a result of vegetation manipulations. Can't those pathways be retraced?

Perhaps, but ecological function does not follow directly from structure. Rather, both structure and function emerge from the underlying process (Bradshaw 1996). In other words, how we get where we wish to go may be just as important as where we wish to go. Silvicultural and prescribed fire management tools can lead to the goal of restoring stand structure and composition, but such restoration may have mixed implications for restoring other aspects of failing ecosystems such as fishes and their habitats. ◆

We do not question the need to manage landscapes and forests in ways more consistent with natural patterns of disturbance (*sensu* Attiwill 1994). We suggest the issue is not whether to do something or nothing, but rather to weigh the risks of our actions as objectively as possible. To that end, we believe it is useful to consider some of the physical and biological processes likely to influence streams, fish habitats, and populations after large wildfires or management intended to reduce the probability of such fires.

Physical Processes

The intensity and distribution of natural disturbance and ecosystem responses to disturbance are strongly influenced by landscape attributes. Swanson et al. (1988) suggest that land form position and slope gradient affect flow of energy and matter (water, solutes, and particulate matter) and, therefore, influence the pattern and intensity of disturbance by flow processes. Thus, natural barriers such as ridges and valley bottoms control the spread of fire, and slope and channel gradients dictate the velocity and travel distance of sediment movement. The result is a somewhat predictable, although dynamic, patchiness of disturbance at the landscape scale, constrained by land form and slope gradients. Disturbance due to management activities likely is less constrained because of human ability to circumvent natural barriers. Swanson et al. (1990) suggest that this has led to creation of new landscape disturbance patterns in the Pacific Northwest with little regard for the ecologic design of management activities. The implications of the changed disturbance regime at the landscape scale are not easily interpreted. However, a wealth of information exists about natural and management induced disturbances as they affect physical processes on mountain slopes.

Temperature—Changes in stream temperature are controlled by the amount of energy (heat) exchange and the mass of water in the stream. While many processes affect energy exchange, including evaporative cooling and conductive transfer with substrate, the controlling factor is solar radiation (Brown 1969). There is anecdotal evidence of dramatic, short-term temperature elevations associated with fire in small streams (Minshall and Brock 1991). In addition to latitude, slope, aspect, frequency of cloudy sky, and time of year, incoming radiation is correlated with shading provided by streamside vegetation and terrain and, therefore, is subject to change from fire (Helvey 1972; Amaranthus et al. 1989) or logging (Brown and Krygier 1970; Meehan 1970; Patton 1973). Loss of shading can cause mean annual maximum temperatures to rise by as much as 15°C (Brown and Krygier 1970). The effects of canopy loss on stream temperature by fire or logging disturbance may be approximately equal if the fire consumes the canopy. However, many fires don't consume canopy foliage, and conifer needles may be retained following crown fires for up to a year. In addition, standing tree boles adjacent to streams provide some shade for many years. Streamside vegetation recovery is usually rapid after logging or a fire unless soil is compacted during timber harvest and yarding. Recovery of streamside vegetation is generally faster in moist coastal ecosystems than in drier inland systems, and recovery is faster at lower elevations than higher ones (Beschta et al. 1987).

Erosion—Disturbance by fire, harvest activities, and road construction invariably results in greater erosion and sediment production; however, the severity and longevity of increase is highly dependent on site properties and the kind of disturbance. At the broad scale, erosional processes differ due to lithology, land form, and local climate. Consequently, landscapes are predisposed to variation in erosional processes and efficiencies of sediment transport during times of relative stability and after disturbance. A thorough discussion that distinguishes these process differences in natural landscapes can be found in Swanston (1991) and Megahan (1991).

Disturbances that change the way a slope handles water are likely to result in more erosion and to persist longer than disturbances that reduce cover. Fire consumption of forest floor or removal of soil cover during log skidding generally increases surface erosion rates. Effects are usually short-lived (a few years), although harsh sites may not reestablish effective ground cover for more than a decade (Megahan et al. 1995). In contrast, fire-induced water repellence (DeBano et al. 1970) increases the risk of surface rilling, "slurry flows" (Rinne 1996), debris flows, and hyperconcentrated flood events by reducing the infiltration capacity of soils. While reduced infiltration due to fire-induced water repellence is brief—usually a few years—channel recovery (reestablishing soil and mature vegetation) after debris flows or floods may take decades (Megahan 1991) or even centuries (Benda 1985). These effects are most common in headwater basins, although structure and habitat in receiving streams may be altered for long periods through deposition and scour (Swanston 1991).

Timber harvest and fire both increase the likelihood of mass erosion on sites. The presence of roots and tree stems imparts a measure of strength to soils, decreasing the likelihood of mass failure (Ziemer and Swanston 1977; Gray and Megahan 1981). This is particularly important in colluvial soils because frequent erosion and deposition in these sites provide little time for soil development and pedogenic processes such as weathering and organic matter accretion to promote soil structure and increase resistance to mass erosion. Soil strength declines as roots decay and typically reaches a minimum in 5–10 years after disturbance, but buttressing by large trees may last for several decades after mortality. Vegetation loss also results in decreased water removal from soil by transpiration. Higher antecedent soil moisture during rainfall or snowmelt increases the probability of flooding and the occurrence of shallow, rapid landslides, and can activate flowing water in ephemeral draws. In cohesionless soils, "sapping" failures (Megahan and Bohn 1989) and channel headcutting may greatly accelerate sediment production from ephemeral draws.

Road construction causes the most severe disturbance to soils on slopes, far overshadowing fire and logging as a cause of accelerated erosion (Swanson and Dyrness 1975; Beschta 1978; Reid and Dunne 1984). Perhaps the major reason is that the excavation required to build a road on a slope results in a disruption of subsurface water transport, bringing water to the surface where flow is concentrated, and velocities are much higher (Megahan 1972). Other reasons include disruption of soil structure and vegetation rooting; sidecasting of loose, unconsolidated roadfill; increased connections between roads and streams by gully formation and slides; and oversteepened cutslopes. Accelerated sediment production from roads depends on how recent construction was and on whether road design features that mitigate erosion such as special drainage features, structures, revegetation, and surfacing treatments were incorporated. Acceleration factors for sediment production range from tens to hundreds of times over natural rates in forested areas, based on various studies in the mountain west (Furniss et al. 1991). Typically, sediment production rates decline by an order of magnitude within three years of construction; however, these accelerated rates remain for the life of the road. Secondary surface erosion on mass failures is often chronic and may be as damaging to stream biota as episodic inputs from mass failures because the sediment particles are finer and are delivered throughout a long time.

Road construction causes the most severe disturbance to soils on slopes, far overshadowing fire and logging as a cause of accelerated erosion.

Surface and mass-erosion risk are increased after fire or logging, and rates and recovery times are generally similar. Fire-induced water repellence may greatly increase the likelihood of flooding, surface soil rilling, and hyperconcentrated flow events in first- and second-order channels. This elevated risk diminishes rapidly with time, but accelerated erosion and channel scour can be dramatic when high-intensity summer rainstorms hit recent burns if soils are water-repellent. Sediment production from logging and fire accelerates at similar rates. Increases may range from 10- to 100-fold and usually decline to natural levels within a decade (Megahan 1980). Rapid erosion from road disturbance is typically greater than accelerated erosion from fire or logging disturbance and persists for the life of the road. Effective road placement and design features can mitigate erosion from roads, but studies clearly identify accelerated sediment production from roads to be larger and more chronic than sediment production from fire or logging.

Streamflow—Water yield and timing of peak and low flows also are affected by disturbance. Harvesting and fire may alter the spatial pattern of snow deposition and redistribution, and change interception, evaporation, rate of snowmelt, and storage of water in soil by decreasing transpiration. Changes in water yield after disturbance depend on climatic factors, vegetation changes, and soil properties. Typically, the largest increases in yield occur on sites that received precipitation or snowmelt during the growing season, sites that were fully vegetated and had a large change in transpiration, and on deep soils with a high water-holding capacity. In the West, water yield increases after vegetation disturbance range from not detectable to 40% in the first year

(Chamberlin et al. 1991); typical recovery to predisturbance levels take 30–50 years. Peak flows average 20% higher after a forest is cut in small patches in a snowmelt-dominated regime (Troendle and King 1985). Effects of roads on streamflow response are varied. Peak flows and flow duration have either increased, decreased, or remained unchanged in basins after road construction (King and Tennyson 1984; Jones and Grant 1996). Maximum effects on fish habitat (both beneficial and detrimental) probably occur when disturbance alters peak flows or low flows.

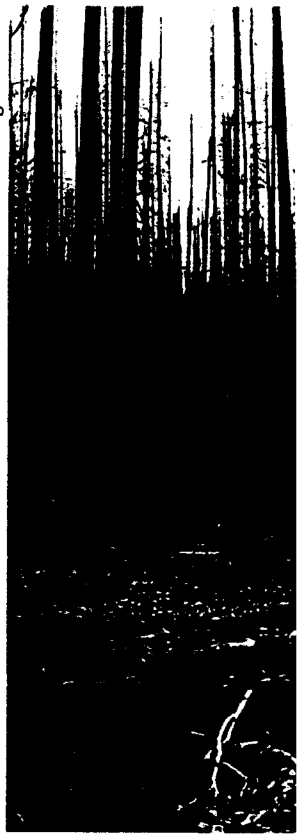
Water Chemistry—Stream chemistry changes have been documented after disturbance by fire or logging. Short-term (1-3 *years)* elevated concentrations of dissolved nitrate, cations, and alkalinity have been reported. With the exception of extreme conditions during or immediately following fires across some small streams (Minshall et al. 1989; Rinne 1996), such changes are inconsequential to aquatic biota. Downstream eutrophication, particularly in lakes and reservoirs, may be a concern that we do not consider in this paper.

Biological Processes and Fishes

The effects of large fires on fishes are direct and indirect. Direct effects may result from changes in water temperature and chemistry (Minshall et al. 1989; McMahon and de Calesta 1990; Minshall and Brock 1991). Direct effects may include mortality and displacement of individuals, but few studies have documented the extent of these responses (McMahon and de Calesta 1990). Clearly, fires can result in immediate mortalities (McMahon and de Calesta 1990; Minshall and Brock 1991; Rieman et al. 1997), but actual mechanisms and interactions are poorly understood (Minshall et al. 1989; Minshall and Brock 1991; Bozek and Young 1994) and may be highly variable. Rieman et al. (1997) noted areas where no live fish and numerous dead fish were found immediately after an intense fire burned through riparian corridors of two streams in Idaho. In a third stream in a similar setting and immediately following a similar fire, fish densities were depressed in intensely burned reaches, but no areas were devoid of fish. Minshall et al. (1989) suggested that changes in water temperature and water chemistry during or immediately after the Yellowstone fires were relatively minor. Minshall and Brock (1991) suggested such effects were more likely to be important in smaller rather than larger streams. Rinne (1996) found no significant reduction in densities of fish as a direct result of a large fire burning across three headwater streams in Arizona.

Indirect effects of large fires are somewhat better understood and can be dramatic. Bozek and Young (1994) noted mortalities of Yellowstone cutthroat trout *(Oncorhynchus clarki bouvieri)* associated with increased suspended sediments in streams two years after large fires in the Greater Yellowstone ecosystem. Floods and debris flows triggered by fires and subsequent storm events have apparently eliminated fish from stream reaches in Montana (Novak and White 1989) and Idaho. Others have documented the local extinction of small populations of salmonids after similar events (Propst et al. 1992; Rinne 1996).

A general conclusion is that large fires can, in the short term, result in substantial mortality and even local extinctions. However, the result may be largely a function of scale. The relative magnitude of an effect is likely to be strongly influenced by the intensity and severity of the fire, which will, in turn, be influenced by patterns of living vegetation, fuel loads, fire weather, and terrain. Small streams are more likely to be influenced stronger than larger streams. Even when large regions are burned, the mosaic of conditions across streams will produce a patchwork of disturbance effects (e.g., Minshall and Brock 1991). The effects of fires on watershed and ecological processes may be evident for decades (Minshall et al. 1989; Minshall and Brock 1991; Huntington 1995), but they may not necessarily be catastrophic.


Rodger L. Nelson

Studies of fish populations after large disturbances or experimental defaunation have concluded that population recovery can occur quickly, frequently within a few years (Niemi et al. 1990; Detenbeck et al. 1992), weeks (Sheldon and Meffe 1995), or even days (Peterson and Bayley 1993). To consider risks and tradeoffs associated with large fires and management designed to mitigate those fires, we should consider characteristics of populations and habitats likely to influence biological responses. Two elements seem important: (1) refounding of affected streams or reaches through dispersal from local or internal refugia and (2) stabilizing effects that emerge from complex life histories and overlapping generations in fish populations.

Refugia—Rieman et al. (1997) followed responses of three rainbow trout and two bull trout populations in three watersheds after two of the largest, most-intense fires recorded in Boise National Forest, Idaho. Despite the large sizes, the fires in these watersheds left an assortment of intensely burned, lightly burned, and unburned areas that immediately and directly influenced fish distributions. In one stream, depauparate reaches were interspersed with areas supporting high densities of fish. In two streams, moderate or high densities of fish were found immediately below long (2 km–5 km) reaches devoid of fish. Fish may have been completely eliminated from some smaller tributary streams. In years after the fires, debris flows occurred in tributary streams of each watershed. Although populations were depressed, fires in 1992 and 1994 did not produce a uniform or complete elimination of fish or disruption of habitat. Rather, fish numbers increased somewhat dramatically, often exceeding those before the fire or those in streams not influenced by fire (Rieman et al. 1997).

Population responses after the Boise fires were apparently influenced by the presence of refugia (i.e., habitats or stream reaches that supported concentrations of fish during and after the fires). In one system where fish persisted in patches throughout the mainstem, numbers and age structure approached preburn conditions in the first year. Fish were reestablished in depauparate reaches by dispersal from both upstream and downstream sources over short distances. In two other streams where fish were apparently eliminated in the upper reaches and some tributaries, recolonization must have occurred through dispersal from downstream sources over longer distances and time.

The mechanism and rate of recolonization are likely to be influenced by the local environment (Sheldon and Meffe 1995). Mechanisms that stimulate recovery or compensate for habitat losses might even be triggered or enhanced by the disturbance (Bisson et al. 1988, 1997; Minshall et al. 1989). Although some scientists have reported degradation of juvenile habitats after fire (Minshall et al. 1989), such changes also may result in increased production. Rieman et al. (1997) found high densities for young-of-the-year trout in several reaches one and two years after the Boise fires in all of the watersheds directly influenced by the fires. In small, very cold streams, increased solar exposure may result in warmer water, increased primary and secondary production (Minshall et al. 1989; Minshall and Brock 1991), and faster growth or higher carrying capacities for juvenile fish (Murphy and Meehan 1991). Similar positive responses in growth and production have been observed for other salmonids after large disturbances (Bisson et al. 1988). Although fires may kill fish and depress production in some life stages for some time (Minshall et al. 1989), they also may create the potential for important compensation and subsequent dispersal within or among populations. The importance of spatially redundant and complex or heterogeneous habitats to the persistence of populations is well established in theory (den Boer 1968; Dunning et al. 1992), and there is growing empirical support (e.g., Pearsons et al. 1992). A complex, well-dispersed network of habitats is likely to be an important element in the persistence of fishes during and after large fires.

Streamside vegetation grows below the badly burned trees in Fall Creek, Payette National Forest, after a fire during summer 1994.

Life History—Many salmonids exhibit a suite of life history forms that include varied patterns of movement, age and timing of maturity, and habitat utilization (Northcote 1992; Thorpe 1994a, 1994b). Multiple forms (i.e., resident and migratory) may occur in proximity to each other, perhaps even in sympatry. The diversity and plasticity of life histories have been viewed as stabilizing mechanisms for populations in variable environments (Gross 1991; Thorpe 1994 b). For example, Rieman et al. (1997) and Novak and White (1989) provide evidence that salmonid populations, severely depressed by the effects of fires, persisted through the presence of a migratory life history form. In essence, although fires caused severe mortality or even eliminated the fish in some streams, populations persisted because some members had migrated outside of the affected area, returning later to spawn. Life history pattern apparently provided both a temporal and spatial hedge against extinction; had these populations been restricted to nonmigratory forms, they might well be extinct. Similar results have been reported for salmonid populations influenced by other disturbances (Titus and Mosegaard 1992; Armstrong et al. 1994).

Complex life histories may be the result of historical patterns of disturbance. For example, spawning and rearing habitats of bull trout are distributed primarily in colder, higher-elevation watersheds (Rieman and McIntyre 1995). Because high-elevation areas throughout the region were more likely to experience mixed or high-intensity fires (Arno 1980), we might expect that, in an evolutionary sense, bull trout and similarly distributed species are well acquainted with large, intense fires. The existence of complex life histories such as the mixed migratory behaviors and overlapping generations found in the Boise River basin could be the expression of strategies that have emerged because of periodic disturbances like fire.

The broad expression of life histories appears dependent on the heterogeneity and connectivity of habitats. Migratory patterns require access to networks of streams that may extend from hundreds of meters to hundreds of kilometers (Bjornn and Mallet 1964; Fausch and Young 1995). Although different forms may occur in the same streams or watersheds, evidence exists that some differences have emerged as adaptations to, or phenotypic expressions against, a template of heterogeneous habitats (Healey 1994; Healey and Prince 1995; Lichatowich and Mobrand 1995). If the spatial and temporal complexity of habitats is lost, the expression of complex life histories may be lost as well.

The Effects of Management—If refugia and the expression of life histories are critical to the recovery and persistence of fishes influenced by large fires, the condition of available habitats and effects of past management should be key elements influencing evaluations of risk. The potential effects of management on the ability of populations to respond to large disturbances must be weighed. Throughout the Pacific Northwest, the effects of chronic watershed disturbance by road-building and timber harvest; introduced species; and barriers such as dams, diversions, and road culverts have resulted in fragmented, isolated salmonid populations and the elimination or restriction of life-history patterns (Rieman and McIntyre 1993; Frissell et al. 1997). The status of native fishes has been negatively associated with indices of human-related disturbance such as the density of roads (Lee et al. 1997). Although the mechanisms are varied, the risks are clear. Through efforts to intensely manage land and associated activities such as timber harvest, we risk expanding the disruption of watersheds and populations of native fishes. In general, attempts to minimize such effects in any single watershed have led to the dispersal of disruptive activities across broad areas (Reeves et al. 1995). The chronic nature of land management and other human-related disturbances has led to lost spatial complexity of stream environments that ultimately may be reflected in the loss of complexity, diversity, and distribution of populations and life histories (Frissell et al. 1993; Reeves et al. 1995). That loss might well erode the ability of populations to respond to the effects of large fires as well as large storms, floods, and other natural events that we cannot hope to control.

Conclusions and Future Direction

The tradeoffs between vegetation management and wildfire are not simple. In some cases short-term rates of erosion and sediment delivery after a fire may be larger than the effects of roads and timber harvest. After a fire, changes in vegetation and watersheds influencing hydrologic and temperature regimes and erosion may persist for years, perhaps decades. However, the long-term legacy can be important. Substantial inputs of large wood and coarse sediments also are likely to follow large fires (Brown 1989; Reeves et al. 1995; Young 1994). The larger materials often store fine sediments and provide the hydraulic complexity necessary for sorting substrates important to fish habitat. After fires, intense debris flow and scour events that generate sediment are often localized (Megahan 1991) and prevail primarily in smaller, high-gradient channels (Swanson et al. 1990). Although the volumes of fine sediments can be large, they may be relatively short-lived and patchy in relation to the effects of other more-chronic disturbances associated with roads and timber harvest (Reeves et al. 1995). Historically, the episodic contribution of coarse debris may have been key to the creation and maintenance of complex instream habitats (Swanson and Lienkaemper 1978; Reeves et al. 1995). Emerging theory supports the idea that natural disturbances have been critical to the maintenance of such habitats, the productivity of associated populations (Reeves et al. 1995; Bisson et al. 1997), and the broad expression of life histories. Recent fire suppression could well have contributed to the overall decline in productivity of fish habitats throughout the region.

Large fires can produce local extinctions of small, isolated populations. However, many species and populations may still have the ecological diversity necessary to persist. Although wildfires may create important changes in watershed processes often considered harmful for fish or fish habitats, the spatial and temporal nature of disturbance is important. Fire and the associated hydrologic effects can be characterized as "pulsed" disturbances (*sensu* Yount and Niemi 1990) as opposed to the more chronic or "press" effects linked to permanent road networks. Species such as bull trout and redband trout appear to have been well adapted to such pulsed disturbance. The population characteristics that provide for resilience in the face of such events, however, likely depend on large, well-connected, and spatially complex habitats that can be lost through chronic effects of other management. Critical elements to resilience and persistence of many populations for these and similar species will be maintaining and restoring complex habitats across a network of streams and watersheds. Intensive land management could make that a difficult job.

Fire and management-related disturbances may cause important, and in some respects similar, changes in physical and biological systems. However, we do not expect conventional timber harvest and road to produce effects equal to fire. Also clear is that attempts to manipulate the structure, composition, and processes of whole ecosystems are largely experimental (Baker 1994; Stanley 1995; Kimmins 1996). In haste, managers have justified actions by citing such concerns as risks to biological communities and sensitive fishes. Undoubtedly, cases exist where the risks of large fires outweigh the risks of intensive management, but those will be clarified only through careful evaluation. Because past management and human disturbance have led to much change in terrestrial and aquatic ecosystems, we suspect that forested landscapes most in need of restoration will often coincide with watersheds and aquatic systems in similar condition (Quigley et al. 1996). That pattern could represent both potential conflicts and opportunities for managers struggling with issues of forest health, wildfire, and sensitive fishes.

Where important, but depressed and small or isolated populations of sensitive fishes persist in landscapes at high risk of uncharacteristic wildfire, populations are threatened both by our management and the lack of it. In these circumstances management should proceed only with the greatest possible care. Silvicultural prescriptions that do not require new or reconstructed road systems and that emphasize prescribed fire or low-impact logging and yarding systems would be clear priorities. Restoring watershed processes through the stabilization and obliteration of roads, hill slope stabilization, and revegetation of riparian areas could be key elements in any plan. Careful watershed and hydrologic analyses are required to weigh any tradeoffs.

Where populations persist in a matrix of healthy watersheds and productive habitats mixed with those in poorer condition, the opportunity for more progressive and adaptive management is greater. Because healthier populations are more likely to persist in the face of even large fires, the need to restructure forests is unlikely to be an immediate priority from an aquatic perspective and, potentially, is a threat. In many cases the extent of the forest health problem is large, and not all lands can be treated in the foreseeable future, even if tradeoffs were clear. We suggest that logical priorities lie in watersheds that are either less or not important in terms of aquatic biological diversity or critical populations, or that have been so strongly altered through past management that further losses are likely to be minor. Prioritizing activities in these areas offers several advantages. First, because management of large systems is largely experimental, new approaches can be tested without placing key aquatic systems at further risk. Second, successfully reestablishing more natural patterns and processes could lead to long-term restoration of more complex, productive aquatic habitats. The long-term restoration of a matrix of whole watersheds could ultimately lead to the wider distribution of habitats necessary to support a broader expression of aquatic biological diversity. The potential for success might be substantially greater if forest restoration projects included the obliteration of particularly damaging roads. Third, working in the matrix would represent progress toward reducing the continuity of fuels, diminishing the risk of uncharacteristically large fires, and restoring biological diversity in terrestrial systems (*sensu* Carey and Curtis 1996).

The approach we suggest will require a broader perspective. As managers attempt to restore more natural patterns in forests (and aquatic ecosystems), both the distribution and intensity of activities should be guided by a landscape perspective of risk and opportunity. A strategic vision of historical condition and variation, current patterns, and long-term potential will be critical. It is unrealistic to hope to return all landscapes to their historical conditions from either an aquatic or forest perspective. Prioritizing efforts that minimize risks to remaining elements while maximizing learning and opportunities to create something more functional will be key to progress. As managers learn to create more natural patterns and restore critical processes through active manipulations of forests and watersheds, landscape management that incorporates planned disturbances could conceivably replace a system of watershed and riparian reserves as the solution for maintaining aquatic diversity and ecological function in forested ecosystems (Reeves et al. 1995; Carey et al. 1996). Ultimately, we believe that healthy forests and healthy aquatic communities should be elements of the same problem (*sensu* Franklin 1992), not competing issues in resource allocation.

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Fire Probability, Fuel Treatment Effectiveness and Ecological Tradeoffs in Western U.S. Public Forests

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Abstract: Fuel treatment effectiveness and non-treatment risks can be estimated from the probability of fire occurrence. Using extensive fire records for western US Forest Service lands, we estimate fuel treatments have a mean probability of 2.0-7.9% of encountering moderate- or high-severity fire during an assumed 20-year period of reduced fuels.

INTRODUCTION

Fuel treatments to reduce fire impacts have been promoted as a public forest restoration priority by policy [1] and the Healthy Forests Restoration Act of 2003. It is difficult to generalize about the effectiveness of fuel treatments under all conditions [2, 3], but treatments are not universally effective when fire affects treated areas [4]. Factors influencing effectiveness include forest type, fire weather [4], and treatment method [5].

However, treatments cannot reduce fire severity and consequent impacts, if fire does not affect treated areas while fuels are reduced. Fuels rebound after treatment, eventually negating treatment effects [3, 6]. Therefore, the necessary, but not sufficient, condition for fuel treatment effectiveness is that a fire affects a treated area while the fuels that contribute to high-severity fire have been reduced. Thus, fire occurrence within the window of effective fuel reduction exerts an overarching control on the probability of fuel treatment effectiveness. The probability of this confluence of events can be estimated from fire records. Although this probability has not been rigorously analyzed, it has often been assumed to be high [7].

The probability of future fire occurrence also abets assessing the ecological risks incurred if fuels are not treated. Therefore, analysis of the likelihood of fire is central to estimating likely risks, costs and benefits incurred with the treatment or non-treatment of fuels.

Assessing fire occurrence and its effect on fuel treatment effectiveness also has merit because treatments can incur ecological costs, including negative impacts on aquatic systems [8], soils [7], and invasion by non-native plants [9, 10]. Here, we use watershed and aquatic systems as a specific context for evaluating tradeoffs involved with treatment and non-treatment of fuels on western public lands. However, the analysis applies to upland ecosystems as well.

The effects of fire on watersheds and native fish vary with several biophysical factors, including watershed and habitat conditions, the condition of affected populations, and fire severity and extent [11]. If treatments reduce the watershed impacts of severe fire, they may provide benefits that outweigh treatment impacts because high-severity fire can sometimes trigger short-term, severe erosion and runoff [12] that can negatively affect soils, water quality, and aquatic populations. However, fuel treatments can also have impacts on aquatic systems. The magnitude and persistence of these treatment impacts vary with treatment methods, location, extent and frequency.

Although some fuel-treatment methods could have lower impacts, ground-based mechanical treatments are often employed because other methods generate activity fuels [7] and are more costly. Ground-based methods and associated machine piling, burning of activity fuels, construction and increased use of roads and landings can increase soil erosion, compact soils, and elevate surface runoff [8, 13, 14]. Although the effects of prescribed fire on watersheds are typically limited and fleeting, it can increase soil erosion and sediment delivery, sometimes significantly and persistently [15], especially if fires escape and burn larger and more severely than planned.

When impacts are extensive, proximate to streams, or in terrain with erosion hazards, treatments can increase runoff and sediment delivery to streams. Road activities that increase sediment production, such as elevated road traffic, often affect stream crossings where sediment delivery is typically efficient and difficult to control [16]. Elevated sediment delivery to streams contributes to water quality degradation that impairs aquatic ecosystems [17].

The extent and frequency of treatments may be significant. Stephens and Ruth [18] suggested treating fuels on 9.4 million ha, or ~53% of USFS lands in the Pacific Northwest and California. Agee and Skinner [7] suggested repeating treatments every 10-20 years, due to transient effects on fuels.

Repeated treatments increase the potential for cumulative effects on aquatic ecosystems due to the persistence and additive nature of watershed impacts over time [19] and may increase the establishment of non-native plants [9]. The chronic watershed impacts from repeated treatments may be more deleterious to native fish than pulsed disturbances from wildfires [8].

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Additional degradation of aquatic habitats on public lands may hamper efforts to protect and restore aquatic biodiversity. These habitats are increasingly important as cornerstones for restoring aquatic ecosystems and native fish [14].

Where fuel treatments might incur soil and watershed impacts, the risks from treatment and non-treatment should be assessed [7]. Although the respective impacts of treatments and fire are influenced by numerous factors, the occurrence of fire strongly affects the net balance between costs and benefits. If fire does not affect treated areas while fuels are reduced, treatment impacts on watersheds are not counterbalanced by benefits from reduction in fire impacts.

We provide a framework for quantitatively bounding the potential effectiveness of fuel treatments and the likelihood of fire affecting untreated watersheds, based on the probability of fire and the duration of treatment effects on fuels. This can be used to help statistically estimate the expected value associated with treatments or non-treatment based on the probability of possible outcomes and their associated costs and benefits [20]. Previous assessments of watershed tradeoffs from treatment and non-treatment [21, 22] did not include these in quantifying risk to aquatic systems associated with treatment versus non-treatment of fuels.

We use geographically-explicit data on fire on public lands in the western US to estimate, at a broad-scale, the probability that fuel treatments will be affected by fire during the period when fuels have been reduced. We also estimate the risk of higher severity fire occurring in watersheds if fuel treatments are foregone. These estimates provide a broad-scale bounding of treatment effectiveness and potential return from the fiscal and environmental costs of fuel treatments.

METHODS

The Analytical Model

Our analysis is based on the simple conceptual framework that unless fire occurs while fuels are reduced, fuel treatments cannot affect fire severity. We examine the probability of discrete classes of fire severity because fire impacts on watersheds vary with severity [11]. For instance, lower-severity fire has minimal, transient watershed impacts [11].

Future fire occurrence in specific locations cannot be predicted with certainty, but its probability can be estimated from empirical data. The probability of fire of a particular severity affecting treated areas can be estimated using the standard formula for the probability of an event occurring during a specific time frame:

$$q = 1 - (1 - p)^n \tag{1}$$

where q is the probability that a fire that would be of a specific severity in the absence of treatment occurs within n years, p is the annual probability of fire of a specific severity at the treatment location, and n is the duration, in years, that treatments decrease fuels and can reduce fire severity. In Equation 1, q provides an estimate of the mean fraction of an analysis area likely to burn at a specific severity within a given time frame in the absence of fuel treatments, which also represents the upper bound of potential effectiveness of treatments in reducing fire, since treatments cannot lower fire severity unless a fire occurs.

Both n and p can be estimated from available data. The duration of post-treatment fuel reduction, n, likely varies regionally with factors affecting vegetation re-growth rates, but fuels in western U.S. forests generally return to pre-treatment levels in 10-20 years [3, 7]. To estimate the upper limit of treatment effectiveness, we assume $n = 20$ years. We estimated the annual probability of fire of various severities, p, for each analysis area based on standard methods [23]:

$$p = (F*r)/(A*D) \tag{2}$$

where p is the annual probability of fire of a specific severity, F is total area burned at any severity within the analysis area over the duration of the data record, r is the estimated fraction of F that burned at the specified severity over the analysis area, A is the total analysis area, and D is the total duration of the data record, in years.

We based our estimates of the annual probability of fire on post-1960 fire records rather than reported natural fire return intervals for two primary reasons. First, evidence indicates that natural fire regimes no longer operate in many forests, because of direct fire suppression and indirect changes in fuels from livestock grazing, logging and fire exclusion [24]. Annual burned area has also increased in some forest types, likely due to climatic warming [25]. Recent fire data ostensibly integrate these alterations, reflecting how fires are likely to burn in the near future under current conditions and management. Natural fire return intervals do not capture these alterations. Second, there is considerable uncertainty regarding the accuracy of reported natural fire intervals [23, 24]. However, we stress that our approach can easily accommodate alternate estimates of annual fire probability using more geographically-refined data or where management changes might alter future fire probability.

We confined analysis to USFS lands in 11 western states, the focus for most proposed fuel treatments on public lands. The probability of fire varies geographically with several factors, including weather, ignition, fuels, and forest types. To bracket this effect, we estimated the annual probability of high-severity fire, p, for (*i*) all landcover types and (*ii*) more frequently burning ponderosa pine (*Pinus ponderosa*) forests at the scale of U.S. Forest Service (USFS) administrative regions that are the finest scale at which extensive data allow estimation of fire severity. We focus on high-severity fire, but also analyze fires of broader severity, including (1) either high- or moderate severity and (2) any severity.

Our estimates represent an initial, broad-scale first approximation of the potential of fire to affect areas within a given time frame, based on the assumption that fire and treatments are random. Although fire is not random, data are insufficient to accurately quantify more local patterns. Our approach provides a valid mean result at our scale of analysis, based on data from more than 40,000 fires across the western U.S. Site-specific data could be used in future, local studies where the probability of fire is known to depart considerably from the regional mean. Ideally, fuel treatments may not be randomly located, but instead focused in areas where fire is most likely. However, this is not assured by current policy [26]. Widely used methods for assessing the risk of high-severity fire may have limited accuracy [27].

Therefore, our analysis assumes random treatment location, as a first approximation.

West-Wide Analysis

To provide a broad-scale perspective of potential fuel treatment efficacy, we estimated mean annual probability, p, of fire for all USFS lands in the 11 western U.S. states, excluding Alaska, for the entire duration that data on total annual fire area are available (1960-2006). Data on fire area from 1993-2003, reported by agency ownership [28], were used to estimate mean annual fraction of total fire area on USFS lands, which was extrapolated to estimate mean annual fire area on USFS lands from 1960-1993 and 2004-2006, for which fire area data were reported [29], but not by agency ownership. Annual fire area on USFS lands in the 11 western states was assumed proportional to the fraction of total USFS area in these states. Total number of fires on western USFS lands from 1960-2006 is not reported, but based on the foregoing areal partitioning, the fire area data are from several hundred thousand fires on western USFS lands. The estimated annual fire area on these western USFS lands from 1960-2006 was summed to yield F in Equation 2.

The fraction of total fire area, r, that burned at high severity and high-moderate severity was estimated from data in USFS burned area emergency rehabilitation reports (BAER) for 470 fires in the 11 western states from 1973-1998 in six western USFS regions [30].

Regional Analysis of Fire in Ponderosa Pine

Because ponderosa pine forests are a key forest with more frequent fire, we estimated the mean annual probability of fire by severity in these forests on USFS lands: 1) on a regional basis, in six western USFS regions; and 2) West-wide. We used geographical information system (GIS) data for 40,389 fires in these forests for the entire period of data availability, 1980-2003 (Fig. **1**). Data were in a GIS point dataset, containing burned area for each fire, maintained by the Bureau of Land Management [31] and derived from a systematic National database [32]. We quality controlled these data for our study area, removing a few duplicate records.

A GIS map of ponderosa pine forests was obtained by selecting codes 5-7 (ponderosa pine) in the Westgap map from the GAP program, which includes national vegetation mapping from satellite imagery [33]. A GIS map of U.S. Forest Service regions is from the agency [34]. We converted all maps to Albers projection, Clarke 1866 datum, then used these to extract all fire records (n = 40,389) for ponderosa pine forests on USFS land in the 11 western states. We used USFS maps to subset fires by region, and then: (*i*) areas of individual fires were summed to yield F in Equation 2; (*ii*) the GIS was used to obtain A, and (*iii*) fire severity data by USFS region from 1973-1998 [30] were used to estimate r by severity.

RESULTS AND DISCUSSION

West-Wide Analysis

For the period 1960-2006, an estimated mean of ~220,000 ha, or a decimal fraction of 0.0037 of USFS western lands burned annually at any severity. Despite the approximations involved, our estimate of the mean annual fraction of areas burning at any severity compares reasonably with independent estimates by falling between them. Fire of any severity annually burned a mean fraction of ~0.0014 of the Deschutes National Forest in Oregon, from 1910-2001 [35], and ~0.0046 of 11 national forests in the Sierra Nevada, California, based on data from 1970-2003 [36].

Together with fire severity data [30], our West-wide estimate yields an estimated mean annual probability, p, of 0.001 and 0.002 for high- and high-moderate severity fire, respectively (Table **1**). Based on these estimates of p, Equation 1 yields a probability, q, of 0.020 and 0.042, respectively, for high- and high-moderate-severity fire. Substituting space for time, our results indicate that, on average, approximately 2.0 to 4.2% of areas treated to reduce fuels are likely to encounter fires that would otherwise be high or high-moderate severity without treatment. In the remaining 95.8-98.0% of treated areas, potentially adverse treatment effects on watersheds are not counterbalanced by benefits from reduced fire severity. These results also provide an estimate of the likelihood of high-severity fire affecting forests, if fuels are untreated. On average, over a 20-year period, about 2.0-4.2% of untreated areas would be expected to burn at high or high-moderate severity, respectively.

Using Equation 1, our results indicate that if treatments were repeated every 20 years across all USFS lands in the West, it would take about 720 years (36 cycles of treatments), on average, before it is expected that high-severity fire affects slightly more than 50% of treated areas while fuels are reduced. Treatments would have to be repeated at 20-year intervals for 340 years (17 cycles of treatments) before high-moderate severity fire is expected to encounter more than 50% of treated areas. Even after this duration of repeated treatments, it is likely that almost 50% of treated areas will be cumulatively affected by repeated treatments without compensatory benefits from reduced fire severity.

These West-wide estimates provide perspective, but include forest types, such as subalpine forests, typified by low-frequency, high-severity fire, where fuel treatments are unlikely to encounter fire [4]. Other forests, such as ponderosa pine, burn more often.

Regional Analysis of Ponderosa Pine

For ponderosa pine forests, the probability, q, of treated areas being affected within their window of effectiveness varies regionally from 0.020 to 0.040 for high-severity fires and from 0.042 to 0.079 for high-moderate severity fires (Table **1**). As expected, q in these forests is higher than for the West-wide analysis of all cover types. The highest probabilities, as expected, are in the Southwest and in the Northern Rockies, with its dry summers (Table **1**).

In these forests with more frequent fire, it is likely that fuel treatments can potentially reduce fire severity on a small fraction of treated areas. The results (Table **1**) indicate that in 92.1-98.0% of treated areas, fuel treatment impacts on watershed processes are not likely to be counterbalanced by a reduction in higher-severity fire.

Across the six regions, treatments would have to be repeated every 20 years for 340 to 700 years (17 to 35 times), on average, before it is expected that high-severity fire affects more than 50% of treated areas during periods of treat-

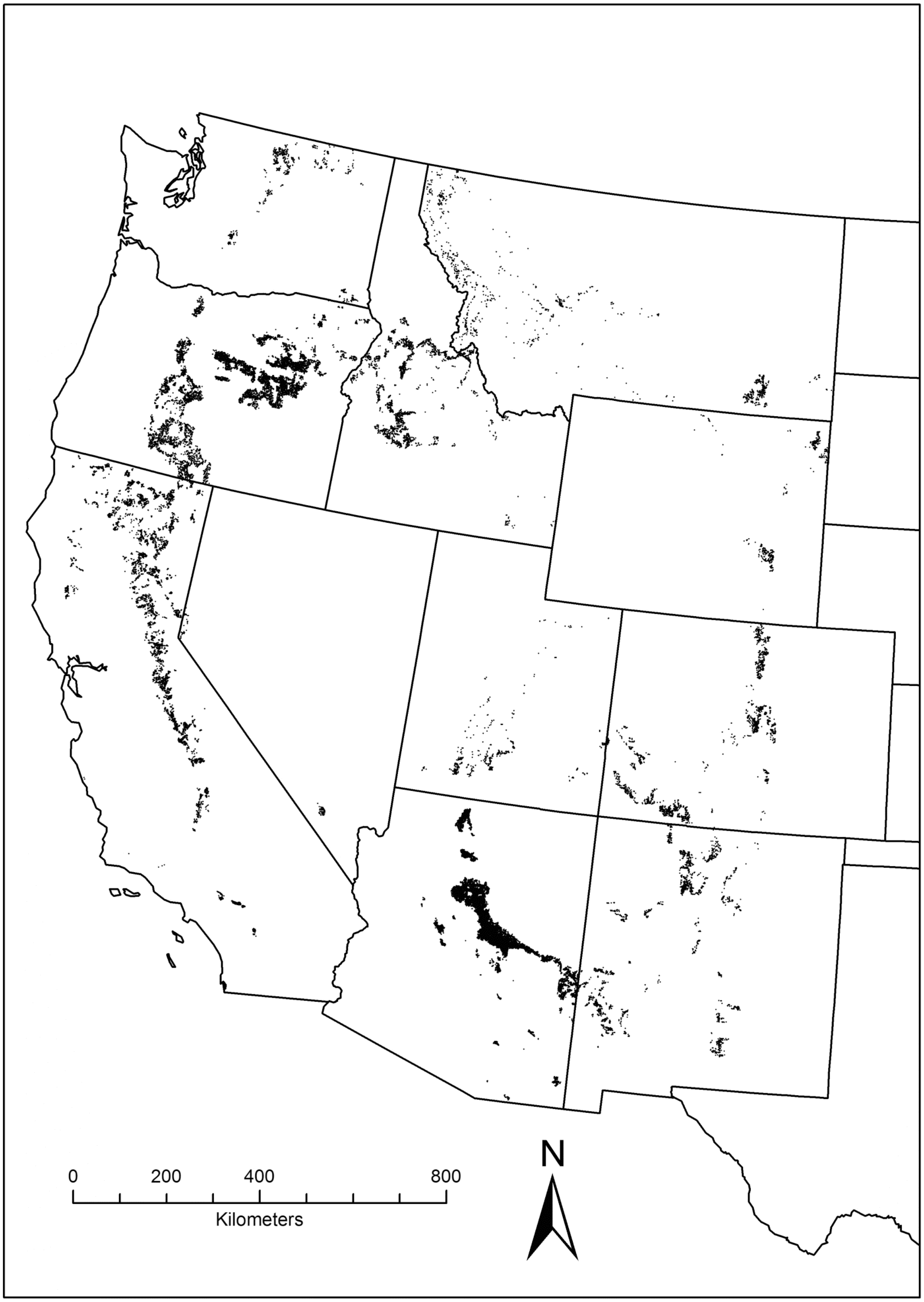

Fig. (1). Ponderosa pine forest fires (n = 40,389) in the western United States from 1980-2003. This is the dataset used in the regional analysis.

ment effectiveness. Treatments would have to be repeated for 180 to 340 years (9 to 17 times) before more than 50% of treated areas are expected to be affected by high-moderate severity fire. On average, these repeated treatments would affect watersheds and, potentially aquatic systems, depending on treatment practices, without providing reduction in fire severity on almost 50% of treated area.

An alternative method for estimating the risk of fire in the absence of fuel treatments is to use the fire rotation rather than mean annual probability of fire. The fire rotation indicates how long it takes, on average, for a particular area to burn one time and how often fire may return to a particular point in the landscape [23]. The fire rotation is calculated by:

$$B = 1/p \tag{3}$$

Table 1. Estimated *p* and *q* for Fires in Ponderosa Pine (PIPO) Forests. Data are Shown for Three Fire Severity Classes by USFS Region, and for All Forests on USFS Lands West-Wide

USFS Region	Any Severity		High-Moderate Severity		High Severity	
	p	q	p	q	p	q
1 N. Rockies	0.0070	0.1311	0.0036	0.0693	0.0020	0.0402
2 C&S Rockies	0.0059	0.1116	0.0041	0.0786	0.0014	0.0269
3 SW	0.0053	0.1008	0.0025	0.0487	0.0016	0.0307
4 Gt. Basin	0.0090	0.1654	0.0037	0.0715	0.0013	0.0257
5 Calif.	0.0046	0.0881	0.0031	0.0603	0.0017	0.0338
6 NW	0.0037	0.0715	0.0022	0.0421	0.0010	0.0198
West-wide: PIPO	0.0054	0.1026	0.0031	0.0602	0.0015	0.0295
West-wide: All types	0.0037	0.0715	0.0021	0.0416	0.0010	0.0203

where B is the fire rotation for fire of a specific severity and p is, again, the mean annual probability of fire of a specific severity.

Based on our analysis, the mean annual probability, p, of high-severity fire in ponderosa pine forests West-wide is 0.0015 (Table **1**), implying a fire rotation, B of about 667 years, varying from 500 to 1,000 years among individual regions. Based on the results in Table **1**, the fire rotation for high-moderate severity fire is about 323 years in ponderosa pine forests West-wide, varying from 244 to 454 years in individual regions, based on data in Table **1**. These results suggest that western ponderosa pine forests are not currently being rapidly burned by high or high-moderate severity fire, counter to other previous work [37].

Relaxing the Assumptions and Some Caveats

In some cases, the occurrence of fire of any severity may be of interest. Such cases include areas where fire of any severity might lead to high-severity fire. In ponderosa pine forests, the probability of fire of any severity encountering treatments within 20 years is approximately 7.15-16.5% across the six regions (Table **1**). Thus, if it is assumed that fuel treatments that encounter fire of any severity might be effective, the results indicate fuel treatments, on average, would not have the potential to reduce fire impacts on aquatic systems in 83.5-92.8% of the area treated. Based on Equation 1 and Table **1**, treatments would have be repeated every 20 years for 80-200 years, on average, before fire of any severity affects more than 50% of the treated areas in ponderosa forests in these USFS administrative regions.

However, the assumption that treatments that encounter low-severity fire convey benefits may not be warranted. Low-severity fires are commonly and easily extinguished under current management whether or not they encounter fuel treatments. Further, low-severity fire has minimal adverse impacts on watershed processes while conveying benefits, including maintenance of forest structure and fuel levels.

Our probabilistic approach does not explicitly address factors that can strongly influence fire area and severity, such as fuel conditions. Although spatially-explicit modeling of fire behavior can directly investigate the effects of such conditions, such models are unlikely to provide accurate estimates of the probability of occurrence of fire of a given severity because a host of other factors that influence fire area and severity cannot be deterministically predicted, including the frequency and location of ignitions and weather conditions during fire. Methods of assessing the risk of high-severity fire that are primarily based on fuel conditions have been shown to be an ineffective predictor of the actual severity at which fires burn [38]. In contrast, extensive recent data from numerous fires, as used in our analysis, does provide a robust estimate of the mean probability of the occurrence of fire of a given severity, because it integrates the many factors that influence fire occurrence and severity.

Our estimates likely represent the upper bound for fuel treatment effectiveness at the scale of analysis. In many cases, less than 4.16-7.86% of treated area is likely to experience high-moderate severity fire during the duration of treatment effectiveness, because q decreases with decreases in n, the duration of treatment effectiveness. This duration is often less than the 20 years assumed in our analysis. In the Sierra Nevada of California, fuels returned to pre-treatment levels within 11 years [39]. At the values of p in Table **1**, reducing n from 20 to 11 years (Eq. 1) reduces the probability that higher-severity fire affects treatments by ~45%.

Moreover, fuel levels rebound after treatment, eventually negating potential treatment effectiveness. If the reduction in effectiveness over time is such that mean effectiveness over the duration, n, is half the initial degree of effectiveness, the probability that fuel treatments reduce high-severity fire is approximately half the value of q for any value of p and n calculated using Equation 1.

Finally, available data indicate that fuel treatments do not always reduce fire severity when fire affects treated areas while fuels are reduced [4]. Our analysis does not address these effectiveness issues. For these combined reasons, Equation 1 likely estimates the upper bound of potential fuel treatment effectiveness in reducing fire impacts on aquatic systems.

Although our analysis focuses on higher-severity fire in bounding the effectiveness of fuel treatments and their net watershed effects, these fires do not have solely negative effects. Higher-severity fire benefits watersheds and aquatic ecosystems in several ways, including providing a bonanza of recruitment of large wood and pulsed sediment supply that can rejuvenate aquatic habitats and increase their productivity [8, 14]. High severity fire is also a key process for the restoration of structural heterogeneity in forests, which is important for biodiversity [27, 40].

Our analysis intrinsically assumes some degree of climatic stationarity, which may not be warranted. Climatic variability influences the area annually burned in forests [25, 41]. However, the relatively recent fire data used in our regional analysis incorporates recent climatic fluctuation and possibly directional change, which would not be reflected in estimates based on natural fire return intervals. For instance, the data in our analysis of ponderosa pine forests come primarily from years in which annual fire area had increased due to climatic warming [25]. However, the analysis framework is flexible enough to accommodate projected values of the mean annual probability of fire, p, based on forecasts of climatic change or changes in fire management.

Current findings suggest treatment effects on fire severity are mostly confined to treated areas [3], but theory suggests a dense network of treatments might slow fire spread and reduce intensity, yielding a landscape-scale effect on fire severity [42]. However, empirical evidence of severity reduction was seen in the lee of only three of several dozen treatments in two Arizona wildfires [43]. Nonetheless, if dense treatment networks are shown to work in the future, our approach can aid in estimating their costs and benefits, because fire must still affect treated areas while fuels are reduced for networks to reduce fire severity.

CONCLUSIONS

Our analysis provides West-wide and regional first approximation of the likely upper bound of fuel treatment effectiveness. While valid at these two scales, they are not applicable to all smaller analysis areas, due to spatial variation in annual fire probability. However, the framework is flexible enough to allow more spatially explicit analyses of q where local estimates of n and p are available. The framework allows analysis of uncertainty, by using a range of plausible values for n and p. The analysis can also estimate the number of treatments to reach a specified q, abetting estimation of cumulative effects on ecosystems from repeated treatments.

Our approach also provides a method for quantitatively assessing the imminence of high-severity fire effects in the absence of fuel treatments and the degree of urgency of response. Based on available data, these are shown to be much lower than previously estimated in some work [37].

Our results and analyses can improve the assessment of risks to watersheds inherent in the treatment or non-treatment of forest fuels, because it accounts for the probability of fire and the transient nature of fuel treatments. For instance, previous work [22], evaluating treatment and non-treatment impacts, assessed the risks associated with fuel treatments based on the assumption that a single treatment significantly reduces fire risk on all treated areas, subsequently reducing consequent watershed impacts from fire. Other evaluations of these tradeoffs [21] compared the erosional effects of fuel treatments with high-severity fire under the explicit assumption that high-severity fire was inevitable without treatment and the implicit assumption that treatments always reduce or eliminate the potential for high-severity fire. Our analysis indicates that these assumptions are unwarranted and likely mischaracterize the outcomes and associated impacts of treatment options.

The approach can be extended to aid in assessing the risk to other ecosystem elements and processes that may be adversely affected by either fuel treatments or high-severity fire. For instance, non-native vegetation can be influenced by high fire severity [44] and some fuel treatments [10], especially if the treatments are repeated [9].

Even in ponderosa pine forests that burn relatively frequently, our regional analysis indicates that after 17 cycles of treatments, only slightly more than 50% of treated areas could potentially have fire severity reduced, on average. Our results indicate that high-severity fire is far from inevitable in areas left untreated and is, instead, expected to affect only a relatively small fraction of such areas at the broad scale of our analysis. Factoring in the probability of fire, using our framework, can significantly improve the assessments of the risks posed to aquatic systems by treating or not treating forest fuels. Where site-specific data on fire probabilities exist, the framework can be used to help locate treatments where they are most likely to encounter higher severity fire, increasing the likelihood of treatment benefits. In fact, our results indicate that such efforts are crucial.

There are several important factors that influence the aquatic tradeoffs among fuel treatments, fire, and aquatic systems that our framework does not address. Although the probability of outcomes is critical to assessing the expected value of options, the ecological costs of the outcomes of treatment *vs* non-treatment are also important in assessing the expected value of these options. With respect to the aquatic context, there is an ongoing need to fully evaluate tradeoffs such as the severity and persistence of the negative and positive impacts on watersheds and aquatic populations from fuel treatments and higher severity fire [8, 45]. An additional related issue is how effective treatments are when they encounter fire under a broad array of conditions affecting fire behavior [3]. While our analysis does not address these factors, it refines evaluation of net impacts of fuel treatment *vs* non-treatment by providing a framework for estimating the likelihood of fire occurrence in a given time frame.

At the scales of our analysis, results indicate that even if fuel treatments were very effective when encountering fire of any severity, treatments will rarely encounter fire, and thus are unlikely to substantially reduce effects of high-severity fire.

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The role of defensible space for residential structure protection during wildfires

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Abstract. With the potential for worsening fire conditions, discussion is escalating over how to best reduce effects on urban communities. A widely supported strategy is the creation of defensible space immediately surrounding homes and other structures. Although state and local governments publish specific guidelines and requirements, there is little empirical evidence to suggest how much vegetation modification is needed to provide significant benefits. We analysed the role of defensible space by mapping and measuring a suite of variables on modern pre-fire aerial photography for 1000 destroyed and 1000 surviving structures for all fires where homes burned from 2001 to 2010 in San Diego County, CA, USA. Structures were more likely to survive a fire with defensible space immediately adjacent to them. The most effective treatment distance varied between 5 and 20 m (16–58 ft) from the structure, but distances larger than 30 m (100 ft) did not provide additional protection, even for structures located on steep slopes. The most effective actions were reducing woody cover up to 40% immediately adjacent to structures and ensuring that vegetation does not overhang or touch the structure. Multiple-regression models showed landscape-scale factors, including low housing density and distances to major roads, were more important in explaining structure destruction. The best long-term solution will involve a suite of prevention measures that include defensible space as well as building design approach, community education and proactive land use planning that limits exposure to fire.

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Introduction

Across the globe and over recent decades, homes have been destroyed in wildfires at an unprecedented rate. In the last decade, large wildfires across Australia, southern Europe, Russia, the US and Canada have resulted in tens of thousands of properties destroyed, in addition to lost lives and enormous social, economic and ecological effects (Filmon 2004; Boschetti *et al.* 2008; Keeley *et al.* 2009; Blanchi *et al.* 2010; Vasquez 2011). The potential for climate change to worsen fire conditions (Hessl 2011), and the projection of continued housing growth in fire-prone wildlands (Gude *et al.* 2008) suggest that many more communities will face the threat of catastrophic wildfire in the future.

Concern over increasing fire threat has escalated discussion over how to best prepare for wildfires and reduce their effects. Although ideas such as greater focus on fire hazard in land use planning, using fire-resistant building materials and reducing human-caused ignitions (e.g. Cary *et al.* 2009; Quarles *et al.* 2010; Syphard *et al.* 2012) are gaining traction, the traditional strategy of fuels management continues to receive the most attention. Fuels management in the form of prescribed fires or mechanical treatments has historically occurred in remote, wildland locations (Schoennagel *et al.* 2009), but recent studies suggest that treatments located closer to homes and communities may provide greater protection (Witter and Taylor 2005; Stockmann *et al.* 2010; Gibbons *et al.* 2012). In fact, one of the most commonly recommended strategies in terms of fuels and fire protection is to create defensible space immediately around structures (Cohen 2000; Winter *et al.* 2009). Defensible space is an area around a structure where vegetation has been modified, or 'cleared,' to increase the chance of the structure surviving a wildfire. The idea is to mitigate home loss by minimising direct contact with fire, reducing radiative heating, lowering the probability of ignitions from embers and providing a safer place for fire fighters to defend a structure against fire (Gill and Stephens 2009; Cheney *et al.* 2001). Many jurisdictions provide specific guidelines and practices for creating defensible space, including minimum distances that are required among trees and shrubs as well as minimum total distances from the structure. These distances may be enforced through local ordinances or state-wide laws. In California, for example, a state law in 2005 increased the required total distance from 9 m (30 ft) to 30 m (100 ft).

Despite these specific guidelines on how to create defensible space, there is little scientific evidence to support the amount and location of vegetation modification that is actually effective at providing significant benefits. Most spacing guidelines and laws are based on 'expert opinion' or recommendations from older publications that lack scientific reference or rationale (e.g. Maire 1979; Smith and Adams 1991; Gilmer 1994). However, one study has provided scientific support for, and forms the basis of, most guidelines, policy and laws requiring a minimum of 30 m (100 ft) of defensible space (Cohen 1999, 2000). The modelling and experimental research in that study showed that flames from forest fires located 10–40 m (33–131 ft) away would not scorch or ignite a wooden home; and case studies showed 90% of homes with non-flammable roofs and vegetation clearance of 10–20 m (33–66 ft) could survive wildfires (Cohen 2000). However, the models and experimental research in that study focussed on crown fires in spruce or jack pine forests, and the primary material of home construction was wood. Therefore, it is unknown how well this guideline applies to regions dominated by other forest types, grasslands, or nonforested woody shrublands and in regions where wooden houses are not the norm.

Some older case studies showed that most homes with non-flammable roofs and 10–18 m (33–ft) of defensible space survived the 1961 Bel Air fire in California (Howard *et al.* 1973); most homes with non-flammable roofs and more than 10 m (33 ft) of defensible space also survived the 1990 Painted Cave fire (Foote and Gilless 1996). Also, several fire-behaviour modelling studies have been conducted in chaparral shrublands. One study showed that reducing vegetative cover to 50% at 9–30 m (30–ft) from structures effectively reduced fireline intensity and flame lengths, and that removal of 80% cover would result in unintended consequences such as exotic grass invasion, loss of habitat and increase in highly flammable flashy fuels (A. Fege and D. Pumphrey, unpubl. data). Another showed that separation distances adequate to protect firefighters varied according to fuel model and that wind speeds greater than 23 km h^{-1} negated the effect of slope, and wind speed above 48 km h^{-1} negated any protective effect of defensible space (F. Bilz, E. McCormick and R. Unkovich, unpubl. data, 2009). Results obtained through modelling equations of thermal radiation also found safety distances to vary as a function of fuel type, type of fire, home construction material and protective garments worn by firefighters (Zárate *et al.* 2008).

Although there is no empirical evidence to support the need for more than 30 m (100 ft) of defensible space, there has been a concerted effort in some areas to increase this distance, particularly on steep slopes. In California, a senate bill was introduced in 2008 (SB 1618) to encourage property owners to clear 91 m (300 ft) through the reduction of environmental regulations and permitting needed at that distance. Although this bill was defeated in committee, many local ordinances do require homeowners to clear 91 m (300 ft) or more, and there are reports that some people are unable to get fire insurance without 91 m (300 ft) of defensible space (F. Sproul, pers. comm.). In contrast, homeowner acceptance of and compliance with defensible space policies can be challenging (Winter *et al.* 2009; Absher and Vaske 2011), and in many cases homeowners do not create any defensible space.

It is critically important to develop empirical research that quantifies the amount, location and distance of defensible space that provides significant fire protection benefits so that guidelines and policies are developed with scientific support. Data that are directly applicable to southern California are especially important, as this region experiences the highest annual rate of wildfire-destroyed homes in the US. Not having sufficient defensible space is obviously undesirable because of the hazard to homeowners. However, there are clear trade-offs involved when vegetation reduction is excessive, as it results in the loss of native habitats, potential for increased erosion and invasive species establishment, and it potentially even increases fire risk because of the high flammability of weedy grasslands (Spittler 1995; Keeley *et al.* 2005; Syphard *et al.* 2006).

It is also important to understand the role of defensible space in residential structure protection relative to other factors that explain why some homes are destroyed in fires and some are not. Recent research shows that landscape-scale factors, such as housing arrangement and location, as well as biophysical variables characterising properties and neighbourhoods such as slope and fuel type, were important in explaining which homes burned in two southern California study areas (Syphard *et al.* 2012; 2013). Understanding the relative importance of different variables at different scales may help to identify which combinations of factors are most critical to consider for fire safety.

Our objective was to provide an empirical analysis of the role of defensible space in protecting structures during wildfires in southern California shrublands. Using recent pre-fire aerial photography, we mapped and measured a suite of variables describing defensible space for burned and unburned structures within the perimeters of major fires from 2001 to 2010 in San Diego County to ask the following questions:

1. How much defensible space is needed to provide significant protection to homes during wildfires, and is it beneficial to have more than the legally required 30 m (100 ft)?
2. Does the amount of defensible space needed for protection depend on slope inclination?
3. What is the role of defensible space relative to other factors that influence structure loss, such as terrain, fuel type and housing density?

Methods

Study area

The properties and structures analysed were located in San Diego County, California, USA (Fig. 1) – a topographically diverse region with a Mediterranean climate characterised by cool, wet winters and long summer droughts. Fire typically is a direct threat to structures adjacent to wildland areas. Native shrublands in southern California are extremely flammable during the late summer and fall (autumn) and when ignited, burn in high-intensity, stand-replacing crown fires. Although 500 homes on average have been lost annually since the mid-1900s (Calfire 2000), that rate has doubled since 2000. Most of these homes have burned during extreme fire weather conditions that accompany the autumn Santa Ana winds. The wildland–urban interface here includes more than 5 million homes, covering more than 28 000 km^2 (Hammer *et al.* 2007).

Property data

The data for properties to analyse came from a complete spatial database of existing residential structures and their

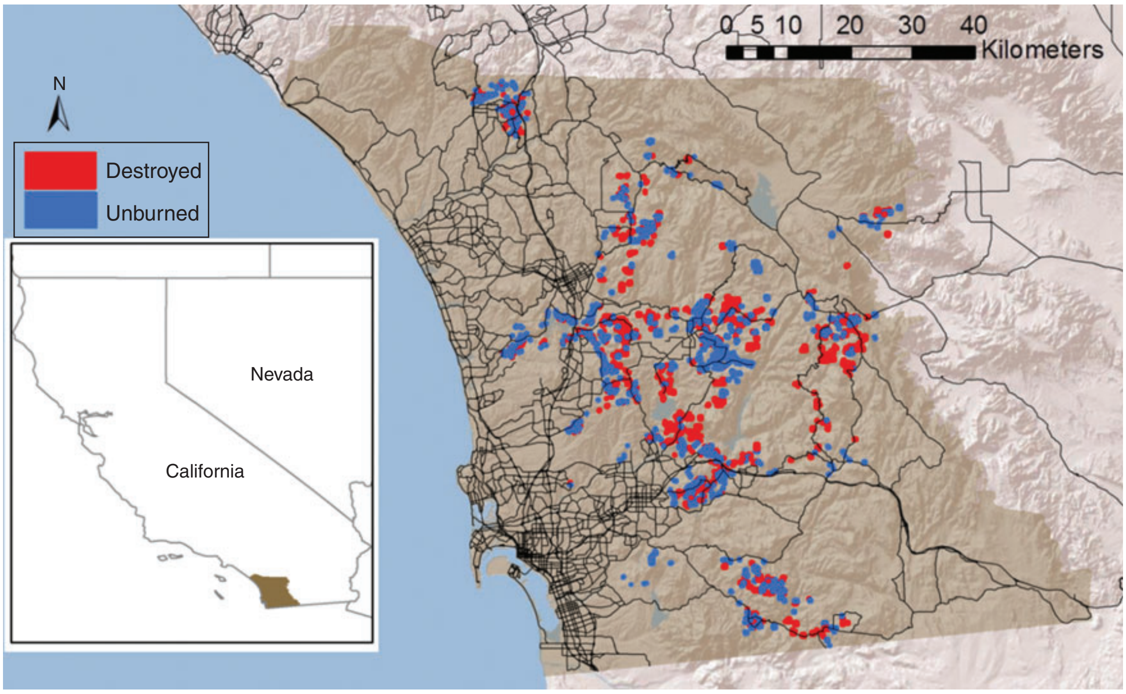

Fig. 1. Location of destroyed and unburned structures within the South Coast ecoregion of San Diego County, California, USA.

corresponding property boundaries developed for San Diego County (Syphard *et al.* 2012). This dataset included 687 869 structures, of which 4315 were completely destroyed by one of 40 major fires that occurred from 2001 to 2010. Our goal was to compare homes that were exposed to wildfire and survived with those that were exposed and destroyed. To determine exposure to fire, we only considered structures located both within a GIS layer of fire perimeters and within areas mapped as having burned at a minimum of low severity through thematic Monitoring Trends in Burn Severity produced by the USA Geological Survey and USDA Forest Service. From these data, we used a random sample algorithm in GIS software to select 1000 destroyed and 1000 unburned homes that were not adjacent to each other, to minimise any potential for spatial autocorrelation. Our final property dataset included structures that burned across eight different fires. More than 97% of these structures burned in Santa Ana wind-driven fire events (Fig. 1).

Calculating defensible space and additional explanatory variables

To estimate defensible space, we developed and explored a suite of variables relative to the distance and amount of defensible space surrounding structures, as well as the proximity of woody vegetation to the structure (Table 1). We measured these variables based on interpretation of Google Earth aerial imagery. We based our measurements on the most recent imagery before the date of the fire. In almost all cases, imagery was available for less than 1 year before the fire.

Our definition of defensible space followed the guidelines published by the California Department of Forestry and Fire Protection (Calfire 2006). 'Clearance' included all areas that were not covered by woody vegetation, including paved areas or grass. Although Google Earth prevents the identification of understorey vegetation, woody trees and shrubs were easily distinguished from grass, and our objective was to measure horizontal distances as required by Calfire rather than assess the relative flammability of different vegetation types. Trees or shrubs were allowed to be within the defensible space zone as long as they were separated by the minimum horizontal required distance, which was 3 m (10 ft) from the edge of one tree canopy to the edge of the next (Fig. 2). Although greater distances between trees or shrubs are recommended on steeper slopes, we followed the same guidelines for all properties. For all structures, we started the distance measurements by drawing lines from the centre of the four orthogonal sides of the structure that ended when they intersected anything that no longer met the requirements in the guidelines. A fair number of structures are not four sided; thus, the start of the centre point was placed at a location that approximated the farthest extent of the structure along each of four orthogonal sides.

We developed two sets of measurements of the distance of defensible space based on what is feasible for homeowners within their properties *v.* the total effective distance of defensible space. We made these two measurements because homeowners are only required to create defensible space within their own property, and this would reflect the effect of individual homeowner compliance. Therefore, even if cleared vegetation extended beyond the property line, the first set of distance measurements ended at the property boundary. The second set of measurements ignored the property boundaries and accounted for the total potential effect of treatment. For all measurements, we recorded the cover types (e.g. structure >3 m (10 ft) long, property boundary, or vegetation type) at which the distance measurements stopped (Table 1). Because property

Table 1. Defensible space variables measured for every structure

Urban veg, landscaping vegetation that was not in compliance with regulations within urban matrix; wildland veg, wildland vegetation that was not in compliance with regulations; orchard, shrub to tree-sized vegetation in rows; urban to wildland, landscaping vegetation that leads into wildland vegetation; structure, any building longer than 3 m (10 ft)

Variable	Definition
Distance defensible space within property	Measure of clearance from side of structure to property boundary calculated for four orthogonal directions from structure and averaged
Total distance defensible space	Measure of clearance from side of structure to end of clearance calculated for four orthogonal directions from structure and averaged
Cover type at end of defensible space	Type of cover encountered at end of measurement (urban veg, wildland veg, orchard, urban to wildland, structure)
Percentage clearance	Percentage of clearance calculated across the entire property
Neighbours' vegetation	Binary indicator of whether neighbours' uncleared vegetation was located within 30 m (100 ft) of the main structure
Vegetation touching structure	Number of sides on which woody vegetation touches main structure (1–4) Structure with more than 4 sides were viewed as a box and given a number between 1 and 4
Vegetation overhanging roof	Was vegetation overhanging the roof? (yes or no)

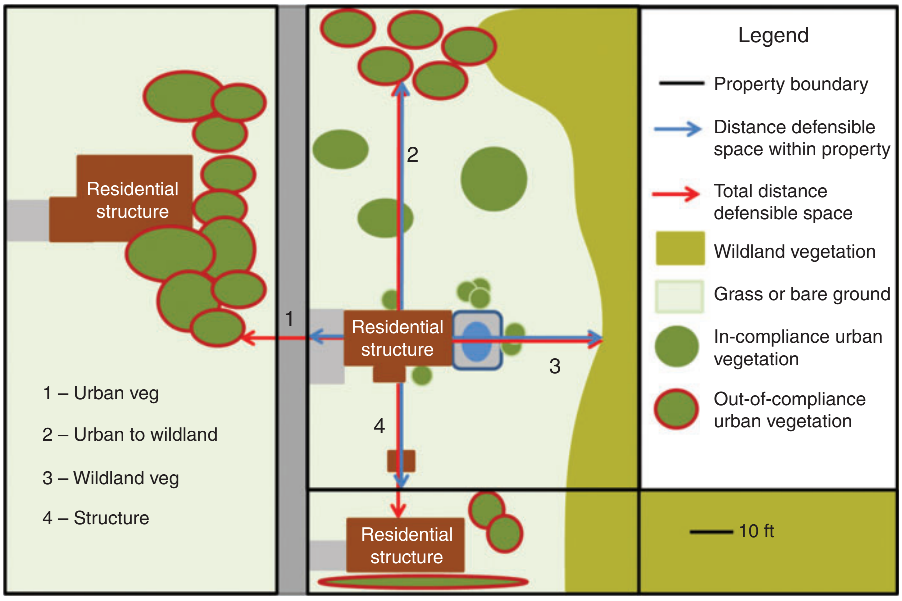

Fig. 2. Illustration of defensible space measurements. See Table 1 for full definition of terms.

owners usually can only clear vegetation on their own land, it is possible that the effectiveness of defensible space partly depends upon the actions of neighbouring homeowners. Therefore, we also recorded whether or not any neighbours' un-cleared vegetation was located within 30 m (100 ft) of the structure.

To assess the total amount of woody vegetation that can safely remain on a property and still receive significant benefits of defensible space, we calculated the total percentage of cleared land, woody vegetation and structure area across every property. This was accomplished by overlaying a grid on each property and determining the proportion of squares falling into each class. Preliminary results showed these three measurements to be highly correlated, so we only retained percentage clearance for further analysis. To evaluate the relative effect of woody vegetation directly adjacent to structures, we also calculated the number of sides of the structure with vegetation touching and recorded whether any trees were overhanging structures' roofs.

In addition to defensible space measurements, we evaluated other factors known to influence the likelihood of housing loss to fire in the region (Syphard et al. 2012, 2013). Using the same data as in Syphard et al. (2012, 2013), we extracted spatial information from continuous grids of explanatory variables for the locations of all structures in our analysis. Variables included interpolated housing density based on a 1-km search radius; percentage slope derived from a 30-m digital elevation model (DEM); Euclidean distance to nearest major and minor road and fuel type, which was based on a simple classification of US Forest Service data (Syphard et al. 2012), including urban, grass, shrubland and forest & woodland.

Analysis

We performed several analyses to determine whether relative differences in home protection are provided by different distances and amounts of defensible space, particularly beyond the legally required 30 m (100 ft), and to identify the effective treatment distance for homes on low and steep slopes.

Categorical analysis

For the first analysis, we divided our data into several groups to identify potential differences among specific categories of defensible space distance around structures located on shallow and steep slopes. We first sorted the full dataset of 2000 structures by slope and then split the data in the middle to create groups of homes with shallow slope and steep slope. We divided the data in half to keep the number of structures even within both groups and to avoid specifying an arbitrary number to define what constitutes shallow or steep slope. The two equal-sized subsets of data ranged from 0 to 9%, with a mean of 8% for shallow slope, and from 9 to 40%, with a mean of 27% for steep slope. Within these data subsets, we next created groups reflecting different mean distances of defensible space around structures. We also performed separate analyses based on whether defensible space measurements were calculated within the property boundary or whether measurements accounted for the total distance of defensible space.

Within all groups, we calculated the proportion of homes that were destroyed by wildfire. We performed Pearson's Chi-square tests of independence to determine whether or not the proportion of destroyed structures within groups was significantly different (Agresti 2007). We based one test on four equal-interval groups within the legally required distance of 30 m (100 ft): 0–7 m (0–25 ft), 8–15 m (26–50 ft), 16–23 m (51–75 ft) and 24–30 m (76–100 ft). A second test was based on three groups (24–30 m (75–100 ft), 31–90 m (101–300 ft) and >90 m (>300 ft) or >60 m (>200 ft)) to evaluate whether groups with mean defensible space distances >30 m (>100 ft) were significantly different from groups with <30 m (<100 ft). When defensible space distances were only measured to the property boundary, few structures had mean defensible space >90 m (>300 ft). Therefore, we used a cut-off of 60 m (200 ft) to increase the sample size in the Chi-square analysis. In addition to the Chi-square analysis, we calculated the relative risk among every successive pair of categories (Sheskin 2004). The relative risk was calculated as the ratio of proportions of burned homes within two groups of homes that had different defensible space distances.

Effective treatment analysis

In addition to comparing the relative effect of defensible space among different groups of mean distances, as described above, we also considered that the protective effect of defensible space for structures exposed to wildfire is conceptually similar to the effect of medication in producing a therapeutic response in people who are sick. In addition to pharmacological applications, treatment–response relationships have been used for radiation, herbicide, drought tolerance and ecotoxicological studies (e.g. Streibig *et al.* 1993; Cedergreen *et al.* 2005; Knezevic *et al.* 2007; Kursar *et al.* 2009). The effect produced by a drug or treatment typically varies according to the concentration or amount, often up to a point at which further increase provides no additional response. The effective treatment (ET50), therefore, is a specific concentration or exposure that produces a therapeutic response or desired effect. Here we considered the treatment to be the distance or amount of defensible space.

Using the software package DRC in R (Knezevic *et al.* 2007; Ritz and Streibig 2013), we evaluated the treatment–response relationship of defensible space in survival of structures during wildfire. To calculate the effective treatment, we fit a log-logistic model with logistic regression because we had a binary dependent variable (burned or unburned). We specified a 2-parameter model where the lower limit was fixed at 0 and the upper limit was fixed at 1. We again performed separate analyses for data subsets reflecting shallow and steep slope, as well as from measurements of defensible space taken within, or regardless of, property boundaries. We also performed analyses to find the effective treatment of percentage clearance of trees and shrubs within the property.

Multiple regression analysis

To evaluate the role of defensible space relative to other variables, we developed multiple generalised linear regression models (GLMs) (Venables and Ripley 1994). We again had a binary dependent variable (burned versus unburned), so we specified a logit link and binomial response. Although the proportion of 0s and 1s in the response may be important to consider for true prediction (King and Zeng 2001; Syphard *et al.* 2008), our objective here was solely to evaluate variable importance. We developed multiple regression models for all possible combinations of the predictor variables and used the corrected Akaike's Information Criterion (AICc) to rank models and select the best ones for each region using package MuMIn in R (R Development Core Team 2012; Burnham and Anderson 2002). We recorded all top-ranked models that had an AICc value within 2 of that of the model with lowest AICc to identify all models with empirical support. To assess variable importance, we calculated the sum of Akaike weights for all models that contained each variable. On a scale of 0–1, this metric represents the weight of evidence that models containing the variable in question are the best model (Burnham and Anderson 2002). The distance of defensible space measured within property boundaries was highly correlated with the distance of defensible space measured beyond property boundaries ($r = 0.82$), so we developed two separate analyses – one using variables measured only within the property boundary and the other using variables that accounted for defensible space outside of the property boundary as well as the potential effect of neighbours having uncleared vegetation within 30 m (100 ft) of the structure. A test to avoid multicollinearity showed all other variables within each multiple regression analysis to be uncorrelated ($r < 0.5$).

Surrounding matrix

To assess whether the proportion of destroyed structures varied according to their surrounding matrix, we summarised the most common cover type at the end of defensible space measurements (descriptions in Table 1) for all structures. These summaries

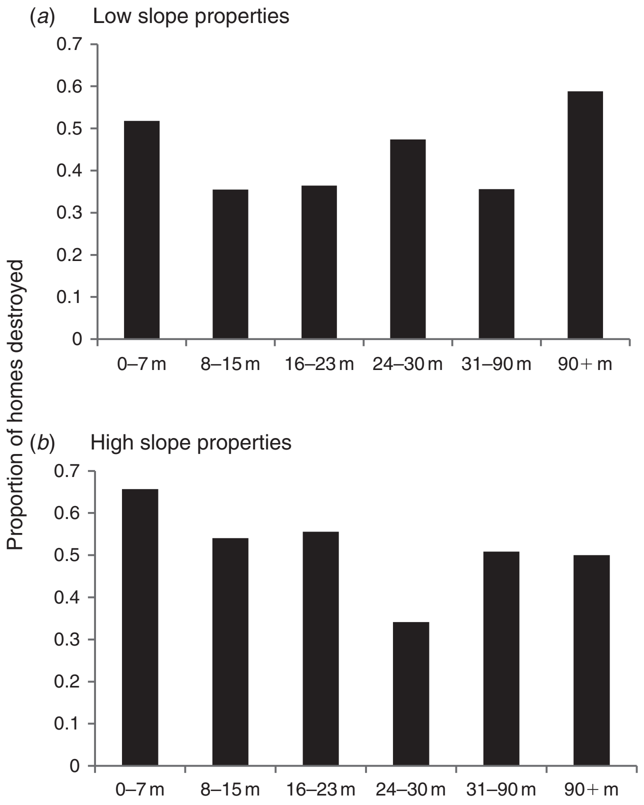

Fig. 3. Proportion of destroyed homes grouped by distances of defensible space based upon total distance of clearance within property boundary, for structures on (*a*) shallow slopes (mean 8%) and (*b*) steep slopes (mean 27%).

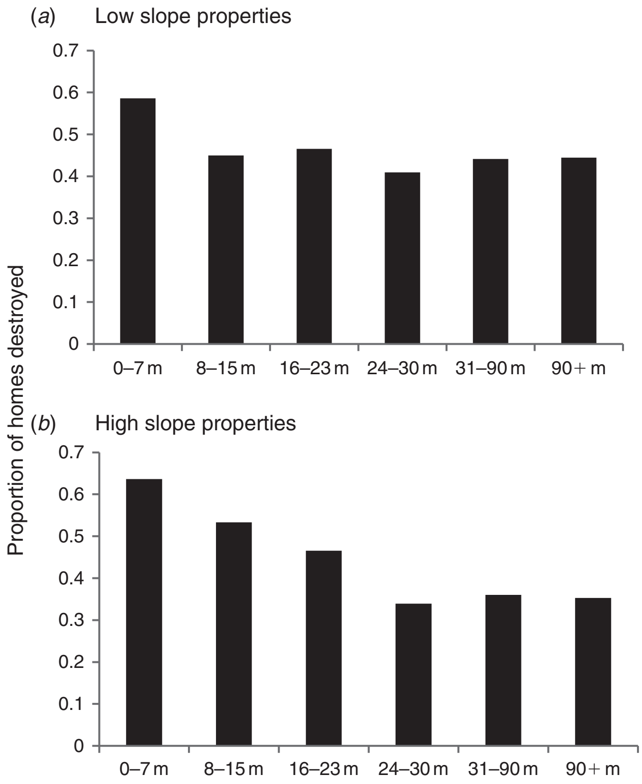

Fig. 4. Proportion of destroyed homes grouped by distances of defensible space based upon total distance of clearance regardless of property boundary, for structures on (*a*) shallow slopes (mean 8%) and (*b*) steep slopes (mean 27%).

were based on the majority surrounding cover type from the four orthogonal sides of the structure. We also noted cases in which there was a tie (e.g. two sides were urban vegetation and two sides were structures).

Results

Categorical analysis

When the distance of defensible space was measured both 'only within property boundaries' (Fig. 3) and 'regardless of property boundaries' (Fig. 4), the Chi-square test showed a significant difference ($P < 0.001$) in the proportion of destroyed structures among the four equal-interval groups of distance ranging from 0 to 30 m (0–100 ft). This relationship was consistent on both shallow-slope and steep-slope properties, although the relative risk analysis showed considerable variation among classes (Table 2) There was a steadily decreasing proportion of destroyed structures at greater distances of defensible space up to 30 m (100 ft) on the steep-slope structures with defensible space measured regardless of property boundaries (Fig. 4*b*). Otherwise, the biggest difference in proportion of destroyed structures occurred between 0 and 7 m (0–25 ft) and 8–15 m (26–50 ft) (Figs 3*a–b*, 4*a*).

When the distance of defensible space was measured in intervals from 24 m (75 ft) and beyond, the Chi-square test showed no significant difference among groups ($P = 0.96$ for shallow-slope properties and $P = 0.74$ for steep-slope properties) (Figs 3, 4), although again, the relative risk analysis showed considerable variation (Table 2).There was a slight increase in the proportion of homes destroyed at longer distance intervals when the defensible space was measured only to the property boundaries (Fig. 3*a–b*). This slight increase is less apparent when distances were measured regardless of boundaries (Fig. 4*a–b*).

The relative risk calculations showed that the ratio of proportions was generally more variable among successive pairs when the distances were measured within property boundaries (Table 2). For these calculations, the risk of a structure being destroyed was significantly lower when the defensible space distance was 8–15 m (25–50 ft) compared to 0–7 m (0–25 ft) on both shallow- and steep-slope properties. On the steep-slope properties, there was an additional reduction of risk when comparing 24–30 m (75–100 ft) to 16–23 m (50–75 ft). However, the risk of a home being destroyed was slightly significantly higher when there was 31–90 m (101–225 ft) compared to 16–23 m (50–75 ft). For distances that were measured regardless of property boundary (total clearance), the only significant differences in risk of burning were a reduction in risk for 8–15 m (25–50 ft) compared to 0–7 m (0–25 ft).

Table 2. Number of burned and unburned structures within defensible space distance categories (m), their relative risk and significance

A relative risk of 1 indicates no difference; <1 means the chance of a structure burning is less than the other group; >1 means the chance is higher than the other group. The relative risk is calculated for pairs that include the existing row and the row above. Confidence intervals are in parentheses

	Distance within property				Total distance			
	Burned	Unburned	Relative risk	*P*	Burned	Unburned	Relative risk	*P*
Shallow slope								
0–7	200	186			162	114		
8–15	109	198	0.69 (0.12)	<0.001	108	132	0.77	0.002
16–23	51	89	1.03 (0.30)	0.850	78	90	1.03	0.770
24–30	36	40	1.30 (0.39)	0.110	50	70	0.90	0.430
31–90	28	47	0.79 (0.24)	0.220	79	99	1.06	0.640
60 or 90+	10	6	1.67 (0.63)	0.040	8	9	1.01	0.830
Steep slope								
0–7	245	128			224	128		
8–15	174	148	0.82 (0.10)	0.001	158	139	0.84	0.008
16–23	85	68	1.03 (0.16)	0.750	73	83	0.87	0.210
24–30	29	56	0.61 (0.17)	0.004	26	50	0.73	0.080
31–	29	28	1.49 (0.48)	0.050	39	68	1.06	0.760
60 or 90+	5	5	0.98 (0.47)	0.950	4	8	0.91	0.830

Table 3. Effective treatment results reflecting the distance (in metres, with feet in parentheses) and percentage clearance within properties that provided significant improvement in structure survival during wildfires

The property mean is the average distance of defensible space or percentage clearance that was calculated on the properties before the wildfires and provides a means to compare the effective treatment result to the actual amount on the properties

	All parcels effective treatment ($n = 2000$)	Parcel mean	Shallow slope (mean 8%) effective treatment ($n = 1000$)	Parcel mean	Steep slope (mean 27%) effective treatment ($n = 1000$)	Parcel mean
Defensible space within parcel	10 (33)	13 (44)	4 (13)	14 (45)	25 (82)	11 (35)
Total distance defensible space	10 (32)	19 (63)	5 (16)	20 (67)	20 (65)	18 (58)
Mean percentage clearance on property	36	48	31	51	37	35

Effective treatment analysis

Analysis of the treatment–response relationships among defensible space and structures that survived wildfire showed that, when all structures are considered together, the mean actual defensible space that existed around structures before the fires was longer than the calculated effective treatment (Table 3). Regardless of whether the defensible space was measured within or beyond property boundaries, the estimated effective treatment of defensible space was nearly the same at 10 m (32–33 ft).

The effective treatment distance was much shorter for structures on shallow slopes (4–5 m (13–16 ft)) than for structures on steep slopes (20–25 m (65–82 ft)), but in all cases was <30 m (<100 ft). Although longer distances of defensible space were calculated as effective on steeper slopes, these structures actually had shorter mean distances of defensible space around their properties than structures on low slopes (Table 3).

The calculated effective treatment of the mean percentage clearance on properties was 36% for all properties, 31% for structures on shallow slopes and 37% for structures on steep slopes (Table 3). In total, the properties all had higher actual percentage clearance on their property than was calculated to be effective. However, this mainly reflects the shallow-slope properties, as those structures on steep slopes had less clearance than the effective treatment.

Multiple regression analysis

When defensible space was measured only to the property boundaries, it was not included in the best model, according to the all-subsets multiple regression analysis (Table 4). However, it was included in the best model when factoring in the distance of defensible space measured beyond property boundaries (Table 5). In both multiple regression analyses, low housing density and shorter distances to major roads were ranked as the most important variables according to their Akaike weights. Slope and surrounding fuel type were also in both of the best models as well as other measures of defensible space, including the percentage clearance on property and whether vegetation was overhanging the structure's roof. The number of sides in which vegetation was touching the structure was included in the best model when defensible space was only measured to the property boundary. The total explained deviance for the multiple regression models was low (12–13%) for both analyses.

Table 4. Results of multiple regression models of destroyed homes using all possible variable combinations and corrected Akaike's Information Criterion (AICc)

Includes variables measured within property boundary only. Top-ranked models include all those ($n = 12$) with AICc within 2 of the model with the lowest AICc. Relative variable importance is the sum of 'Akaike weights' over all models including the explanatory variable

Variable in order of importance	Relative variable importance	Model-averaged coefficient	Number inclusions in top-ranked models
Housing density	1	−0.003	12
Distance to major road	1	−0.0005	12
Percentage clearance	1	−0.02	12
Slope	1	0.03	12
Vegetation overhang roof	1	0.5	12
Fuel type	0.67	Factor	9
Vegetation touch structure	0.49	0.07	6
Distance defensible space within property	0.45	−0.0002	5
South-westness	0.36	−0.0007	3
Distance to minor road	0.28	−0.0002	1
D^2 of top-ranked model			0.123

Table 5. Results of multiple regression models of destroyed homes using all possible variable combinations and corrected Akaike's Information Criterion (AICc)

Includes variables measured beyond property boundary. Top-ranked models include all those ($n = 6$) with AICc within 2 of the model with the lowest AICc. Relative variable importance is the sum of 'Akaike weights' over all models including the explanatory variable

Variable in order of importance	Relative variable importance	Model-averaged coefficient	Number inclusions in top-ranked models
Housing density	1	−0.003	6
Distance to major road	1	−0.0005	6
Total distance defensible space	1	−0.004	6
Percentage clearance	1	−0.01	6
Vegetation overhang roof	0.99	0.4	6
Slope	0.99	0.03	6
Fuel type	0.86	Factor	4
South-westness	0.42	−0.0009	2
Distance to minor road	0.36	−0.0009	2
Neighbours' vegetation	0.27	0.08	1
Vegetation touch structure	0.27	0.18	1
D^2 of top-ranked model			0.125

Surrounding matrix

The cover type that most frequently surrounded the structures at the end of the defensible space measurements was urban vegetation, followed by urban vegetation leading into wildland vegetation, and wildland vegetation (Fig. 5). Many structures were equally surrounded by different cover types. There were no significant differences in the proportion of structures destroyed depending on the surrounding cover type. However, a disproportionately large proportion of structures burned (28 *v.* 9% unburned) when they were surrounded by urban vegetation that extended straight into wildland vegetation.

Discussion

For homes that burned in southern Californian urban areas adjacent to non-forested ecosystems, most burned in high-intensity Santa Ana wind-driven wildfires and defensible space increased the likelihood of structure survival during wildfire. The most effective treatment distance varied between 5 and 20 m (16–58 ft), depending on slope and how the defensible space was measured, but distances longer than 30 m (100 ft) provided no significant additional benefit. Structures on steeper slopes benefited from more defensible space than structures on shallow slopes, but the effective treatment was still less than 30 m (100 ft). The steepest overall decline in destroyed structures occurred when mean defensible space increased from 0–7 m (0–25 ft) to 8–15 m (26–50 ft). That, along with the multiple regression results showing the significance of vegetation touching or overhanging the structure, suggests it is most critical to modify vegetation immediately adjacent to the house, and to move outward from there. Similarly, vegetation overhanging the structure was also strongly correlated with structure loss in Australia (Leonard *et al.* 2009).

In terms of fuel modification, the multiple regression models also showed that the percentage of clearance was just as, or more important than, the linear distance of defensible space.

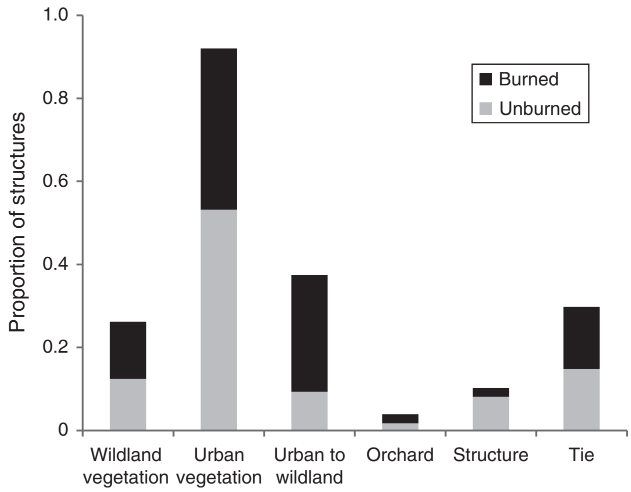

Fig. 5. Proportion of destroyed and unburned structures based on the primary surrounding cover type at the end of defensible space measurements. There were no significant differences in the proportion of burned and unburned structures within cover types ($P = 0.14$). Cover types are defined in Table 1.

However, as with defensible space, percentage clearance did not need to be draconian to be effective. Even on steep slopes, the effective percentage clearance needed on the property was <40%, with no significant advantage beyond that. Although these steep-slope structures benefited more from clearance, they tended to have less clearance than the effective amount, which may be why slope was such an important variable in the multiple regression models. Shallow-slope structures, in contrast, had more clearance on average than was calculated to be effective, suggesting these property owners do not need to modify their behaviours as much relative to people living on steep slopes.

Although the term 'clearance' is often used interchangeably with defensible space, this term is incorrect when misinterpreted to mean clearing all vegetation, and our results underline this difference. The idea behind defensible space is to reduce the continuity of fuels through maintenance of certain distances among trees and shrubs. Although we could not identify the vertical profile of fuels through Google Earth imagery, the fact that at least 60% of the horizontal woody vegetative cover can remain on the property with significant protective effects demonstrates the importance of distinguishing defensible space from complete vegetation removal. Thus, we suggest the term 'clearance' be replaced with 'fuel treatment' as a better way of communicating fire hazard reduction needs to home owners.

The percentage cover of woody shrubs and trees was not evenly distributed across properties, and we did not collect data describing how the cover was distributed. Considering the importance of defensible space and vegetation modification immediately adjacent to the structure, it should follow that actions to reduce cover should also be focussed in close proximity to the structure. The hazard of vegetation near the structure has apparently been recognised for some time (Foote *et al.* 1991; Ramsey and McArthur 1994), but it is not stressed enough, and rarely falls within the scope of defensible space guidelines or ordinances.

In addition to the importance of vegetation overhanging or touching the structure, it is important to understand that ornamental vegetation may be just as, if not more, dangerous than native vegetation in southern California. Although the results showed no significant differences in the cover types in the surrounding matrix, there was a disproportionately large number of structures destroyed (28% burned *v.* 9% unburned) when ornamental vegetation on the property led directly into the wildland. Ornamental vegetation may produce highly flammable litter (Ganteaume *et al.* 2013) or may be particularly dangerous after a drought when it is dry, or has not been maintained, and species of conifer, juniper, cypress, eucalypt, *Acacia* and palm have been present in the properties of many structures that have been destroyed (Franklin 1996). Nevertheless, ornamental vegetation is allowed to be included as defensible space in many codes and ordinances (Haines *et al.* 2008).

One reason that longer defensible space distances did not significantly increase structure protection may be that most homes are not destroyed by the direct ignition of the fire front but rather due to ember-ignited spot fires, sometimes from fire brands carried as far as several km away. Although embers decay with distance, the difference between 30 and 90 m (100 and 300 ft) may be small relative to the distance embers travel under the severe wind conditions that were present at the time of the fires. The ignitability of whatever the embers land on, particularly adjacent to the house, is therefore most critical for propagating the fire within the property or igniting the home (Cohen 1999; Maranghides and Mell 2009).

Aside from roofing or home construction materials and vegetation immediately adjacent to structures (Quarles *et al.* 2010; Keeley *et al.* 2013), the flammability of the vegetation in the property may also play a role. Large, cleared swaths of land are likely occupied at least in part by exotic annual grasses that are highly ignitable for much of the year. Conversion of woody shrubs with higher moisture content into low-fuel-volume grasslands could potentially increase fire risk in some situations by increasing the ignitability of the fuel; and if the vegetation between a structure and a fire is not readily combustible, it could protect the structure by absorbing heat flux and filtering fire brands (Wilson and Ferguson 1986).

The slight increase in proportion of structures destroyed with longer distances of defensible space within parcel boundaries was surprising. However, that increase was not significant in the Chi-square analysis, although there were some significant differences in the pairwise relative risk analysis. Nevertheless, the largest significant effect of defensible space was between the categories of 0–7 m (0–25 ft) to 8–15 m (26–50 ft), and it may be that differences in categories beyond these distances are not highly meaningful or reflect an artefact of the definition of distance categories. These relationships at longer distances are likely also weak compared to the effect of other variables operating at a landscape scale. Although the categorical analysis allowed us to answer questions relative to legal requirements and specific distances, the effective treatment analysis was important for identifying thresholds in the continuous variable.

The multiple regression models showed that landscape factors such as low housing density and longer distances to major roads were more important than distance of defensible space for explaining structure destruction, and the importance of these variables is consistent with previous studies (Syphard *et al.* 2012, 2013), despite the smaller spatial extent studied here. Whereas this study used an unburned control group exposed to the same fires as the destroyed structures, previous studies accounted for structures across entire landscapes. The likelihood of a fire destroying a home is actually a result of two major components: the first is the likelihood that there will be a fire, and the second is the likelihood that a structure will burn in that fire. In this study, we only focussed on structure loss given the presence of a fire, and the total explained variation for the multiple regression models was quite low at ~12%. However, when the entire landscape was accounted for in the total likelihood of structure destruction, the explained variation of housing density alone was >30% (Syphard *et al.* 2012). One reason for the relationship between low housing density and structure destruction is that structures are embedded within a matrix of wildland fuel that leads to greater overall exposure, which is consistent with Australian research that showed a linear decrease of structure loss with increased distance to forest (Chen and McAneney 2004). That research, however, only focussed on distance to wildland boundaries and did not quantify variability in defensible space or ornamental vegetation immediately surrounding structures. Thus, fire safety is important to consider at multiple scales and for multiple variables, which will ultimately require the cooperation of multiple stakeholders.

Conclusions

Structure loss to wildfire is clearly a complicated function of many biophysical, human and spatial factors (Keeley *et al.* 2009; Syphard *et al.* 2012). For such a large sample size, we were unable to account for home construction materials, but this is also well understood to be a major factor, with older homes and wooden roofs being most vulnerable (Franklin 1996; Cohen 1999, 2000). In terms of actionable measures to reduce fire risk, this study shows a clear role for defensible space up to 30 m (100 ft). Although the effective distances were on average much shorter than 30 m (100 ft), we recognise that additional distance may be necessary to provide sufficient protection to firefighters, which we did not address in this study (Cheney *et al.* 2001). In contrast, the data in this study do not support defensible space beyond 30 m (100 ft), even for structures on steep slopes. In addition to the fact that longer distances did not contribute significant additional benefit, excessive vegetation clearance presents a clear detriment to natural habitat and ecological resources. Results here suggest the best actions a homeowner can take are to reduce percentage cover up to 40% immediately adjacent to the structure and to ensure that vegetation does not overhang or touch the structure.

In addition to defensible space, this study also underlines the potential importance of land use planning to develop communities that are fire safe in the long term, in particular through their reduction to exposure to wildfire in the first place. Localised subdivision decisions emphasising infill-type development patterns may significantly reduce fire risk in the future, in addition to minimising habitat loss and fragmentation (Syphard *et al.* 2013). This study was conducted in southern California, which has some of the worst fire weather in the world and many properties surrounded by large, flammable exotic trees. Therefore, recommendations here should apply to other non-forested ecosystems as well as many forested regions.

Acknowledgements

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