Appendix A

SCIENCE & RESEARCH

Quantifying the Range of Variability in Wilderness Areas:

A Reference When Evaluating Wilderness Candidates

ABSTRACT The US Forest Service recently began revising forest management plans under the 2012 forest planning rule. The forest plan revision process includes a wilderness inventory and evaluation that can lead to some lands being recommended as wilderness. During this process, the Forest Service evaluates the wilderness character of candidate roadless lands. This evaluation can result in the disqualification of areas for wilderness recommendations based on degraded qualities of wilderness character. However, it is unknown how the wilderness character of candidate lands compares to conditions within the existing National Wilderness Preservation System (NWPS). Without such an evaluation and comparison, candidate areas for wilderness recommendation could be held to a higher standard of wilderness character than lands currently protected as wilderness. Here, four national mapped datasets representing qualities of wilderness character (human modification, distance to roads, light pollution, and noise pollution) of candidate roadless lands (also called wilderness candidates) were compared to the existing NWPS. The number of candidate areas that were more degraded than the most degraded wilderness area was counted. In addition, the distribution of values of wilderness candidates was compared with the distribution of values from areas of the NWPS. Data were analyzed at two scales: among all wilderness areas and within Forest Service regions. Among all wilderness areas, no wilderness candidate was more degraded than the range of conditions within the existing NWPS. Within regions, very few candidates were more degraded than current wilderness areas, irrespective of the metric evaluated. These results suggest that most candidates for wilderness recommendation fall within the range of conditions observed within the current NWPS. A similar approach to quantifying the range of conditions within existing wilderness could be used in local evaluations to ensure that candidates for wilderness are not held to a higher standard of wilderness character than that of the existing NWPS.

By R. Travis Belote

PEER REVIEWED

Wilderness areas of the National Wilderness Preservation System (NWPS) serve as core units of a national system of conservation reserves in the United States (Aycrigg et al. 2013, 2016a; Belote et al. 2016). As important as the existing NWPS is, additional reserves are needed to better represent ecological diversity (Dietz et al. 2015) and establish an ecologically-connected network of protected areas (Aycrigg et al. 2016b, Belote et al. 2017). Fortunately, a process exists whereby lands can be recommended to the US Congress for inclusion as new legislated wilderness areas. For example, during land management planning under the 2012 Planning Rule of the National Forest Management Act (USDA 2012), the Forest Service evaluates wilderness character of lands under Chapter 70 of the planning rule directives. Through this local inventory and evaluation process, the agency determines which candidate roadless lands (generally greater than 2,023 ha/4,999 acres) on each national forest maintain outstanding wilderness character and which areas should be recommended as wilderness.

Wilderness character is based on concepts outlined in the Wilderness Act including naturalness, undeveloped condition, untrammeledness, and outstanding opportunities for solitude or primitive recreation (Landres et al. 2015). Wilderness character is associated with the concept of wildness (Aplet 1999) and generally describes ecological conditions (e.g., integrity) and the degree of control humans assert on natural processes (e.g., through fire suppression and plant and animal management). Quantifying and mapping wilderness character of existing wilderness areas has occurred throughout the country using spatial data representing human impacts to qualities of land (Tricker et al. 2012, 2013, 2016, 2017; Burrows et al. 2016). Datasets used in these analyses have included features that represent built structures, distance to roads, and sights and sounds that impact wilderness character. These qualities are also closely related to conditions outlined for assessment in the directives of the wilderness inventory and evaluation process. In developing forest plan revisions, such wilderness inventory and evaluation are usually conducted at local scales across a single (or several adjacent) national forests.

Such local evaluations in conservation planning are a critical step for determining high-priority lands to include in formal ecological reserves (Pressey and Bottrill 2009). Local evaluations provide data necessary for managers to identify lands suitable or unsuitable for wilderness recommendations. Data on local roads, trails, structures, historical timber management, the presence of nonnative species, and other qualities must be evaluated to identify and prioritize places with high wilderness character. The importance of local evaluations notwithstanding, mapped national and global datasets increasingly provide opportunities to evaluate the importance of land based on a national or global perspective (Pouzols et al. 2014; Belote and Irwin 2017).

In some instances, local evaluations may result in the national or global significance of areas being overlooked (sensu Noss et al. 2015). For instance, features that degrade wilderness character at a local level (e.g., an old cabin or a patch of invasive species) could result in managers downgrading or disqualifying areas for consideration as new wilderness areas. However, the same candidate lands – when compared to all other lands in the nation – may be of extremely high value and represent some of the wildest remaining lands in the country. Only through a national or regional evaluation of candidates can such determination of relative quality be made (e.g., Belote and Irwin 2017).

Moreover, through the wilderness inventory and evaluation process some candidates for recommended wilderness may be downgraded because of features that reduce wilderness character (e.g., cabins, bridges), even though such features may occur in existing wilderness areas. The NWPS is widely regarded as a national and global treasure for maintaining America's remaining wildlands (Cordell et al. 2005). Yet, candidates for future consideration may be held to a higher standard than the lands that currently make up the NWPS. During local evaluations of wilderness character, it is critical to evaluate candidates for future wilderness recommendations at a national scale and compare wilderness character of candidates with that of the existing NWPS.

Candidates for future consideration may be held to a higher standard than the lands that currently make up the NWPS

With the above concepts in mind, two primary questions were asked: (1) What is the range of variability in various metrics of wilderness character within the NWPS? (2) How do candidates for recommended wilderness compare to this range of variability?

Methods

Four metrics that serve as proxies for wilderness character were evaluated: human modification (Theobald 2013); distance to roads (National Park Service 2013); light pollution (Monahan et al. 2012); and noise pollution (Mennitt et al. 2014). These four metrics (Figure 1) are among several national datasets that represent qualities associated with wilderness character and wildness (sensu Aplet 1999; Aplet et al. 2000; Watson et al. 2016). These metrics have been used to map wilderness character in several wilderness areas in the United States, generally provide estimates of gradients in wilderness character (Tricker et al. 2012, 2013, 2016, 2017; Burrows et al. 2016), and represent nationally mapped data of relatively high resolution. Other national mapped datasets representing estimates of biological diversity priorities are also available (Dietz et al. 2015; Jenkins et al. 2015; Belote et al. 2017) and could be used to evaluate the importance of these candidate lands as means of better representing biodiversity (Aycrigg et al. 2015; Belote and Irwin 2017). The intent here was to focus on nationally available mapped data representing four qualities closely associated with measures of wilderness character (Landres et al. 2015).

Human modification data is based on land cover, human population density, roads, and other mapped metrics of ecological condition (Theobald 2013). Data are scaled from 0 (no measured human modification) to 1 (high degree of human modification). Distance to roads was calculated as the geographic distance (in meters) from all roads using Topologically Integrated Geographic Encoding and Referencing (TIGER) data available from the US Census (US Census Bureau 2015). This analysis was conducted with the EUCLIDEAN DISTANCE tool in ArcGIS 10.5, which resulted in gridded data where distance is assigned to each 90-meter resolution pixel for the contiguous US. Wilderness character is assumed to increase with distance from roads, although the decay of impact likely varies among ecosystems. Different types of roads are not differentiated here, as these differences are accounted for within the human modification data. Smaller distances are related to likely degraded wilderness character, based on increased human access, pervasive sights and sounds from the roads, and other ecological impacts associated with roads (Tricker et al. 2012; Burrows et al. 2016; Ibisch et al. 2016).

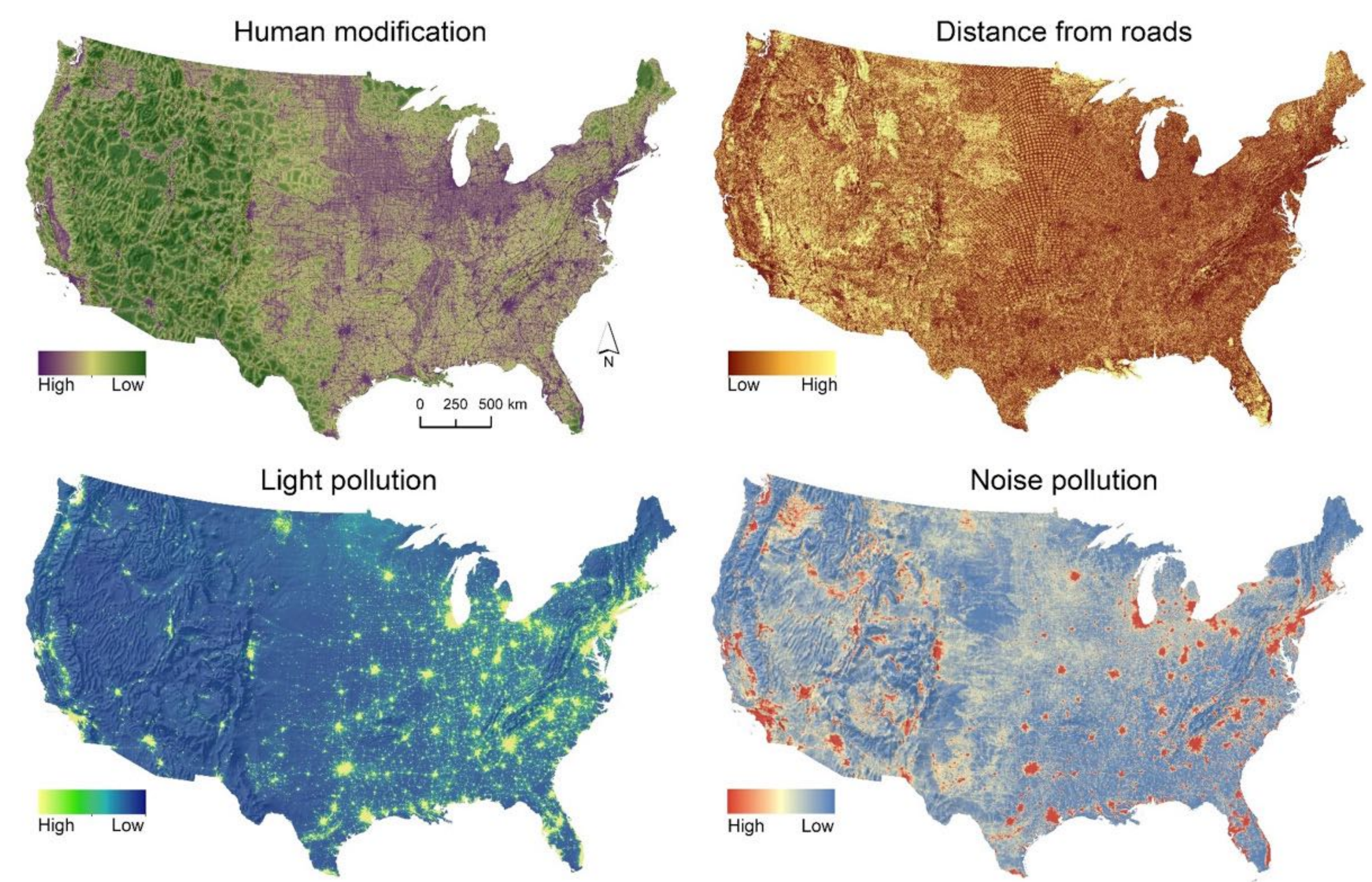

Figure 1 – Four datasets used as measures of wilderness character: human modification, distance to roads, light pollution, and noise pollution.

Light pollution represents satellite-measured light intensity during the night from the Visible Infrared Imaging Radiometer Suite (VIIRS) nighttime lights data (Nelson et al. 2015). This mapped dataset serves as a measure of the intactness of the night sky. Higher values represent more intense light pollution and thus lower wilderness character (Tricker et al. 2012). Similarly, mapped data of human-generated noise pollution is based on field observations and a spatial model using landscape features that influence sound propagation (Mennitt et al. 2014; Nelson et al. 2015). Greater intensity of human noises (higher predicted dBA) is associated with reduced wilderness character. The Forest Service 2012 Planning Rule directives on wilderness evaluations suggests that "sights and sounds from outside the area" should be considered when evaluating wilderness suitability of candidates. Data on light and sound pollution provide a national dataset to evaluate these qualities of wilderness character.

For each of the four qualities, data were extracted for all 683 existing wilderness areas of the NWPS within the contiguous United States, along with 300 wilderness inventory areas among 9 national forests. Summary statistics were then calculated for each unit (Figure 2). Hereafter, wilderness inventory areas are referred to as wilderness candidates, as they are among a pool of areas currently being considered for wilderness recommendations in national forest planning. The nine national forests included the Flathead and Helena-Lewis and Clark from the Northern Region (R1); the Cibola and Santa Fe from the Southwestern Region (R3); the Rio Grande from the Rocky Mountain Region (R2); the Sierra, Sequoia, and Inyo from the Pacific Southwest Region (R5); and the Nantahala-Pisgah from the Southern Region (R8). These areas were chosen because the Forest Service had completed identification of their candidate wilderness areas in these national forests at the time of our analysis. Candidate wilderness boundaries were obtained from local Forest Service staff. The mean distribution of each quality from all NWPS units in the lower 48 states was plotted using kernel density plots from the ggplot2 package in R. In addition, values from individual units were added as "rug plots" to evaluate the range for each quality. Data were also stratified based on Forest Service regions so that existing wilderness in different regions of the country were compared to candidates from the roadless area inventories in those same regions.

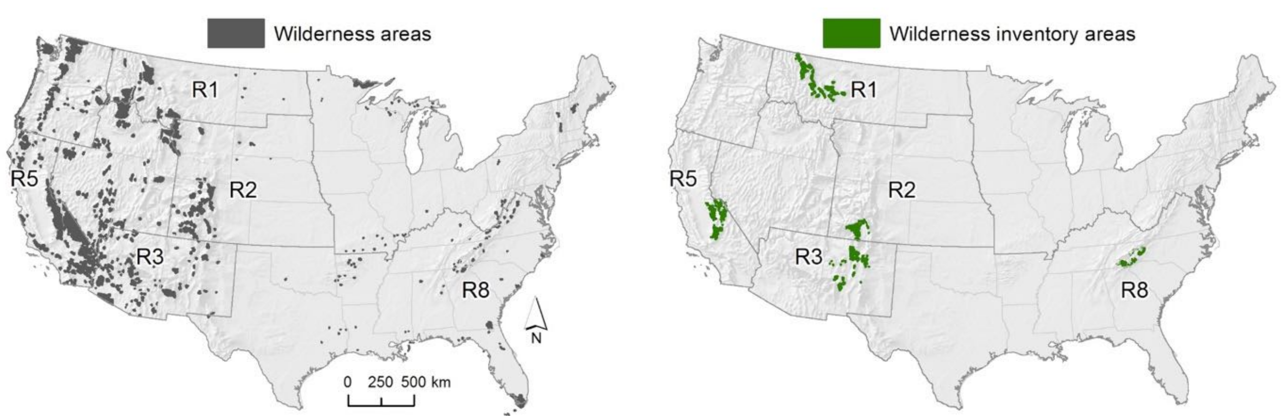

Figure 2 – The existing National Wilderness Preservation System and wilderness inventory areas (wilderness candidates) currently being evaluated among different Forest Service regions.

Because these data represent a census of all areas, inferential statistics (e.g., using analysis of variance) were not conducted. Instead, visual comparisons were made of the distributions of data using the kernel density plots. The number of wilderness candidate units with characteristics more degraded than the range observed within the NWPS among and within regions were also evaluated. In other words, the number of wilderness candidates with greater human modification, were closer to roads, or were exposed to higher levels of light and noise pollution than existing wilderness areas were counted. Finally, as a post hoc analysis, mean elevation of wilderness areas and wilderness candidates was compared using a national 30-meter resolution digital elevation model to potentially explain observed patterns in wilderness character metrics.

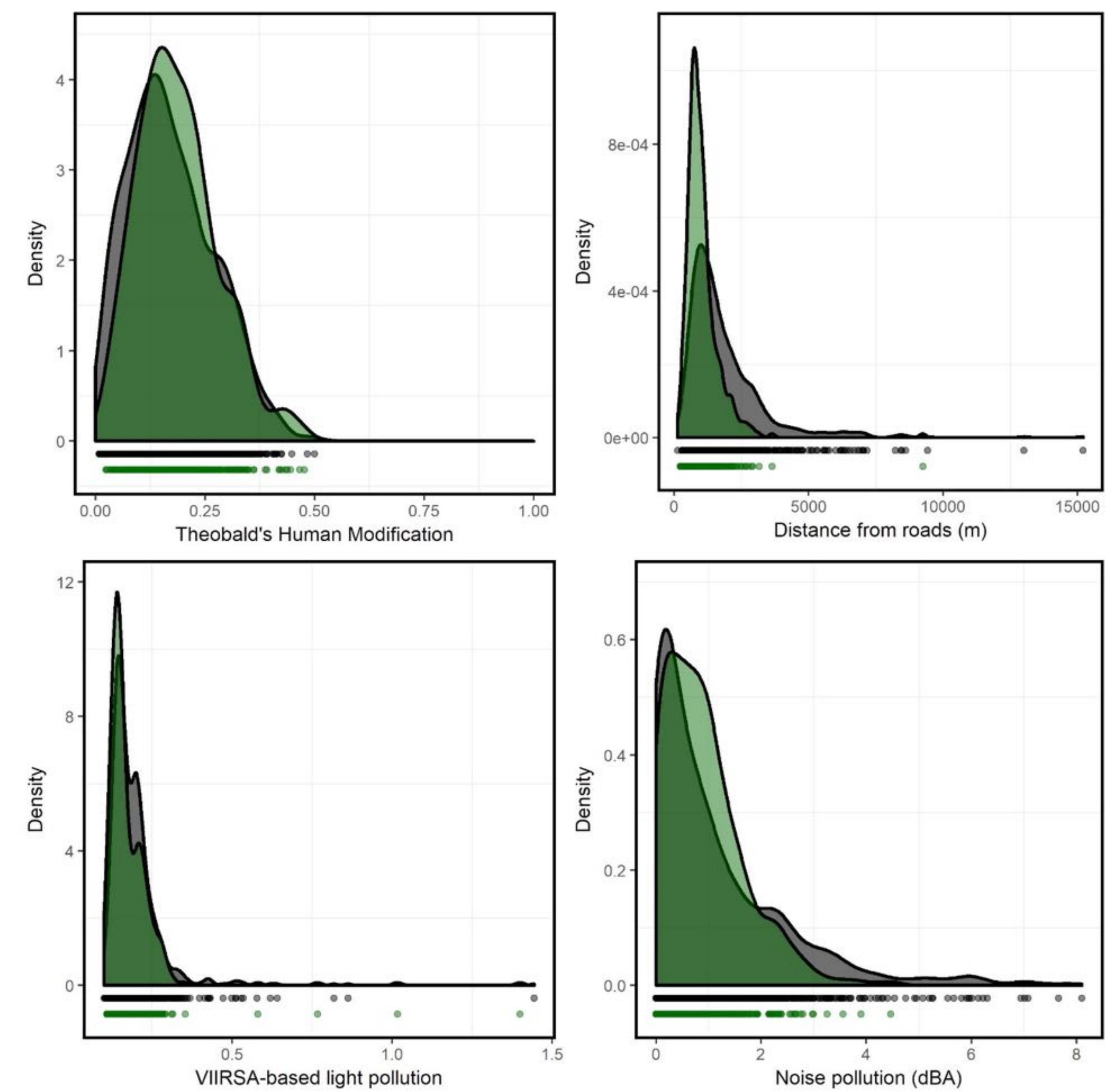

Figure 3 – Human modification, distance from roads, light pollution, and noise pollution for all wilderness areas (black) and all wilderness candidate areas to date (green). The overlap registers as dark green.

Results

When compared across the lower 48 states, all candidates for recommended wilderness were within the range of values observed within existing wilderness areas of the NWPS (Figure 3; Table 1). The kernel density distribution of values for each quality varied little when comparing wilderness areas with wilderness candidate areas, although wilderness areas tended to be farther from roads than wilderness candidate areas (Figure 3).

Region	Number of NWPS areas	Number of wilderness candidate areas > 2023 hectares	Number of units more degraded with respect to human modification	Number of units more degraded with respect to distance to roads	Number of units more degraded with respect to light pollution	Number of units more degraded with respect to noise pollution
Across all regions	683	300	0	0	0	0
Northern (R1)	25	58	1	1	0	0
Rocky Mountain (R2)	62	46	0	1	0	1
Southwestern (R3)	117	92	4	21	4	0
Pacific Southwest (R5)	158	72	1	0	0	0
Southern (R8)	101	32	0	1	0	0

Table 1 – Number of wilderness candidates among five US Forest Service Regions and the number of those units that are more degraded compared to the range of values within existing wilderness areas of the NWPS.

When comparing human modification values within region, 6 (2% of total) wilderness candidate areas among all regions were outside the range of values observed within wilderness areas (Figure 5; Table 1). The distribution of human modification in wilderness candidate areas overlapped wilderness areas for nearly all regions. The wilderness candidate areas were slightly more modified than the NWPS units in R1, R3, and R5 (Figure 4). When comparing distance from roads within region, 24 (8% of total) wilderness candidate areas among all regions were outside the range of values observed within wilderness areas (Figure 5; Table 1). Based on the distribution of values, wilderness candidate areas tended to occur closer to roads compared to wilderness areas in nearly all regions.

When comparing light and noise pollution within region, 4 (1.3% of total) and 1 (<0.5%) of wilderness candidates among all regions, respectively, were outside the range of values observed within wilderness areas (Figure 5; Table 1). Based on the distribution of values of light pollution, wilderness candidate areas tended to be very similar to wilderness areas in nearly all regions (Figure 6). However, R2 wilderness candidate areas tended to experience less light pollution, whereas R8 wilderness candidate areas tended to experience more light pollution compared to wilderness in those regions (Figure 6). Based on the distribution of values of noise pollution, wilderness candidate areas tended to be very similar to wilderness areas in nearly all regions (Figure 7). Finally, average wilderness candidate areas were slightly higher in elevation (2,198 meters/7,211 ft. above sea level) than wilderness areas (1,413 meters/4,636 ft. asl) in the regions assessed here.

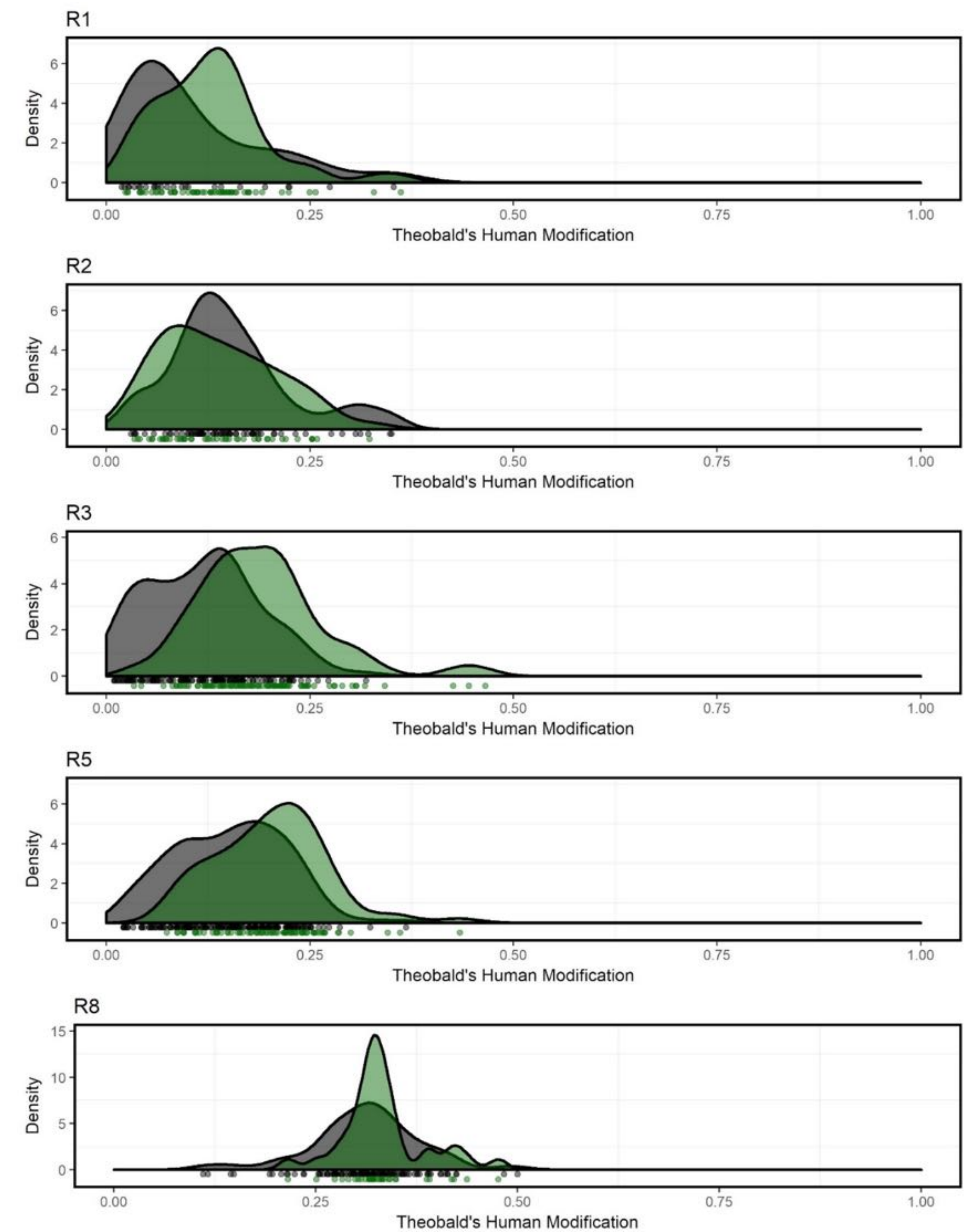

Figure 4 – Distribution of human modification by region within wilderness areas of the National Wilderness Preservation System (black) and wilderness candidate areas (green). Individual units are shown as a "rug plot" and used to count units outside the range of conditions within the NWPS.

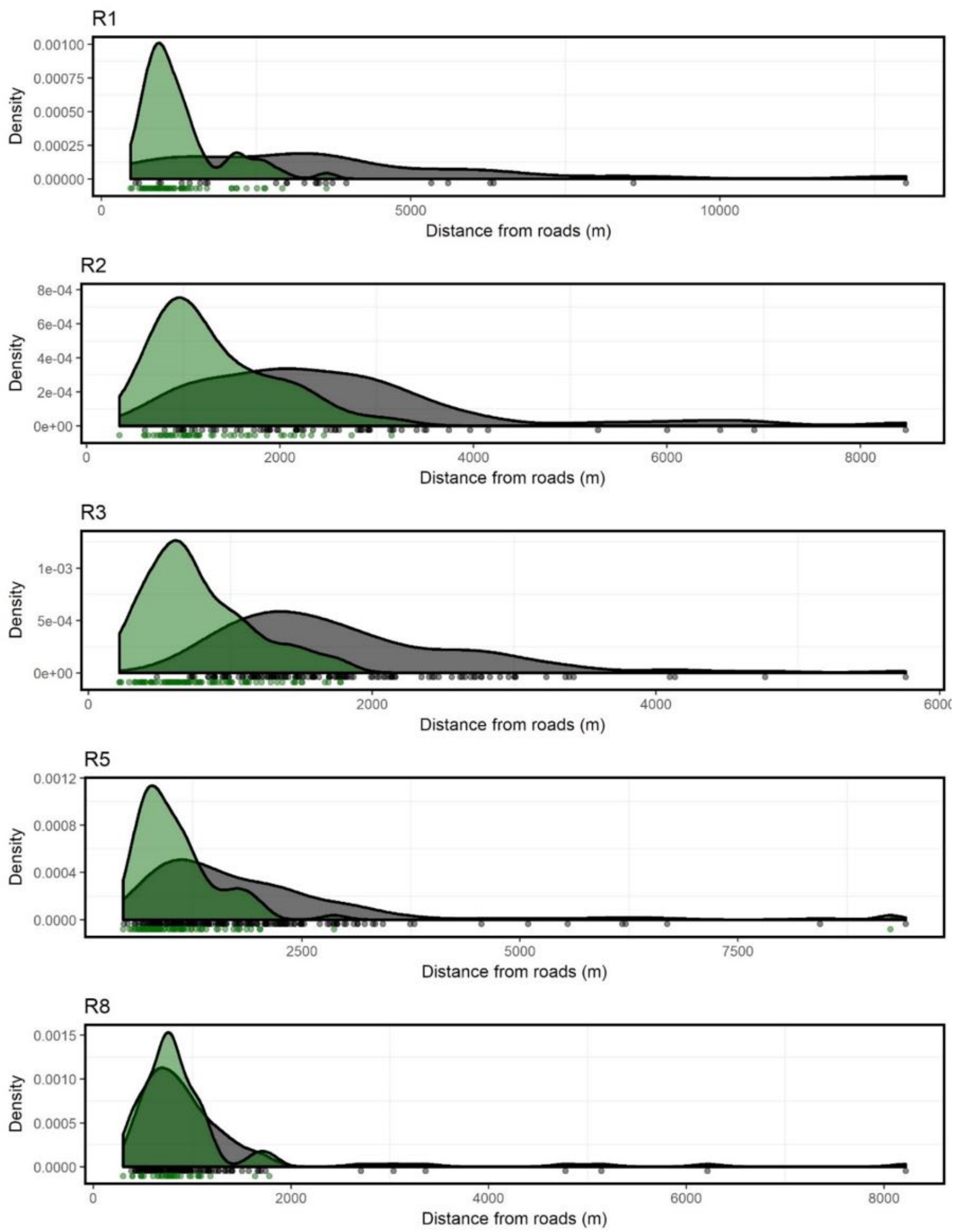

Figure 5 –Distance from roads by region within wilderness areas of the National Wilderness Preservation System (black) and wilderness candidate areas (green). Individual units are shown as a "rug plot" and used to count units outside the range of conditions within the NWPS.

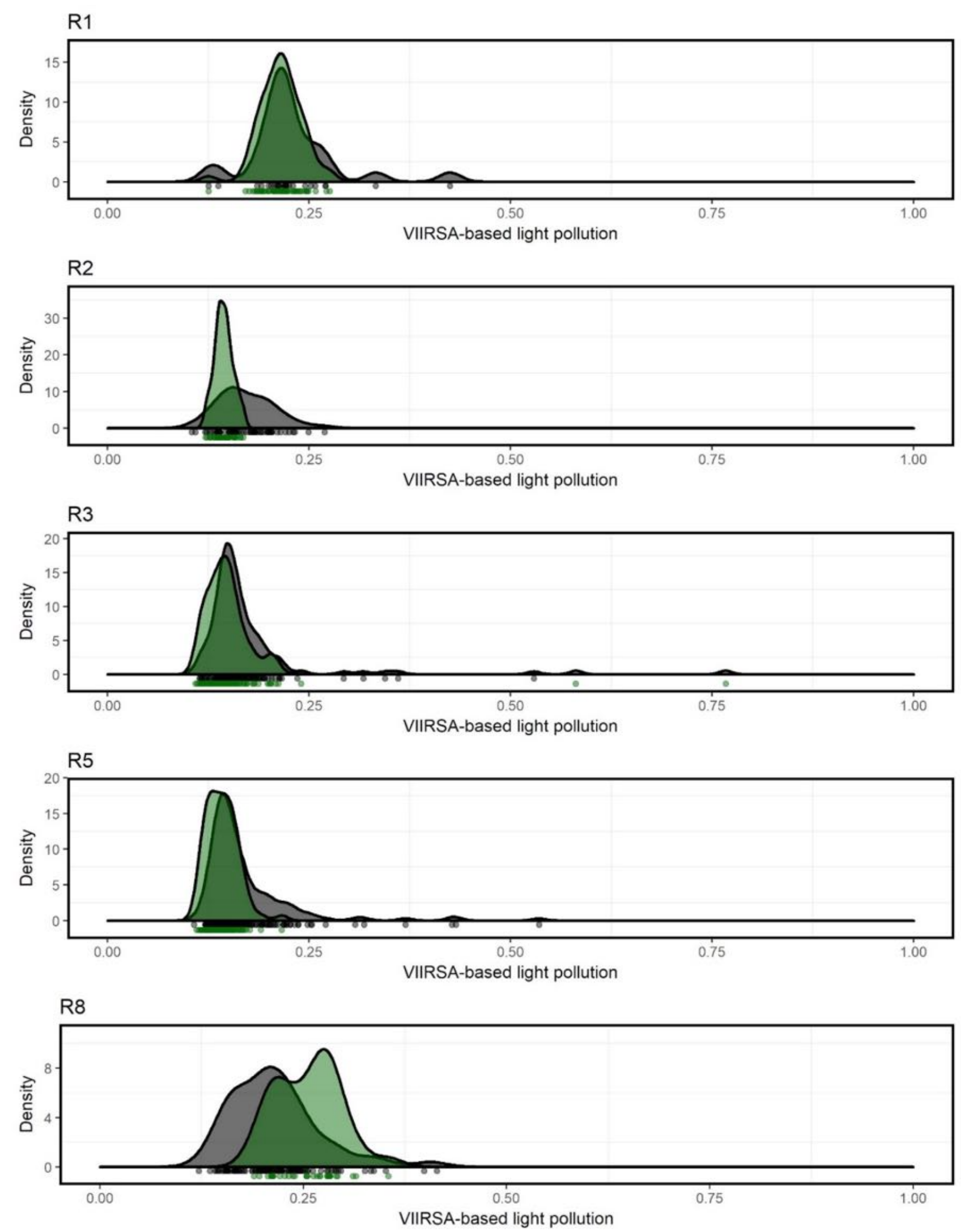

Figure 6 – Light pollution by region within wilderness areas of the National Wilderness Preservation System (black) and wilderness candidate areas (green). Individual units are shown as a "rug plot" and used to count units outside the range of conditions within the NWPS.

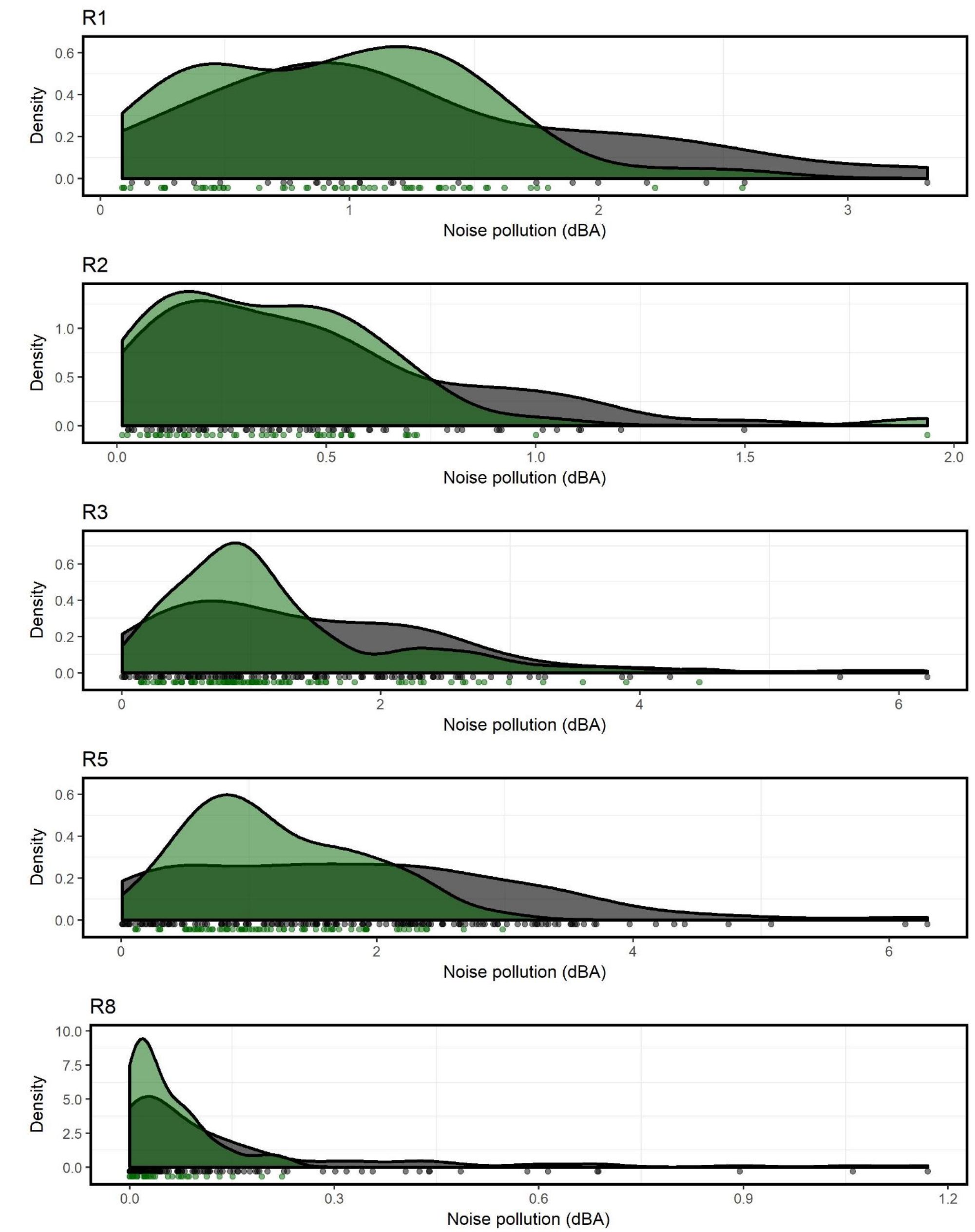

Figure 7 – Noise pollution by region within wilderness areas of the National Wilderness Preservation System (black) and wilderness candidate areas (green). Individual units are shown as a "rug plot" and used to count units outside the range of conditions within the NWPS.

Discussion

The wilderness character of wilderness candidates was almost always within the range of the existing NWPS. As citizens, stakeholders, and agency personnel evaluate candidates for future wilderness recommendations, this kind of national assessment of wilderness character should be an important step to ensure that candidates for wilderness are not held to a higher standard than lands within the existing NWPS. In fact, when pooled nationally, all wilderness candidate areas were within the range of values observed in the NWPS for each of the four metrics.

As a sample of areas, wilderness candidates did tend to have higher degree of human modification and lower distance to roads compared to wilderness areas in some regions (e.g., R1, R3, and R5) based on the distributions of values. It was hypothesized that this was because wilderness candidates were lower in elevation than wilderness areas. Wilderness and other protected areas typically occur higher in elevation (Aycrigg et al. 2013) with steeper slopes compared to unprotected lands, which has provided easier access for building roads and harvesting timber (Belote and Aplet 2014), or otherwise converting land to agricultural commercial, or residential land uses. Contrary to expectations, wilderness candidates were slightly higher in elevation compared to existing wilderness among and within the regions studied here. Despite this pattern, wilderness candidates do tend to be closer to human development and roads compared to existing wilderness.

Human modification and distance to roads both serve as measures of ecological integrity, degree of trammeledness, and outstanding opportunities for solitude, all of which represent wilderness character (Aplet et al. 2000; Theobald 2013; Landres et al. 2015). Despite the general patterns between wilderness candidates and wilderness areas in these qualities, no individual wilderness candidate could be considered more degraded than the range of the existing NWPS, and only a few could be considered more degraded than the range of values within the region. Wilderness candidates in the Southwestern Region (R3), however, had the greatest number of areas (23% of units) outside of the range of existing wilderness in that region with respect to distance from roads.

Light and noise pollution varied little between wilderness candidates and wilderness areas. In fact, in the Rocky Mountain Region (R2), wilderness candidates were characterized by darker night skies (less light pollution) than existing wilderness. Like distance to roads, the largest number of wilderness candidates that had more light pollution than the existing NWPS occurred in the Southwestern Region (R3), which may reflect proximity to urban or developed areas around Santa Fe and Los Alamos, New Mexico, and the surrounding national forests. Light and noise pollution impact ecological systems (Longcore and Rich 2004; Mennitt et al. 2014; Shannon et al. 2016) and can erode wilderness character (Tricker et al. 2012). Dark night skies with intact star-viewing opportunities and quiet outdoor experiences free from human-generated noises all represent important qualities of wilderness character and wildness (Aplet et al. 2000).

Taken together, these results suggest that in most cases, candidates for recommended wilderness represent lands that are as wild as the existing NWPS. As human populations increase and land use expands (Sohl et al. 2014), protecting the remaining wildlands is increasingly recognized as a key global, national, and local conservation priority (Venter et al. 2016; Watson et al. 2016; Belote et al. 2017). Wilderness areas and the NWPS represent a critical tool used to protect the remaining wildlands. The process by which agencies evaluate lands for potential wilderness recommendations is central to adding lands to this system. Although local evaluations will continue to be essential to assessing wilderness character (sensu Landres et al. 2015), national and global datasets increasingly allow for broad-scale analyses to evaluate lands across larger areas (Belote and Irwin 2017).

Ultimately, only the US Congress has the authority to legislatively designate new wilderness areas, which requires social and political processes. Agency recommendations to Congress, however, are an important aspect of designating new wilderness areas, as well as administratively maintaining the wilderness character of lands classified as recommended wilderness. Given the loss of wildlands globally (Watson et al. 2016) and nationally (Theobald et al. 2016), it is imperative that a national context is applied to decisions of how wilderness character is protected and managed on federal lands.

In conclusion, four nationally available mapped datasets representing measures of wilderness character were used to compare wilderness candidates with existing wilderness areas. This analysis was used to identify whether and how many wilderness candidates fell outside the range of the existing NWPS. Local evaluations could use the same framework to compare wilderness character of candidates with the existing system. If local assessments measure features that erode wilderness character (e.g., old roads, cabins, historical timber harvests) within lands serving as candidate for wilderness recommendations, managers could compare these qualities to nearby wilderness areas. Although wilderness areas represent some of the wildest and most intact lands in the country, they are not without human impacts (Cole and Yung 2010). Candidates for future wilderness should not be held to a higher standard than the existing NWPS. IJW

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Appendix B


Stateline Trail in Meadow Creek-Upper North Fork Roadless Area

Dear Lolo Forest Planning Team & Lolo National Forest Service Staff,

Please accept this wilderness character data to inform the Lolo Forest Plan revision process. We appreciated the opportunity to provide input on the wilderness inventory data, and hope you will take into consideration this information for your wilderness evaluation analysis. Using the wilderness evaluation worksheet your team created and protocols from the Wilderness Institute, we collected data regarding the Sliderock/Quigg Roadless Area, the Cube Iron/Sundance Ridge Roadless Areas, and a few of the important roadless areas that make up the Great Burn ecosystem. We believe each of these unique roadless areas, as well as other areas not included in this assessment, possess incredible wilderness character and should be seriously considered for recommended wilderness designation in the new forest plan. We will provide additional data during the 2024 scoping period.

Please let us know if you have any questions or would like to discuss any of our findings. Thank you for all your work on the planning process thus far. We look forward to continuing to work with you.

Sincerely,

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Wilderness Characteristics Overview

The following report summarizes data collected during a wilderness character inventory study led by Wild Montana in the summer of 2023. Unless otherwise noted, every photo in this report was taken by Wild Montana staff: Maddy Munson, Mattea Prison, or Erin Clark. For each inventoried roadless area studied, we measured the following qualities:

I. Untrammeled and Natural Quality

Natural quality reflects the extent to which "ecological systems are substantially free from the effects of modern civilization." The natural quality is assessed by monitoring attributes that reflect the integrity of ecological systems, such as species composition and physical characteristics.

II. Undeveloped Quality

Undeveloped quality is one of the primary elements of wilderness character found within the language of the 1964 Wilderness Act. This quality refers to the extent to which "wilderness retains its primeval character and influence, and is essentially without permanent improvement or modern occupation." Non-recreational developments such as installations and signs are considered to affect the undeveloped quality of wilderness character. Recreationally-focused developments, such as trails, campsites, shelters, etc. are considered in the next section, under the solitude or primitive and unconfined quality of wilderness character.

III. Solitude or Primitive and Unconfined Recreation Quality.

Solitude or primitive and unconfined recreation quality is another of the primary elements of wilderness character found within the language of the 1964 Wilderness Act. This quality refers to the extent to which "wilderness provides outstanding opportunities for solitude or primitive and unconfined recreation," and assesses recreational developments such as trails, restrooms, shelters, and campsites. Attributes included in the protocols that reflect this quality are: trail width, non-system trails, evidence of motorized or mechanized use, encounters with other users on trails, motorized noise, visual intrusions from developments outside the Forest Service boundary, and campsite characteristics and impacts.

This quality evaluates the degree to which an area has outstanding opportunities for solitude or for a primitive and unconfined type of recreation. The word "or" means that an area only has to possess one or the other. The area does not have to possess outstanding opportunities for both elements, nor does it need to have outstanding opportunities on every acre.

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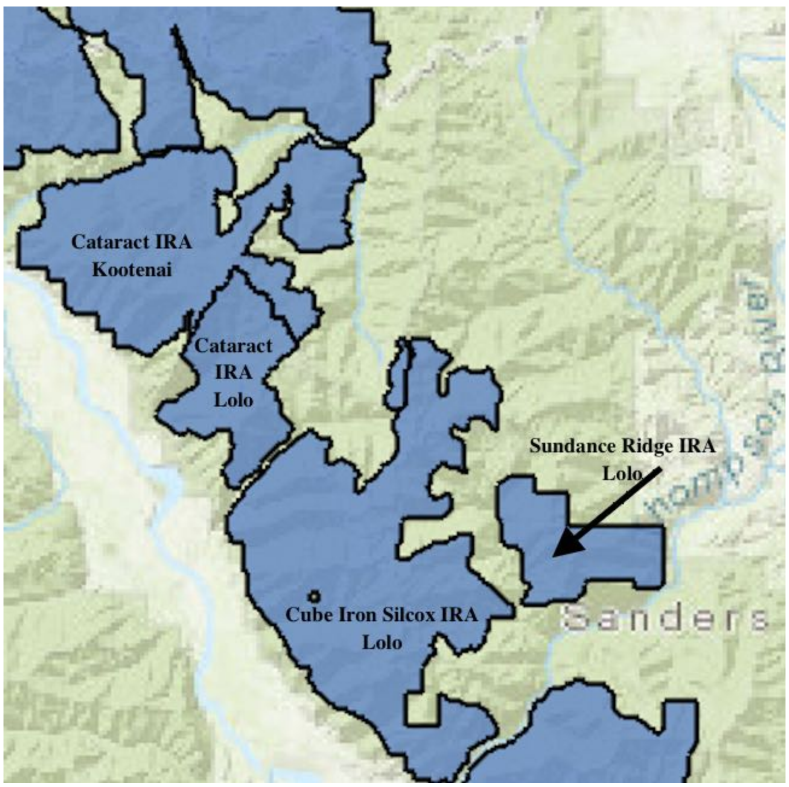
Cube Iron-Silcox and Sundance Ridge Inventoried Roadless Areas

July 2023

Sundance Ridge Inventoried Roadless Area

Summary:

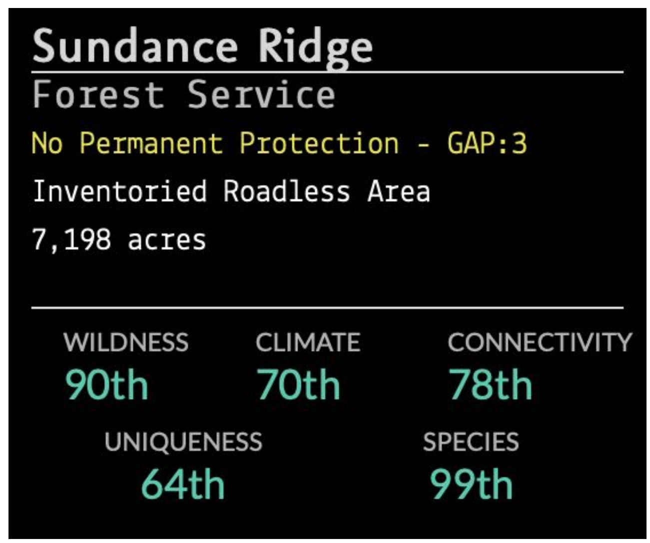
In July 2023, Wild Montana inventoried the Sundance Ridge and Cube Iron-Silcox Inventoried Roadless Areas (IRAs). The Cube Iron-Silcox IRA (36,955 acres) is located between the adjacent Cataract and Sundance Ridge IRAs. The Sundance Ridge IRA (7,198 acre) is in between the Cube Iron-Silcox IRA and parcels of Montana State Trust Lands. This provides excellent connectivity within the immediate landscape as well as contiguous wildlands connected to the Cabinet Wilderness Area in the Kootenai National Forest to the north and other roadless areas in the Lolo National Forest. In the 1986 Forest Plan, the Forest Service said this area was a large roadless areas that is “distinguished primarily by [its] natural environmental character” and would be managed to provide for recreation activities in a “near-natural setting and for old-growth dependent wildlife species,” and the Forest Service designated this area as essential grizzly bear habit.



Inventoried Roadless Area:	Days in area:	Trail Miles Covered:	Roadless Area Acreage:
Sundance Ridge Inventoried Roadless Area	1 day	5.6 miles (one way)	7,198 acres
Cube Iron-Silcox Roadless Area	1 day	9.2 miles	36,955 acres

Sundance Ridge Inventoried Roadless Area

Although the turn-off to access this remote and unique roadless area is inconspicuous, the Sundance Ridge IRA certainly deserves attention. This IRA offers challenging and ambitious opportunities for day hikes and multi-day backpacks. This IRA possesses ample wild character– everything from the terraced cliff edges along the road leading to the Priscilla Peak trailhead to the lush sedge grasses and huckleberries along the ridge to the subalpine meadows and fir forests directly preceding the rocky ridgeline where the historic Priscilla Peak lookout stands watch paint a picture of a truly rich backcountry experience. Due to the limited human modifications and connectivity provided between adjacent IRAs, Wild Montana's modeling shows that this roadless area ranks in the 90th percentile for wildness and 99th percentile for species intactness. [1]



View from Priscilla Peak looking into the valley below.

[1] Wild Montana's Conservation Value analysis defines the values mentioned above as follows: Wildness: The relative wildness of areas by mapping human modifications such as land use, land cover, distance from roads, and darkness of night skies. Species intactness: Measuring where carnivores of conservation concern (red wolves, gray wolves, mountain lions, lynx, grizzly bears, black bears, wolverine, fisher, black-footed ferrets, and swift foxes) are currently versus historical distributions. Climate importance: Modeled predicted migration paths of species and ecosystems in response to climate change. This model evaluates climate connectivity areas and identifies landscapes where the largest number of species can shift from their current climate types to their future analogs. Connectivity: Areas that have been modeled to fulfill connectivity priorities between protected areas in Montana and surrounding states and provinces.

I. Untrammeled and Natural Quality

Plant and Animal Communities

On the trail to Priscilla Peak, we encountered a whitetail doe and many ground squirrels. Towards the top of the ridge right below the old lookout, we found bear scat close to the trail that appeared older. This area is just southeast of the Cabinet-Yaak and serves as a connectivity corridor for grizzly bears.

Weeds

No weeds were encountered on the inventoried trails.

Water Erosion

No human caused erosion was seen along the inventoried trails in the Sundance Ridge area.

II. Undeveloped Quality

Installations and Developments

When you first pull up to the trailhead for Priscilla Peak/Sundance Ridge, it would be understandable for you to assume there is no trailhead. There are no signs and only a faint trail that curves around the steep hill. The biggest giveaway is a hitch post for horses.

At the very top of the peak is a dilapidated, historic lookout with a fire ring situated on the north side. The trail continues on and links this unique IRA all the way to the northern edge of the Cataract IRA.


Priscilla Peak Lookout.

Along the trail we also observed one old mining apparatus.

Signs

There are very few trail markers and most of them are barely legible.

Priscilla Peak Sign. *Unreadable sign.*

Trail Closure

No trail closures were observed during monitoring.

III. Solitude or Primitive and Unconfined Recreation Quality

Just less than 10 miles off Highway 200, the Sundance Ridge IRA offers anyone who makes the short journey immense feelings of quietude. Even on the established trail to Priscilla Peak and beyond, you are unlikely to run into other visitors. The area encompasses terraced cliffs, alpine springs, recovered wildfire swaths, talus slopes, and rocky ridgelines. The views along the ridge are spectacular in all directions, looking northwest to the Cabinet-Yaak and east to the formidable Mission Mountains. One feels part of a primitive and undisturbed landscape while traversing this IRA.

System Trail Conditions

Once on the trail, if you don't have paper maps or maps downloaded, you might find yourself getting off route. There are very few trail markers, and the trail is overgrown. Especially when you hike out of the old burn area into the dense undergrowth and ridgeline trees, the trail barely cuts through the foliage. There is little evidence of human use outside of several very old horse manure droppings along the trail.

Evidence of Mechanized and Motorized Use on Trails

No evidence of mechanized or motorized use on the trail.

Non-system Trails

No user created trails were observed.

Trailheads

Use of trailheads was documented by recording the number of vehicles, horse trailers and ORV trailers parked at the trailhead. The vehicles Wild Montana arrived in are *not included* in this tally.

Priscilla Peak Trail - 0 vehicles

Encounters with People

No other people were seen on any section of the inventoried trails.

Noise

Even with the decently trafficked dirt road at the trailhead, I heard no traffic and saw no other human developments while on Sundance Ridge.

Visual Intrusions

There is no evidence of logging within the IRA itself. Once you get up higher on the cliffs past the trailhead, you can see an old burn and fire roads to the south. There was a logging operation on Thompson River Road many miles back towards the highway, but not up near the Sundance IRA.

Along the ridgeline trail, we encountered an old 4x4 road that appears to have not been used in many years due to the overgrown vegetation. *See photo on the right.*

Campsites

We encountered two small fire rings along the trail, however they look like they have not been used in many years due to overgrown vegetation. No other camping infrastructure was present.

Sensitive Plants

No sensitive plant species were observed during monitoring.

<u>Cube Iron-Silcox Inventoried Roadless Area</u>

After spending a quiet night camped along the Thompson River, we ventured up the dirt road to the Four Lakes Trailhead nestled amongst a cedar grove. This loop is one of the most popular trails in the area, especially the first section from the trailhead to Cabin Lake. However, the area is still remote and you'll likely only see a handful of people all day. The lake offers an excellent opportunity to camp in all the beauty this IRA has to offer without having to hike multiple days. Traveling farther south towards Cube Iron Mountain, the forest becomes more and more densely populated with abundant sedge grasses and other understory growth. The beargrass guides you along the path towards Cube Iron Pass, where you can make your way up switchback to the saddle before Cube Iron Mountain. It is without a doubt that this section of public lands is diverse and offers an incredibly unique experience to any who take the time to explore.

Beargrass along the Four Lakes Loop.

I. Untrammeled and Natural Quality

This IRA is special for its diversity in forest types. The trail starts out in an old burn scar, traversing along cliff faces and boulder filled slopes to the wooded campsites around Cabin Lake. Climbing up to the basin beyond that is populated with many smaller lakes, you'll find the forest composition changes significantly. Lush sedge grass and thick trees carpet the floor around the lakes. That is until you come upon the start of Cube Iron Pass. Alpine boulder fields give way to the rocky, burned ridgeline of Cube Iron Pass up to the unique rock formations that give Cube Iron Mountain its name. Even within a small radius, you are sure to encounter many types of forest flora.

View of Cube Iron Pass Credit: Chris Sawicki

When beginning up the first climbs of this trail, you'll traverse through several burn areas before reaching Cabin Lake itself. These are smaller swaths of burn that do not extend farther past Cabin Lake into the other basins where the other lakes sit. If you take the trail up to Cube-Iron Pass to summit Cube Iron Mountain, you will find yourself stepping into a large burn area that extends along the southwest side of the pass. This burn area is extensive and can be seen flowing down the majority of the mountain side.

Plant and Animal Communities

We did not have any direct encounters with wildlife, although it is apparent that this is a corridor for many different species. In the muddy edges of the trail, we found tracks from deer, bighorn sheep, moose, and elk. Apart from four-legged wildlife, the birds were plentiful and active, providing us with a constant soundtrack to our adventure. And although unconfirmed, we startled a large animal in the bushes along the basin floor that could either be a moose or a bear. The variety of wildlife in this wild space is an indication of a rich ecosystem that supports this diversity.

Weeds

No weeds were encountered on the inventoried trails.

Water Erosion

No human caused erosion was seen along the inventoried trails.

II. Undeveloped Quality

Installations and Developments

There is a well-maintained, built bridge at the start of the trail and evidence of recent trail maintenance.

Signs

The handful of trail signs and trail improvements hardly impact the experience of exploring this area. All wooden signs in good legible condition.

Trail Closure

No trail closures were observed during monitoring.

III. Solitude or Primitive and Unconfined Recreation Quality

This trail is cherished for its access to Cabin Lake, however, once beyond Cabin Lake, you'll find it difficult to run into anyone else. This is reflected in the change in trail width and the amount of overgrowth of grasses over the trail.

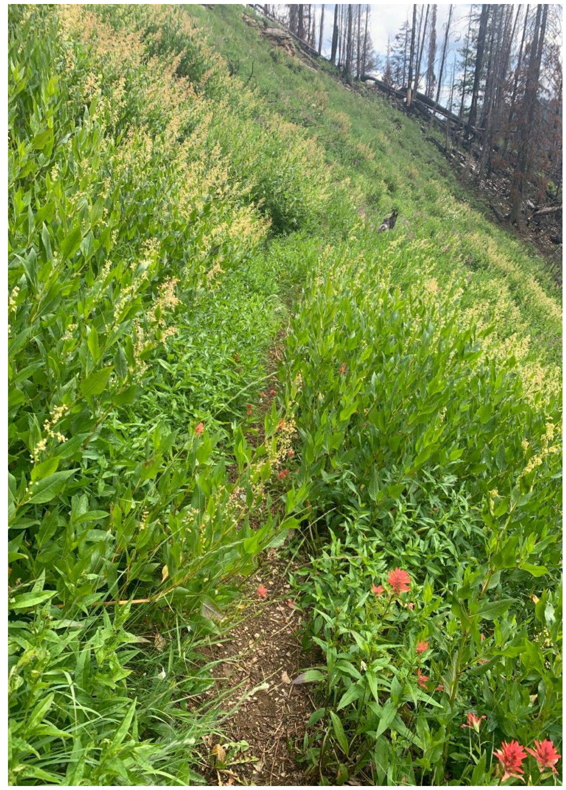
Evidence of Mechanized and Motorized Use on Trails

No evidence of mechanized or motorized use.

System Trail Conditions

While this trail is one of the more well-traveled areas of this IRA, its apparent naturalness is still very much intact. It is clear you are still part of a pristine, wild landscape whilst hiking along the well-established trail. This IRA is close to towns and roads, yet maintains its stillness and undisturbed peace. Once past Cabin Lake, the trail becomes more narrow and overgrown.

Switchback trail to Cube Iron Pass



Further into the IRA, the trail becomes more overgrown.

Non-system Trails

There were a few non-system trails along the main trail that led to water sources. Other than those, no other user created trails were observed.

Trailheads

Use of trailheads was documented by recording the number of vehicles, horse trailers and ORV trailers parked at the trailhead. The vehicles Wild Montana arrived in are *not included* in this tally.

Cabin Lake Trailhead - 3 vehicles.

Encounters with People

We encountered one fisherman at the trailhead and one group of two backpackers at Cabin Lake.

Noise

There are no sightings of towns, nor sounds of trains, planes, or vehicles while exploring in this IRA.

Visual Intrusions

No visual intrusions were observed on the inventoried trails. While you will find evidence of a logging operation farther down the main Thompson River Road heading towards town, you will not see or hear any of the effects of said operation. Within the IRA itself, there is no evidence of past timber activities.

Campsites

There were three established campsites and fire rings at Cabin Lake along with a pit toilet.

Sensitive Plants

No sensitive plant species were observed during monitoring.

Lolo National Forest:

Sliderock/Quigg Inventoried Roadless Area

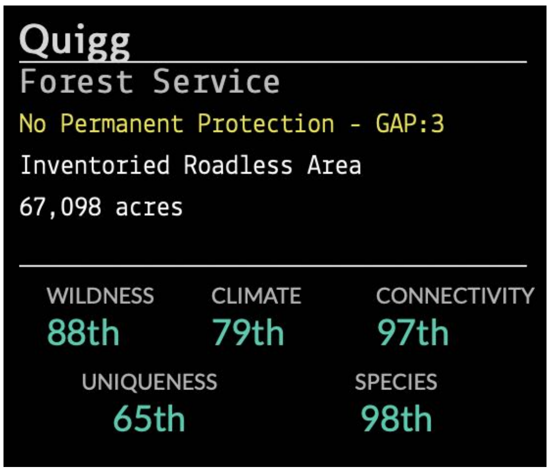
June 2023

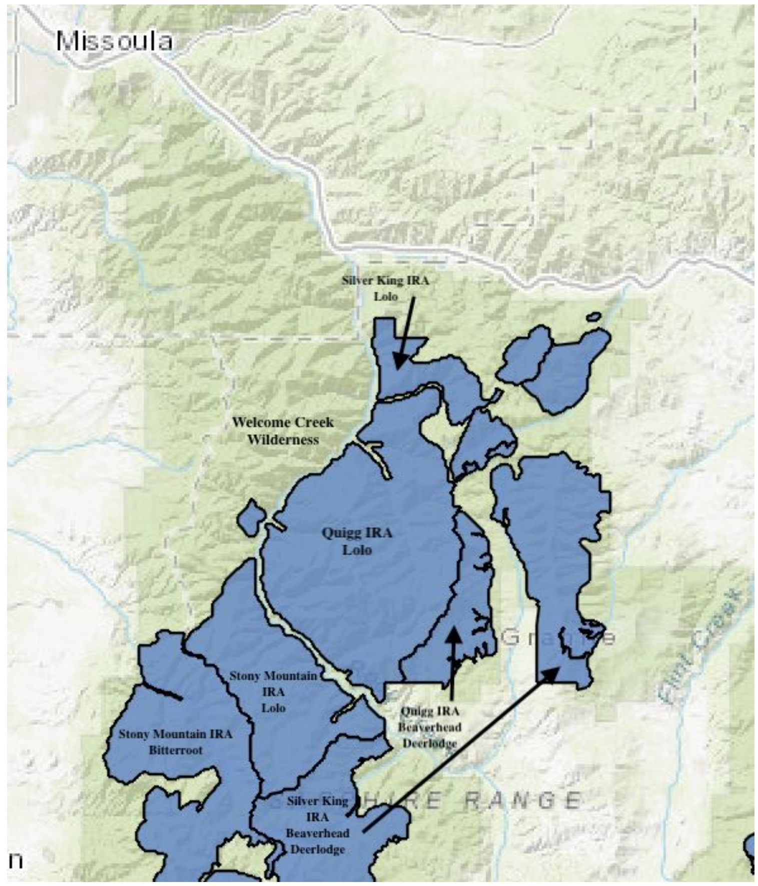
View of Quigg Peak from Butte Cabin Ridge Trail

Summary:

In mid-June 2023, Wild Montana inventoried three areas of the Sliderock/Quigg inventoried roadless area (IRA). We inventoried three main trails in the IRA: *Ranch Creek Trail, Butte Cabin Creek/Quigg Peak Trail to Butte Cabin Ridge Trail, Hogback Ridge Trail*. No other people were on the trails or at the trailhead, and each trail was very overgrown with little sign of human activity (see images below). Therefore, this area demonstrates superb apparent naturalness and provides ample opportunity for solitude and primitive/unconfined recreation. The terrain in this inventoried roadless area is moderately rugged, with many talus fields. The roadless area is adjacent to Rock Creek, a blue ribbon trout stream. The area also provides connectivity between other landscapes on the Bitterroot National Forest, Beaverhead-Deerlodge National

Forest, Welcome Creek Wilderness Area, and a BLM Wilderness Study Area. Under the 1986 Lolo Forest Plan, the Forest Service manages the area as recommended wilderness. There were no signs of motorized or mechanized intrusions. There were also no signs of past timber harvest, mining activities, or private inholdings.





Inventoried Roadless Area:	Days in Area:	Trail Miles Covered:	Roadless Area Acreage:
Quigg (Lolo)	3 days	17.1	67,098 acres

View from Butte Cabin Ridge overlooking Quigg Peak, the Sliderock Roadless Area, and part of the Welcome Creek Wilderness.

I. Untrammeled and Natural Quality

Plant and Animal Communities

Six bighorn sheep lambs and one adult bighorn sheep were seen along the road just before reaching the trailhead for Butte Cabin Creek trail. On the inventoried trails there were no direct or indirect encounters with any mammals. Two hawks were observed on the Butte Cabin Ridge Trail.

Weeds

No weeds were encountered on inventoried trails in the Quigg area.

Water Erosion

No human caused erosion was seen along the inventoried trails in the Quigg area.

II. Undeveloped Quality

<u>Installations and Developments</u>

A total of two developments were recorded during monitoring. One old metal fence post was seen on the Quigg Peak trail. Additionally, a portion of the Butte Cabin Ridge Trail was built into a talus field with an apparent rock wall.

<u>Signs</u>

A total of six signs were observed during monitoring. Signs were primarily at trailheads and trail junctions. All wooden signs in good legible condition.

Trail Closure

No trail closures were observed during monitoring.

III. Solitude or Primitive and Unconfined Recreation Quality

Evidence of Mechanized and Motorized Use on Trails

No evidence of illegal mechanized or motorized use was observed.

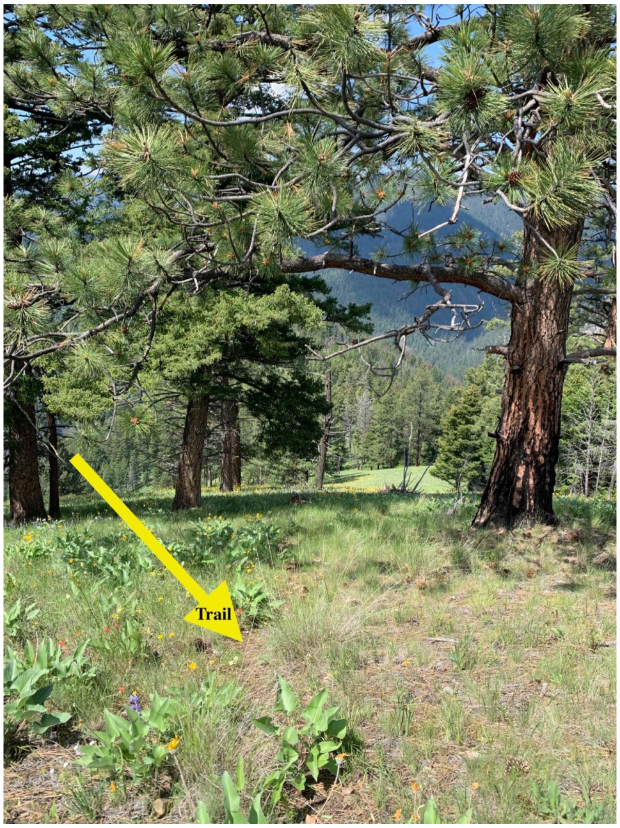
System Trail Conditions

The inventoried trails were overgrown and there were no signs that they are frequently used by humans.

Ranch Creek Trail

Hogback Ridge Trail

Butte Cabin Ridge Trail:



Quigg Peak Trail:

Non-system Trails

No user created trails were observed.

Trailheads

Use of trailheads was documented by recording the number of vehicles, horse trailers and ORV trailers parked at the trailhead. The vehicles Wild Montana arrived in are *not included* in this tally.

Ranch Creek Trail - 0 vehicles
Butte Cabin Creek/Quigg Peak Trail - 0 vehicles
Butte Cabin Ridge Trail - 0 vehicles
Hogback Ridge Trail - 0 vehicles

Encounters with People

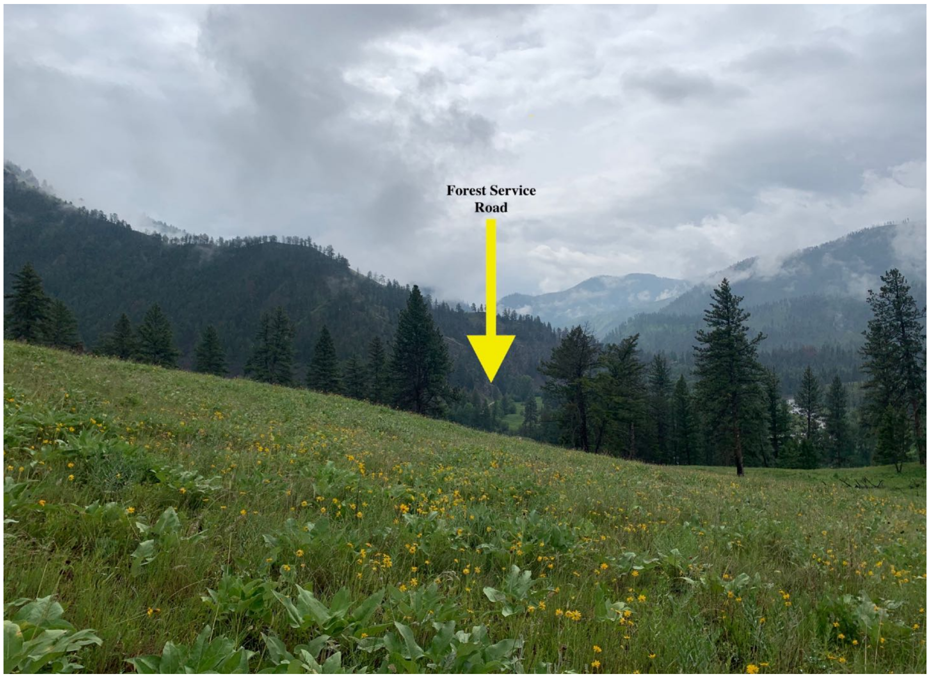
No other people were seen on any of the inventoried trails.

Noise

We heard and saw one airplane and one helicopter overhead on the Butte Cabin Ridge Trail. Additionally for the first mile of the Hogback Ridge trail, you could occasionally see and hear the Forest Service road.

Visual Intrusions

For the first mile of the Hogback Ridge trail, you could occasionally see and hear the Forest Service road and the Hogback Homestead building.



Campsites

One campsite was encountered and was only noticed because of an old fire ring with little evidence of other use observed. The campsite had high opportunity for solitude, as no other camps were seen nearby.

Sensitive Plants

No sensitive plant species were observed during monitoring.

Lolo National Forest

The Great Burn Roadless Areas: Meadow Creek-Upper North Fork, Ward Eagle, Sheep Mountain-Stateline, and Hoodoo Inventoried Roadless Areas.

July–September 2023

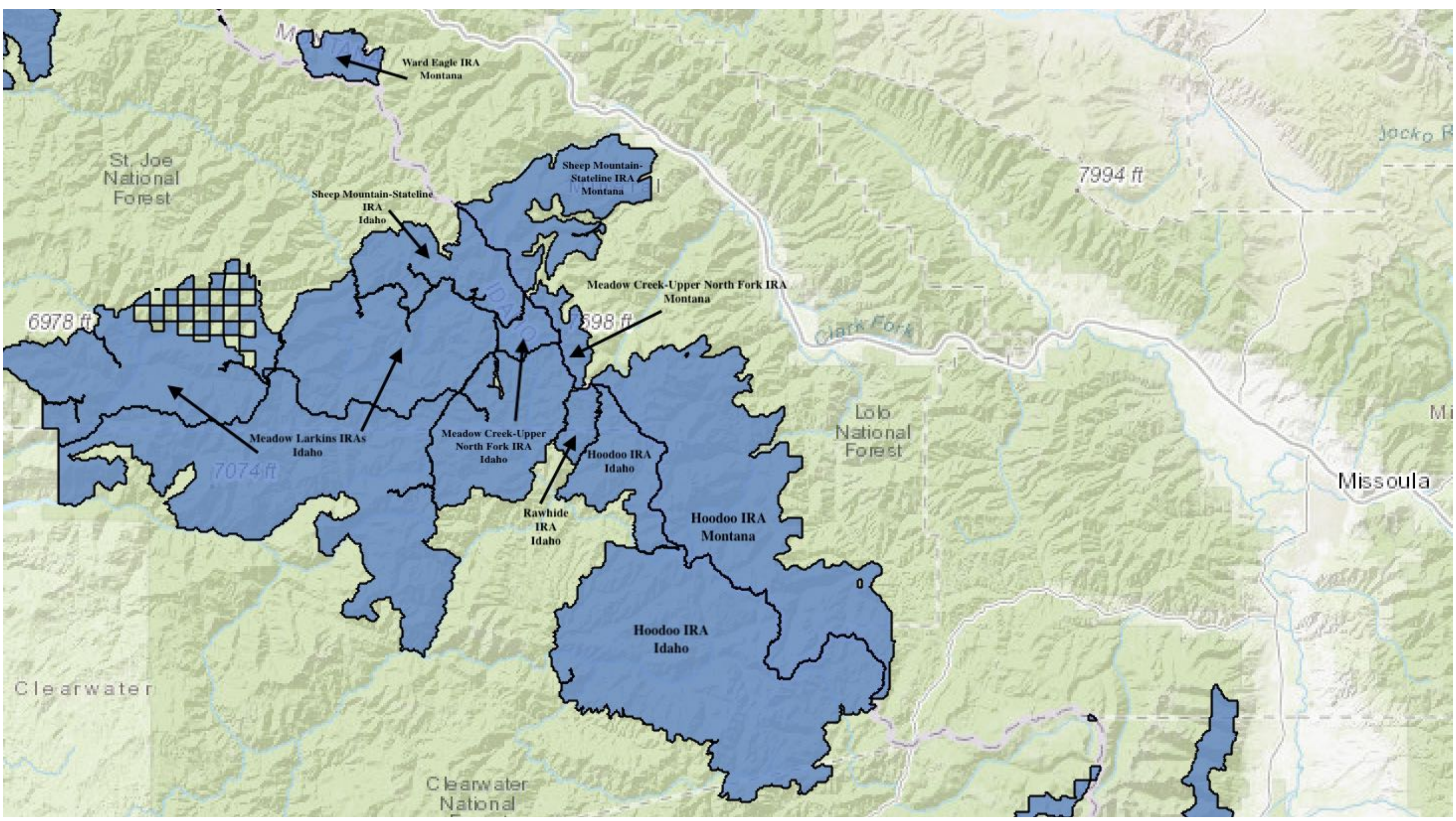
Ridge in the Meadow Creek-Upper North Fork Roadless Area

Summary:

In July through September 2023, Wild Montana inventoried three Inventoried Roadless Areas (IRAs) in the Great Burn landscape of the Lolo National Forest–Meadow Creek-Upper North Fork IRA, Ward Eagle IRA, and the Sheep Mountain-Stateline IRA. As shown in the map below, the roadless areas that comprise the Great Burn ecosystem are connected to additional roadless areas on the Idaho side of the stateline. This creates a larger wild complex that must be taken into consideration as a whole when looking at the incredible values each area holds.

Wild Montana staff and volunteer leaders regularly visit and monitor the natural character and wilderness characteristics of the Hoodoo IRA (also known as the Great Burn Recommended Wilderness), so we did not conduct a focused inventory there this year. Our 2023 inventory effort visited the other inventory roadless areas that make up the Great Burn ecosystem.

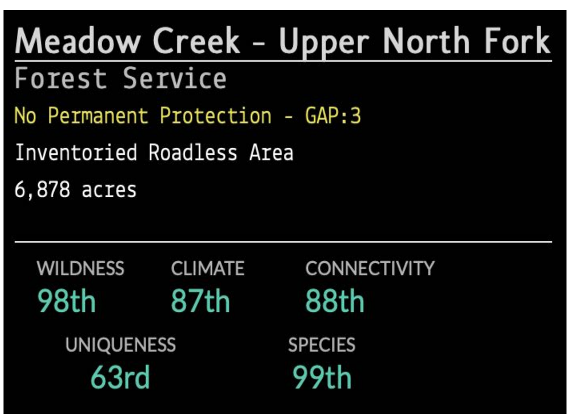
The 1986 Forest Plan designated the Hoodoo IRA as recommended wilderness and for the other IRAs, the Forest Service said these were large roadless areas “distinguished primarily by their natural environmental character” and would be managed to provide for recreation activities in a “near-natural setting and for old-growth dependent wildlife species.”



Inventoried Roadless Area:	Days in area:	Trail Miles Covered:	Roadless Area Acreage:	
			Lolo NF	NPCNF
Meadow Creek-Upper North Fork	3 days	15.4 miles (one way)	6,878	46,438
Ward Eagle	3 days	14 miles (one way)	8,542	N/A
Sheep Mountain-Stateline	1 day	4.7 miles (one way)	37,673	26,926
Hoodoo: This data was collected between 2018–2023 on multiple trips into the IRA.				

Meadow Creek-Upper North Fork Inventoried Roadless Area

The Meadow Creek-Upper North Fork IRA Is bounded on three sides by wild IRAs. The Lolo National Forest portion of the Meadow Creek-Upper North Fork IRA is immediately adjacent to the Idaho Nez Perce-Clearwater National Forest IRA by the same name. There is no road on this state boundary. Together they represent nearly 60,000 acres of public lands that provide high quality wildlife habitat, outstanding opportunities for quiet recreation, and incredible ecosystem connectivity. Further, to the north of the Montana Meadow Creek-Upper North Fork IRA is the 37,673 acre Montana Sheep Mountain Stateline IRA and to the south is the 104,901 acre Hoodoo IRA. Together these IRAs across Idaho and Montana constitute a 231,358 acre wild complex, and it is important to evaluate these areas in combination with one another, even though they are managed by multiple forests, cross state boundaries, and in some cases do have roads between them. Wild Montana's modeling shows this IRA as ranking in the 98th percentile for wildness and 99th percentile for species intactness.



I. Untrammeled and Natural Quality

Hikers possessing Bob Marshall's stamina can travel on foot from one end of the Meadow Creek - Upper North Fork IRA to the other end of this 11,000 acre IRA in a single day, but the relatively small size of this IRA does not account for the diversity found here. This long and skinny inventoried roadless area includes rolling, sinuous ridges straddling the Montana-Idaho border, subalpine lake basins, and deep valleys that hold creeks that capture snowmelt from north-facing basins. There are dark, moist groves of western red cedars, majestic assemblages of mountain hemlock at the higher elevations, thick subalpine fir stands, and large and healthy whitebark pine on the ridges. This tree diversity supports a lot of wildlife. Our inventories encountered or saw sign of mountain goats, pika, red fox, deer, sharp-shinned hawks, Clark's nutcrackers, pileated woodpeckers, wolves, and bobcats. The terrain and topography suggests that this IRA also is wolverine habitat and some of its north facing bowls are good candidates for maternal winter denning habitat. There are likely moose in the lower elevations. The habitat would also support elk, although no sign was detected. Views from this IRA's ridgelines make it clear that there is a tremendously wild, natural landscape surrounding you. Ridgeline after ridgeline cascade away to the south and east, most without any indications of human modification (visible roads, timber treatments, structures, etc.). This largely was true to the north, until this summer when timber harvests were executed right on the northern border of the IRA (Cedar Thom-Cedar South Project), now marring the otherwise wild and natural view to the north where the craggy, above treeline Cabinet Mountains are the furthest visible sight. Trail infrastructure throughout the IRA is minimal and primarily consists of waterbars, and even trail signage is minimal, with major trail intersections, such as the Stateline-St. Joe Lake-Illinois Peak three way trail junction not having any signs.

"*This small IRA punches well above its weight class for naturalness, and it shouldn't come as a surprise that Wild Montana's model puts it in the 98th and 99th percentile respectively for wildness and species richness.*" Wild Montana Organizing Director Erin Clark, on the Meadow Creek-Upper North Fork IRA.

Plant and Animal Communities

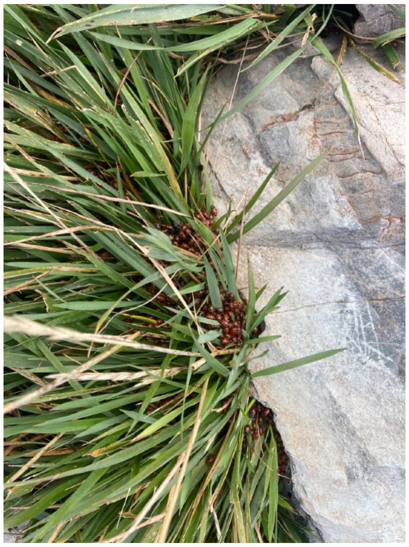
Tree and vegetation health was good, there was evidence of some tree disease, but not beyond a natural/normal level. Very healthy, old whitebark pine present on ridgelines. Old growth hemlock groves. 200+ year old cedars in Trail Creek bottom (age determined by branching pattern).

Whitebark Pine on Trail 169

Water flowing off portions of the Stateline ridgeline in this IRA flow into St. Joe Lake and then into the St. Joe River. From its beginnings at St. Joe Lake, this river is designated wild for roughly 26 miles and then for recreation until being joined by the North Fork St. Joe. It provides habitat for westslope cutthroat trout and bull trout.

Stateline Trail View of St. Joe Lake:

We observed numerous signs of wildlife in our two days of inventory: a mountain goat on Illinois Peak, deer, wolf, bobcat, and pileated woodpecker. We also saw multiple species during the inventory, including pika (heard in nearly every scree slope passed, at least six distinct scree slopes along #738 currently have pika), grouse, frogs, sharp-shinned hawk, dark morph red fox, Clark’s nutcracker, and significant ladybug congregations in the rocks at the top of Illinois Peak.



Ladybugs on Illinois Peak. *Trail Lake Trail Bobcat Track.* *Frog at Oregon Lakes.*

Weeds

No weeds were encountered on inventoried trails in the Meadow Creek-Upper North Fork area, although spotted knapweed was present at several trailheads, such as the Trail Lake Trailhead. There were few patches of non-native clover along Stateline #738 and Trail Creek Trail #156. There is currently work being done by organizations such as the Great Burn Conservation Alliance to reduce the spread of invasive species/weeds in this roadless area complex. Please see the report submitted by the Great Burn Conservation Alliance on campsites and noxious weeds during the assessment phase of this forest plan revision process

Water Erosion

There was significant rainfall the day and week prior to our inventory. Wooden water bars were present on all trails. All trails could be improved to reduce erosion.

II. Undeveloped Quality

Installations and Developments

Hoodoo Pass trailhead, just outside of the IRA, contains infrastructure for horses (hitch rails, no corral, ample trailer parking and space for a trailer to turn around). Further on Illinois Peak, there are some remnants from an old fire lookout.

There were no bridges on Trail Lake Trail #156 creek crossings, but a primitive log bridge is present at the first crossing and a human hewn wooden block is at the second crossing. The Oregon Lakes trail had three different bridge features.

Trail Lake Trail Wooden Block.

Log bridge on the Oregon Lakes Trail.

Signs

Trail signage is minimal, with major trail intersections, such as the Stateline-St. Joe Lake-Illinois Peak three-way trail junction not having any signs. We only encountered one trail sign on the entire stretch of Stateline Trail between Hoodoo Pass and Illinois Peak. The limited signage requires users to use their navigation skills. Signs along the Oregon Lakes trail were primarily at the lakes. All signs were wooden and in good legible condition.

Trail Closures

No trail closures were observed during monitoring.

III. Solitude or Primitive and Unconfined Recreation Quality

Although the Hoodoo Pass trailhead often will fill on weekends July through August, very few users choose to travel north on the Stateline trail and most will head into the Lolo (and Region 1)'s largest IRA, Hoodoo, by traveling south on the Stateline trail. Word is not out that the spectacular features found in the Hoodoo IRA are also available in the Meadow Creek - Upper North Fork IRA. That means that the incredible opportunities for primitive recreation available in the Hoodoo can be found there as well, but with considerably higher levels of solitude. The summit register at Illinois Peak demonstrated that there is regular visitation, but rarely more than one party visiting per day. The mellow ridgeline and subalpine grassy meadows along the Stateline trail provide outstanding opportunities for off trail rambles, even if just to stop to enjoy a short break and watch for wildlife. Evidence of horse use was minimal on this IRA's trails, but the Hoodoo Pass trailhead is well equipped for stock and stock trailers and most of the trails in this IRA are well suited for travel by horseback.

The Stateline trail through this IRA is part of the Idaho Centennial Trail (ICT) and is used by long-distance hikers traveling the ICT. This IRA is bounded by other IRAs known for their outstanding solitude and primitive recreation opportunities (Hoodoo IRA (202,000 acres) and the Nez Perce-Clearwater National Forest Meadow Creek–Upper North Fork IRA (46,438 acres)) and there is excellent trail access from this IRA into those IRAs.

<u>Evidence of Mechanized and Motorized Use on Trails</u>

We observed evidence of motorized trespass on Trail #169 (approx. one mile, likely a motorbike) and the Stateline #738 (less than a half mile).

<u>System Trail Conditions</u>

The inventoried area contained high naturalness and very little sign of human impact. There is no history of timber harvest or significant forest management on this IRA. There were obvious clearing of trees to maintain the trails, but also several instances of recent hand-cutting of trees for non-management purposes (cutting of trees on the edge of a small burn below Graves Peak and unnecessary felling of trees at two campsites on Trail Lake). There has been some trail reconstruction on Trail Lake Trail #156 that utilized rocks in a wire mesh frame to stabilize the bank. This minor visual eyesore could be easily improved.

Trail Lake Trail

Stateline Trail view of Graves Peak

Oregon Lakes Trail.

<u>Non-system Trails</u>

No user created trails were observed. User created trails on Illinois Peak have been rehabilitated, and while they can still be detected there was no indication of current use. Some trail braiding present below Graves Peak on the Stateline trail.

Trailheads

Use of trailheads was documented by recording the number of vehicles, horse trailers and ORV trailers parked at the trailhead. The vehicles Wild Montana arrived in are *not included* in this tally.

Freezeout Pass - 0 vehicles
Trail Lake Trail - 0 vehicles
Hoodoo Pass - 2 vehicles. Those users presumably were recreating in the Hoodoo IRA, as it had recently rained and there were no footprints on the Stateline trail north of Hoodoo Pass.
Oregon Lakes: 1 vehicle

Encounters with People

Very few recreationists opt to head into this IRA. During three days of inventory surveys, no other people were seen on any of the inventoried trails. A review of the Illinois Peak register suggested that half a dozen people visit this peak each week throughout the summer hiking season.

Noise

We heard and saw approximately two planes per hour. There was an active logging operation happening adjacent to the Freezeout Pass trailhead that could be heard for a mile along the Trout Creek Driveway trail (#169) towards Illinois Peak.

Visual Intrusions

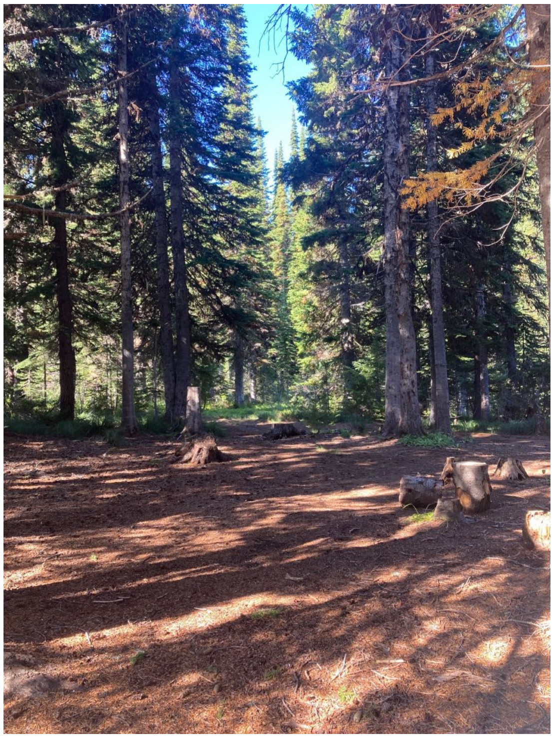
There are three very new timber harvest units visible to the north. The Cedar Thom #920 and 921 units are larger than 50 acres, have visible sky lining scars, and from Trail #169 it appears that almost no understory vegetation survived.

Trail 169, Northeast View: *Trail 169, North View, Cedar Thom Units #920 and 921*:

Campsites

We removed one fire ring on Trail 169 and one cairn on the Stateline Trail while conducting our inventory, and left fire rings at campsites intact.

Trail 169 Campsite *Trail Lake Trail Campsite*



Stateline Trail Campsite

We encountered two campsites and fire rings at Lower Oregon Lake.

Ward Eagle Inventoried Roadless Area

This spectacularly lush and diverse IRA provides excellent opportunities for solitude and recreation in what seems like a never-ending wild landscape. This IRA boasts an old-growth cedar forest replete with mossy banks along quiet streams and talus fields with the chirps of pikas to accompany your travels.

Looking down onto Hub Lake at the base of Ward Peak.

I. Untrammeled and Natural Quality

Besides the roads leading to this IRA's trailheads, it would be unlikely to find any sort of human disturbance within the Ward-Eagle IRA. There are no visuals of roads, clearcuts, or towns, nor auditory machinery or vehicles. The only disturbance noted was an airplane flying overhead once a day at the same exact time. At first, it was difficult to tell what the noise was due to how faint it was. It does not dramatically impact the feeling of disconnection from the human developed world.

Plant and Animal Communities

It would be surprising if anyone visited this IRA without hearing and/or seeing the pika that dominate the talus fields. These small creatures, along with marmots, can be heard throughout this IRA. We did not see any larger wildlife species during the inventory, but there was evidence of wolves, mountain goats, deer, and bears (scat, tree markings, etc.).

Cedar Tree along the trail to Hub & Hazel Lakes.

Weeds

There is some work being done currently by organizations such as the Great Burn Conservation Alliance to reduce the spread of invasive species/weeds in these areas, including Oxeye Daisy, which was spotted at the trailhead of Hub and Hazel Lakes. Please see the report submitted by the Great Burn Conservation Alliance on campsites and noxious weeds during the assessment phase of this forest plan revision process.

Water Erosion

Hub and Hazel Lakes Trail and Crystal Lake Trail would benefit from basic trail maintenance. Water bars are needed to reduce water erosion.

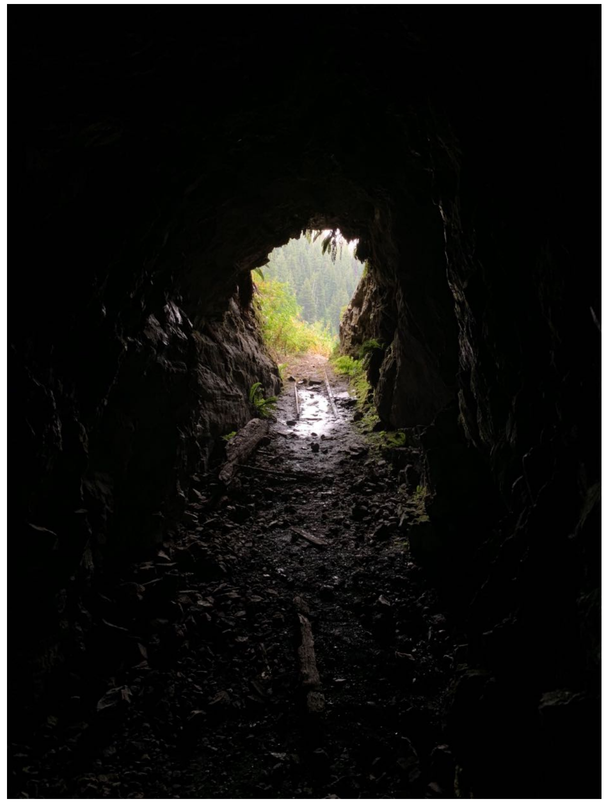
Existing water bar on Hub & Hazel Lake Trail.

II. Undeveloped Quality

Installations and Developments

Since this area is an IRA, there is no evidence of road construction for commercial logging within the area, however there are some operations on the landscape adjacent to the IRA.

Ward Eagle has a long mining history. At Hub Lake, a trail ascends to an old mining shaft that tunnels into the heart of Ward Peak. This tunnel serves as a reminder of the historic uses of this landscape.



Signs

Although we encountered wooden signs along the trail, some were barely legible or in poor condition.

Trail Closures

No trail closures were observed during monitoring.

III. Solitude or Primitive and Unconfined Recreation Quality

Even though some of these trails within this IRA are becoming more trafficked, it is incredibly easy to find space for solitude away from human development and other users in this IRA. The farther you travel into the IRA, the closer you are to quietude. This IRA offers visitors peace amongst the landscape and wildlife.

As one Wilderness Walk participant, Sarah Bates, eloquently put, "The opportunity to traverse different forest types, enjoy quiet and abundant water, and end with a spectacular vista at a beautiful lake surrounded by cliffs - it just doesn't get much better."

Evidence of Mechanized and Motorized Use on Trails

We encountered no evidence of mechanized or motorized use.

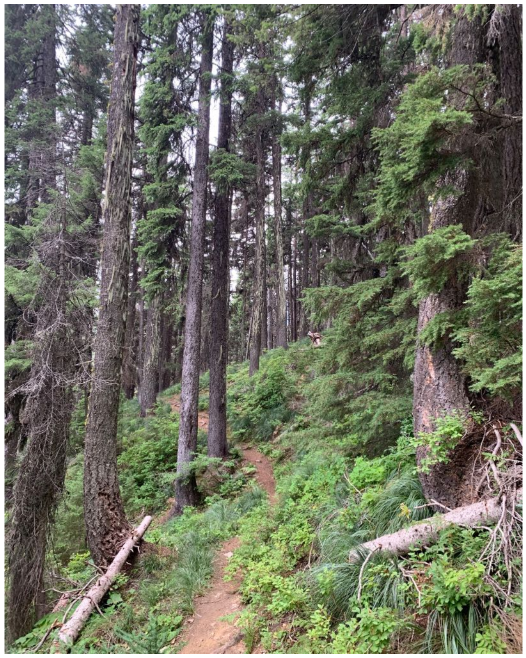
System Trail Conditions

The inventoried trails were overgrown and faint in spots, with the trail to Hub & Hazel Lakes being the most well defined.

Crystal Lake Overgrown Trail:

Up Up Ridge Trail:

Hub & Hazel Trail:



Non-system Trails

In the Ward-Eagle IRA, there are a handful of non-system trails, or social trails. These trails do not travel far and are usually in place to reach a water source near the trail.

Social trail to water source.

Trailheads

Use of trailheads was documented by recording the number of vehicles, horse trailers, and ORV trailers parked at the trailhead. The vehicles Wild Montana arrived in are *not included* in this tally.

Crystal Lake Trailhead - 1 vehicle

Up Up Trailhead - 0 vehicles.

Hub & Hazel Lakes Trailhead - 2 vehicles.

Encounters with People

We saw one other hiking party of two hikers and their dog at Hub and Hazel Lakes and one fisherman at Crystal Lake.

Noise

We heard an airplane once a day, approximately at the same time at 12:42 p.m..

Visual Intrusions

The topography of this IRA is moderately rugged with talus fields, cliffs, and alpine lakes. During our time in the IRA, there were no visuals of roads, towns, or vegetation treatments.

Campsites

At Hub & Hazel Lakes we observed one campsite with a makeshift corral and an additional campsite near the lake with a campfire ring.

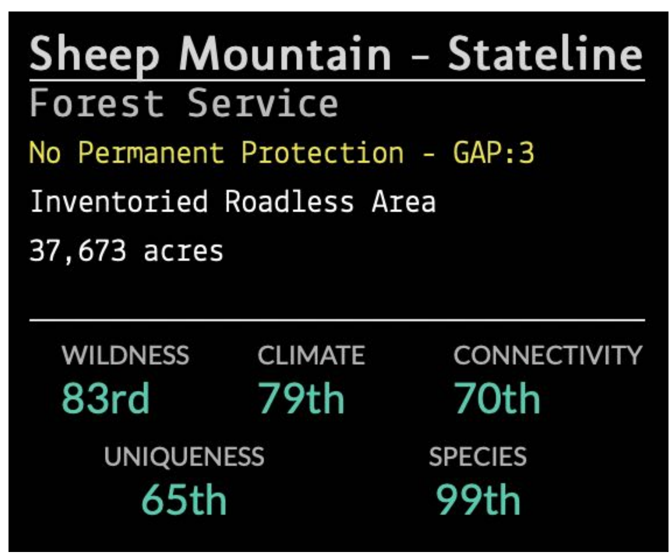
There was one established campsite at Crystal Lake.

Sheep Mountain - Stateline Inventoried Roadless Area

The Lolo National Forest portion of the Sheep Mountain-Stateline IRA (37,673 acres) is immediately adjacent to the Idaho Nez Perce-Clearwater National Forest IRA (26,925 acres) by the same name. Further, the Meadow Creek-Upper North Fork IRAs discussed above are directly adjacent to the south. Like the Meadow Creek-Upper North Fork IRA, it is important to think of this IRA in the context of the larger complex it is a part of.

Stateline Trail looking into the Idaho IRA:

Bonanza Lakes Trail looking out into the Sheep Mountain - Stateline IRA



I. Untrammeled and Natural Quality

Plant and Animal Communities

While hiking through talus fields on the Stateline Trail, we encountered pika and saw one hawk. On the Bonanza Gulch Trail leading to the lakes, we saw evidence of deer.

Weeds

No weeds were encountered on inventoried trails in this area. There is currently work being done by organizations such as the Great Burn Conservation Alliance to reduce the spread of invasive species/weeds in this roadless area complex and has found St. John's wort near Bonanza Lakes. Please see the report submitted by the Great Burn Conservation Alliance on campsites and noxious weeds during the assessment phase of this forest plan revision process.

Water Erosion

No water erosion on this IRAs trails was observed.

II. Undeveloped Quality

Signs

Signs were primarily at trailheads and trail junctions. All wooden signs were in good legible condition. There is a sign and degraded rock cairn at the Stateline Trail junction for Bonanza Lakes:

Trail Closures

No trail closures were observed during monitoring.

III. Solitude or Primitive and Unconfined Recreation Quality

Upper Bonanza Lake in the Sheep Mountain IRA

Evidence of Mechanized and Motorized Use on Trails

No evidence of illegal mechanized or motorized use was observed.

System Trail Conditions

The Stateline Trail #738 in this IRA is well maintained. Bonanza Gulch Trail #616 from the Stateline Trail to the lakes as well as the Bonanza Gulch Trail #616 from Forest Service Road 7763 to the lakes were more overgrown and did not appear to get significant use.

Stateline Trail near Cascade Pass:

More Overgrown Bonanza Lake Trail:

Non-system Trails

No user created trails were observed.

Trailheads

Use of trailheads was documented by recording the number of vehicles, horse trailers and ORV trailers parked at the trailhead. The vehicles Wild Montana arrived in are *not included* in this tally.

Cascade Pass - 2 vehicles. Saw the owners of one vehicle heading south on the trail toward the Meadow Creek- Upper North Fork IRA.

Encounters with People

We encountered one group of two backpackers on the trail to Bonanza Lakes. They camped at Lower Bonanza Lake. We also encountered one hiker at Upper Bonanza Lake.

Noise

We did not hear any airplanes or vehicle noises.

Visual Intrusions

Within the IRA, there was no evidence of vegetation treatments. From a high point, looking southeast towards National Forest Road 320 and non-IRA forest lands, one could see recent vegetation treatment units:

Campsites

There was evidence of camping off the Stateline trail due to the trampled vegetation and obvious sites where tents have been. Further, at each of the Bonanza Lakes, there was an established campsite.

Campsite on the Stateline Trail:

Campsite at Upper Bonanza Lake:

Campsite at Lower Bonanza:

Hoodoo Inventoried Roadless Area

The Hoodoo Inventoried Roadless Area is the largest roadless and unprotected landscape managed by the USFS in Region One. Spanning elevations of 3,000 to 8,000 feet, this roadless area's landscapes are a mosaic of old-growth forests, lush meadows, alpine tundra, dramatic cliff faces, and crystal-clear lakes and streams. In short, a haven for fish and wildlife and a dreamland for backcountry travelers. National Geographic has called the Hoodoo landscape a "gem of wild beauty" and a "quintessential wilderness". For several decades, the Arthur Carhartt National Wilderness Training Center and Ninemile Wildlands Training Center have used the Hoodoo IRA as one of American's premier proving grounds for wilderness skills trainings.

The Hoodoo IRA is often referred to as being gem-like given its summer emerald and sapphire colors.

The Hoodoo IRA is used and beloved by families and individuals for hiking, travel with stock, backpacking, great fishing and hunting, camping, solitude and wildlife viewing. This IRA contains over 50 miles of the Idaho-Montana Stateline trail (also known as the Idaho Centennial Trail) between Granite Pass and Hoodoo Pass. The Hoodoo area is seemingly remote, yet very close to Missoula and the 700,000+ residents of the Spokane and Coeur d'Alene metro area. You can leave your home in the morning and by afternoon experience alpine lakes, pristine wilderness, open ridges with amazing vistas, and solitude. For these areas, the Hoodoo is their premier backyard wildland. Trailheads will with Washington and Idaho plates all summer long.

The Hoodoo Roadless Area is co-managed by the Nez Perce-Clearwater and Lolo National Forests. To the north of the Hoodoo IRA are the Idaho and Montana Meadow Creek-Upper North Fork IRAs. Together these IRAs across Idaho and Montana constitute a 231,358 acre wild complex, and it is important to evaluate these areas in combination with one another, even though they are managed by multiple forests, cross state boundaries, and in some cases do have roads between them. Wild Montana's modeling shows this IRA as ranking in the 93rd percentile for wildness and 99th percentile for species intactness.

There are large and passionate constituencies of recreationists in Montana, Idaho, and, even, eastern Washington that value the quality of human-powered experience they can have in the Great Burn. These users are eager to support protections for the Great Burn and to make sure the next bill that proposes Wilderness designation for the Great Burn gets across the finish line.

This report contains inventories for over 250,000 acres of wild, unroaded lands on the Montana-Idaho border managed by the Lolo National Forest. The Hoodoo IRA represents the beating heart of this entire roadless complex.

The Hoodoo roadless area exhibits different spectacular colors in the fall. Here Heart Lake dazzles in reds, oranges, and yellows in late September.

I. Untrammeled and Natural Quality

The Hoodoo IRA has received one of the highest wilderness ratings of any area managed by the Forest Service nationwide, and has been recommending that Congress designate the area as Wilderness since the 1970s. Recently, Appendix E of Nez Perce-Clearwater DEIS emphasized that "the outstanding scenery, the variety and abundance of wildlife species (elk, black bears, mountain goats, and moose) and the high quality westslope cutthroat trout fishery in Idaho are major attractions." These qualities are all abundantly present and true on the Montana side of the roadless area as well.

This area was subject to the impacts of the history-making and shaping fires of 1910 and since that time this area has been allowed to recover without the trammeling of humans, with the exception of trail maintenance. For this reason, the Hoodoo IRA is an incredible natural laboratory demonstrating what natural regeneration and recovery from severe and large-scale fire looks like after a century. This laboratory supports diverse and healthy populations of plants and animals, while providing truly superlative backcountry recreation opportunities.

The history of fire, past and recent, is visible throughout the Hoodoo roadless area. The severe fires of 1910 created the conditions that support the area's alpine vegetation communities.

> *"The Great Burn may be especially valuable for scientific study and wildland education precisely because of its freedom from human influences. Because of the 1910 fire this austere landscape holds awareness to questions about natural plant succession… I sat alone on the top of a remote, windblown peak deep within the Great Burn and I wondered. I wondered what it is that really sets this land before me apart from other wild places. Below me stretched a "graveyard" of ghost-white snags–remnants of the great fire. Images came to mind of cascading waterfalls, clear mountain lakes nestled in deep cirques, blazing yellow larch in the fall, crimson heather adorning the slopes of glacial headwalls, the striking subalpine tundra of high, open ridges, of elk summering in lush hanging valleys, of goats hopping along the sheer rock face of Shale Mountain. Then I knew. The Great Burn, unlike so many wildernesses where one looks down on towns, farms, and roads, induces a feeling of total wilderness. I've had the same feeling in the middle of the Bob Marshall."* - Bill Cunningham, The Great Burn, Up From The Ashes

Plant and Animal Communities

The Hoodoo roadless area contains an intact, healthy assemblage of plant and animal communities that is one of the most diverse in western Montana. In compiling this report we are going to focus only on a few species of particular note, but this area is renowned for its western red cedar refugia, ridgeline western hemlock stands, abundant moose population, and so much more that we don't address here, such as the wolves we heard howling while camped at French Lake.

A moose eats lakeside vegetation at one of the Hoodoo roadless area's alpine lakes. Credit: Brian Christianson

Mountain Goats

There are two herds of mountain goats that utilize habitat in the Hoodoo. One herd primarily resides on the Nez Perce-Clearwater NF managed portion of the area, around the Blacklead area. Idaho Fish & Game's monitoring of this herd shows significant declines (possibly as high as 80%) in the last two decades,[2] which they have indicated may be due to illegal winter motorized use happening in this area.

[2] Nez Perce-Clearwater DEIS, p 3.2.3.4-31.

A mountain goat below the Stateline in summer. Credit: Brian Christianson

Another herd resides primarily on the Montana side, wintering near Landowner Mountain and dispersing in the summer to alpine lake basins along the Stateline trail. Montana Fish, Wildlife, and Parks has reported more stability in this population than the Blacklead herd, but this population is currently experiencing increasing human impact, especially around Heart, Pearl, and Dalton Lakes and the adjacent portions of the Stateline trail.

Declines in native mountain goat populations across western Montana and throughout Idaho warrant listing the mountain goat as a Species of Conservation Concern. We were pleased to see the Regional Forester include the mountain goat in the Lolo's revised Species of Conservation Concern list.

Wolverine

The wolverine is pending listing as a proposed threatened or endangered species by the U.S. Fish and Wildlife Service. The Hoodoo IRA provides unique, high-quality habitat worthy of special consideration.

Population surveys have not been conducted in the Hoodoo, but wolverine are present and modeling has shown that this area is an important connectivity area and meets requirements for maternal denning. The presence of persistent spring snowpack is a necessary component of wolverine habitat and the Copeland et al. model utilized by the Nez Perce-Clearwater NF in their forest planning analysis identified the high elevation zones along the Idaho-Montana border as having persistent snowpack in at least five years out of seven. Research has shown that wolverine exhibit strong avoidance of both motorized and non-motorized winter recreation.[3]

These images depict the persistent, long lasting snowpack that makes this outstanding wolverine maternal habitat. Left photo: Snowpack in early September at Straight Lake. Right photo: Stateline above Heart Lake in July.

Grizzly Bears

While this portion of the Northern Bitterroot Range does not currently support a resident population of grizzly bears, this area is currently important for habitat connectivity between the Bitterroot Ecosystem and Northern Continental Divide Ecosystem grizzly bear recovery units.

In the fall of 2007, a grizzly bear was shot by a black bear hunter in the Kelly Creek area on the Idaho side of the Hoodoo IRA. The bear was genetically identified as having originated in the Selkirk Mountain population

[3] Heinemeyer et al., 2019. Wolverines in winter: indirect habitat loss and functional responses to backcountry recreation. Ecosphere 10(2)e:02611. 10.1002/ecs2.2611.

of North Idaho.[4] It is likely only a matter of time, probably within the next two to three decades, that grizzly bears will again reside in or regularly pass through the Hoodoo. Management of the Hoodoo as recommended Wilderness provides habitat security and meets habitat and management requirements as outlined for Bear Management Units (BMUs) by the Interagency Grizzly Bear Management Team.

Connectivity

The large, unroaded, high quality, high elevation habitat provided by the Great Burn not only provides critical habitat for resident wildlife populations, but also serves an important role as a connectivity zone between other large, wild areas in this region. The Hoodoo is the largest recommended wilderness and roadless area in Region 1. Opportunities to protect landscapes of this scale, and in a position to link other large roadless areas, are few across our region.

The Hoodoo area contains highly mosaiced habitat, and outstanding connectivity to other wild areas in all directions. The Cabinet Mountains are visible to the north, and the enormous Selway-Bitterroot Frank Church River of No Return Wilderness to the south.

Weeds

There is currently work being done by organizations such as the Great Burn Conservation Alliance to reduce the spread of invasive species/weeds in this roadless area complex largely focused on John's wort, Canada thistle, knapweed, and Oxeye Daisy. The group frequents the sub-alpine meadows around Pearl Lake. Please see the report submitted by the Great Burn Conservation Alliance on campsites and noxious weeds during the assessment phase of this forest plan revision process.

Water Erosion

There is a wide range of trail maintenance levels throughout the Hoodoo IRA, which is not surprising given the sheer number of miles of trail within the IRA. As such, some trails are well maintained to manage water, others are not. At this time, we do not have an adequate analysis of enough trails of the IRA to provide specific examples of trails needing attention. The high traffic areas of the Hoodoo IRA (Heart Lake loop, trails out of Clearwater Crossing) all have installed structures (bridges, boardwalks, etc.) to manage water erosion and reduce human impacts to moist soils.

II. Undeveloped Quality

Human use of the Hoodoo IRA goes back centuries, and there has been extensive use since colonial settlement. There are historical cabins, mine sites, cultural sites, and more within this expansive IRA. The historic Cedar Creek Mine District is adjacent to the Hoodoo IRA and there

[4] Servheen, et al, *A Sampling of Wildlife Use in Relation to Structure Variables for Bridges and Culverts Under I-90 between Alberton and St. Regis, Montana,* 2004.

are several historic mining sites within the roadless area. In spite of the range of developments and installations, this area is very wild and undeveloped.

Even though this area around Cedar Log Lakes experienced historical mining use, it still is predominantly wild and feels undeveloped.

<u>Installations and Developments</u>

Many of the Hoodoo IRAs trailheads, which are located on the perimeter of the IRA, have been significantly developed and feature well-maintained pit toilets, hitch rails and other infrastructure for stock, trail signage, and extensive parking. This is true not only of very accessible trailheads like Heart Lake, but also more remote trailheads like Schley.

The Heart Lake trailhead is one of the most popular trailheads on the Lolo National Forest. Heavy use at Heart, Pearl and Dalton Lakes is creating a range of resource issues, many of which could be alleviated with public education, but has also been partially addressed through installations and developments. This high use area receives both overnight and day use. Dispersed camping in sensitive areas (ie. Dalton Lake inlet), trash, human waste, and mountain goat/human interactions are just some of the management issues the area faces. In order to minimize issues related to human-waste at Heart and Pearl Lakes, two low-profile toilets have been installed at Heart Lake for use by overnight backcountry campers, as well as day hikers. This infrastructure has positively reduced waste issues in this high-use area.

All three lakes – Heart, Pearl, and Dalton – have campsites near their shorelines. As a result, the three lakes are absorbing most of the overnight use in the area. We anticipate an increase in user impacts, such as newly created fire rings and impacts from spike camps, in the more primitive area that surrounds the Heart Lake high use area in the coming decade. The kiosks discussing camping in mountain goat country at the Heart Lake Trailhead and Heart Lake are a good start to public education.

Beyond the Heart Lake area, most of the named lakes within the Hoodoo IRA have campsites, many of which include primitive campfire rings and primitive log stools.

Major stream and creek crossings in the Hoodoo IRA typically have wooden bridges. The Heart Lake Trail also includes boardwalks to minimize trail erosion and impacts through several wet and marshy sections. A significant bridge is present near the Clearwater Crossing Trailhead to allow foot and stock traffic to cross the West Fork of Fish Creek.

Signs

There is regular trail signage throughout the Hoodoo IRA, mainly marking junctures and trail intersections. Most signage is wooden and complies with Forest Service standards.

Trail Closures

The West Fork Fish Creek bridge was recently replaced, which reopened the West Fork of Fish Creek trail to foot traffic, which was not possible during late spring and early summer when the creek was running high. The previous burned in the 2015 West Fork Fish Creek Fire.

Wild Montana previously distributed a map that depicted a trail that connected Pearl Lake with Lightning Peak. This trail had not been maintained, however, and was no longer discernible on the ground. We no longer distribute this map, but understand that this trail may be in the process of being reconstructed.

III. Solitude or Primitive and Unconfined Recreation Quality

In the last decade there has been much discussion and awareness about growing recreational use of the Hoodoo IRA, but it is still very easy to find and relish in solitude in this IRA. Our Organizing Director spent four days in the roadless area backpacking a large loop from Clearwater Crossing to the Stateline to Fish Lake and back during Labor Day weekend. She did not share a campsite with a single other party, and only encountered two other parties in four full days on the trail (two individuals at Straight Creek falls and a party camping on the other side of Goose Lake). This was on a holiday weekend with outstanding weather.

While the growing popularity of the Heart Lake loop may benefit from additional management and public education, even this 'high-use' area provides an important backcountry and unconfined recreation opportunity in this area, and the amount of use received does not preclude it from consideration for Wilderness designation.

The wilderness evaluation criteria regarding recreation evaluates the degree to which the area has outstanding opportunities for solitude or for a primitive and unconfined type of recreation. The word "or" means that an area only has to possess one or the other. The area does not have to possess outstanding opportunities for both elements, nor does it need to have outstanding opportunities on every acre.

Backcountry users who want to experience stunning alpine lakes and a high probability of wildlife sightings find what they are looking for in the Hoodoo roadless area, likely without encountering any other people.

> *It is difficult to imagine regretting the decision to protect the Great Burn in 100 years. A protected Great Burn will ensure that future generations of hunters, anglers, hikes, skiers, and backpackers will be able to enjoy the rich experience of recreating in an intact ecosystem. - Brian Christianson, Missoula Current, April 2020*

Evidence of Mechanized and Motorized Use on Trails

The Lolo National Forest manages the Hoodoo IRA as closed to all motorized recreation use, including over-snow vehicles (OSVs). The 2012 Clearwater Travel Management Plan closed the Idaho side of the recommended wilderness area to all motorized *and* all mechanized (i.e. mountain bike) recreation use. In spite of this, Wild Montana has regularly seen and received reports of mechanized and motorized use on Hoodoo IRA trails managed by the Lolo.

Snowmobiles

Illegal winter OSV use currently occurs in the Great Burn—mostly by 'extreme users' who use the most modern equipment to 'high mark' in the backcountry. Increasingly there is evidence of snow bike use in the Great Burn as well. In 2019, Wild Montana members hiking near Kid Lake found an abandoned snowmobile from the previous winter. The owner must have known that snowmobiles were not allowed in this area, because all VIN and serial numbers had been removed from the snowmobile before it was abandoned. Wild Montana and the Great Burn Conservation Alliance attempted to arrange pack stock to remove the snowmobile, but it was too difficult to saw the machine into pieces and in October 2019 the Nez Perce-Clearwater National Forest removed it by helicopter. In March 2020, Wild Montana's Organizing Director backcountry skied into the Granite Peak area and found snow bike tracks on both the Lolo and Nez Perce-Clearwater sides of this peak. From Granite Peak, snowmobile and snowbike tracks in the drainage below were visible all the way to Cache Saddle. Illegal winter use has been reported to us in the Hoodoo Pass area as well.

Mountain Bikes

There is increasing use by mountain bikes in the Hoodoo IRA. Originally this use was concentrated on the Stateline trail and around Heart Lake. There is now, however, frequent evidence of mountain bike use from the Clearwater Crossing trailhead. Recent efforts have been made to improve signage at trails indicating that mechanized use is not allowed, such as at the Hoodoo Pass trail on the Stateline. Use continues to occur, however.

A Wild Montana staff member encountered two mountain bikers on the Stateline Trail above Hidden Lake on July 25, 2020. Both bikers denied seeing the 'No Mechanized Vehicles' sign at the Hoodoo Pass trailhead where they began their ride, declined to return to the trailhead, and continued their ride with the intention of descending to Heart Lake.

In September 2020, Wild Montana staff encountered mountain bike tracks on the North Fork Fish Creek Trail from the Trio Lakes trail junction to the French Lake trail junction.

System Trail Conditions

Wild Montana frequently receives reports from users that trails in the southern half of the Hoodoo IRA (Lolo NF side) are not adequately brushed or maintained. These reports are received both from hikers and stock users. As a result, many of our members have indicated that they avoid trails south of the Clearwater Crossing area.

A backpacker enjoys classic Stateline Trail ridge terrain. This section of trail is between Fish Lake and Goose Lake.

Non-system Trails

The Stateline Trail has many short, spur social trails to spots that overlook alpine lakes. These trails travel through alpine vegetation that is difficult to rehabilitate, and it is likely that these trails would be recreated if reclaimed.

Trailheads

Three trailheads serve as the primary access points to the Hoodoo IRA for the majority of users: Heart Lake, Hoodoo Pass, and Clearwater Crossing. These trailheads are often nearly full (or full in the case of Heart Lake) on summer weekends. Even lesser trailheads accessed from the Fish Creek corridor often are nearly full during summer months.

Access to the Hoodoo IRA from the Idaho side is challenging and requires a high clearance vehicle and a willingness to travel many miles on difficult dirt roads. For example, travel from Missoula to the Blacklead Trailhead in Idaho takes nearly four hours. In contrast, a Missoulian can access the Heart Lake trailhead in just over an hour. As a result, most Hoodoo IRA enter the area through Montana access points, not from Idaho.

Most of the Hoodoo trailheads include pit toilets, except notably Hoodoo Pass. Almost all have hitch rails or other infrastructure to support stock users. In spite of heavy use, these trailheads are in good condition and generally trash-free.

Encounters with People

Users visiting the Heart Lake Basin can expect to encounter and interact with a large number of parties, especially on weekends. Unfortunately, users on the Stateline Trail above the Heart Lake basin may also encounter users illegally mountain biking on this trail. In three visits to this portion of the trail in 2020, our Organizing Director encountered mountain bikers two out of three visits. This caused Wild Montana to request that the Lolo National Forest issue a special closure order for mechanized use in the Lolo National Forest managed portion of the Lolo NF. This order was also requested by the Great Burn Conservation Alliance, and a letter of concern was also submitted by the Idaho Conservation League. These requests were not responded to, and illegal mechanized use continues to be frequent in and above the Heart Lake Basin.

Outside of this area, solitude is easy to find and encounters with other users are sparse. All alpine lakes in the Great Burn that are accessible by trails have campsites on their shores, and it is at these lakes that users are most likely to encounter other people.

Photo: On the day we ascended to Chilcoot Pass from Straight Creek we passed two people enjoying Straight Falls and otherwise had more than 10 miles of trail all to ourselves – and plentiful, delicious thimbleberries. We were the only ones to camp that night at Lower Siamese Lake, other than a family of mountain goats, that is.

There is a small amount of winter human-powered backcountry use of the Hoodoo IRA, and these users are highly unlikely to encounter other users.

Winter in the Great Burn is a time of high wildlife habitat security. For the few humans who travel into this area when snow is on the ground, solitude is abundant and views are endless, like this one frpm Granite Peak in March. Credit: Brian Christianson

<u>Noise</u>

There is no noise incursion from uses outside the IRA into the Hoodoo area, as the topography of the area protects it from such sources. The main noise present is from overhead flights and the volume of flights is similar across all of the Great Burn roadless areas. At present there is motorized access allowed to Fish Lake on the Idaho-side of the Hoodoo IRA, and users can sometimes hear this motorized use while hiking the Stateline trail in Montana adjacent to Fish Lake.

<u>Visual Intrusions</u>

There are a very limited set of trail locations from which a user in the Hoodoo IRA can see developments or roads outside of the Hoodoo. These locations are mostly in the first two miles of the Stateline Trail for users traveling south from Hoodoo Pass.

Night sky darkness in this area is excellent, as the nearby community of Superior creates very little light pollution and Missoula is far enough away to not have significant impacts on sky brightness. The Stateline trail was an excellent location for viewing the NEOWISE comet the summer of 2020, and the trail would be an outstanding location for viewing most meteorological phenomenons visible from western Montana.

Campsites

All alpine lakes in the Hoodoo roadless area that are accessible by trail have campsites on or near their shorelines. Most of these campsites feature rough hewn log stools, benches, and primitive campfire rings. Some of these campsites are too close to lakeshores and should be relocated. This has been an issue at Dalton Lake in particular. Major trails throughout the Hoodoo also have campsites at fairly regular intervals. Campsites not located at lakes appear to be very infrequently used and some could be reclaimed.

Campsite at French Lake

Campsite at Heart Lake:

Appendix C

August 20, 2020

Carolyn Upton, Forest Supervisor
Lolo National Forest
24 Fort Missoula Road
Missoula, Montana 59804

cc: Cheryl Probert, Forest Supervisor, Nez Perce-Clearwater National Forest
Leanne Marten, Regional Forester, Northern Region
Jimmy Gaudry, Wilderness and Wild & Scenic River Program Manager, Northern Region
Carole Johnson, Superior District Ranger, Lolo National Forest

Dear Supervisor Upton,

It was a pleasure to have the opportunity to discuss, as well as visit, the Great Burn (Hoodoo Roadless Area) with you at the end of July. As the largest recommended wilderness area (RWA), and largest roadless area, in Region 1, this landscape is a very important component of our regional wild public lands. The Great Burn is notable not only for its size, but also for the quality of its wilderness characteristics and the unique wildlife security and connectivity zones it contains. The Great Burn is superlatively suited for consideration for Wilderness designation. In addition, the area is, as we discussed, increasingly valued by local communities for the beauty of its landscapes, wildlife viewing opportunities, and hence the quality of the human-powered recreation opportunities it provides, which presents both opportunities and challenges.

The Lolo National Forest portion of the Hoodoo Roadless Area is currently managed as MA12, i.e. recommended wilderness. The Lolo National Forest Plan of 1986 states management of MA12 areas that are not yet designated as Wilderness should "protect wilderness characteristics pending a decision as to their classification." Furthermore, RWAs across the Forest Service system must be managed for social and ecological characteristics that preserve and enhance wilderness character over time, as required by the 2012 Planning Rule, US Forest Service guidance, and case law.

As forests across Region 1 engage in planning processes and release new plans, such as the 2018 Flathead National Forest Plan, these plans have acknowledged that allowing non-conforming uses in recommended wilderness would degrade wilderness characteristics[1], and therefore non-conforming uses should not be

[1] "I have included plan components to protect and maintain the ecological and social characteristics that provide the basis for each area's suitability for wilderness recommendation. One of these plan components indicates mechanized transport and motorized use are not suitable (MA1b-SUIT-06) in recommended wilderness areas. I have included this plan component in my final decision because **I believe it is necessary to protect and maintain the ecological and social characteristics that provide the basis for their wilderness recommendation . . . The Forest Service has an**

allowed in RWAs. In addition to the Flathead plan, the plans released for the Helena-Lewis & Clark and Custer-Gallatin National Forests in the last six months also prohibit mechanized use in RWAs. Comments Montana Wilderness Association submitted on the Nez Perce-Clearwater National Forest Plan DEIS in April 2020 encourage the same approach to RWA management for the Nez Perce-Clearwater portion of the Great Burn, and we encourage that approach for the portion of the Great Burn managed by the Lolo National Forest in Montana as well.

The 2012 Clearwater National Forest travel management plan prohibits motorized and mechanized use within the Great Burn recommended wilderness. A Nez Perce-Clearwater Forest Plan outcome that prohibits non-conforming recreational uses in RWAs would be consistent with this current travel management plan.

Increasingly, however, there is evidence that mechanized use, mountain biking in particular, is becoming common in the Great Burn. Mountain bike users are violating the 2012 Clearwater Travel Plan by biking the Stateline Trail from Hoodoo Pass. The Stateline Trail, particularly from Hoodoo Pass to the Heart/Pearl Basin overlooks, passes in and out of Montana, into Idaho, numerous times, as we discussed while on the trail together. Each time a rider enters Idaho, they are in violation of the 2012 Clearwater Travel Plan. These mechanized users frequently drop off of the Stateline Trail to complete loop or point-to-point rides, exiting the Great Burn via the Heart Lake Trailhead or the Clearwater Crossing Trailhead, in the Lolo National Forest.

Case studies from across Region 1 (Beaverhead-Deerlodge and Flathead National Forests in particular) show that allowing non-conforming uses has directly precluded previously recommended RWA acreage from the possibility of inclusion in the National Wilderness Preservation System in the future; failing to uphold a desired condition where RWAs maintain their potential for future Wilderness designation. Details about these case studies are not included here, but I will be happy to provide them if requested.

To maintain the wilderness characteristics of the Great Burn and uphold MA12 management standards, the Montana Wilderness Association requests that a special closure order be issued for mechanized uses on the portion of the Hoodoo Roadless Area managed by the Lolo National Forest.

At minimum, this closure would ensure consistency with the Clearwater Travel Plan, by helping to address mechanized trespass on the Stateline Trail, which contains segments in both Idaho and Montana. At maximum, this closure will ensure that

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affirmative obligation to manage recommended wilderness areas for the social and ecological characteristics that provide the basis for their recommendation until Congress acts. The land management plan does not allow for continued uses that would affect the wilderness characteristics of these areas and possibly jeopardize their designation as wilderness in the future." Flathead National Forest, Forest Plan Record of Decision (2018), p. 26.

wilderness characteristics are being upheld on the portion of the Hoodoo Roadless Area managed by the Lolo National Forest in keeping with the MA12 standard.

In addition to protecting wilderness character and the potential for future designation, this closure will protect public health and safety, as well as wildlife security, as anecdotal evidence from users in the Great Burn indicate that mechanized use, particularly in the Heart and Pearl Lake basins, is inconsistent with providing safe experiences for foot users, especially those with children. The Heart and Pearl Lake basin is widely known as an ideal area for family backpacking. The single-track nature of this trail system, as well as the presence of several sections of trail with thick vegetation and sharp drop offs (particularly as the trail ascends/descends from the lake basin to the Stateline Trail), creates hazard-prone areas where dangerous conflicts could occur between mechanized users and horseback or foot-based users.

Mechanized use may also have negative impacts on mountain goat populations that reside in this area. Native mountain goat herds in western Montana have declined in recent decades[2], and this has also been true for the Blacklead herd on the Idaho-side of the Great Burn. Idaho Fish & Game recently estimated that this Great Burn herd may have declined by up to 80% over the last decade alone.[3] Although the Montana-side Great Burn mountain goat herd has not yet shown such a trend, this population should be considered sensitive. Research has shown that mountain goats are highly sensitive to both motorized and non-motorized recreational disturbance and demonstrate behavioral changes (increased vigilance and decreased foraging time), reduced reproductive success, and changes in spatial distribution (reducing presence in or abandoning desired habitat) as a result of this use.[4]

USFS rules state that a Forest Supervisor may issue a special closure order to limit certain trail uses “for the protection of…public health or safety.” Special closure orders on National Forest System trails may include prohibitions on “any type of vehicle” or “any type of traffic or mode of transport”. We respectfully request that you use your authority to issue this Great Burn Recommended Wilderness special closure order, allowing MA12 management standards to be upheld and wilderness characteristics of this area to be protected and secured until Congress acts.

Montana Wilderness Association has worked closely with our partner organization, the Great Burn Conservation Alliance, in crafting this request. Together, we request that you consider this issue promptly. And together, we look forward to providing any support that would be helpful in issuing, upholding, and communicating about

[2] Smith, B. and DeCesare, N., 2017, Status of Montana’s mountain goats: A synthesis of management data (1960–2015) and field perspectives.
[3] Nez Perce-Clearwater DEIS, p 3.2.3.4-31.
[4] Joslin, G., 1986. *Mountain goat population changes in relation to energy exploration along Montana’s Rocky Mountain front*. Biennial Symposium of the Northern Wild Sheep and Goat.
Council 5:253–269; Hurley, K. 2004.

this special closure, both with our memberships and with the larger communities that use and value the Great Burn Recommended Wilderness.

Thank you for your consideration. I look forward to discussing this request further with you and working together to address this matter.

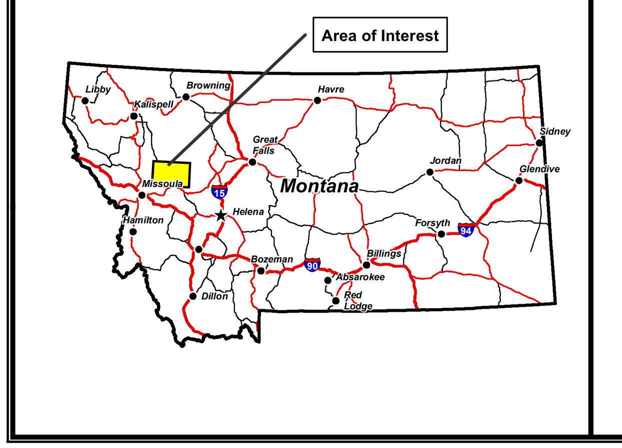
Sincerely,

Erin Clark, western Montana field director

Amy Robinson, conservation director

Appendix D

Vicinity Map



References

- Wilderness and Recreation Area boundaries were delineated by Sen. Tester's office
- USFS ownership, PLSS, Forest, County, Reservation and existing Wilderness boundaries from R1 LSRS data
- Non-USFS ownership from Montana Cadastral
- All other map data was adapted from the USFS Secondary Base Series
- Acreages expressed on this map were calculated using the R1 Albers projection
- This map was intended to be printed at 44" x 34"
- Data used in creation of this map are on file with R1 Land Status, Missoula, MT

Disclaimer

The USDA Forest Service makes no warranty, expressed or implied regarding the data displayed on this map, and reserves the right to correct, update, modify, or replace this information without notification.

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The Blackfoot Clearwater Stewardship Act will add approximately 4,463 acres to the Mission Mountain Wilderness, 47,207 acres to the Bob Marshall Wilderness, and 27,392 acres to the Scapegoat Wilderness, for a total of 79,062 new Wilderness acres. It will also designate the 2,247 acre Otatsy Recreation Management Area and the 3,835 acre Spread Mountain Recreation Area.

N

Contour Interval: 120 feet

0 1 2 3 4 Miles

- Wilderness Addition
- Otatsy Recreation Management Area
- Spread Mountain Recreation Area
- Forest Boundary
- County Boundary
- Existing Wildnerness Area
- Boundary Description Break Point

General descriptions of the boundary delineation between break points are shown on the map.
Example: ***Drainage*** *or* ***Metes and Bounds***

- National Forest System Land
- Flathead Reservation
- County Government
- Montana Fish, Wildlife, and Parks
- Montana State Trust Lands
- US Bureau of Land Management

Bob Marshall, Mission Mountains and Scapegoat Wilderness Additions, Otatsy Recreation Management Area and Spread Mountain Recreation Area

Legislative Map, 2/22/2017
Blackfoot Clearwater Stewardship Act, Senator Tester, Montana

U.S. Forest Service - Northern Region
Lolo National Forest

Lewis and Clark, Missoula and Powell Counties, Montana

Montana At-Large Congressional District

Map prepared by U.S. Forest Service, Region 1, at the request of Senator Tester, Montana

Copies of this map are available for public inspection in the Office of the Regional Forester, Northern Region, Missoula, MT

Appendix E

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Combining resource selection and movement behavior to predict corridors for Canada lynx at their southern range periphery

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ABSTRACT

Maintaining connectivity with source populations is especially important for populations of boreal species at the southern edge of their distributions, where anthropogenic disturbance and climate change can be a threat. In the conterminous United States, Canada lynx *Lynx canadensis* is a federally threatened boreal species that may require connectivity with northern populations to persist. Connectivity is a function of movement between patches and the likelihood that patches are suitable for resident populations. Therefore, we combined resource selection, step selection, and least-cost path models to define empirically movement corridors for lynx in the Northern Rocky Mountains. We used telemetry data for 64 lynx monitored during 1998–2007 to create a broad-scale resource selection model that predicted probable lynx habitat across the species' distribution in the Northern Rocky Mountains. Our model indicated that lynx selected home ranges at mid-elevations with low surface roughness and high canopy cover. Based on a subset of 37 (16 females, 21 males) adult lynx fitted with GPS collars from 2005 to 2007, we then tested the extent to which remotely-sensed indices of environmental heterogeneity, including greenness, normalized difference vegetation index, surface roughness, and a principal component that indexed stand age, could characterize landscape connectivity for lynx. We found that connectivity between lynx habitat in Canada and that in the conterminous US is facilitated by only a few putative corridors that extend south from the international border. Maintaining the integrity of these connectivity corridors is of primary importance to lynx conservation in the Northern Rockies.

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1. Introduction

Populations at the periphery of species' ranges are important for long-term conservation due to a greater potential for speciation and potentially greater survivorship than core populations when species experience sharp range contractions (Lesica and Allendorf, 1995; Channell and Lomolino, 2000; Carroll, 2007). Peripheral populations often occupy suboptimal habitats (Brown, 1984), making them vulnerable to a loss of connectivity with larger source populations (Root, 1998; Thomas et al., 2001). Moreover, peripheral populations may be particularly at risk where they face high levels of anthropogenic disturbance (Channell and Lomolino, 2000; Schaefer, 2003). Canada lynx (*Lynx canadensis*), listed as threatened under the Endangered Species Act in the contiguous US, exhibits population dynamics that lag those in their range core in Canada (McKelvey et al., 2000). Thus, lynx conservation in the contiguous US hinges in part on maintaining population connectivity between Canada and the US. However, maintaining connectivity for lynx may become increasingly difficult due to climate and anthropogenic change, as evidenced by reduced connectivity of other boreal species (van Oort et al., 2011). Preserving connectivity throughout the northern Rocky Mountains (hereafter Northern Rockies) is central to the conservation of many boreal species that are listed or proposed as threatened or endangered under the Endangered Species Act including Canada lynx, wolverine (*Gulo gulo*), grizzly bear (*Ursus arctos*), and woodland caribou (*Rangifer tarandus caribou*). Long-term population recovery of these species requires maintenance of short and long-distance connectivity (Clark et al., 2002). Thus, managers need approaches and tools that identify and maintain connectivity for such species across differing spatial scales (Carroll et al., 2010).

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Important conservation efforts focus on identifying and maintaining corridors that connect local carnivore populations (Chetkiewicz and Boyce, 2009) and their habitats (Rabinowitz and Zeller, 2010). However, identifying conservation corridors requires several stages of analyses that link species distributions to suitable habitat across scales (Beier et al., 2009). For example, patches of habitat that likely contain resident populations can be generated from a broad-scale spatial gradient in habitat suitability. Next, a probabilistic surface that predicts fine-scale movement decisions can depict the functional connectivity between these populations, as defined by resource-movement relationships (Beier et al., 2009; Richard and Armstrong, 2010; Dancose et al., 2011). Reviews of movement ecology (Fahrig, 2007) and corridor conservation (Chetkiewicz et al., 2006) recommend the integration of both disciplines using least-cost path techniques. Thus, incorporating these broad- and fine-scale species-habitat relationships is useful to identify areas most important for species connectivity (i.e. corridors) (Fahrig, 2007; Dancose et al., 2011).

In this study, we integrate three commonly used spatial modeling approaches that combine patterns of broad-scale habitat residency and fine-scale movement behavior into a single depiction of connectivity for Canada lynx in the Northern Rockies. First, we use resource selection functions (RSFs) to identify patches of suitable habitat for population residency (Mladenoff et al., 1995). Second, we complement this broad-scale RSF with fine-scale analyses of step selection functions (SSF; Fortin et al., 2005); the SSF relates animal movement to fine-scale habitat heterogeneity. Last, we use least-cost path analysis to translate our multi-scaled habitat models into a spatial depiction of lynx habitat connectivity across the Northern Rockies.

To facilitate practical application of our results for conservation planning, we characterized habitat resources of lynx using spatial data layers that are widely available and represent climatic, topographic, and vegetative heterogeneity. We included data layers that index important characteristics of lynx resource use, such as horizontal vegetative cover (Squires et al., 2010). Because horizontal cover decreases during winter (after deciduous leaf-fall), we expected to observe seasonal differences in lynx movement relative to vegetative indices (Squires et al., 2010). Finally, in an effort to prioritize conservation efforts, we quantified the relative likelihood of lynx crossing major highways, as roads are one of the major hypothesized anthropogenic threats to lynx connectivity (Carroll et al., 2001) in their southern distribution. Evaluating highway crossings is an important conservation application given the potential impacts of increased vehicle traffic on road networks in the Northern Rockies (Carroll et al., 2001).

2. Methods

2.1. Study area

Our study area encompassed the occupied range of lynx within the Northern Rockies as estimated from a compilation of lynx distribution data collected from 1998 to 2007. The study area border followed natural topographic and vegetative boundaries to generally encompass all forested regions with recent evidence of lynx presence, including all telemetry locations we documented for resident lynx from 1998 to 2007 ($N = 81{,}523$ locations, Fig. 1); this study area represented our best estimate of the current distribution of lynx in western Montana. The study area spanned a total of 36,096 km^2 and included private lands, federal- and state-managed multiple-use public lands, the Bob Marshall Wilderness Complex, and Glacier National Park. The Northern Rockies is home to a diverse boreal carnivore community, many of whom are also of special concern, including gray wolves (*Canis lupus*), grizzly bear, wolverine, and fisher (*Martes pennanti*). Elevation on the study area ranged from 530-3190 m and forests varied from dry ponderosa pine (*Pinus ponderosa*) and Douglas fir (*Pseudotsuga menziesii*) stands at lower elevations to lodgepole pine (*Pinus contorta*), western larch (*Larix occidentalis*), subalpine fir (*Abies lasiocarpa*), and Engelmann spruce (*Picea engelmannii*) at high-elevation sites.

2.2. Lynx capture and monitoring

To identify lynx habitat at a broad spatial scale, we used location data from 64 lynx that were monitored as adults and had been located at least 20 times within a consistent home range (median $N_{\text{locations}} = 561$). Lynx were captured from 1998 to 2007 using a combination of box traps, foothold traps and foot snares following Kolbe et al. (2003). Animals were fitted with very high frequency (VHF) radio-collars (Advanced Telemetry Systems, Isanti, Minnesota, USA), some of which also included Argos platform transmitter terminals (PTTs; Sirtrack Ltd., Havelock North, New Zealand) or store-on-board global positioning system (GPS) units (Lotek Wireless, Newmarket, Ontario, Canada). We used VHF ($N = 23$), Argos ($N = 6$), and GPS ($N = 35$) data to estimate home ranges; location accuracy varied among these 3 types of telemetry, but all were sufficient for identifying home ranges at a broad scale (Appendix A). To study movement at a fine scale, we used a subset of 37 (16 females, 21 males) adult lynx fitted with GPS collars that were captured from 2005 to 2007. We programmed GPS collars to obtain locations every 30 min throughout discrete 24-h periods, every other day during both winter (December–April) and summer (May–September).

2.3. Predicting resident habitat patches using RSF

To define lynx habitat, we calculated a resource selection function based on logistic regression (Manly et al., 2002) that quantified the environmental characteristics of resident lynx home ranges relative to those available across their range in the Northern Rockies (second-order habitat selection; Johnson, 1980). We used 80% fixed kernel lynx home ranges ($N = 64$; Rodgers et al., 2007) to characterize lynx use for comparison with random circular home ranges ($N = 1000$) equal in area to the median lynx home range (39.6 km^2; Katnik and Wielgus, 2005) that characterized home range availability across the study area. Random home ranges were sampled within the species' range as defined in the Northern Rockies (see Study Area description; Fig. 1).

We used a combination of categorical maps and continuous indices based on satellite imagery to capture vegetative heterogeneity across the study area, hypothesizing that each may serve as an index to factors important to lynx ecology. We considered environmental variables that characterized vegetative, topographic, and climatic spatial heterogeneity (Table 1). Specifically, we were interested in remotely sensed vegetation indices that may serve as surrogates for field-based measures of vegetative heterogeneity found to be important in previous studies, such as horizontal cover or stand age (Squires et al., 2010). For continuous variables we calculated the average value in each used and random home range, and for categorical values we calculated the proportion of each used and random home range within each category (Table 1).

We used logistic regression in SYSTAT 11.0 (Systat Software, Inc., Richmond, California, USA) to compare used to random home ranges. We weighted random-used cases as 0.0064:1 to provide a balanced comparison of 64 used to 64 available home ranges and avoid inflating statistical precision while still allowing a large and representative random sample of habitat availability. We constructed multivariate logistic models of resource selection from important ($P < 0.25$) variables that we identified using univariate logistic regression according to Hosmer and Lemeshow (2000).

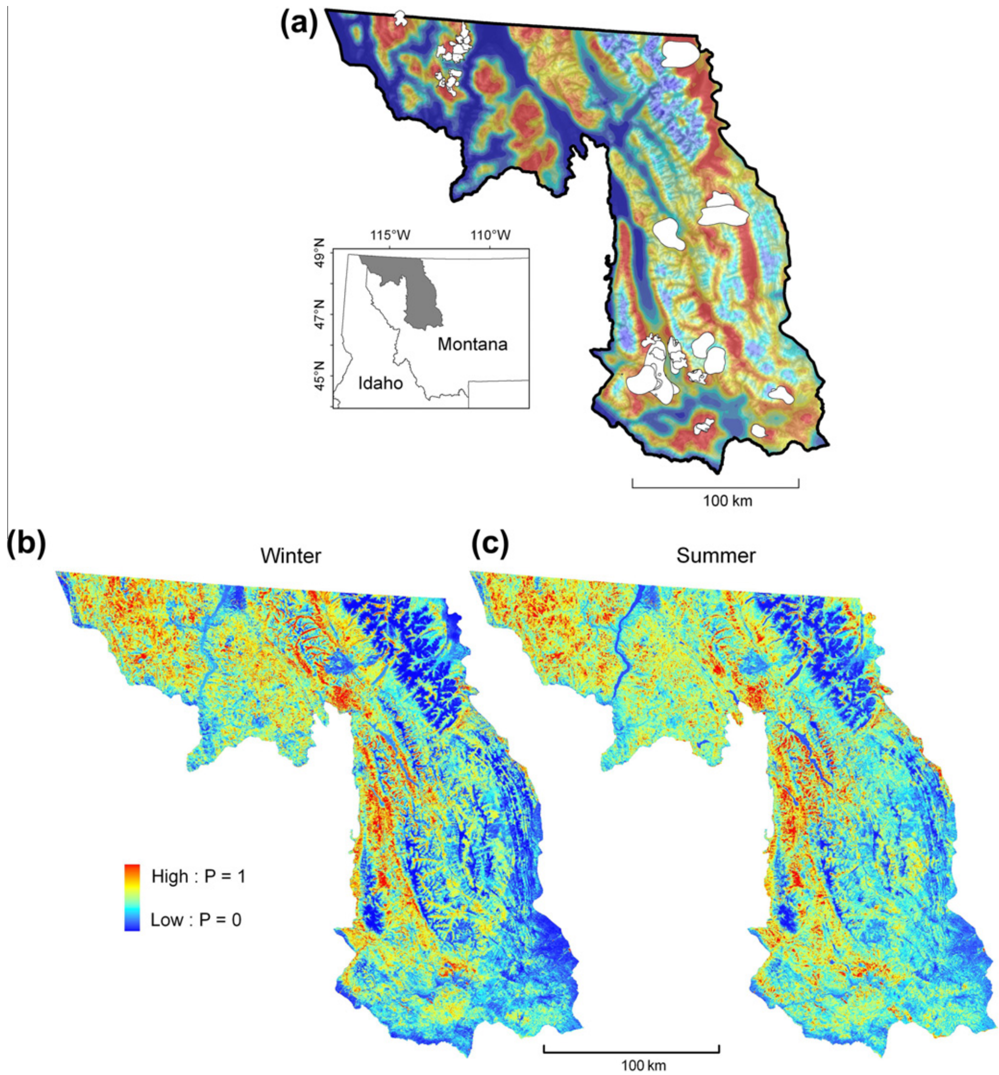

Fig. 1. Selection probability surfaces for: (a) home-range level resource selection function of Canada lynx (white polygons indicate lynx home ranges) within the northern Rocky Mountains; winter (b) and summer (c) population-level models of step selection functions for Canada lynx movement in the Northern Rocky Mountains, 2005–2007.

Before inclusion in a final model, we further evaluated variables in terms of their stability, collinearity, biological meaningfulness, interpretability, and their contribution to the model log-likelihood (Squires et al., 2010). We added variables to multivariate models using a manual forward-stepping procedure based on the strength of univariate relationships as measured by Wald statistics. We then evaluated whether to retain added variables to multivariate models according to biological reasoning and statistical likelihood ratio tests (Hosmer and Lemeshow, 2000). We selectively added and removed variables from multivariate models to see if this changed the sign or standard errors of variable coefficients to ensure that our final model was stable and with low collinearity among predictor variables (Hosmer and Lemeshow, 2000).

For model evaluation, we used 2-fold cross-validation to assess model predictions by randomly dividing our sample of lynx ($N = 64$) into two subsets and re-estimating the coefficients of our best model for each subset. We then assessed predictive capacity of each subset model with Spearman-rank correlation statistics that essentially tested if withheld lynx home ranges were indeed concentrated in areas of high predicted probabilities of use (Boyce et al., 2002). We then spatially applied the multivariable RSF across the study area using:

$$w(x) = \exp(\beta_1 x_1 + \beta_2 x_2 + \ldots + \beta_i x_i), \qquad (1)$$

where β_i is the RSF coefficient for each predictor variable (i), x_i is the value of each predictor variable (i), and $w(x)$ is a predicted value relative to the probability of use as lynx resident habitat (Boyce et al., 2002). To estimate a binary surface of low- versus high-probability habitat for resident lynx, we used a conservative cut-off value equal to the lowest predicted value of the observed sample of lynx home ranges.

2.4. Lynx movement modeling using step-selection functions

Similar to RSFs (Manly et al., 2002), step selection functions are based on case-control logistic regression of used and available steps and provide a powerful method for quantifying how organisms respond to their environment using biologically meaningful scales of availability (Fortin et al., 2005). To study the behavior of lynx specifically when moving, we used a hierarchical set of rules to remove GPS locations collected when lynx were likely stationary according to Olson et al. (2011). This involved using step length and turn angle data from movement paths to distinguish 'active' from 'resting' GPS locations (Morales et al., 2004). To distinguish true movement from that induced by GPS error, we compared the step length and turn angles leading to each GPS location to the distribution of step lengths and turn angles from test collars

Table 1
Variables used to quantify resource selection and movement behavior of Canada lynx in response to environmental heterogeneity.

Type	Variable name	Description	Source
Vegetation variables	CC.open	Canopy closure < 10%	VMAP (Brewer et al., 2004)
	CC.low	Canopy closure ⩾ 10% & < 25%	
	CC.mod	Canopy closure ⩾ 25% & < 60%	
	CC.high	Canopy closure ⩾ 60%	
	VEG.alpine	Rock, ice and grasslands above 2000 m	
	VEG.grass	Grasslands and open clear cuts	
	VEG.shrub	Shrub	
	VEG.xeric	Ponderosa pine, Douglas-fir, and western larch forests	
	VEG.mesic	Engelmann spruce, subalpine fire and lodgepole pine forests	
	VEG.other	Water and developed	
Vegetation indices	NDVI	Normalized differenced vegetation index	Pettorelli et al. (2005)
	Bright	Tasseled-cap brightness	Crist et al. (1986)
	Green	Tasseled-cap greenness	
	Wet	Tasseled-cap wetness	
	PCA	Single band principle component analysis	Schriever and Congalton (1995)
Topography	Elev	Elevation (kilometers)	USGS
	$Elev^2$	Elevation (kilometers) squared	
	Slope	Slope (degrees)	
	Roughness	A ratio of 3-dimensional to 2-dimensional terrain surface area	Jenness, 2004
	Aspect	Cosine transformed into a linear variable between southwest (215°) and northeast (35°)	Squires et al. (2008)
	TPI	Topographic position index (500 m scale)	Weiss, 2001; Jenness, 2006;
	Dist_H20	Distance to water	US Bureau of the Census (2000)
Climate	T.avg	Average daily temperature	PRISM Climate Group (2006)
	Precip	Average annual precipitation	
	Snow	Average winter snow depth	NOHRSC 2004

known to be in a stationary position. Segments that had a length or turn-angle within the 70th percentile of the stationary test collar's segment distribution were classified initially as 'resting'; the remaining segments were classified as 'active'. Among the remaining 'active' segments, we removed GPS points which spiked abruptly away from clusters of consecutive 'resting' points. We used a non-linear curve fitting procedure (Johnson et al., 2002) to determine that a 2-state model (distinguishing stationary and moving states) provided the best fit to observed lynx movements (Appendix B). We used matched case-control logistic regression to compare environmental features associated with observed steps between sequential lynx GPS locations to those associated with 5 control steps, with each case identified using a stratifying variable (Fig. 2; Hosmer and Lemeshow, 2000). Case-control logistic regression minimizes variance associated with the stratified variable and the associated autocorrelation inherent in spatial data collected along a track over short time intervals (Craiu et al., 2008). We generated control steps by randomly sampling step lengths and turn angles from their respective distributions in lynx GPS data (Fortin et al., 2005).

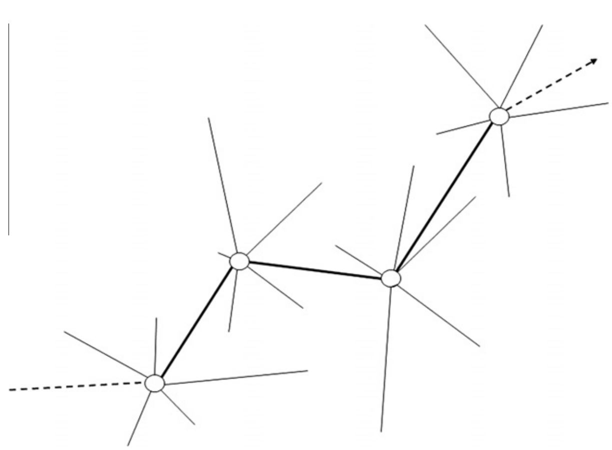

Fig. 2. Depiction of step-selection function that compared used (▬▬▬) lynx movement steps to 5 controls (———) at each GPS location.

We treated marked animals as the experimental unit, thus addressing the most common problems associated with resource selection analyses including the pooling of data across individuals (Thomas and Taylor, 2006). We constructed individual SSF models for each animal and season (winter [December–April] and summer [May–August]) using SAS (SAS Institute, Cary, NC). We then averaged logistic coefficients across individual lynx as an estimate of the population-level effect of predictor variables on the relative probability of use (Sawyer et al., 2009). We used a t-statistic to test if coefficients averaged across individuals were significantly different from zero ($\alpha \leqslant 0.1$), and included only significant variables in the population-level model for each season (Hosmer and Lemeshow, 2000). We considered the same suite of vegetative, topographic, and climatic variables as when estimating the lynx habitat RSF. Prior to modeling, we identified and removed predictor variables with high ($|r| > 0.50$) multicollinearity based on Pearson's pairwise correlation analyses. We did not use an information theoretic approach such as Akaike's Information Criterion (Burnham and Anderson, 2002) for model selection because these methods lack standardized approaches to retain the animal as the experimental unit and build a population-level model from common predictor variables (Sawyer et al. 2009).

We mapped seasonal (winter and summer) projections of spatially referenced use surfaces of lynx movement using the coefficients from the population-level SSF model in Eq. (1). We then re-scaled SSF predicted values to probability of use values between 0 and 1 by dividing each raster cell value by the maximum predicted value. To remove the effect of a few extreme outliers, we included only the range of predicted values contained within the 5th – 95th percentiles for the final SSF probability surface.

2.5. Mapping lynx connectivity

We integrated our multi-scale analyses of lynx habitat and movement behavior with a least-cost path analysis to assess connectivity across the species' distribution in the Northern Rockies

(Chetkiewicz et al., 2006; McKelvey et al., 2011). We used the "Cost Distance" and "Cost Path" functions in ArcGIS® Desktop 9.2 (ESRI, Redlands, CA, USA) to determine least cost paths from source points in the north to destination points in the south (Cushman et al., 2009). We spaced potential source points uniformly at 7 km intervals along the Canadian border in high probability lynx habitat and we randomly located destination points within all patches of high-probability lynx habitat in the Northern Rockies. The western section of the study area was divided from the rest of the study area by a large reservoir (Lake Koocanusa), which creates a geographical barrier to movement between east and west sections. Therefore, we restricted western source points ($N = 9$) to western destination points ($N = 25$) and eastern source points ($N = 12$) to eastern destination points ($N = 200$). To create least-cost paths that reflect connectivity of ecologically meaningful areas for lynx, we used the binary RSF model of lynx habitat to identify habitat likely to be occupied by lynx and limited source and destination points to those areas. We converted the SSF probability surface to a resistance surface for least-cost path analysis using the reciprocal of the probability values, so that areas with high probabilities of use had low resistance values, and areas with low probabilities of use had high resistance values.

Once least cost paths were generated, we determined the routes most likely used by lynx by summing the total number of paths and calculating the percent of this total for routes in which multiple paths overlapped. To evaluate where highways potentially impacted connectivity, we counted the number of putative path crossings per km for 10-km segments of highway throughout the species' distribution in the Northern Rockies.

3. Results

3.1. Lynx habitat in the northern rockies

In the northern Rocky Mountains, Canada lynx selected home ranges at mid-elevations ($X = 1681$ m, SD = 116 m, range = 1425–1998 m) with low surface roughness, high canopy cover, and little open grassland vegetation ($X_5^2 = 94.482$, $P < 0.001$; Table 2). The spatial predictive surface resulting from the RSF model indicated lynx habitat in the Northern Rockies is distributed in patches at a landscape scale (Fig. 1). Spearman-rank correlation statistics from 2-fold cross-validation of ranked model prediction bins and the frequency of values for withheld home ranges were 0.845 and 0.941 for each fold of data, suggesting good predictive fit.

3.2. Lynx movements from step-selection functions

From 2005 to 2007, we modeled how lynx responded to landscape heterogeneity based on 33 lynx (22,401 GPS locations) during winter and 28 lynx (20,615 GPS locations) during summer. During winter, SSF coefficients averaged across individuals were significantly different from zero ($\alpha \leqslant 0.1$) for all significant predictor variables except for aspect and TPI (Table 3). Lynx during winter preferentially traversed habitats characterized by high greenness and NDVI compared to available movement segments, but they generally avoided habitats characterized by high PCA (Table 3, Fig. 1). During summer, coefficients averaged across individuals were significantly different from zero ($\alpha \leqslant 0.1$) for all significant predictor variables except for aspect (Table 3). Lynx preferred habitat for movement in summer was generally characterized by a greater distance from water (drainages) and with high greenness, NDVI, PCA, and TPI (Fig. 1). Overall lynx were consistently selective of high values of greenness and NDVI and low values of surface roughness regardless of season. Lynx did not exhibit selection ($P = 0.127$) for PCA during summer, but did prefer low PCA values during winter (Table 4). The spatial application of SSF predictive models revealed patterned responses of lynx movement behavior to habitat fragmentation, as indexed by continuous, remotely-sensed vegetation metrics greenness, NDVI, and PCA (Fig. 3). Additionally, population SSF models resulted in spatial movement surfaces that were generally similar seasonally with the exception of some contraction in preferred movement habitats at the southeastern extent of the species' range in the Northern Rockies (Fig. 1).

3.3. Lynx connectivity

We generated 2625 least cost paths between all habitat patches identified with the RSF model across a resistance surface defined by the SSF model. We found a primary putative corridor for connectivity of lynx from Canada to the Northern Rockies that extended from the Whitefish Range in the north, along the western front of the Swan Range and ended near Seeley Lake, MT (Fig. 4). The majority of paths (up to 64%, $N = 1673$) followed all or a portion of this route, before branching off to destination points in high-probability lynx habitat identified in our RSF model. A second putative corridor extended along the east side of Glacier National Park to the Bob Marshall Wilderness Complex. In general, connectivity paths did not vary seasonally, but there were some seasonal differences in paths in mountainous areas near Glacier National Park (Fig. 4). Paths that were located in the western portion of the study area ($N = 225$) were less concentrated, which may be partially explained by our treatment of the Lake Koocanusa reservoir as a barrier, the relatively smaller total area of resident patches within this isolated portion of the study area, and the close proximity to lynx habitat across the Canadian border.

The majority of least cost paths crossed the US Highway 2 transportation corridor to the north of the Hungry Horse reservoir near the town of Hungry Horse, MT (Fig. 5). In both summer and winter, the 10 km stretch of US Highway 2 near the town of Hungry Horse

Table 2
Resource selection function coefficients (β), standard errors (SE), Wald statistics (Z) and probability values comparing Canada lynx ($N = 64$) home ranges in the northern Rocky Mountains to randomly available home ranges ($N = 1000$) with multivariate logistic regression, 1998–2007.

Variable	β	SE	Z	P
Elevation	128.898	34.047	3.786	<0.001[a]
Elevation2	−36.277	9.852	−3.682	<0.001
Surface roughness	−50.051	12.105	−4.135	<0.001
High canopy cover (⩾60%)	3.102	2.154	1.440	0.150
Grass Cover (%)	−11.147	7.152	−1.559	0.119
Intercept	−59.641	28.002	−2.130	0.033

[a] Global Likelihood Ratio Test relative to null model: $X_5^2 = 94.482$, $P < 0.001$.

Table 3
Numbers of Canada lynx with negative and positive relationships to predictor variables in the Northern Rocky Mountains based on step-selection coefficients from case-control logistic regression, 2005–2007.

	Aspect	Dist_H20[a]	Green[a]	NDVI[a]	PCA[a]	Rough[a]	TPI[b]
	Winter ($N = 33$)						
Negative	15	23	1	8	25	21	14
Positive	18	10	32	25	8	12	19
	Summer ($N = 28$ lynx)						
Negative	17	6	2	3	9	22	8
Positive	11	22	26	25	19	6	20

[a] Predictor variables with averaged coefficients across individuals significantly ($\alpha \leqslant 0.1$) different than zero for inclusion in population-level models.
[b] TPI predictor variable was significant only for inclusion in summer population model.

Table 4
Coefficients for population-level, step-selection function models of Canada lynx in the Northern Rocky Mountains by season, 2005–2007.

	Summer			Winter		
Variable	*β*	*SE*	*P*	*β*	*SE*	*P*
Dist_H20	0.301	0.099	0.005	−0.269	0.120	0.033
Green	0.020	0.003	<0.001	0.033	0.003	<0.001
NDVI	2.243	0.368	<0.001	1.343	0.348	0.001
PCA	0.005	0.003	0.127	−0.014	0.003	<0.001
Rough	−2.296	0.731	0.004	−3.059	0.808	0.001
TPI	0.004	0.002	0.025			

had the largest number of simulated lynx paths (154.4 and 126.8 paths per km, respectively) connecting northern populations to destination points in the study area. Relatively fewer predicted paths crossed other 2-lane highways, though minor crossing areas were identified along Montana Highways 83, 89, 93, 141, and 200 across the study area (Fig. 5).

4. Discussion

4.1. Lynx movement corridors

We used empirical models of both broad-scale resident habitat and fine-scale movement behavior to collectively identify functional corridors for lynx conservation. We proposed that connectivity of lynx in the Northern Rockies is maintained by a primary north–south corridor that extends from the Canadian border and proceeds south along the west side of the Bob Marshall Wilderness Complex. We also identified a putative corridor that traverses the

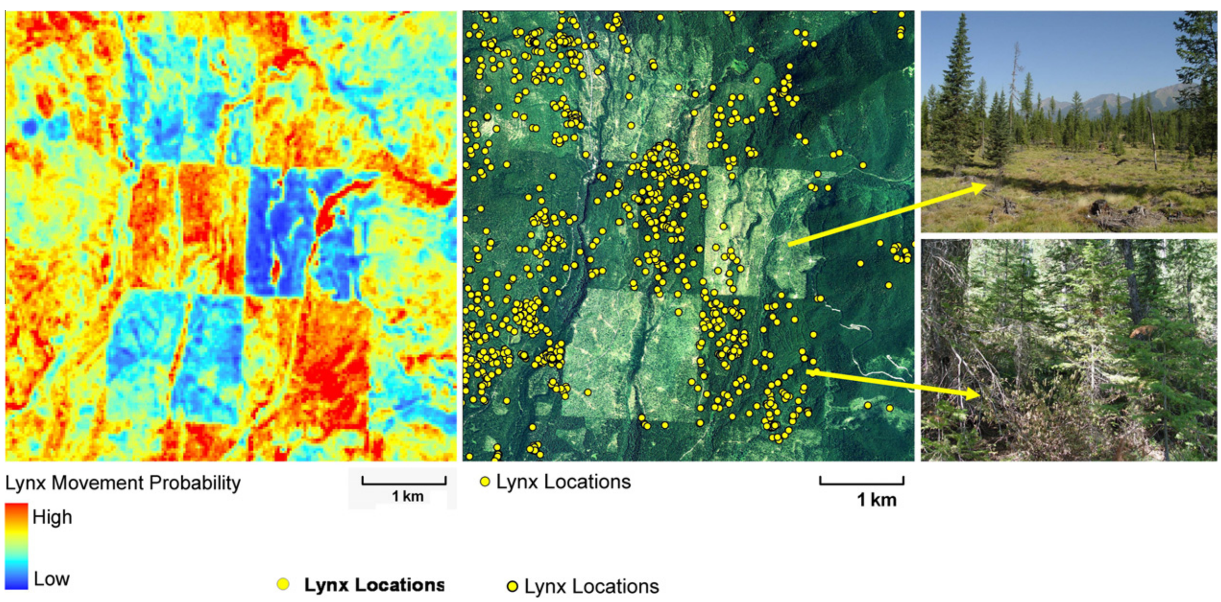

Fig. 3. Fragmentation from forest thinning decreased the probability of Canada lynx movements based on a population-level, step-selection function; 2005–2007.

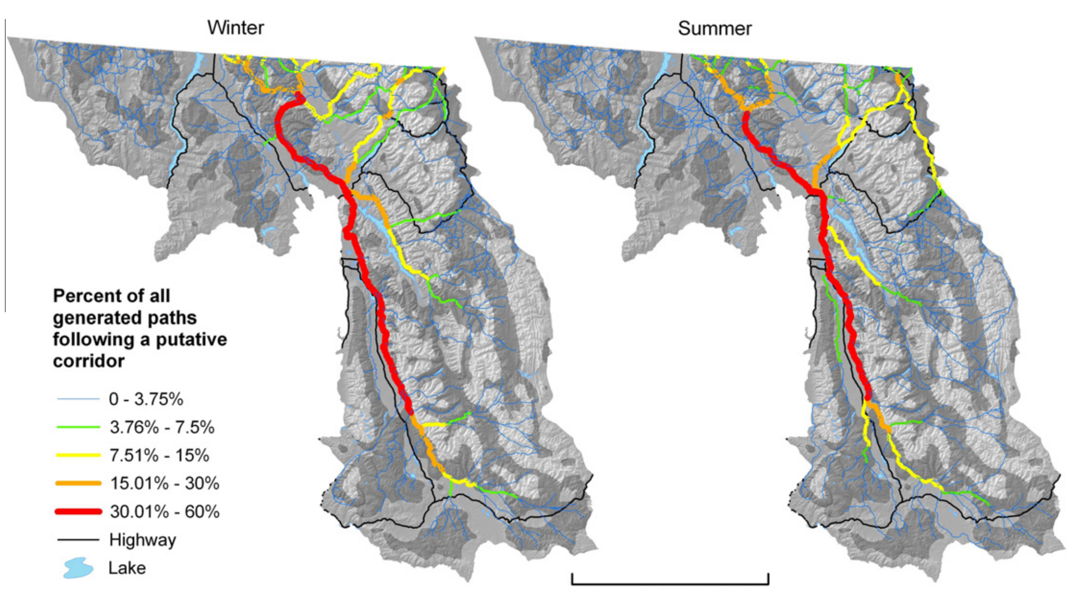

Fig. 4. Putative corridors facilitating dispersal from northern populations to patches capable of supporting Canada lynx (shaded areas) in the Northern Rocky Mountains based on least-cost path analysis of movement surfaces empirically defined using population-level, step-selection models, 2005–2007.

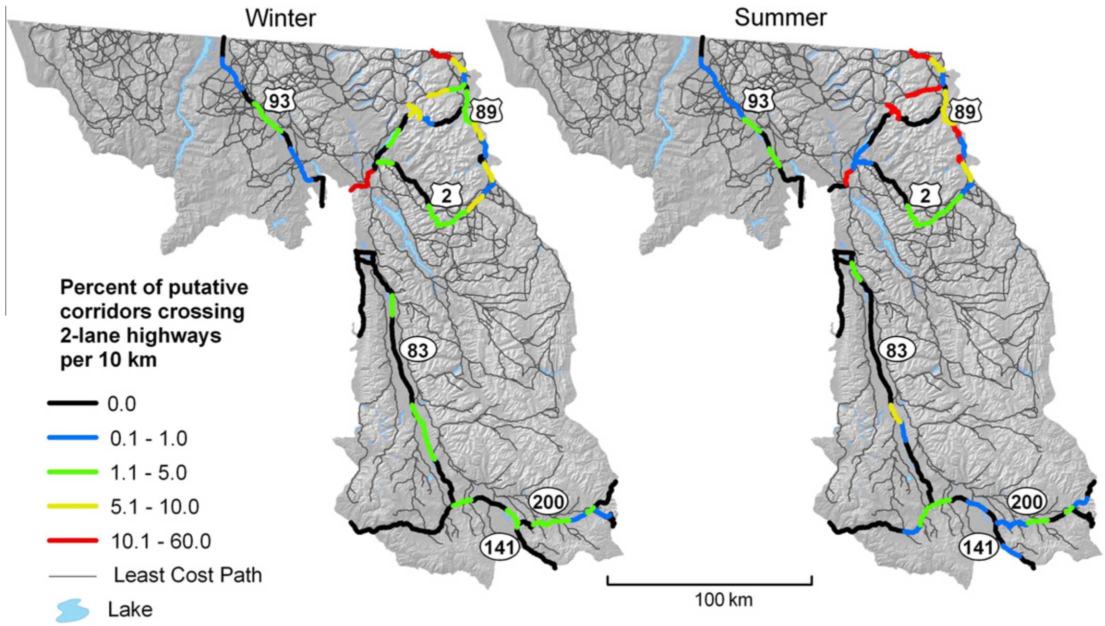

Fig. 5. Percent of all putative corridors out of all possible paths that crossed given 10 km stretches of 2-lane highway based on least-cost path analysis within the distribution of Canada lynx in the Northern Rocky Mountains by season, 2005–2007.

east side of Glacier National Park that connects Canada to northern portions of the Bob Marshall Wilderness. The putative movement corridors we identified for lynx also show reasonable correspondence with previously published models for wolverine (Schwartz et al., 2009), wolves (Oakleaf et al., 2006) and grizzly bears (Mace et al., 1999). Thus, considerable conservation value may be gained by combining habitat selection and movement analyses to identify corridors in the Northern Rockies for other focal species, such as wolves, grizzly bears, fisher and wolverine (i.e., Carroll et al., 2001).

Some species follow unique patterns of habitat selection during dispersal movements (Soulsbury et al., 2011), whereas others do not (Newby, 2011). We assumed when we defined putative corridors that lynx during dispersal would respond similarly to a resistance surface derived from within home-range movements. We initially hoped to test formally whether dispersing and resident lynx responded similarly to landscape heterogeneity, but this was impossible due to a small sample of dispersal movements. We assumed that broad-scale data layers adequately quantified environmental heterogeneity for SSF and RSF modeling (Brambilla et al., 2009). The models we developed were based on data layers that are widely available to landscape managers, but these model covariates only coarsely quantify the underlying environmental heterogeneity. For these reasons, the putative corridors that we present may be treated as testable hypotheses for further study using both spatial and genetic methods.

Rates of movement have direct biological importance in how organisms respond to their environment (Johnson et al., 2002). Many factors affect an organism's movement rates, including physiological constraints, environmental factors, and behavior. Overall, lynx movement rates in our study area averaged 6.9 km/day (Appendix C), which is considerably higher than those reported at northern latitudes during periods of high hare density but similar to those during cyclic lows (Ward and Krebs, 1985). It seems likely that southern lynx, with lower hare densities and higher movement rates in general, would be more vulnerable to factors negatively affecting connectivity. We found no statistical evidence for an "inter-patch" or dispersal movement state (Appendix B).

4.2. Response to environmental heterogeneity

Our prediction that lynx would exhibit seasonal differences in their response to environmental heterogeneity was only partially supported. In addition to consistent selection for high NDVI and greenness regardless of season, lynx appeared to conserve energy by preferentially selecting travel routes with low topographic heterogeneity, as observed for other mammals (Bruggeman et al., 2007), including carnivores (Dickson et al., 2005). We found no selection ($P = 0.127$) for areas with increased PCA values during summer; however, lynx avoided these areas during winter (Table 4). Principal components analysis of visible and near-infrared light is correlated with leaf-area index, and used to discriminate between vegetation types such as coniferous forests, shrubs, and grasslands (Wang et al., 2001). In our study area, PCA values tended to decrease in mature forests and increase in young, regenerating forest stands. Patterns of PCA selection support previous evidence of the reliance on older forests during winter and younger forests during summer by both lynx (Koehler et al., 2008; Squires et al., 2010) and snowshoe hares (Griffin and Mills, 2009). Although lynx corridors were generally similar seasonally, their respective role for conservation may depend on seasonal patterns of lynx dispersal. During the breeding season in late winter, males may exhibit extra-home range movement when they seek females. However, lynx in southern populations often make significantly longer exploratory or dispersal movements when prey availability is highest during summer (Apps, 2000; Aubry et al., 2000; Squires and Oakleaf, 2005). Thus, the winter corridors we identify may best provide for local connectivity of neighboring breeding populations, whereas summer corridors may facilitate long-distance dispersal such as those from range core to periphery.

Ideally, movement studies elucidate the behavioral response of organisms to environmental heterogeneity (Schick et al., 2008). We predicted that remotely-sensed vegetation indices would serve as broad-scale surrogates adequate for distinguishing lynx movement behaviors likely associated with an important fine-scale component of Canada lynx habitat, horizontal cover. Implicit in our approach is that animals are able to select "best" least-cost paths rather than turning around after starting down a poor quality path or starting down a poor quality path knowing that conditions improve down the track. Previous research has emphasized the importance of horizontal cover for both lynx (Moen et al., 2008; Fuller and Harrison, 2010; Squires et al., 2010) and their primary prey, snowshoe hares (*Lepus americanus*; Griffin and Mills, 2009). When building SSF models from strictly satellite-derived indices, we found that lynx were most consistently sensitive to positive values of NDVI and greenness when traversing landscapes (Table 4). High values of NDVI correlate with dense vegetation cover, such as evergreen trees in winter or dense shrubs and regenerating forests in summer, while low values correlate with barren areas (Gamon et al., 1995). Greenness also provides an index of the density of green vegetation and correlates with plant biomass and net primary productivity; like NDVI, greenness values often increase and then decrease as forests age (Crist et al., 1986; Carroll et al., 2001). The relationship between remotely-sensed indices and horizontal cover has not been explicitly tested, but the consistent predictive capacity of these indices in lynx movement models suggest them as candidate surrogates for this typically field-measured variable.

4.3. Fragmentation and highway crossings

Habitat fragmentation is clearly detrimental to some taxa (Crooks, 2002; Laurance, 2008), but the impact of fragmentation on meso-carnivores is not well understood. Results from our population-level model indicate that changes to vegetation structure can increase landscape resistance to lynx movements (Fig. 3), however, there is no evidence that this currently is causing genetic isolation (Schwartz et al., 2002). Although lynx are capable of crossing hundreds of kilometers of unsuitable habitat, as evidenced by verified locations in prairie ecosystems (McKelvey et al., 2000), lynx in the Northern Rockies are sensitive to changes in forest structure and tend to avoid forest openings (Koehler, 1990; Squires et al., 2010). The extent to which fragmentation from roads and urbanization can impact connectivity of meso-carnivore populations likely depends on the physical design of highway improvements, the surrounding environmental features, the density of increased urbanization, and the increased traffic volume (Clevenger and Waltho, 2005; Grilo et al., 2009). Carnivores are especially vulnerable to highway-caused mortality in areas with dense and high-traffic volume roadways (Clevenger et al., 2001). For example, 20% of mortalities (13 out of 65) of reintroduced lynx in Colorado were due to vehicle collisions (Devineau et al., 2010), as well as 19% (16 out of 83) of reintroduced lynx in the Adirondack Mountains of New York (Aubry et al., 2000). In Germany, 45% of the mortalities of subadult Eurasian lynx (*Lynx lynx*) are caused by traffic accidents (Kramer-Schadt et al., 2004). In adjacent southeastern British Columbia, lynx avoided crossing highways within their home ranges (Apps, 2000).

We documented 44 radiocollared lynx with home ranges within an 8 km buffer of 2-lane highways; only 12 of these individuals crossed the highway (Squires, unpublished data). Although the exact crossing locations were unknown, straight lines between subsequent telemetry locations all bisected the highway within a 10 km stretch predicted by our model as a likely crossing area. These observations increase our confidence in our predicted crossing zones of highways that bisect lynx putative corridors in the Northern Rockies. Given that increased traffic and urbanization are projected for the Northern Rockies (Hansen et al., 2002), mitigation such as land purchases and conservation easements may be necessary to preserve connectivity among lynx populations. If traffic volume greatly increases across corridors, the construction of wildlife crossing structures may be an appropriate conservation strategy; however the degree to which these structures effectively connect lynx populations is currently unknown (Clevenger and Waltho, 2005).

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.biocon.2012.07.018.

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Appendix F

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Influence of biotic interactions on the distribution of Canada lynx (*Lynx canadensis*) at the southern edge of their range

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The persistence of cold-adapted species along their equatorial range edge (i.e., southern range edge for species in the Northern Hemisphere and northern range edge for species in the Southern Hemisphere) is threatened by climate change. These species will be challenged not just by unfavorable climatic regimes, but also by changing biotic interactions, which may be more intense along equatorial edges. However, we currently have a poor understanding of the nature of biotic interactions at range edges and how climate may mediate those interactions, particularly for cold-adapted mammals. We studied the distribution of threatened Canada lynx (*Lynx canadensis*) at their southern range edge in northern Washington, United States from 2014 to 2016. Using data collected from 397 camera-trap stations in snow-on and snow-off seasons, and single- and 2-species occupancy models, we investigated seasonal patterns of habitat selection and spatial association of lynx with their primary prey (snowshoe hares, *Lepus americanus*) and potential competitors (bobcats, *Lynx rufus*; cougars, *Puma concolor*). Single-species occupancy models revealed lynx distribution was strongly associated with snowshoe hare abundance and topographic variables related to lower temperatures and increased moisture. In contrast, bobcats and cougars were more generalized in their habitat associations or displayed the reverse response to environmental variables. Spatial overlap of the 3 felid species increased during snow-off seasons. Two-species occupancy models showed a decrease in use of camera sites by lynx when bobcats were present, suggesting lynx were avoiding their warm-adapted competitor. Taken together, these results suggest that biotic interactions are partly shaping large-scale lynx distribution patterns along their southern range edge. Increasing temperatures and loss of snow may result in a combination of habitat isolation and potential for increased competitive interactions for lynx at the margins of their range.

Key words: biotic interactions, bobcat, camera trapping, Canada lynx, climate change, occupancy, range edge

Species declines, local extinctions, and range shifts are occurring as a result of climate change and may accelerate over the coming decades (Chen et al. 2011; Zhang et al. 2013). Despite climate change being one of the key conservation challenges of the next century, our ability to predict species' responses to changing climates remains inadequate (Araújo et al. 2005). This knowledge gap is particularly important for montane or cold-adapted species, which may be some of the most vulnerable to climate change and most in need of informed conservation strategies (Sekercioglu et al. 2008; La Sorte and Jetz 2010).

A multitude of factors may affect how a species will respond to climate change (HilleRisLambers et al. 2013); however, current climate-based niche models that have formed the basis for range-shift forecasts and conservation policy guidelines focus primarily on how temperature and precipitation affect species distributions (Franklin 2010; Early and Sax 2011; Schwartz 2012). Although these models provide a useful starting point to understanding how climate change will influence species, they ignore a potentially key factor impacting the nature of the response of species to climate change: their interactions with other species. In fact, changing biotic interactions may be more important in determining response to climate change than the effect of altered temperature and precipitation (Schwartz 2012; Urban et al. 2012; Cahill et al. 2013; Milazzo et al. 2013).

Accordingly, recent calls have been made for improving our understanding of how biotic interactions change along climatic gradients and shape range dynamics and large-scale distribution patterns (HilleRisLambers et al. 2013; Alexander et al. 2016). Although experimental studies may ultimately be needed to address this objective (Alexander et al. 2016), observational field studies of species interactions along climatic gradients at the boundary of ranges can provide insight into how interactions shape use of habitat at range limits. Such empirical work remains rare but may be particularly important at the equatorial range edge of species (i.e., southern range edge for species in the Northern Hemisphere and northern range edge for species in the Southern Hemisphere), where biotic interactions may be most intense (MacArthur 1972).

Canada lynx (*Lynx canadensis*, hereafter "lynx") are a cold-adapted mammal whose southern (equatorial) range just enters the northern regions of the United States. Due to their restricted range within United States borders, lynx in the United States are federally listed as threatened under the Endangered Species Act and southern populations are at greater risk of extirpation than their northern counterparts. Climate change is expected to negatively impact lynx and simple niche models predict spatially variable contractions in the southern range limit for lynx as the climate warms (Peers et al. 2014). However, several biotic interactions may be important for lynx persistence along the southern edge of their range that may alter response to changing climatic conditions. Lynx are generally dependent on snowshoe hares (*Lepus americanus*) as prey, with morphological adaptations for pursuit of this prey in deep snow (Murray and Boutin 1991; Squires and Ruggiero 2007). Although other prey may be utilized along the southern edge of the lynx's range (Ruggiero et al. 1999; Roth et al. 2007; Ivan and Shenk 2016), snowshoe hares may remain an important prey item. If that is the case, the distribution of snowshoe hares could influence large-scale range shifts of lynx in response to climate change (Peers et al. 2014). In addition, several potential competitors may influence lynx distribution in marginal, range-edge environments. Bobcats (*Lynx rufus*) are closely related to lynx and similar in size, but lack adaptations for movement through deep snow. Bobcats appear to be expanding northward (Lavoie et al. 2009). Past modeling work and anecdotal evidence suggest the potential for competitive interactions between lynx and bobcats (Parker et al. 1983; Peers et al. 2013). Cougars (*Puma concolor*) also compete with lynx through interference competition, but such competition may be less intense during winter when deep snow limits cougar movement (Ruggiero et al. 1999). Interactions between these 3 felid species remain poorly understood and, in particular, how these interactions may be mediated by environmental and seasonal variation.

Lynx, cougars, and bobcats are sympatric in only a few locations in North America, which are located at the southwestern edge of the geographic range of lynx. Within one of these unique locations in northern Washington state, we utilized a large-scale camera-trap array, and single-species and conditional 2-species occupancy models, to assess broad-scale habitat associations of these 3 species, seasonal differences (snow-on versus snow-off) in spatial and temporal overlap, and evidence for negative interactions between lynx and their potential competitors.

We tested several predictions related to how biotic interactions influence lynx distribution patterns. Due to the general dependence of lynx on snowshoe hares as a prey item, we predicted that broad-scale lynx occupancy patterns will be influenced by the distribution of snowshoe hares. Given that bobcats and cougars are potential competitors with Canada lynx, we predicted that, in areas of overlap, there will be evidence of spatial or temporal avoidance of bobcats and cougars by lynx. Lastly, due to the morphological advantages of lynx in deep snow, we predicted that the distributional overlap between lynx and their potential competitors (bobcats and cougars) will be greatest in snow-off seasons.

Materials and Methods

Study area.—We conducted the study within a 551-km^2 landscape located in Loomis State Forest, north-central Washington, United States (Fig. 1). Loomis State Forest is managed by the Washington Department of Natural Resources and contains one of the last remaining lynx populations in Washington, where lynx are listed as State Endangered. Elevation in the study area ranged from 330 to 2,520 m (average = 1,266 m). The area had hot dry summers, with less than one-half of the precipitation falling as rain, and cold winters, where the majority of annual precipitation was snow. The average annual precipitation was < 50 cm and the temperature ranged from −23°C to 31°C (NOAA 2018). As a managed forest, the study area was impacted by extractive forestry, cattle grazing (in summer and fall), and recreational activities.

Data collection.—We collected data using remote infrared-sensing cameras between July 2014 and August 2016 following the guidelines of the American Society of Mammalogists (Sikes et al. 2016) and Washington State University IACUC protocol No. 04748-002. Within our study site, we used a random point generator in ArcMap 10.4.1 (ESRI 2016) to choose locations for camera placement with the criteria that the cameras would be > 500 m apart and placed along roads and trails (Fig. 1). A few areas of Loomis State Forest were devoid of roads or trails and were omitted from camera placement. We placed cameras on the tree or other vertical structure (e.g., fence post) nearest to the randomly selected point, provided that the tree was set back > 3 m from the road and had a clear view of the road or trail. We used placement of cameras along roads and trails to increase detection probability for the 3 carnivores, given much greater detection probabilities for on versus off roads and trails (Harmsen et al. 2010; Kays et al. 2017). We placed cameras in both snow-off (May–October) and snow-on (December–April) seasons. Only a single camera was placed at each site. We did not place an attractant or lure of any kind in front of the camera, due to concerns that this might create differential positive or negative responses by the study species and heterogeneity in detection as the lure decayed.

We created detection matrices (1 = detected, 0 = not detected) for each species (lynx, bobcat, cougar, snowshoe hare), for

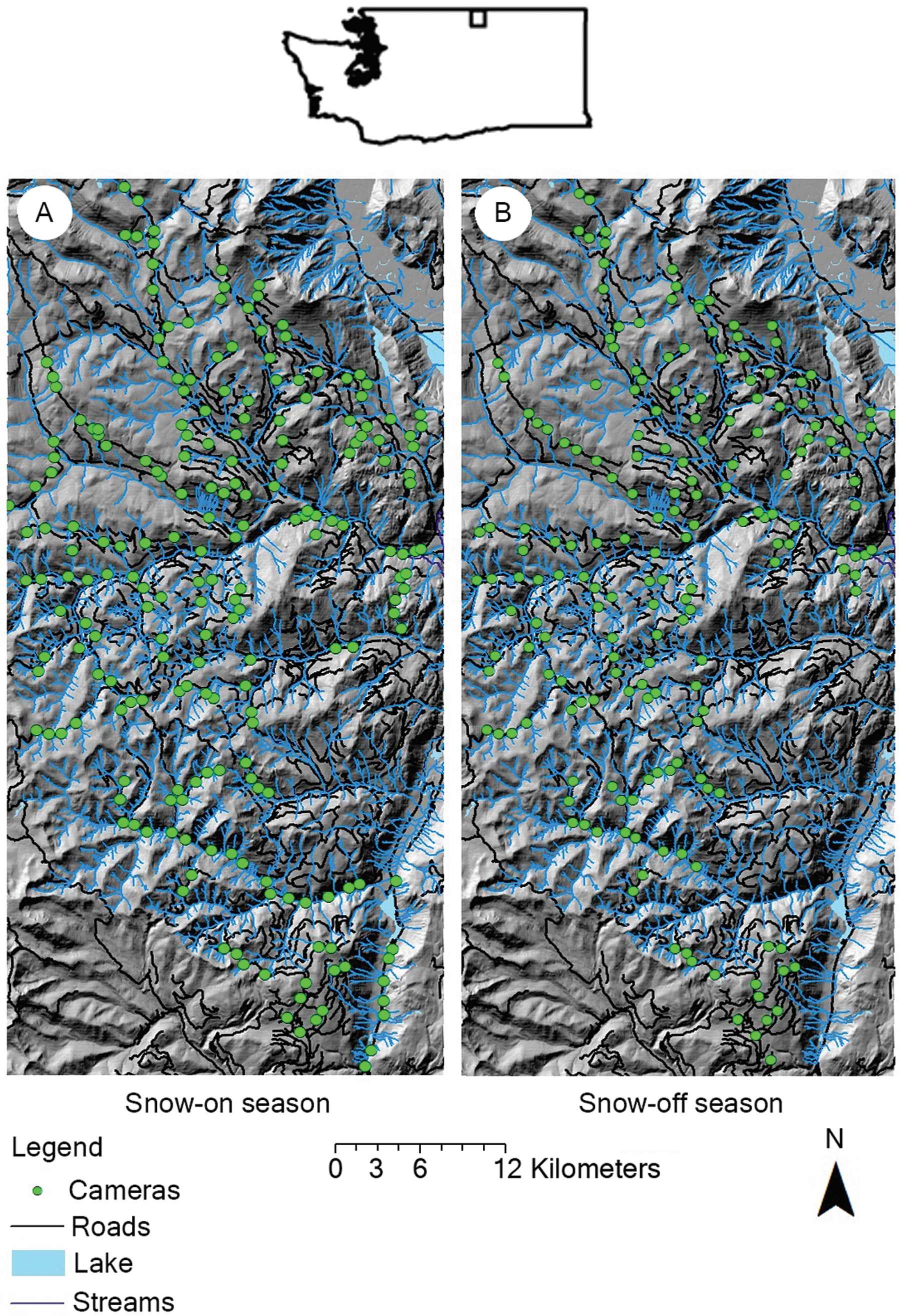

Fig. 1.—Inset shows location of the study area in Washington. Main figure shows the location of camera-trap stations (green dots) in snow-on (A) and snow-off (B) seasons. Background is a hillshade indicating the topography of the study area, with roads and streams indicated in black and blue lines, respectively.

each camera, per 5-day sampling intervals within a camera deployment. As cameras were > 500 m apart, often located on distinct trail systems, and detections were combined into 5-day intervals, detections at neighboring cameras, even when of the same animals, were considered to be independent observations of how each species used and co-occurred within the large study area. Because photos were time stamped, we also extracted information about time of day from the images for use in activity modeling.

Occupancy and detection covariates.—We calculated several covariates that we hypothesized would influence occupancy or detection in the immediate vicinity of each camera site (including a 50-m buffer around the camera location). Covariates for detection included trail type (primary roads that were heavily used by vehicles versus secondary roads and hiking trails that had lower vehicle use) and season (snow-off versus snow-on). Season was included as a detection covariate due to the possible differential use of trails during seasons with low versus high vegetative cover on the landscape. For occupancy covariates, we selected a small number of variables that were known or suspected to influence lynx distribution based on previous studies, and that also likely related strongly to occupancy of bobcats and cougars. We purposely limited the number of variables to reduce correlations between predictors and decrease the number of parameters to be estimated in the single- and 2-species occupancy models. At each camera site (with a 50-m buffer), we calculated average values of several variables reflecting the abiotic environment ("abiotic" model). These included elevation, slope, and aspect from the National Map Viewer (National Map 2015). Elevation and aspect relate strongly to temperature and snow accumulation and retention as well as overstory association on the landscape (Romano and Palladino 2002). Increasing elevation as well as north-facing slopes (particularly in snow-off seasons) have been found to correlate with increasing use by lynx (Koehler 1990). Slope (the ratio of elevation change to horizontal distance) relates to ease of mobility across a landscape, moisture retention, and overstory association; less steep slopes have been found to be an important covariate of lynx use (Koehler et al. 2007). Climatic and topographic conditions reflected in these abiotic variables also likely exert strong effects on bobcat and cougar occupancy (Koehler and Hornocker 1991; Ruggiero et al. 1999; Dickson and Beier 2007; Peers et al. 2014).

We also calculated 2 variables related to the vegetative characteristics of the environment ("vegetation" model), including canopy cover from LANDFIRE datasets (LANDFIRE 2012) and the normalized difference vegetation index (NDVI) from the National Agriculture Imagery Program. Increasing canopy cover gives lynx and their prey safety while moving, resting, and hunting or foraging and lynx have been found to select habitats with increasing canopy cover (Squires et al. 2013). The NDVI gives a measure of live green vegetation and its condition. The greater the amount of healthy, productive vegetation (high NDVI values), the more browse and cover available, even in areas of low canopy cover. The NDVI has been found to be a positive indicator of use by lynx (Carroll et al. 2001). Bobcats and cougars also have been associated with heavier cover (Koehler and Hornocker 1991; Holmes and Laundré 2006; Tucker et al. 2008; Thornton and Pekins 2015). Finally, we calculated the ratio of snowshoe hare detections (no more than 1 per hour per camera) to camera-trapping days to get a variable reflecting the availability of hares at each camera station ("hare" model). Although we expected this final model to be a primary influence on lynx distribution, we kept it as a model for the other 2 species for comparison with lynx. We did not attempt to model occupancy of snowshoe hares, but rather used the rate of detections of hares as an index of prey abundance at camera sites, which is a common approach (e.g., Kays et al. 2017; Rich et al. 2017). All continuous covariates were standardized prior to analysis. We found no evidence of correlation among the predictors used in the analysis (all correlation coefficients were < |0.35|; Table 1).

Single-species occupancy models.—Occupancy models are the preferred approach for dealing with large-scale data on species presence and absence, because they can account for the fact that species will not be detected 100% of the time in sites where they are present (MacKenzie et al. 2002). Through repeated surveys of a site (i.e., 5-day intervals in our study), detection

Table 1.—Correlation matrix of occupancy covariates used in the single- and 2-species occupancy models for Canada lynx (*Lynx canadensis*) at the southern edge of their range in Washington, United States. Correlations were less than 0.4 for all comparisons, indicating that multicollinearity was not a concern for our analysis. CAN = percent canopy cover; ELE = elevation; HARE = snowshoe hare ratio; NDVI = normalized difference vegetation index; SLO = slope.

	NDVI	CAN	ELE	SLO	HARE
NDVI		−0.004	−0.345	0.113	0.079
CAN	−0.004		−0.010	0.115	0.021
ELE	−0.345	−0.010		−0.170	0.223
SLO	0.113	0.115	−0.170		−0.134
HARE	0.079	0.021	0.223	−0.134	

histories can be used to estimate probability of occupancy (ψ) and detection (p). Lynx in our study may have had more than 1 camera trap within their home range, and thus moved between those sites during sampling. Therefore, the occupancy probabilities estimated in our study are related to the local probability of use of a camera location (MacKenzie 2006), and should be interpreted as estimates of "use" instead of traditional "occupancy." We analyzed data for the 2 seasons together using what MacKenzie et al. (2006) referred to as the implicit dynamics approach. We combined data from snow-off and snow-on seasons into one dataset, and created a final covariate indicating the status of the season (snow-off versus snow-on) prior to analysis. Given that snow may be an important factor in niche separation of these species, combining data allowed us to determine whether occupancy and the relationships between occupancy and habitat covariates changed between seasons when snow was present on the landscape versus absent, by including the interaction between season and habitat covariates in all models (see below).

We built occupancy models in a sequential manner. We first determined the parameters that best predicted detection in single-species models, holding occupancy constant. We tested several different detection models, where we included neither, one, or both detection covariates. We used the model that had the lowest value of the Akaike Information Criterion corrected for small sample size (AICc—Burnham and Anderson 2002) to determine which detection covariates to include in subsequent steps. Using the parameters that best predicted detection, we then fit a series of single-species occupancy models for each species to determine the covariates that best predicted occupancy. We tested abiotic, vegetative, and hare models singly and in combination (Table 2). We compared models using AICc and selected the model with the lowest AICc as the most parsimonious model. Model goodness-of-fit was calculated on global models using the MacKenzie-Bailey goodness-of-fit test implemented in program PRESENCE 10.1 (Hines 2006).

Two-species occupancy models.—As a last step in the model-building process, we fit conditional 2-species occupancy models (Richmond et al. 2010) for pairs of potential competitors with lynx: lynx–bobcat and lynx–cougar. Conditional 2-species occupancy models are a recently developed extension of occupancy models that allow for the

Table 2.—Model comparison set for Canada lynx (*Lynx canadensis*), bobcat (*Lynx rufus*), and cougar (*Puma concolor*) single-season occupancy models at the southern edge of the range of lynx in Washington, United States. Columns indicate the model tested, and AICc weights that indicate the likelihood of each model being the best model, given the overall model set. Note that a metric of snowshoe hare abundance was included in the top-ranked model only for Canada lynx. AICc = Akaike Information Criterion corrected for small sample size.

Canada lynx		Bobcat		Cougar	
Model[a]	AICc weight	Model	AICc weight	Model	AICc weight
Abiotic + hare	0.9100	Abiotic	0.4352	Abiotic	0.4796
Abiotic + vegetative + hare	0.0895	Abiotic + hare	0.2500	Abiotic + hare	0.2855
Abiotic	0.0002	Abiotic + vegetative	0.2116	Season	0.1386
Abiotic + vegetative	0.0002	Abiotic + vegetative + hare	0.0730	Hare	0.0300
Vegetative + hare	0.0000	Vegetative	0.0139	Abiotic + vegetative	0.0281
Hare	0.0000	Null	0.0064	Abiotic + vegetative + hare	0.0273
Null	0.0000	Season	0.0050	Vegetative	0.0052
Season	0.0000	Vegetative + hare	0.0033	Null	0.0043
Vegetative	0.0000	Hare	0.0016	Vegetative + hare	0.0013

[a] The abiotic model includes elevation, slope, and aspect; the vegetative model includes percent canopy cover and normalized difference vegetation index; the hare model includes the hare detection ratio; the season model includes a variable denoting snow-on or snow-off season; the null model did not include any covariates. All interactions between season and other covariates are included in each model, in order to allow the influence of covariates on occupancy to vary according to the presence of snow on the landscape.

assessment of positive or negative interactions between species by allowing the probability of occupancy or detection of a subordinate competitor to be dependent on occupancy or detection of a dominant competitor at that site (Richmond et al. 2010; Robinson et al. 2014). For our analysis, we designated lynx as the subordinate competitor, based on previous anecdotal evidence of species interactions (Parker et al. 1983; Squires and Laurion 2000; Murray et al. 2008), recent modeling work (Peers et al. 2013), and differential patterns in recent range shifts (McKelvey et al. 2000a; Lavoie et al. 2009; Koen et al. 2014). Two-species occupancy models allow for the inclusion of habitat covariates of importance to each species for occupancy and detection. By including habitat covariates in the models, we decrease the likelihood that avoidance of the dominant species is confused with differential habitat selection (as habitat relationships independent of competition can influence co-occurrence). Our analyses are, therefore, a conservative test of interactions between the species pairs, as habitat associations may themselves result because of competitive exclusion in suboptimal habitats. We included the habitat covariates that appeared in best single-species models for each species in the 2-species models. We used a model selection approach to determine if lynx occupancy or detection were influenced by the dominant species. We ran 6 different models: 1) "Dependent," where both occupancy and detection of lynx are dependent on the occupancy and detection of the dominant species; 2) "Occupancy-Independent," where lynx occupancy is independent of the dominant species; 3) "Detection-Independent," where detection of lynx is independent of the dominant species; 4) "Detection-in-Interval-Independent," where occupancy and detection of lynx is dependent on occupancy and detection of the dominant species, but independent of recent detection; 5) "Occupancy-Detection-in-Interval-Independent," where occupancy and detection of lynx within a sampling interval is independent of the dominant species; and 6) "Independent," where detection and occupancy of lynx are independent of the dominant species.

All models were initially fit by allowing the effect of the competitor to vary between seasons, and holding that effect constant. However, the models where the competitor effect was allowed to vary between seasons were not competitive and are thus not considered further. Two-species models were compared using AIC to see which model best predicted occupancy and detection of lynx. All 2-species occupancy models were fit in program PRESENCE 10.1 (Hines 2006).

Daily activity patterns.—We used the package "overlap" (Meredith and Ridout 2016) to estimate the activity patterns and temporal overlap of lynx, bobcats, and cougars during snow-on and snow-off seasons. To confirm that we had enough detections to sufficiently represent the activity patterns of the species of interest, we used hourly accumulation curves from the package "vegan" (Oksanen et al. 2016), which indicated sufficient activity data for the analysis for all species in both seasons (Supplementary Data SD1).

Seasonal spatial overlap.—We calculated the amount of seasonal spatial overlap between lynx, bobcats, and cougars in several ways: 1) we determined the number of cameras that were jointly occupied by each species in each season; 2) we derived a minimum convex polygon around occupied camera stations and overlapped those polygons for lynx–bobcat and lynx–cougar in each season; and 3) we derived a kernel density surface in ArcMAP 10.4.1 (ESRI 2016) for each species, with number of detections as the Z axis, and overlapped the density surfaces for each species pair in each season. We made this latter comparison because the number of detections at a site has been related to increased intensity of use of a location (Rowcliffe et al. 2008). Overlap of the minimum convex polygons was measured in area and overlap of the kernel density surface was measured in volume. We note that because we were not adjusting for detection in these analyses, but simply used presence locations to derive our overlap estimates, our results for spatial overlap should be viewed with some caution. However, overall detection rates at cameras stations were quite high for each species (see below), and therefore should not have resulted in a substantial bias in our estimates of overlap.

Results

Single-species occupancy models.—We surveyed 205 camera stations during the snow-on season (16,259 camera-trapping days) and 192 camera stations during the snow-off season (10,940 camera-trapping days). We report effects from occupancy models in terms of odds ratios, where odds of success = probability of success (e.g., occupancy) / probability of failure (e.g., absence), and odds ratio = odds of success at one level of a covariate compared to another level of a covariate, where 1 indicates equal odds of success at both levels. Best detection covariates were season for lynx and trail and season for bobcats and cougars (Supplementary Data SD2). All 3 species had lower probabilities of detection in snow-on seasons, where the odds of detection in snow-on seasons decreased by a factor of 0.65, 0.39, and 0.37 at the cameras for lynx, bobcats, and cougars, respectively. Bobcats and cougars were less likely to be detected at cameras located on secondary versus primary trails (odds of detection on secondary trails decreased by a factor of 0.51 and 0.50 compared to primary trails for bobcat and cougars, respectively). Overall detection probabilities were fairly high for all 3 species, demonstrating the efficacy of our sampling method (the probability of detecting a lynx, bobcat, or cougar in a 90-day period, given its presence in the sampling area of the camera, was 0.87, 0.82, and 0.80, respectively)

Best occupancy models for lynx included abiotic and hare covariates whereas best occupancy models for bobcats and cougars included only abiotic covariates (Table 2). Based on large parameter estimates relative to *SE*s, and in accordance with our 1st prediction, use by lynx was highly associated with availability of hares (Table 3; Fig. 2). Elevation also was highly influential (Table 3), with odds of use increasing by a factor of 2.44 and 2.71 for a 1 *SD* increase in elevation in snow-off and snow-on seasons, respectively. There also was a substantial interaction between hares and season, and south-facing slopes and season, with the hare detection ratio and south-facing slopes exerting a more positive effect on use in snow-on seasons (Table 3). In comparison to lynx, use of camera sites by bobcats was not associated with the hare detection ratio, and responses to abiotic variables were largely opposite to those of lynx (Fig. 3; Table 3). For cougars, use of camera sites was influenced strongly by an interaction with elevation and season, with elevation exerting a positive influence on use in snow-off, and a slight negative effect in snow-on seasons (Fig. 3).

Two-species occupancy models.—The top-ranked 2-species model for lynx–bobcat was a model in which use of a camera site by lynx depended on whether a bobcat also was present at that site, but detection of lynx was unaffected by presence or detection of bobcats (Table 4). In accordance with our 2nd prediction, parameter estimates of this model indicated that lynx were negatively influenced by the presence of bobcats at a camera station, where odds of use increased by a factor of 2.71 when bobcats were not present, although there was substantial variability in this effect (95% *CI* of the odds ratio just included 1.00; 0.99–7.46). However, contrary to expectations, avoidance was not more pronounced in snow-off versus snow-on seasons, as models that allowed the effect of bobcat to vary per season were not competitive. We reran the lynx–bobcat model without the habitat covariates and found a strong negative association between lynx and bobcats. By including the habitat covariates, we are performing a more conservative test of interactions, as our habitat associations explain some of the segregation observed in the species distribution. Best models for lynx–cougars indicated that presence and detection of lynx was independent of presence and detection of cougars (Table 4).

Daily activity patterns.—Lynx had a mainly nocturnal activity pattern during the snow-off seasons and a constant activity throughout the day and night in the snow-on seasons (Supplementary Data SD3). Bobcats and cougars had very similar nocturnal activity patterns to lynx but both displayed more activity throughout the day in snow-on seasons with less midday activity than lynx (Supplementary Data SD3). Estimates of activity overlap between lynx and bobcats changed insignificantly between seasons (overlap = 0.81; 95% *CI* = 0.63–0.84 during snow-on and 0.86; 95% *CI* = 0.76–0.93 for snow-off). Overlap between lynx and cougars remained constant at 0.78 for both seasons.

Seasonal spatial overlap.—Obvious patterns of spatial segregation of the 3 species in both seasons are apparent from the detection data (Fig. 4). In accordance with our 3rd prediction, spatial overlap of lynx and other felids increased during snow-off seasons. The number of cameras with dual occupancy

Table 3.—Occupancy and detection parameter estimates from the best single-season occupancy models for Canada lynx (*Lynx canadensis*), bobcats (*Lynx rufus*), and cougars (*Puma concolor*) at the southern edge of the range of lynx in Washington, United States. Continuous variables are on a standardized scale, and thus parameter estimates indicate the influence on the log odds of occupancy for a 1 *SD* increase in that variable. *SE* for each estimate is shown in parentheses. Note that all models were tested with an interaction term included between snow and the other variables in the model, to allow the influence of habitat variables to change between snow-on and snow-off seasons. Description of occupancy variables is given in main text. Asp (S) = south-facing aspect; S: = snow-on interaction with covariate; S (det) = snow-on as a detection covariate; trail (det) = secondary trail as a detection covariate.

	Parameters										
Species	Snow-on	Asp (S)	Elevation	Slope	Hare	S:elevation	S:slope	S:asp (S)	S:hare	S (det)	Trail (det)
Lynx	−0.705 (0.650)	−0.504 (0.432)	0.891 (0.348)	−0.233 (0.239)	0.457 (0.186)	0.106 (0.519)	−0.33 (0.408)	2.426 (0.882)	2.06 (0.902)	−1.04 (0.236)	
Bobcat	−0.875 (0.651)	−0.616 (0.4523)	−0.413 (0.406)	0.152 (0.241)		0.121 (0.492)	0.875 (0.539)	0.724 (0.665)		−0.491 (0.194)	−0.681 (0.196)
Cougar	−1.9713 (0.616)	0.354 (0.487)	0.7163 (0.385)	0.3911 (0.294)		−1.0344 (0.470)	−0.3531 (0.379)	1.0847 (0.729)		−0.468 (0.288)	−0.702 (0.241)

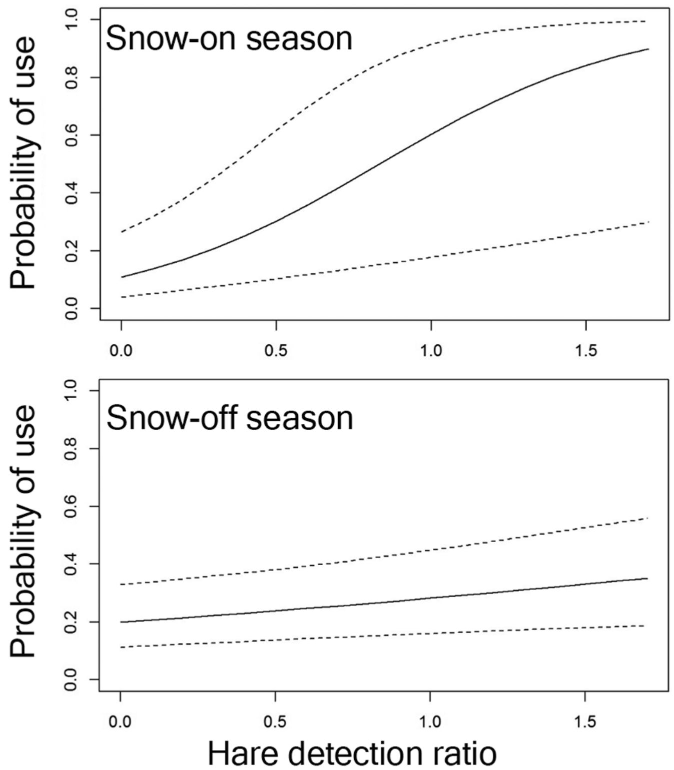

Fig. 2.—Influence of hare detection ratio on probability of use by Canada lynx (*Lynx canadensis*) in both snow-on and snow-off seasons obtained from the top-ranked single-species occupancy model. Dotted lines represent 95% *CI*s on the effect. Hares are influential on probability of use of camera sites by lynx in both seasons, but with a more marked effect in snow-on seasons, perhaps because fewer alternative prey are available in winter.

of lynx–bobcat and lynx–cougar increased from snow-on to snow-off seasons from 8 to 10 and 9 to 14, respectively. Area of overlap of minimum convex polygons also increased in snow-off seasons for lynx–bobcat and lynx–cougar comparisons (increase in overlap of 91 and 56 km^2, respectively) and kernel density overlap increased by 63.9% and 93.1% from snow-on to snow-off seasons for lynx–bobcat and lynx–cougar.

Discussion

Our analysis demonstrates that habitat use patterns of lynx at the southern edge of their range are driven in part by biotic interactions. Lynx responded strongly to prey availability even after accounting for other potential habitat and topographic covariates. Moreover, the negative influence of bobcats on use of camera sites by lynx also suggested that competitive interactions with more warm-adapted competitors may play a role in shaping lynx distribution on this landscape. While this influence appeared to be variable, there was a large decrease in probability of use of camera sites by lynx when bobcats were present. Because overlap of bobcat and lynx increased in snow-off seasons, these results indicated the potential impact of changing climatic conditions on the intensity of biotic interactions on this landscape. However, given the high variability in the effect of bobcat on lynx and the correlative nature of our analysis, evidence for or against the role of species interactions in our system would be strengthened by examining interactions at larger scales (e.g., at the level of the home range occupancy) where the effect of biotic interactions may be less intense or discernable (Araújo and Rozenfeld 2014), in more detail through telemetry, or taking advantage of natural experiments such as range expansions or reintroductions that would provide stronger causal inference (Alexander et al. 2016). Additional studies in other systems with different environmental characteristics (e.g., less topographic variability) also are needed to investigate the generality of the lynx–bobcat interactions that we documented. At the landscape scale over which we are working, however, our approach provided an excellent starting point for exploring how distribution patterns may be influenced by competitors in range-edge environments, and our results added to the small but growing literature regarding the importance of biotic interaction in shaping distribution patterns and potentially range limits (Urban et al. 2012; Wisz et al. 2013), which may be especially prevalent at the equatorial edge of a species' range (MacArthur 1972; Normand et al. 2009; Schemske et al. 2009).

Limited evidence suggests that resource–consumer interactions may influence range shifts or broad-scale distribution patterns, and such interactions may be most important for dietary specialists that could be most at risk of spatial mismatch with key resources (Schweiger et al. 2008; Hof et al. 2012; Peers et al. 2014). Our work supported this idea, as the availability of snowshoe hares was positively associated with use of the cameras sites by lynx, and that association was stronger in snow-on seasons. Dependence of lynx on snowshoe hares, as well as their responses to declines in hares, is well documented (e.g., Koehler 1990; O'Donoghue et al. 1997). Even at the southern edge of their range, where densities of snowshoe hares are comparatively low (Wirsing et al. 2002) and lynx diet may be more diverse (Ruggiero et al. 1999; Roth et al. 2007; Ivan and Shenk 2016), hares can exert a strong influence on landscape-level distribution patterns (Vashon 2007; Fuller and Harrison 2010; this study). The increased influence of hares on lynx in the winter may be a function of lesser availability of alternative prey or fewer intraguild competitors. Given that lynx have a predation advantage in deep snow (Murray and Boutin 1991), if snow-on seasons are reduced in length or intensity due to climate change, the ability of lynx to specialize on snowshoe hares may be jeopardized, or exploitative competition with other carnivores may be more likely. Bobcats and cougars were not strongly associated with snowshoe hares on this landscape, therefore the potential for competition for food between felids may be diminished. However, the degree to which exploitative competition influence lynx remains a knowledge gap along their southern range edge (Murray et al. 2008). Furthermore, snowshoe hare populations along the southern margin may decline or shift northward as the climate warms and snow cover decreases (Diefenbach et al. 2016; Sultaire et al. 2016; Burt et al. 2017), affecting the southern distribution of lynx even in the absence of increased competition for food.

Evidence of negative interactions between lynx and bobcats supports research showing that bobcats may be displacing lynx,

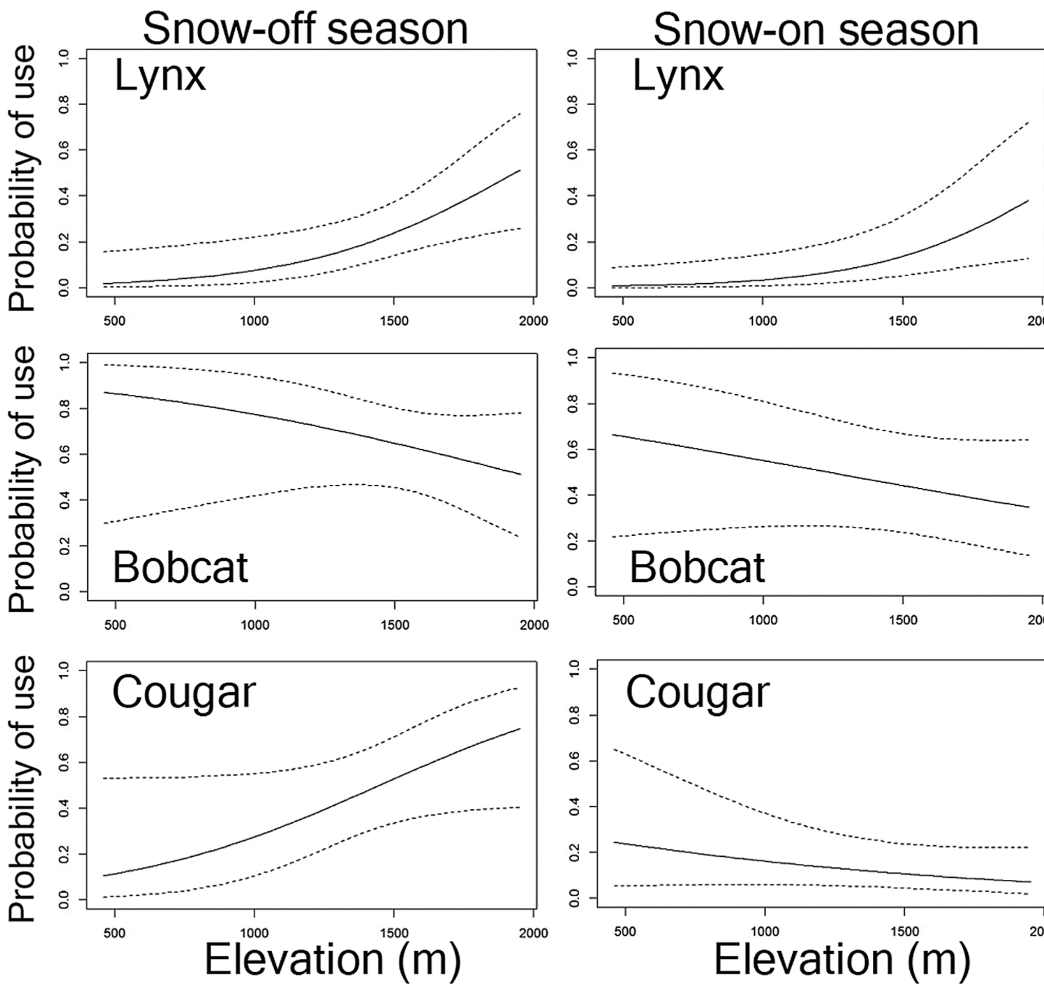

Fig. 3.—Influence of elevation on probability of use by Canada lynx (*Lynx canadensis*), bobcats (*Lynx rufus*), and cougars (*Puma concolor*) in both snow-off and snow-on seasons based on the top-ranked single-species occupancy models for each species. Dotted lines represent 95% *CI*s on the effect. Note that the cold-adapted lynx have the reverse response to elevation compared to their warm-adapted congeneric competitor (bobcats), and the strong interaction between season and response to elevation seen for cougars.

Table 4.—Model comparison table for 2-species occupancy models for Canada lynx (*Lynx canadensis*) and bobcats (*Lynx rufus*) or cougars (*Puma concolor*) at the southern edge of the range of lynx in Washington, United States. In both comparisons, lynx are designated as the subordinate competitor and either bobcats or cougars as the dominant competitor. Several cougar models did not converge and were discarded. We, therefore, present a reduced model set for cougars. The best model for Canada lynx and bobcats indicated that lynx occupancy, but not detection, was affected by presence of bobcats at a camera site, whereas the best model for Canada lynx and cougars indicated that neither occupancy nor detection of lynx was affected by presence of cougars at a camera site. AICc = Akaike Information Criterion corrected for small sample size.

Model	ΔAICc	AICc weight	Model likelihood	# of parameters	–2*log-likelihood
Bobcat-lynx					
Detection-Independent	0.00	0.3714	1.0000	25	2981.05
Independent	1.23	0.2008	0.5406	24	2984.28
Detection-in-Interval-Independent	1.63	0.1644	0.4426	26	2980.68
Occupancy-Independent	2.13	0.1280	0.3447	26	2981.18
Dependent	2.89	0.0876	0.2357	27	2979.94
Occupancy-Detection-in-Interval-Independent	4.10	0.0478	0.1287	25	2985.15
Null	58.18	0.000	0.0000	11	3067.23
Cougar-lynx					
Independent	0.00	0.5195	1.0000	24	2348.31
Occupancy-Independent	1.17	0.2894	0.5571	26	2345.48
Occupancy-Detection-in-Interval-Independent	2.00	0.1911	0.3679	25	2348.31
Null	60.00	0.0000	0.0000	11	2434.31

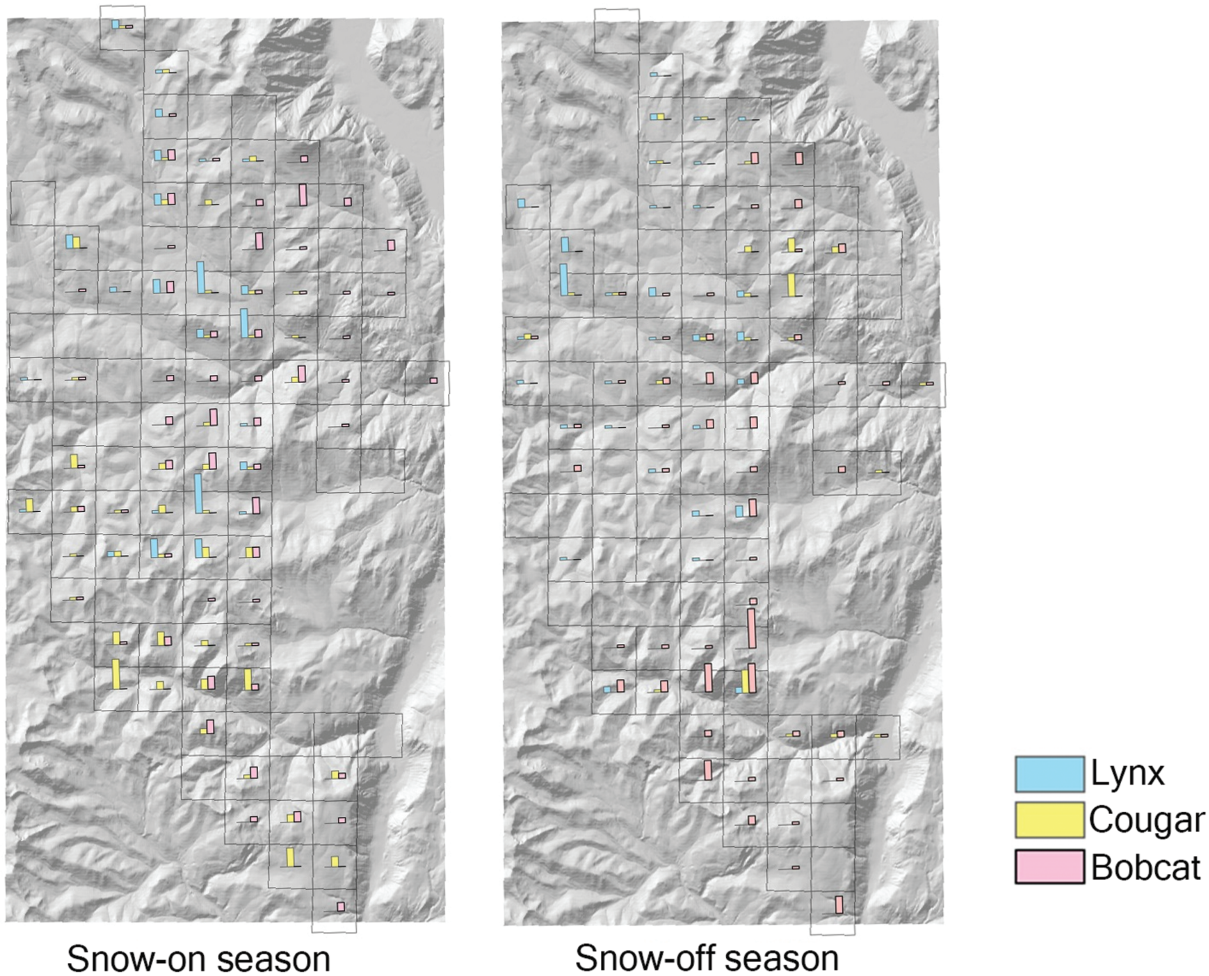

Fig. 4.——Canada lynx (*Lynx canadensis*), cougar (*Puma concolor*), and bobcat (*Lynx rufus*) detections at camera-trap stations during snow-on and snow-off seasons. Results are summarized for all cameras within 2 × 2 km grid cells (hollow squares) for ease of display. Height of the bar represents the number of detections of each species within that grid.

or altering their niche, in areas of overlap (Parker et al. 1983; Peers et al. 2013). Lynx and bobcats are virtually the same size on this landscape, and bobcat densities appear to be quite high in our study area (A. E. Scully, pers. obs.), which could be a factor in lynx avoidance of bobcats. Although spatial overlap of these 2 species increased in snow-off seasons, there was limited overlap during winter as well, which may have been facilitated by presence of roads or snowmobile trails that allowed access to deep-snow sites for the more warm-adapted bobcats (as has been found for coyotes, *Canis latrans*—Bunnell et al. 2006; Dowd et al. 2014). A failure to find any negative effect of cougars on lynx occupancy was surprising; previous research has found that cougars will kill smaller felids when they are sympatric (Koehler and Hornocker 1991), could easily scare a lynx off a recent kill (Ruggiero et al. 1999), and have been found to be a significant source of lynx mortality (Squires and Laurion 2000). Lack of an interaction with lynx could be driven by lower cougar densities on our landscape, or be a methodological artifact of using cameras to assess interactions across a landscape that likely only encompassed a small number of cougar home ranges.

Single-species models reveal the difficulties lynx may face as the climate warms. Lynx responded positively to higher elevations in all seasons, northern aspects in summer and southern aspects in winter, and gentle slopes. These factors are tied to moisture retention, colder temperatures, and deeper snow, and have been found to be influential to habitat use by lynx in other studies (Koehler 1990; Koehler et al. 2007; Squires et al. 2010). Although the interaction between season and slope and elevation was not strong, parameter estimates were in the direction expected based on previous work, with elevation being a stronger influence in summer and slope in winter (McKelvey et al. 2000b). These same covariates, though in the opposite direction of influence, were important to habitat use by bobcats (Table 3), and cougars displayed a marked negative response to elevation in winter. Interestingly, the vegetative model did not come out as important in our analysis for any of the 3 species, but the effect of the cover variables we used may have been much less important for landscape-scale distribution than the topographic variables that were strongly associated with both climate and distinct habitat types (e.g., subalpine fir) on our landscape. The single-species models and topographic patterns of use in snow-on and snow-off seasons do suggest that as climate change progresses, habitat available to lynx will be reduced to high-elevation enclaves with increasing encroachment from their potential competitors.

One caveat to our analysis is that placement of cameras along roads and trails meant that we may have been documenting more than one kind of "use" of the landscape, including territorial patrolling, movement from one foraging location to another, or active foraging along the trail. Because linear features like roads often serve as territorial markers and pathways for rapid movement of carnivores (Tucker et al. 2008), placement of cameras along roads and trails may have been more likely to detect the former 2 types of use than the later. Given that the influence of environmental variables on use patterns of lynx that we found were consistent with other studies, we doubt placement of cameras along trails strongly biased our conclusions, and trail-based camera studies are often used to make inferences about habitat use and occupancy (e.g., Kelly and Holub 2008; Jenning et al. 2015; Lewis et al. 2015; Rich et al. 2017) as well as species interactions (Harmsen et al. 2008; Farris et al. 2017). If placement of cameras along trails is indeed more likely to detect rapid or directed movement of carnivores from one location to the next rather than intensive foraging, such a design may be less likely to see evidence of fine-scale avoidance behavior and thus our multispecies models may have been conservative. In addition, placement of cameras was close enough that the same individuals could be observed at many points. We believe the observations represent independent choices about habitat use by animals given the distance and time needed to travel between locations. However, our results likely include repeated measures of animals and further work in other locations should validate whether patterns we observed hold for other populations.

Compelling evidence for temporal niche partitioning was not suggested during any season. We found high levels of activity overlap throughout the year (ranging from 78% to 86%). Moreover, all 3 species switched from strongly nocturnal activity patterns during the snow-off seasons to a more even pattern of activity throughout the entire day (e.g., smaller nighttime peaks), suggesting climatic and seasonal life history considerations were the main determinants of activity. Other studies of southern lynx populations indicate a seasonal shift to relatively more daytime, and in particular, late afternoon activity, during the snow-on season with substantial overlap with potential competitors (Kolbe and Squires 2007).

Our results indicated that changing biotic interactions could impact cold-adapted species along their southern range edge. Of particular note, in the context of climate change, was that spatial overlap of lynx and potential competitors was more pronounced in snow-off seasons. This is a likely scenario for many cold-adapted species at their southern range edge that will be exposed to increasingly greater overlap with warm-adapted competitors as climate barriers are removed. Given that consumer–resource associations also were partly dependent on presence of snow on the landscape, we suggest that incorporation of consumer–resource and competitor interactions in predictive models of responses to climate change is essential to ensuring proper conservation strategies for sensitive cold-adapted species (e.g., Trainor et al. 2014). In the specific case of lynx, developing detailed predictions of responses of hares to climate change and forestry practices that encourage robust snowshoe hare populations (Stenseth et al. 1997) will be highly beneficial to lynx at their southern range edge. Forestry practices also could be relevant to the ability of potential competitors of lynx to occupy high-elevation forested environments, given the relatively greater tolerance of many generalist competitors to habitat fragmentation (Buskirk et al. 2000). Furthermore, the environmental associations we document suggest that protecting habitat in high-elevation environments that will be the most resistant to climate change and invasion by competitors, as well as paths of connectivity between high-elevation environments, may be important to persistence of lynx along their southern margin. High-resolution data on current and future characteristics of snow could help further refine distribution models for lynx, as well as predict changes in and consequences of interactions with other carnivores. Finally, we encourage more work to document the habitat and biotic associations in range-edge environments.

Supplementary Data

Supplementary data are available at *Journal of Mammalogy* online.

Supplementary Data SD1.—Hourly accumulation curves showing that the asymptote of the number of the observations needed to represent activity in all hours was reached, indicating that we had sufficient data for the activity analysis.

Supplementary Data SD2.—Model comparison for single-species detection models for Canada lynx, bobcat, and cougar. These models were fit holding occupancy constant (no covariates for occupancy included in the models). Best models for each species were then used in subsequent modeling efforts for the single and 2-species models.

Supplementary Data SD3.—Overlap of daily activity patterns of Canada lynx with bobcat and cougar during snow-on and snow-off seasons. Overlap of the density functions of the 2 species are indicated by the gray shading. Note that there is substantial overlap between the species in both seasons, and that all species shift to more daytime activity in winter.

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Appendix G

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Article

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Mammal responses to global changes in human activity vary by trophic group and landscape

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Wildlife must adapt to human presence to survive in the Anthropocene, so it is critical to understand species responses to humans in different contexts. We used camera trapping as a lens to view mammal responses to changes in human activity during the COVID-19 pandemic. Across 163 species sampled in 102 projects around the world, changes in the amount and timing of animal activity varied widely. Under higher human activity, mammals were less active in undeveloped areas but unexpectedly more active in developed areas while exhibiting greater nocturnality. Carnivores were most sensitive, showing the strongest decreases in activity and greatest increases in nocturnality. Wildlife managers must consider how habituation and uneven sensitivity across species may cause fundamental differences in human–wildlife interactions along gradients of human influence.

With the global human population size now past 8 billion and the associated human footprint covering much of the Earth's surface[1], survival of wild animals in the Anthropocene requires that they adapt to physical changes to the landscape and to increasing human presence. Animals often perceive humans as threats and subsequently adjust behaviours to avoid people in space or time[2]. Conversely, some animals are attracted to people to obtain resource subsidies or protection from predators[3,4]. These contrasting responses to humans shape the prospects for human–wildlife coexistence, with consequences for the capacity of human-influenced ecosystems to support robust animal populations and communities.

Variation in animal responses to human activity can be driven by intrinsic factors such as species' ecological and life-history traits (Table 1)[5]. For instance, small-bodied generalist species may be more tolerant of human presence, as they can be less conspicuous than larger species and more capable of shifting resource use within their broader niches than are specialists[6]. Wide-ranging, large-bodied carnivores face considerable risk of mortality from humans[7] and so may exhibit more risk-averse responses to human activity. Animal responses may also be heavily influenced by the type of human activity (for example, hunting versus hiking[8]) and by extrinsic factors such as landscape context. Animals may be warier of people in open or human-modified environments relative to areas with abundant vegetation cover or minimal human landscape modification[9]. Conversely, animals in heavily modified landscapes could habituate to human presence and thus be less likely to respond to changes in human activity. Our ability to resolve such hypotheses about the interacting influences of species traits and landscape characteristics has been limited by the focus of previous studies on few species and contexts, with indirect measures of human activity and weaker correlative inferences. Ultimately, anticipating and managing impacts to wild animals requires stronger inferences from experimental manipulations of human activity and concurrent monitoring of people and animals across a range of species and environmental contexts.

Government policies during the early months of the COVID-19 pandemic (henceforth, pandemic) resulted in widespread changes to human activity that provided a quasi-experimental opportunity to study short-term behavioural responses of wild animals[10]. Early observations of animal responses to this 'anthropause'[11] relied on qualitative or opportunistic sightings prone to bias (for example, contributed by volunteers[12]), or focused on small spatial scales and few species, reporting a mix of positive and negative responses that make it difficult to reach more general conclusions[13]. Furthermore, measures of human activity have typically been coarse and indirect[14], yet changes to human activity during the pandemic appeared highly variable at the fine scales that affect animal behaviour (Fig. 1). For example, some natural areas experienced increases in human visitation while others were closed to visitors[15] and the strength of government restrictions changed over time[14]. It is thus important for studies using

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Table 1 | Predictor variables hypothesized to explain variation in species responses to higher human activity, with greater reductions in amount of activity or increases in nocturnality predicted for more sensitive species (further details in Supplementary Information)

Class	Variable	Prediction	Range
Species trait	Body mass	Large-bodied species will be more sensitive	Small (1–20 kg; *n*=101); large (20–4,600 kg; *n*=62)
Species trait	Trophic level	Higher trophic levels will be more sensitive	Carnivore (*n*=59), omnivore (*n*=27), herbivore (*n*=77)
Species trait	Diet breadth	Specialists with narrower diet will be more sensitive	1–4 diet categories
Species trait	Habitat breadth	Specialists with narrower habitat preference will be more sensitive	1–9 habitat categories
Species trait	Diel activity	Diurnal species will be most sensitive, cathemeral species intermediate and nocturnal species least sensitive	Diurnal (*n*=13), cathemeral (*n*=91), nocturnal (*n*=59)
Species trait	Hunting status	Hunted species (within projects) will be more sensitive to increased human activity than their non-hunted counterparts	Yes (*n*=486), no (*n*=491) (total=977 project–species)
Species trait	Relative brain size	Small-brained species will be more sensitive	0.006–5.3 kg
Habitat structure	Openness	Animals will be more sensitive in open habitat types relative to closed habitats	Open (*n*=31), closed (*n*=71)
Land-use disturbance	Human modification index	Animals will be more sensitive in landscapes with more human modification	0.005–0.834
Magnitude of human change	Global stringency index	Animals will show stronger responses where lockdowns were more stringent	38.9–96.0 stringency units
Magnitude of human change	Mean change in human detections (at camera traps)	Animals will show stronger responses where change in human activity greater	1–100-fold changes

For continuous variables we show the range (minimum–maximum); for categorical variables we show the sample size for each level, which sum to 163 species for species-level variables or 102 projects for project-level variables (unless otherwise stated). Body mass and trophic level were combined in a new variable 'trophic group'.

the pandemic as an unplanned experiment to have localized information on human activity that matches their animal data and to tackle context-dependency by using robust, standardized methods across several species and landscapes.

The widespread use of camera traps to survey terrestrial mammals[16] provides a unique opportunity to take advantage of the pandemic experiment and improve our understanding of animal responses to changes in human activity. Thousands of cameras are deployed around the world[17], providing standardized animal sampling while simultaneously quantifying local human activity[15,18]. We harnessed this opportunity to examine relationships between detections of people and mammals across gradients in land use and habitat type—spanning 102 survey sites (projects) in 21 countries (predominantly in Europe and North America) with 5,400 camera-trap locations sampling for 311,208 camera-days before and during the pandemic (Fig. 1; Methods). Some sites experienced a decrease in human activity during the pandemic, consistent with the notion of an anthropause, while there was an increase or no change at others. We focused our analysis on those sites with some change in human activity (either increase or decrease) and standardized our comparisons to be between periods of relatively lower to higher human activity (either across years or within 2020; Fig. 1; Methods) to mimic the general trend of increasing human presence in the Anthropocene. We examined site-level changes in animal detection rates and nocturnality across populations of 163 mammal species (body mass ≥ 1 kg; range 1–65 populations per species; Supplementary Table 1) as measures of the relative amount and timing of animal activity (Methods). We then used meta-analytic mixed-effects models to quantify the extent to which variation in animal responses across sites was explained by species traits, landscape modification and other site characteristics and the magnitude of change in human activity (Table 1; Methods).

Results and discussion

Our camera-trap measures of human activity varied widely under COVID-19 lockdowns (occurring between March 2020 and January 2021), from 100-fold decreases to 10-fold increases within sites between comparison periods (Fig. 1 and Supplementary Fig. 1). These changes were not predicted by coarser measures of human activity based on the stringency of lockdowns (Supplementary Fig. 1), highlighting the complementary value of finer-scaled monitoring of human activity.

Changes in amount of animal activity

Animals did not show consistent, negative responses to greater human activity; instead, responses were highly variable among species and sites (Figs. 2 and 3). Across 1,065 estimated responses (one per species per project, that is, population), changes in animal detection rates (reflecting the intensity of habitat use; Methods) varied from 139-fold increases to 36-fold decreases, with a near-zero mean change overall (−0.04, 95% confidence interval (CI) = −0.11–0.03; Fig. 2b). Trophic group (combining body mass and trophic level) was the strongest predictor of changes in animal activity in response to increasing human use, with large herbivores showing the largest increases in activity and carnivores showing the strongest decreases (Fig. 2c, Supplementary Table 2 and Supplementary Fig. 3). This is consistent with carnivore avoidance of higher mortality risk from encounters with people[7] and with increased herbivore activity due to either more frequent disturbance by people or attraction to human activity driven by reduced risk of predation (human shield hypothesis[3]).

Animal activity in more developed areas (that is, higher human modification index (HMI) measured at the site level; Table 1) generally increased (+25%) with higher levels of human activity, while animals in less-developed areas decreased their activity (−6%) when human activity was higher (Fig. 2c; coefficient = 0.077; 95% CI = −0.001–0.156). This contrast highlights an important interaction between human modification of a landscape and human activity therein—between human footprint and footfalls—which we posit could be the result of two factors. First, local extirpations of sensitive species (species 'filtering'[19]) would result in only human-tolerant species persisting in developed areas—for example, sensitive wolverine (*Gulo gulo*) were absent from sites with intermediate to high human modification. Second, species found across the gradient, such as mule deer (*Odocoileus hemionus*), could become habituated to benign human presence in more developed landscapes and therefore be less fearful of human activity than their conspecifics in less-developed areas[20]. Notably, this

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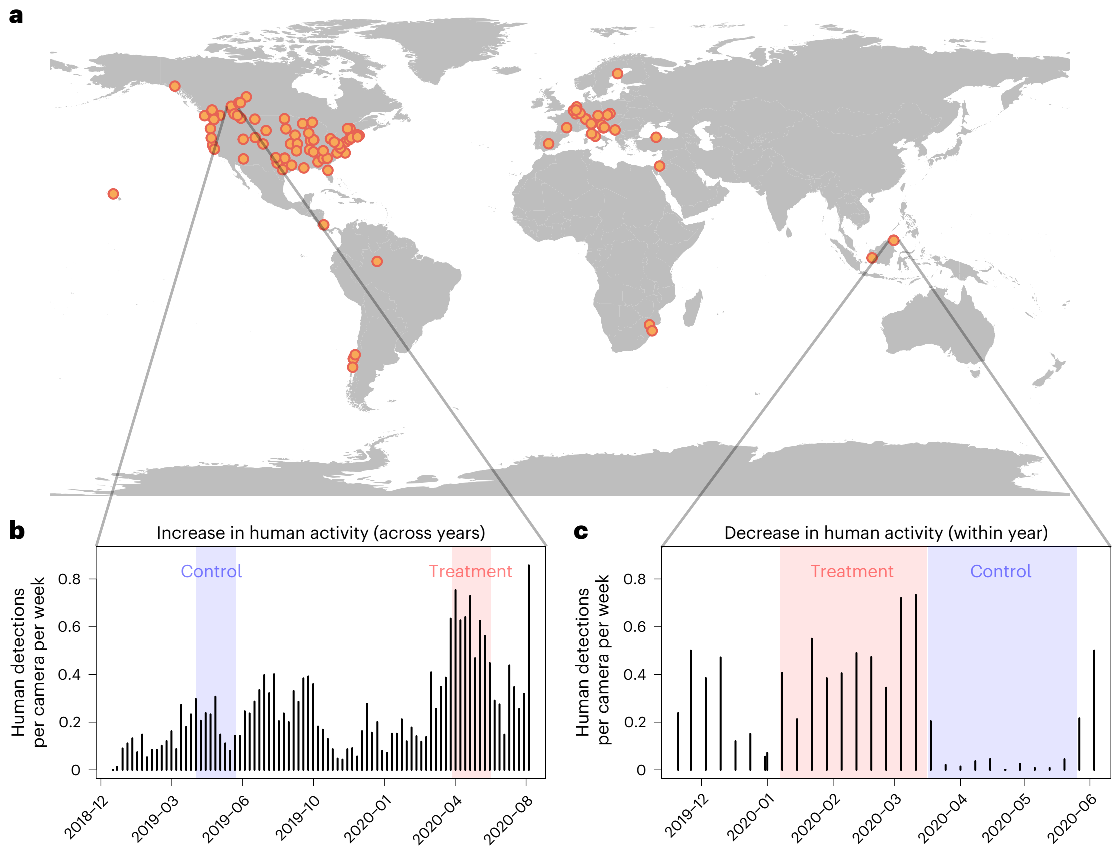

Fig. 1 | Camera-trap sampling of contrasts between periods of higher versus lower human activity. a, Location of camera-trap projects included in the analysis ($n = 102$). **b**,**c**, Examples for two projects: Edmonton, Canada (**b**) and Danum Valley, Malaysia (**c**) showing time series of human detections for the two types of comparisons used to assess the effects of higher human activity on animals. **b**, A between-year comparison with increased human activity during the COVID-19 pandemic (treatment, red shading) relative to the same time period the year before (control, blue shading). **c**, A within-year comparison with decreased human activity during the pandemic (control, blue shading) relative to the prepandemic period (treatment, red shading).

relationship with landscape modification varied predictably across trophic groups (Fig. 2d and Supplementary Table 3). Small and large carnivores, small herbivores and small omnivores increased their activity with higher human activity in developed areas (increasing by an average of 54%), while the response was much weaker for large herbivores and in fact opposite for large omnivores, which decreased activity when human activity increased in more modified landscapes (50% decrease; Fig. 2d). This negative response was common across all of the frequently detected large omnivores—wild boar (*Sus scrofa*), American black bear (*Ursus americanus*) and brown bear (*Ursus arctos*)—and could be driven by their attraction to anthropogenic food resources (for example garbage and fruit trees) that may be less risky to access when human activity is reduced[21].

Animal detections were also more likely to decline with higher human activity in more open habitat types such as grasslands or deserts, relative to closed habitats such as forests (Fig. 2c; coefficient = −0.172; 95% CI = −0.3428 to −0.0018). This is consistent with predictions under the landscape of fear framework that suggest that animal perceptions of risk are influenced by availability of cover[22]. Contrary to our expectations, we did not find strong evidence that the magnitude of change in human activity (measured by camera traps or the stringency index; Table 1) affected animal responses or that hunted populations changed their amount of activity more than non-hunted ones (Supplementary Tables 2, 4 and 5). We also did not find strong support for the hypothesis that species with relatively larger brains—as an index of behavioural plasticity[23]—would show more pronounced responses to changes in human activity (Supplementary Table 5).

Changes in timing of animal activity

Whether or not animals change their intensity of use of an area, they could shift their timing of activity to minimize overlap with increasing human activity (Fig. 3a)[24]. We measured changes in animal nocturnality (proportion of night time detections) across 499 populations (Methods) and found considerable variation in animal responses to increasing human activity (though generally less than for amount of activity): from fivefold increases in nocturnality to sixfold decreases (mean change in proportion of nocturnal detections = 0.008; 95% CI = −0.02–0.04; Fig. 3b). The strongest predictor of changes in nocturnality was the degree of landscape modification (HMI): in more developed areas, animals tended to become more nocturnal as human activity increased (19.3% increase in nocturnality; Fig. 3c, coefficient = 0.047; 95% CI = 0.026–0.069; Supplementary Table 6). This is consistent with previous evidence of increasing wildlife nocturnality in the face of growing human impacts[24] and highlights the importance of the temporal refuge provided by night time cover for human–wildlife coexistence in increasingly human-dominated environments[25].

Paralleling our findings about changes in the amount of animal activity, trophic group was also an important predictor of changes in nocturnality, with large carnivores becoming notably more nocturnal than other groups (+5.3%; Fig. 3c and Supplementary Table 6). Again, we found support for an interaction between human modification and trophic group: most groups had stronger increases in nocturnality along the disturbance gradient as human activity increased (mean +22.6%), whereas the increases in nocturnality for large carnivores did not vary with land-use disturbance (Fig. 3d and Supplementary Table 7). This finding could reflect greater

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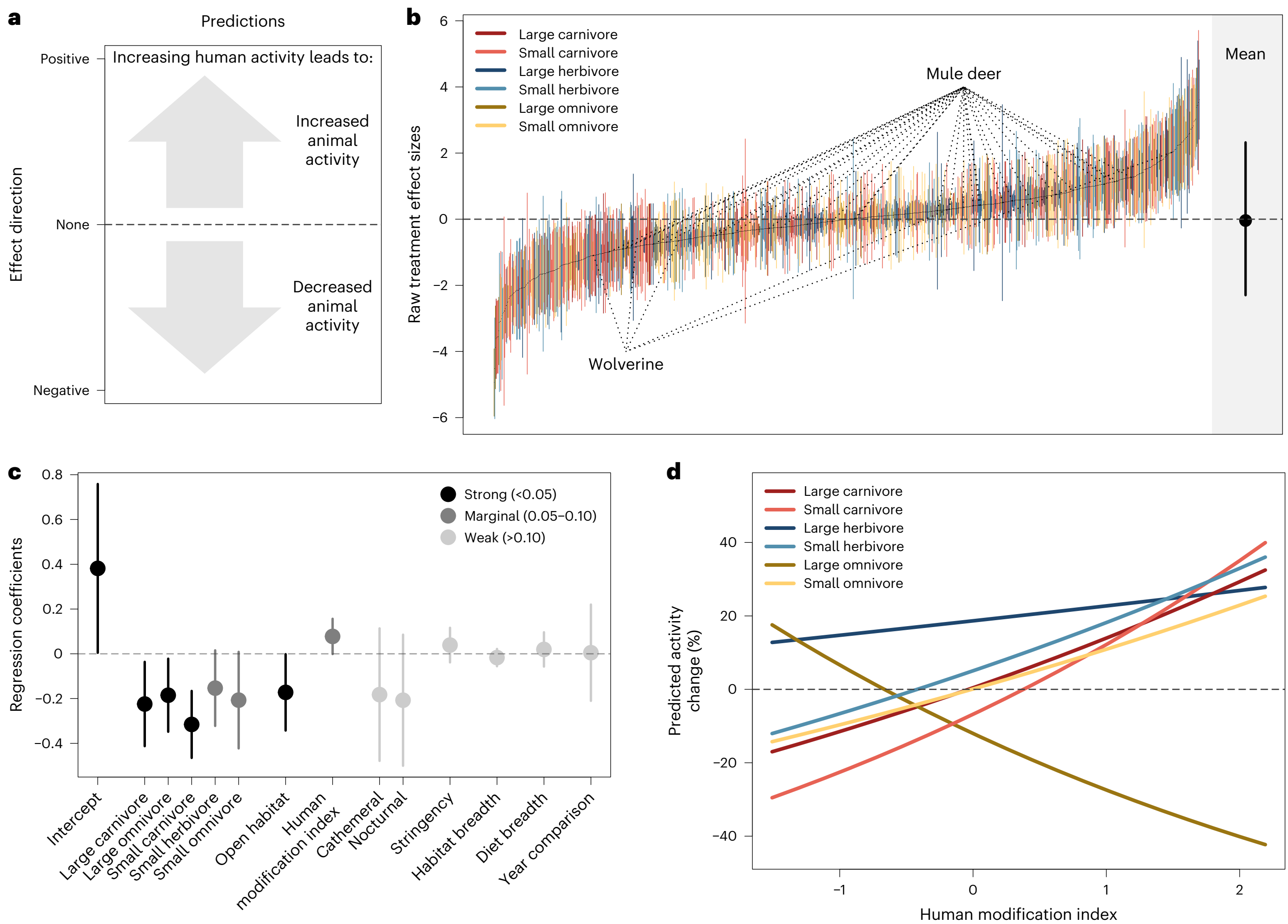

Fig. 2 | Changes in the amount of animal activity in response to increasing human activity. a, Interpretation of effects. **b**, Estimated effect sizes (black points) and variances (coloured lines) for all populations included in the analysis ($n = 1,065$ project–species combinations from 102 independent projects; two example species highlighted) with the global mean (and 95% quantiles) plotted in black to the right. **c**, Estimated model coefficients (points) and 95% CIs (lines; $n = 1,065$ project–species combinations from 102 independent projects) for additive factors (with complete data; Methods) hypothesized to influence changes in the amount of animal activity when human activity is higher, where: intercept is diurnal, large herbivore in closed habitat type with a seasonal comparison and all other effects are contrasts. **d**, Model predictions for the interaction between trophic group and HMI.

sensitivity of large carnivores to the increased risk of conflict associated with more human presence[26], such that they shift timing of activity to minimize overlap regardless of landscape context. Other groups increased night time activity only in landscapes with higher risk of human encounters (that is, more modification), which may in turn enable the increases in amount of activity observed for many of these species (Fig. 2d).

Unlike for the amount of activity, changes in the timing of animal activity were mediated by the hunting status of species in an area, whereby hunted animals showed stronger increases in nocturnal behaviour at higher levels of landscape modification (+26.6%) relative to their non-hunted counterparts (+13.5%; Fig. 3e and Supplementary Table 8). We did not find strong evidence that relative brain size was associated with shifts in animal nocturnality, nor that the magnitude of change in the amount of human activity explained variation in animal responses (Fig. 3c and Supplementary Tables 6 and 9). We did find an effect of our comparison type such that, on average, comparisons between years showed larger shifts in nocturnality than within-year comparisons (Fig. 3c and Supplementary Table 6), underscoring the importance of temporal matching to minimize influence of other factors such as seasonal changes in activity patterns.

Implications for human–wildlife coexistence

Contrary to popular narratives of animals roaming more widely while people sheltered in place during early stages of the COVID-19 pandemic, our results reveal tremendous variation and complexity in animal responses to dynamic changes in human activity. Using a unique synthesis of simultaneous camera-trap sampling of people and hundreds of mammal species around the world, combined with a powerful before–after quasi-experimental design, we quantified how animals change their behaviours under higher levels of human activity across gradients of human footprint. As the human population continues to grow, the persistence of wild animals will depend on their responses to increasing human presence in both highly and moderately modified landscapes. It may thus be encouraging that many animal populations did not show dramatic changes in the amount or timing of their activity under conditions of higher human activity. Indeed, mean changes across all populations assessed were close to zero, suggesting that there was no global systematic shift in animal activity during the pandemic, consistent with other recent observations of highly variable animal responses[13,27]. Nevertheless, we saw stronger responses to human activity for certain species and contexts and these patterns can help us better understand and mitigate negative impacts of people on wildlife communities.

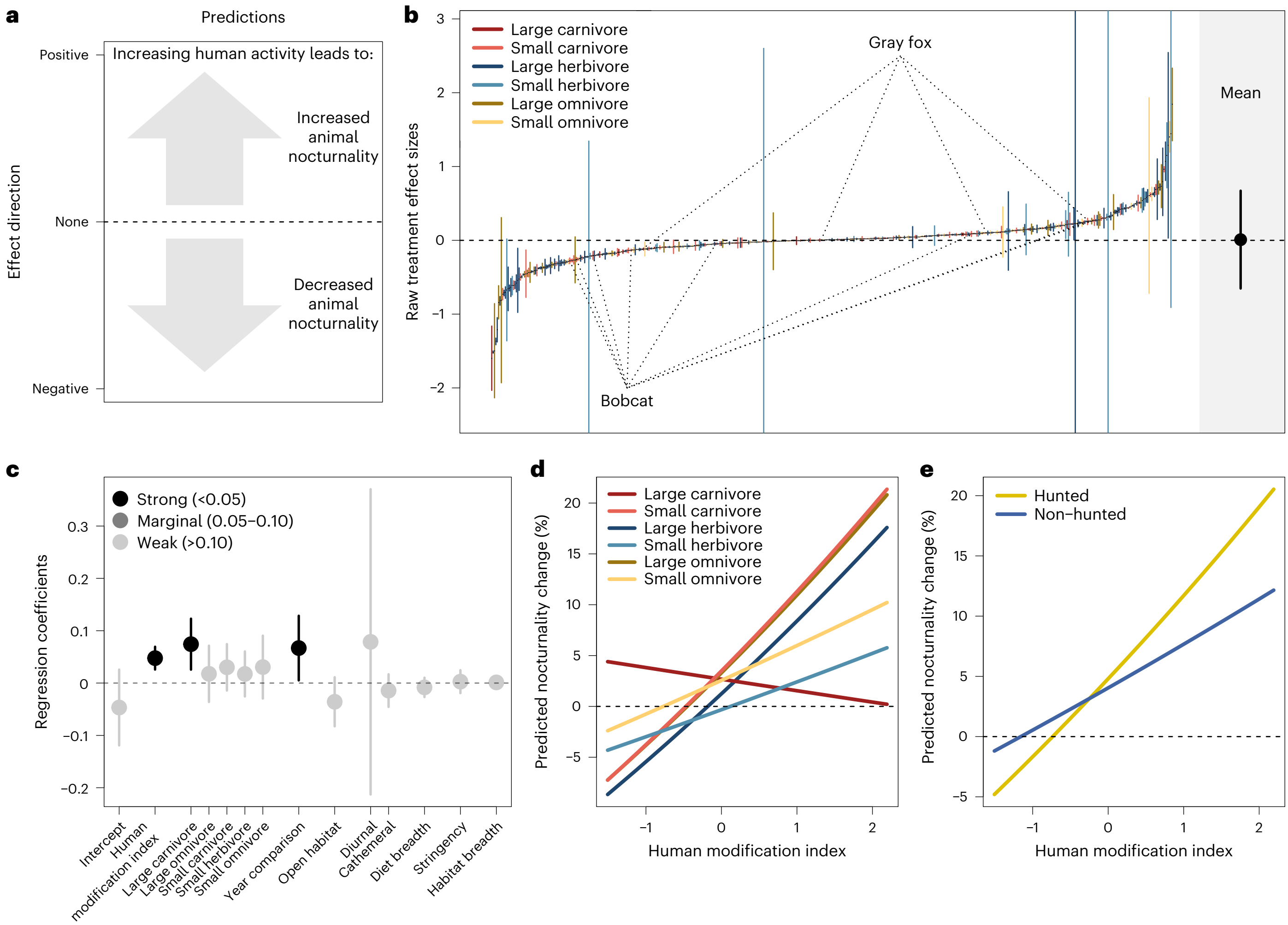

Fig. 3 | Changes in animal nocturnality in response to increasing human activity. a, Interpretation of effects. **b**, Estimated effect sizes (black points) and variances (coloured lines) for all populations included in the analysis (n = 499 project–species combinations from 100 independent projects; two example species highlighted) with the global mean (with 95% quantiles) plotted in black to the right. **c**, Estimated model coefficients (points) and 95% CIs (lines; n = 499 project–species combinations from 100 independent projects) for additive factors (with complete data; Methods) hypothesized to influence changes in animal nocturnality when human activity is higher, where: intercept is nocturnal, large herbivore in closed habitat type with a seasonal comparison and all other effects are contrasts. **d**, Model predictions for interaction between trophic group and human modification index. **e**, Model predictions for interaction between hunting and HMI.

One striking pattern is that animal responses to human activity varied with the degree of human landscape modification. Our results imply that risk tolerance and associated behaviours vary between wildlife in more- versus less-developed contexts. As human activity increased, many species in more modified landscapes surprisingly had higher overall activity, although this activity was more nocturnal, suggesting that animals persisting in these developed environments may be attracted to anthropogenic resource subsidies but still seek ways to minimize encounters with people through partitioning time[28]. Wildlife managers in such modified environments should anticipate some animal habituation and manage the timing of human activity to protect night time refuges that promote human–wildlife coexistence—particularly for hunted species that showed the strongest shifts toward nocturnality. On the other hand, regulating the amount of human activity may be more important in less-developed landscapes where we detected the greatest declines in animal activity with increasing human activity. Such remote landscapes are often spatial refuges for sensitive species that may be filtered out as human modification increases; yet these areas face increasing demands from popular pursuits, such as outdoor recreation and nature-based tourism[18], and may also be more difficult to protect from illegal hunting, encroachment or resource extraction[29].

The sensitivity of species to human footprint and footfalls varied by trophic group and body size, as did the interplay of space and time in behavioural responses. Both large and small carnivore species were among the more sensitive to changes in human activity, generally reducing their activity levels and exhibiting more nocturnality with higher human activity. This motivates a continued emphasis on carnivore behaviour and management as a key challenge for human–wildlife coexistence, given the threatened status of many carnivores, the risk of negative outcomes of human–carnivore encounters and the ecological importance of carnivores as strongly interacting species[7,30]. Avoidance of people by carnivores could be beneficial if it reduces human–carnivore conflict[25,28] but it could also lead to different types of conflict if it results in lower predation rates on herbivores near people, as seen in overbrowsing by habituated deer[4]. Indeed, large herbivores showed the strongest increases in activity with higher human activity in our study, consistent with habituation and increased risk of conflict. Large omnivores, such as bear and boar, were unique in both spatially and temporally avoiding higher human activity in more developed environments, underscoring that management efforts to regulate human activity and create spatial or temporal refuges may lead to outcomes that differ by species and setting. Managers must pay particular attention to the prospect that such differential responses can alter species interactions and cause knock-on effects with broader consequences for ecosystem functions and services[31,32].

Our study highlights the value of learning from unplanned 'experiments' caused by rapid changes in human activity[33] and other extreme events (for example, ref. 34). These insights are enabled by sampling methods, such as camera trapping, that facilitate standardized, continuous monitoring of diverse animal assemblages and humans across varied landscape contexts. While many studies of the anthropause focused on wildlife observations by volunteers in more accessible urban environments (for example, ref. 35), our results emphasize that animal responses to changes in human activity differ between more- and less-developed landscapes. This context-dependency should be a focus of further research, including expanded assessment of contexts and species under-represented in our sample, such as those in tropical regions subjected to different pressures during the pandemic[36]. Many geographic and taxonomic gaps in global biodiversity monitoring remain and must be filled by cost-effective networks that gather reliable evidence across several scales; standardized camera-trap programmes and infrastructure are helping to do so[37,38]. As the cumulative effects of the human enterprise put pressure on ecosystems worldwide[39], bending the curve of biodiversity loss will require context-specific knowledge on ecological responses to human actions that can guide locally appropriate and globally effective conservation solutions.

Methods

Data collection

We issued a call in September 2020 to camera-trap researchers around the world for contributions of camera-trap data from before and during the onset of the COVID-19 pandemic and associated restrictions on human activity[10,11]. This initial call included a social media post (Twitter, now X) and targeted emails to 143 researchers in 37 countries. We requested datasets that adhered to global camera-trap metadata standards (Wildlife Insights[38]) and received submissions from 146 projects. Submitted data were summarized using a standardized script and evaluated according to the following key criteria: (1) most or all camera-trap stations were deployed in the same area of interest (hereafter site) before and during COVID-19-related restrictions; (2) a minimum of seven unique camera-trap deployment locations (stations) were sampled; (3) a minimum sampling effort of at least 7 days per camera period (see below); and (4) trends in human detections were recorded from camera-trap data (that is, detections of humans) or human activity for a given sampling area was available from other sources (for example, lockdown dates and local knowledge).

We only included detections of wild mammal species ≥1 kg (mean species body mass in kg obtained from ref. 40; we excluded domestic animals, which represented only 6% of overall detections and were associated with humans) and humans (excluding research personnel servicing cameras). Our full dataset for the next step of analysis included 112 projects sampling across 5,653 cameras for 329,535 camera-days (see below for data included in specific models). The mean number of camera locations per project was 42 (range 6–300) and mean camera-days per project was 2,945 (range 348–27,986). Camera locations were considered independent within projects, as no paired cameras were included (see Supplementary Table 10 for more details on camera deployments and spacing).

Experimental design

For each project, we first reviewed site-level trends in independent detection events of humans (using a standardized 30 min interval: that is, a detection was considered independent if >30 min from previous detection at the same camera station) to identify whether there were changes in human activity associated with COVID-19 restrictions in 2020. We sought to identify two comparable sampling periods that differed in human activity but were otherwise similar (for example, in camera locations and sampling effort) and thus could be used as a quasi-experimental comparison to assess wildlife responses to the change in human activity. We initially anticipated that human activity would be reduced during COVID-19 lockdowns (that is, the anthropause[11]) but observed a wide variety of patterns of human detections across datasets, including decreases, increases and no change in human detections between sampling before and during COVID-19 (Supplementary Fig. 1). Since our primary interest was in evaluating wildlife responses to changes in human activity and in general we anticipate increases in human activity during the Anthropocene, we standardized our treatments to represent increases in human activity. In other words, we defined a 'control' period as one with lower human activity and a 'treatment' period as one with higher human activity, regardless of which occurred before or during the COVID-19 pandemic (Fig. 1).

We identified start and end dates for each period on the basis of clear changes in human detections (determined from visual inspection of daily detections; Fig. 1). For some projects, dates corresponded to known dates of local COVID-19 lockdowns or changes in study design (for example, dates of camera placement or removal). We prioritized comparison between years when data were collected in similar periods in years before 2020 (n = 95 projects). If multiyear data were not available, we selected comparison periods before and after the onset of lockdowns around March 2020 (with specific dates chosen according to local lockdown conditions; n = 17). If there were several potential treatment periods, we prioritized periods on the basis of the following ordered criteria: (1) the fewest seasonal or ecological confounds; (2) the most similar study design; (3) the greatest sampling effort; and (4) the most recent time period. Of the 95 projects for which we made comparisons between 2020 and a previous year, we used 2019 for 88 projects, 2018 for 6 and 2017 for 1.

In cases where there was no noticeable difference in human detections between candidate periods, or there were insufficient human detections from camera traps, we used other data or local knowledge of changes in human activity (for example, lockdown dates and visitor use data) from co-authors responsible for the particular project. Of the 112 projects included in our initial analyses, 15 used this expert opinion to determine changes in human activity. After completing our initial categorization of comparison periods, we shared details with all data contributors for review and adjustment, if necessary, based on expert knowledge of a given study area. Contributors were asked whether our delineation of sampling periods as being high versus low in human activity corresponded with their knowledge of the study system. We also asked them to consider whether other sources of environmental variation (for example, fire, drought, seasonal or interannual variation) or sampling design could confound the attribution of changes in wildlife detections to changes in human activity. After this evaluation and review, we retained 102 project datasets that had a detectable change in human activity between a treatment and control period for subsequent statistical modelling. These projects spanned 21 countries, mostly in North America and Europe but with some representation from South America, Africa and Southeast Asia (Fig. 1 and Supplementary Table 10).

Our paired treatment–control design makes several assumptions. For instance, we assumed that either: (1) changes in human activity occurred in the same direction throughout the entire study area within the treatment period; (2) the direction of the average effect was more important than variation in direction across camera sites; (3) variation in human activity within a study area was lower than differences in human activity between the treatment (higher activity) and control (lower activity) periods. By standardizing our treatment to be the period of higher human activity, we also assumed that the temporal direction of change did not affect animal responses.

Data analysis

We compared two response variables between treatment and control periods to assess wildlife responses to changes in human activity: the

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amount of animal activity and the timing of animal activity (described below). We used a two-stage approach in which we first estimated the direction and magnitude of change in these responses between periods for each species and then used a meta-analytical approach to evaluate the degree to which a set of candidate predictor variables explained variation in estimated responses. All data manipulation and analysis were done using R statistical software (v.4.1.3; ref. 41).

Amount of animal activity. To evaluate changes in the amount of animal activity, we quantified detection rates for each mammal species (and humans) at each camera for the treatment and control periods of each project. Specifically, we calculated the number of independent detections for a given species and camera station using a standardized 30 min interval (that is, detection was considered independent if >30 min from previous detection of the same species at the same camera station), while controlling for variation in sampling effort (log of camera-days included as an offset in models). We assumed that this detection rate (sometimes termed relative abundance index[16]) measured the relative intensity of habitat use by a species at a camera station, which reflects both the local abundance of the species (number of individuals in sampled area) and the movement patterns of individuals.

To quantify the magnitude of change in the amount of animal activity, we first ran single-species models to estimate changes in detection rates for species and humans between the comparison periods for each project. The response variable was the count of independent detection events, modelled as negative binomial, with an offset for active camera-days. Treatment was included as a fixed effect and a random intercept was included for camera station where the same camera locations were sampled in both periods (no random effect was included if a project used different camera locations between periods). All models were implemented using the glmmTMB package[42]. These models produced a regression coefficient (effect size) for each project–species population (humans and animals) representing the estimated magnitude of change in the amount of activity between the control period and the treatment period (and its corresponding sampling variance).

Timing of animal activity. To assess changes in timing of animal activity, we first classified each independent detection of a given species within a given project as 'day' or 'night'. We used the lutz package to convert all local times to UTC[43]. We calculated the angle of the sun at the time of the first image in each detection using the sunAngle function in the oce package[44], based on the UTC time and latitude and longitude of the camera deployment location. Negative sun angles corresponded to 'night' (between sunset and sunrise) and positive sun angles to 'day' (between sunrise and sunset). Following ref. 24, we calculated an index of nocturnality, N, as the proportion of independent camera-trap detections that occurred during the night (N = detections during night/ (detections during night + detections during day)) for all species which had ten or more detections in both the control and treatment periods. We then calculated the log risk ratio, RR and its corresponding sampling variance (weighted by sample size) between the treatment and control periods, pooled across all camera traps within a given study using the escalc() function within the metafor package[45]. This effect size compared the percentage of animal detections that occurred at night with high human activity (N_h) to night time animal activity under low human activity (N_l), with $RR = \ln(N_h/N_l)$). A positive RR indicated a relatively greater degree of nocturnality in response to human activity, while a negative RR indicated reduced nocturnality.

Hypothesized explanatory variables. We identified and calculated a set of variables that we hypothesized would affect species responses to changes in human activity. These fell into four general classes: (1) species traits, (2) habitat (that is, vegetation) structure, (3) anthropogenic landscape modification and (4) magnitude of human change (Table 1). We did not include any covariates reflecting differences in camera-trap sampling protocols between projects, as our estimates of species responses were made within projects (that is, comparing treatment versus control periods) and thus sampling methods were internally consistent within projects (for example, camera placement and settings).

Species traits. We hypothesized that species with the following traits would be more sensitive to changes in human activity (that is, more vulnerable or risk averse): larger body mass[46], higher trophic level[46], narrower diet and habitat breadth[47], diurnal activity[46] and smaller relative brain size[48]. We extracted variables for each species from the COMBINE database[40], the most comprehensive archive of several mammal traits curated to date (representing 6,234 species). Given that some traits in the database were imputed, we reviewed the designations for plausibility and cross-referenced the traits with other widely used databases—specifically Elton Traits[49] and PanTHERIA[50]—and made the following corrections to the 'activity cycle' trait (diurnal, nocturnal and cathemeral): diurnal to cathemeral—*Mellivora capensis, Neofelis nebulosa, Neofelis diardi*; diurnal to nocturnal—*Meles meles;* nocturnal to diurnal—*Phacochoerus africanus;* nocturnal to cathemeral—*Ursus americanus*. To calculate relative brain size we divided log-transformed brain mass by log-transformed body mass (as in ref. 48). We combined body mass and trophic level into a new variable 'trophic group' (consisting of small- or large-bodied categories for each of the three trophic levels, Table 1). Dietary and habitat breadth are described in ref. 40.

We further hypothesized that animals in hunted populations would be more sensitive to changes in human activity. We requested that all data contributors complete a survey indicating whether a given species was hunted within their project survey area, from which we created a binary factor representing hunting status for each population (1 = hunted; 0 = not hunted).

Habitat structure. Camera-trap surveys included in our analysis covered an extensive range of biogeographic areas and habitat types. We made the simplifying assumption that species responses to changes in human activity would be most influenced by the degree of openness of habitat (that is, vegetation structure) in a sampling area. More specifically, we hypothesized that areas with more open habitat types would have higher visibility and thus less security cover for animals and thus that animals in these open habitats would be more sensitive to increases in human activity than would animals in more closed habitats with greater security cover[51]. We used the Copernicus Global Land Cover dataset (100 m resolution[52]) via Google Earth Engine to extract land cover class at each camera station. We then used the percentage canopy cover of the mode class across all cameras in a given project to define if the survey occurred in primarily closed (>70% canopy cover) or open habitat types (0–70% canopy cover).

Land cover disturbance. We posited that animal responses to changes in human activity would differ according to the degree of anthropogenic landscape modification (that is, human footprint[1,53]). More specifically, we identified two hypotheses that could underlie variation in species responses as a function of land cover disturbance. On the one hand, our 'habituation hypothesis' predicts that animals in more disturbed landscapes may be less sensitive to changes in human activity (relative to animals in undisturbed landscapes) and thus show less of a negative response or even a positive response as they have already behaviourally adapted to tolerate co-occurrence with people[22]. On the other hand, our 'plasticity hypothesis' predicts that the ability of animals to coexist with people in disturbed landscapes may be dependent on plasticity in animal behaviour[22], such that animals in these landscapes may show more pronounced and rapid responses to changes in human activity (for example, avoidance of areas and times with greater chance of encountering people).

We initially characterized landscape disturbance using three variables accessed via Google Earth Engine: Gridded Population of the World (1 km resolution[54]), road density (m km^{-2}, 8 km resolution; Global Roads Inventory Project[55]) and HMI (for 2016 at 1 km resolution), which represents a cumulative measure of the proportion of a landscape modified by 13 anthropogenic stressors[53]. Point values were extracted for each camera station in each site, then the project-level medians were used in analysis. As the median values of these three variables were highly correlated across projects (Supplementary Fig. 2), we only used HMI in our subsequent models.

Magnitude of human change. We expected that animal responses would be more pronounced in areas that underwent greater changes in human activity and we used two measures to assess the magnitude of those changes. At a coarse scale, we used the COVID-19 stringency index[14], which characterizes the policies restricting human activities within a given geographic region at a daily time scale and has been widely used in studies of COVID-19 on human mobility and the environment (for example, ref. 13). We used the finest-scale regional data available for each project, which was usually at the country level, with the exception of three countries with province- or state-level data (Brazil, Canada and the United States). When projects spanned several countries, provinces or states, we used the stringency index for the region in which most cameras were located. For each region, we calculated the median stringency for the treatment and control sampling periods.

At a finer scale, we used the effect size for the modelled change in camera-trap detection rates of humans across all cameras in a project (as described above under 'amount of animal activity'). Models with this variable excluded 15 projects that either did not detect humans with camera traps or the number of humans detected on cameras was not perceived by the data contributor to be an accurate reflection of change in human use for the sampled area.

Meta-analysis models. To understand which factors mediated the effect of increasing human use on animal activity, we ran mixed-effect meta-analytic models using the rma.mv() function of the metafor package[45] on the effect sizes and sampling variances of the two response variables described above (amount and timing of animal activity). Our unit of observation for modelling was the estimated response for each project–species combination (that is, each animal population) and we included random intercepts for project and for species nested within family, to account for repeated observations within each of those higher-level groups and for phylogenetic relatedness within families. All continuous predictor variables (Table 1) were standardized to unit variance with a mean of zero using the stdize function in the MuMIn package[56]. We tested pairwise correlations among all predictor variables and found that none were highly correlated (that is, all below a threshold of Pearson $|r| < 0.6$; Supplementary Fig. 2) and thus all were retained for modelling.

We performed our analysis in three steps for each of the two wildlife response variables. First, we fit a global model including all hypothesized predictor variables for which we had complete data (excluding hunting status, relative brain size and empirical magnitude of human change, for which we had incomplete data and thus included in analysis of subsets of data, described below). Second, we used model selection to test for plausible interactions and nonlinear effects. Third, we used model selection on subsets of the full data to compare the global and interactions models with candidate models adding three more predictor variables with incomplete data.

Global model. As all of our predictor variables were independent, we used a global model approach that included additive fixed effects for all predictor variables (Table 1). We interpreted the P value of each effect contrast to indicate statistically significant support (at $P < 0.05$ or marginal support at $P < 0.10$) for a consistent effect direction of a given predictor and we used the estimated effect size as a measure of effect magnitude. We calculated the pseudo-R^2 to estimate the total variation explained by our global models. We also calculated the I^2 (ref. 57) of each global model to determine the amount of heterogeneity observed between the random effect levels; consistent variation in the response terms between projects, families and species would result in higher I^2 values compared to the null model with no fixed effects. To aid interpretation, we present effect sizes in terms of the proportional change (%) in model-predicted responses across lowest-to-highest values for continuous predictors (for example, HMI) or between two categories of interest (for example, trophic groups).

Model selection of plausible interactions and nonlinear terms. To explore the possibility of context-specific effects of the predictors of wildlife responses to changes in human activity, we assessed a suite of ecologically plausible interaction and nonlinear (quadratic) terms through adding them in turn to the global model and using Akaike's Information Criterion (corrected for small sample size, AICc) to find the most parsimonious model. We assessed the following terms: (1) 'HMI * habitat_closure', to evaluate the potential for habitat structure to mediate responses to human landscape modification; (2) 'trophic_group * HMI', to evaluate the potential for different trophic groups to respond to human modification in different ways; (3) 'trophic_group * habitat_closure', to evaluate the potential for different trophic groups to respond to habitat structure in different ways; and (4) HMI^2, to assess nonlinear effects of wildlife responses to human modification. Models including the candidate interaction or nonlinear terms were compared to the global model without interaction terms using AICc (in the MuMIn package[56]) and were discussed above if they were within 2 AICc of the best-supported model and there was no simpler, nested model with more support.

Model selection on subsets of data. We had a small amount of missing information in the data available for assessing the effects of population hunting status, species relative brain size and empirical (that is, camera-trap-based) magnitude of change in human activity (91.7%, 98.8% and 86.5% of project–species had data for these variables, respectively). Therefore, we ran the same global model used for the full dataset on the subsetted data along with candidate models including each of these predictor variables and all plausible interactions of interest (as above). These additional candidate models were compared to the global model (run on the same partial dataset) using AICc and were discussed in the results if they resulted in a lower AICc value (that is, had more support than the global model, which was a simpler nested model).

Reporting summary

Further information on research design is available in the Nature Portfolio Reporting Summary linked to this article.

Data availability

The data used in this paper are available in Figshare, with the identifier: https://doi.org/10.6084/m9.figshare.23506536.

Code availability

The code used to analyse the data and create the figures in this paper are available in Figshare, with the identifier: https://figshare.com/articles/software/Analysis_R_Code/23506512.

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Author contributions

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Competing interests

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☐	☒	For hierarchical and complex designs, identification of the appropriate level for tests and full reporting of outcomes
☐	☒	Estimates of effect sizes (e.g. Cohen's *d*, Pearson's *r*), indicating how they were calculated

Our web collection on statistics for biologists contains articles on many of the points above.

Software and code

Policy information about availability of computer code

Data collection	No software was used to collect data
Data analysis	R Statistical Software, version 4.1.3. Code for data analysis are available on FigShare (link provided in Code Availability statement in paper).

For manuscripts utilizing custom algorithms or software that are central to the research but not yet described in published literature, software must be made available to editors and reviewers. We strongly encourage code deposition in a community repository (e.g. GitHub). See the Nature Portfolio guidelines for submitting code & software for further information.

Data

Policy information about availability of data

All manuscripts must include a data availability statement. This statement should provide the following information, where applicable:

- Accession codes, unique identifiers, or web links for publicly available datasets
- A description of any restrictions on data availability
- For clinical datasets or third party data, please ensure that the statement adheres to our policy

All data and code have been made available on FigShare with the links provided in the Data Availability and Code Availability statements in the paper.

Human research participants

Policy information about studies involving human research participants and Sex and Gender in Research.

Reporting on sex and gender	NA (our study did not involve human research participants)
Population characteristics	NA
Recruitment	NA
Ethics oversight	NA

Note that full information on the approval of the study protocol must also be provided in the manuscript.

Field-specific reporting

Please select the one below that is the best fit for your research. If you are not sure, read the appropriate sections before making your selection.

☐ Life sciences ☐ Behavioural & social sciences ☒ Ecological, evolutionary & environmental sciences

For a reference copy of the document with all sections, see nature.com/documents/nr-reporting-summary-flat.pdf

Ecological, evolutionary & environmental sciences study design

All studies must disclose on these points even when the disclosure is negative.

Study description	Comparison of amount and timing of animal activity between paired treatment (higher human activity) and control (lower human activity) time periods using detections from 5400 camera traps for 163 species across 102 survey areas. Unit of comparison was the population (species-project), with 1065 for amount of activity and 499 for timing of activity.
Research sample	Sample of terrestrial mammals detected at motion-triggered camera traps across 102 survey areas around the world. Surveys were identified opportunistically as those with active camera trap sampling before and during the COVID-19 lockdowns in 2020. Within surveys, cameras were deployed randomly or systematically to detect medium- and large-bodied terrestrial mammals (≥ 1 kg), including humans.
Sampling strategy	We included surveys for which: most or all camera trap stations were deployed in the same area of interest before and during COVID-19-related restrictions; a minimum of 7 unique camera trap deployment locations were sampled; a minimum sampling effort of at least 7 days per camera period; and trends in human detections were recorded from camera trap data or human activity was available from other sources.
Data collection	Mammals photographed by camera traps were identified from images by researchers from each project. The date and time of each detection was recorded, as was the location of each camera trap.
Timing and spatial scale	The spatial scale includes the entire world as we considered camera trap surveys from anywhere that met our criteria (listed above under sampling strategy). The timing of sampling varied by project, and across all projects spanned from 2017-2020, with most sampling between 2019-2020.
Data exclusions	We received data submissions from 146 projects, of which 112 met our sampling criteria (described above). We analyzed data from 102 projects, excluding 10 projects that did now show any change in human activity (i.e., no treatment effect).
Reproducibility	This was not a controlled experiment but rather a quasi-experiment based on changes in human activity in response to COVID-19 policies. Our samples were therefore not reproducible.
Randomization	Our comparisons of human and animal activity were between time periods within survey areas, thus controlling for variation between survey areas.
Blinding	Animals are detected by passive infrared cameras triggered by animal motion and body heat. There is no researcher bias in detecting animals.

Did the study involve field work? ☒ Yes ☐ No

Field work, collection and transport

Field conditions	Our study covered 102 survey areas around the world, and compared animal detections between time periods. Field conditions thus varied substantially
Location	Our study covered 102 survey areas around the world.
Access & import/export	Camera trap sampling is non-invasive and does not involved any capture, handling or collection of animal specimens.
Disturbance	Camera trap sampling is non-invasive and causes minimal disturbance to animals.

Reporting for specific materials, systems and methods

We require information from authors about some types of materials, experimental systems and methods used in many studies. Here, indicate whether each material, system or method listed is relevant to your study. If you are not sure if a list item applies to your research, read the appropriate section before selecting a response.

Materials & experimental systems

n/a	Involved in the study	
☒	☐	Antibodies
☒	☐	Eukaryotic cell lines
☒	☐	Palaeontology and archaeology
☐	☒	Animals and other organisms
☒	☐	Clinical data
☒	☐	Dual use research of concern

Methods

n/a	Involved in the study	
☒	☐	ChIP-seq
☒	☐	Flow cytometry
☒	☐	MRI-based neuroimaging

Animals and other research organisms

Policy information about studies involving animals; ARRIVE guidelines recommended for reporting animal research, and Sex and Gender in Research

Laboratory animals	The study did not involve laboratory animals.
Wild animals	No animals were captured or handled. Animals were observed through photographic records obtained by passive infrared remotely triggered cameras
Reporting on sex	Both sexes for all species were recorded by photographs. There is no reason to expect any sex bias in sampling.
Field-collected samples	Study did not involve samples collected from the field (only photographs)
Ethics oversight	No ethical approval is required for non-invasive photographic sampling by passive, remote infrared cameras

Note that full information on the approval of the study protocol must also be provided in the manuscript.

Appendix H

Status of Montana's mountain goats: A synthesis of management data (1960–2015) and field biologists' perspectives

Final report: 1 May 2017

Photo credit: Bruce Smith

Bruce L. Smith
U. S. Fish and Wildlife Service (retired)

Nicholas J. DeCesare
Montana Fish, Wildlife and Parks

Montana Fish, Wildlife & Parks

Suggested citation: Smith, B. L., and N. J. DeCesare. 2017. Status of Montana's mountain goats: A synthesis of management data (1960–2015) and field biologists' perspectives. Final report, Montana Fish, Wildlife and Parks, Missoula.

EXECUTIVE SUMMARY

We synthesized population survey and harvest data collected by Montana Fish, Wildlife and Parks (MFWP) staff over the past 60 years for the state's mountain goat (*Oreamnos americanus*) populations. In addition, we surveyed 18 MFWP biologists who manage goats in Regions 1–5 to learn more about the populations for which they have management responsibility. We summarized their written questionnaire responses to evaluate the current status and management circumstances of Montana's mountain goats.

Mountain goats distributions in Montana include historic ranges as well as mountainous areas into which goats have expanded from introductions of animals to non-native habitat. In 2016 an estimated 3,685 mountain goats were managed by MFWP, 2,526 (69%) in introduced populations, and 1,159 (31%) in native populations. Another 2,225 goats inhabited the Montana portions of Glacier and Yellowstone National Parks. The most important finding of this work was the dichotomy between native and introduced mountain goats. Compared with population estimates from the 1940s and 1950s, numbers of goats across native ranges (outside Glacier National Park) are 3–4 times fewer today than the 4,100 estimated from surveys during the 1940s. Our survey of MFWP biologists confirmed this decline of native goats. Many of the populations are small and isolated demographically and genetically. Furthermore, both hunting licenses issued for and annual harvests of native populations have declined nearly 10-fold from the 1960s to present. On the other hand, the majority of introduced populations are prospering, with some notable exceptions. Introduced populations now provide the majority of Montana's hunting opportunity. Total goat harvest has declined from the 1960s when 300–500 animals were harvested annually to a relatively stable ≈210 goats annually over the past 30 years. Twelve of Montana's 52 hunting districts (9 with native populations) have been closed to hunting in recent years.

Area biologists provided insights into how they survey and establish harvest prescriptions for populations. They also identified a wide range of management and research needs from which they would benefit in managing and conserving mountain goats. We provide full details of the biologists' answers to a 25-item questionnaire in the attached Appendix.

We identified multiple avenues of management and research for MFWP to consider in future planning efforts: evaluation of statistical power associated with various monitoring protocols, continued maintenance of centralized databases, design of monitoring approaches for long-term consistency, potential development of a statewide species management plan, and research into habitat factors, population dynamics, and causes of mortality of mountain goats.

INTRODUCTION

Among North American native big game species, mountain goats (*Oreamnos americanus*) present many challenges for wildlife management and conservation. They live in remote and harsh environments where traditional monitoring techniques are challenging; they often occur in small isolated populations which are, by definition, more difficult to monitor and face increased risk of declines; and they exhibit life history characteristics that make them particularly susceptible to over-harvest and slow to recover from population declines (Toweill et al. 2004, Festa-Bianchet and Côté 2008). Potentially as a result of some of these challenges, mountain goats have suffered recent population declines across much of the southern portion of the species' native range over the past 50–70 years (Côté and Festa-Bianchet 2003, Festa-Bianchet and Côté 2008, Smith 2014). For example, goat populations in British Columbia have declined by half from an estimated 100,000 in 1960 to 39,000–63,000 in 2010 (Mountain Goat Management Team 2010). Abundance of mountain goats in Washington has declined by 60 percent since 1950 (Rice and Gay 2010). Due to concerns about declines in Alberta, wildlife officials closed the entire province to goat hunting in 1987. Only in 2001 were conservative harvest quotas reinstated there (Hamel et al. 2006).

In Montana, the status of mountain goats is complicated. The western portion of the state supports native populations. To the east, additional populations were established by translocating goats into prehistorically unoccupied habitat (Figure 1). License numbers to hunt native goats have generally been reduced over the past three or four decades, indicating population declines in some areas. Carlsen and Erickson (2008) concluded, "The decline in mountain goat populations is alarming and deserves investigation by Montana Fish, Wildlife and Parks [MFWP]. When goat populations decline, it appears they don't recover."

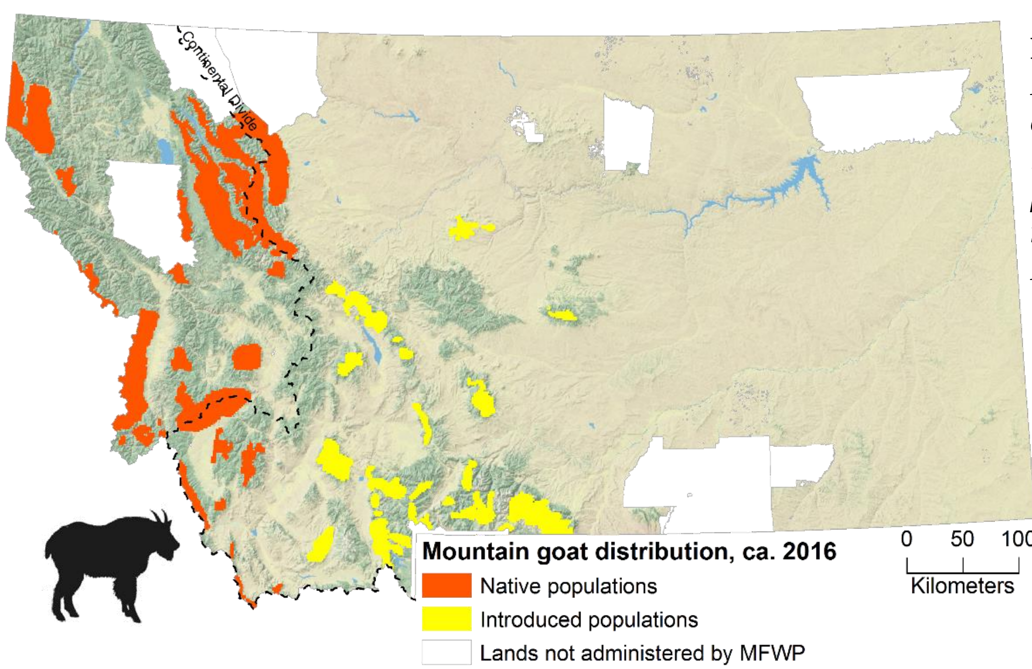


Figure 1. *Distribution of extant native and introduced populations of mountain goats in Montana, 2016.*

Concern over declines in native mountain goat populations are also supported by findings in Alberta, British Columbia, and Washington, which indicate that the mountain goat's natural history may make it particularly sensitive to harvest (and other factors, such as motorized vehicle disturbance) relative to other big game species (Gonzalez-Voyer et al. 2003, Hamel et al. 2006, Mountain Goat Management Team 2010, Rice and Gay 2010).

Contrary to the decline of Montana's native mountain goats, substantial increases have been observed in some introduced populations (Williams 1999, Lemke 2004, Flesch et al. 2016). The transplanting of goats into southwestern and central Montana began over 70 years ago. From 1941 to 2008, 495 animals were transplanted to 27 different sites, with some ranges receiving multiple introductions (Picton and Lonner 2008). Introduced herds in some locations have grown in both numbers and geographic range, while other introductions appeared to have failed, whether immediately or after a period of time.

Carlsen and Erickson (2008) reported that the statewide total goat harvest has been relatively stable over the past 30 years, although this summary may mask markedly different trends occurring among native and introduced populations. A synthesis of historic harvest and monitoring data from each hunting district (HD), and aggregated at larger scales, would elucidate potential shifts in population trends among native and introduced populations, with implications for future conservation of mountain goats and the recreational opportunities they afford.

Montana has a rich history of research into the biology, ecology, and conservation requirements of mountain goats, beginning with the work of Casebeer et al. (1950). Studies during the 1970s and '80s provided the most comprehensive biological information on Montana's native goat populations (Chadwick 1973, Rideout 1974, Smith 1976, Thompson 1980, Joslin 1986). Several studies in the Crazy Mountains provided information on that introduced population's ecology and growth during the 1950s and 1960s (Lentfer 1955, Saunders 1955, Foss 1962). Changes in numbers and distributions of other introduced populations were closely monitored in recent years by MFWP (Swenson 1985, Williams 1999, Lemke 2004). Most recently, Flesch et al. (2016) described range expansion and population growth of introduced goats in the Greater Yellowstone Area.

The aim of this study was to compile and synthesize mountain goat harvest and population information at a statewide scale across Montana over the past 50–60 years, with particular attention to comparing and contrasting dynamics of native and introduced mountain goat populations. We also developed and distributed an expert-opinion survey to solicit the insights and opinions of MFWP personnel (area biologists and/or regional wildlife managers whose jurisdictions include mountain goats) regarding population trends, limiting factors, monitoring practices, and future research and management needs. Summarized results from this survey of MFWP biologists represent the current state of knowledge about Montana's mountain goats, with potential to guide future research, monitoring, and planning efforts aimed at filling information gaps and sustaining or enhancing mountain goat populations and hunting opportunity.

Project Objectives

1. Compile and digitize historical harvest and population monitoring data from MFWP records and reports into a statewide database.

2. Assess trends in mountain goat populations and hunter harvest across Montana, with attention to differences in dynamics among native and introduced populations.

3. Use an expert-opinion questionnaire sent to MFWP personnel to assess the state of knowledge regarding population trends, monitoring practices, limiting factors, and management and research needs for Montana's mountain goats.

OBJECTIVE 1: COMPILE HISTORICAL DATA

We began this project by compiling as much historical data as we could find regarding mountain goat harvest and monitoring. Data sources included:

1. MFWP's internal website databases
    a. Wildlife Information System (WIS), aerial survey data
    b. Wildlife Information System (WIS), hunting and harvest survey data – per HD
    c. Mandatory Reporting Response Entry (MRRE), harvest data – per animal
2. Various electronic data files and reports from area biologists
3. Archived MFWP Survey & Inventory reports from regional office libraries or archives in:
    a. Kalispell
    b. Missoula
    c. Butte
    d. Bozeman
    e. Helena

We organized these data in an electronic database for our analyses. The database will be archived and/or distributed within FWP upon the project's completion. After completing the database, we sent data subsets to each area biologist for review and/or editing of hunting, harvest, and population survey data within their respective jurisdictions. Thus, nearly all of these data have been reviewed by FWP biologists with knowledge about each local area.

The compilation of mountain goat harvest data included >2,200 district-years of data concerning quantities of licenses issued, total numbers of goats harvested, and numbers harvested according to sex. Some data were available as far back in time as 1948 for some HDs. Data for most regions were more consistently available during the period of 1960–2015. Information on the sex, age, and horn measurements for >5,100 individuals was also available via mandatory checking of harvested goats, which began in 1982 and continued through 2015. Other harvest data, such as hunter-days, goats observed, and days per goat seen or harvested, were inconsistently collected over space and time and not deemed suitable for summary in this report.

Population survey data presented challenges to compile because they were not necessarily collected or summarized in reports every year in a way similar to harvest data. We were able to compile data from many population surveys by reading regional survey and inventory reports. Review of population survey data by current FWP area biologists allowed us to fill in many data gaps, although we may still be missing data for certain areas and time periods. To date, we have compiled >700 individual goat population surveys spanning 1942–2016.

OBJECTIVE 2: TRENDS IN HARVEST AND POPULATION SURVEY DATA

Hunter harvest data

We analyzed mountain goat hunter harvest data for the period spanning 1960–2015 (Figure 2). The availability of hunting licenses during this period peaked in 1963 at 1,371 licenses, primarily for hunting of native populations (Figure 2a). Unlimited licenses were available for several native populations in Region 1 at the beginning of the study period in 1960, although regulations for these HDs were gradually switched to limited-draw-based hunting during the subsequent decade. The last unlimited hunting occurred in 1971 in a portion of the Bob Marshall Wilderness, after which only limited licenses were offered in all HDs. In 2015, 16,643 hunters applied to the lottery for 241 goat licenses, with a 1.4% chance of successfully drawing.

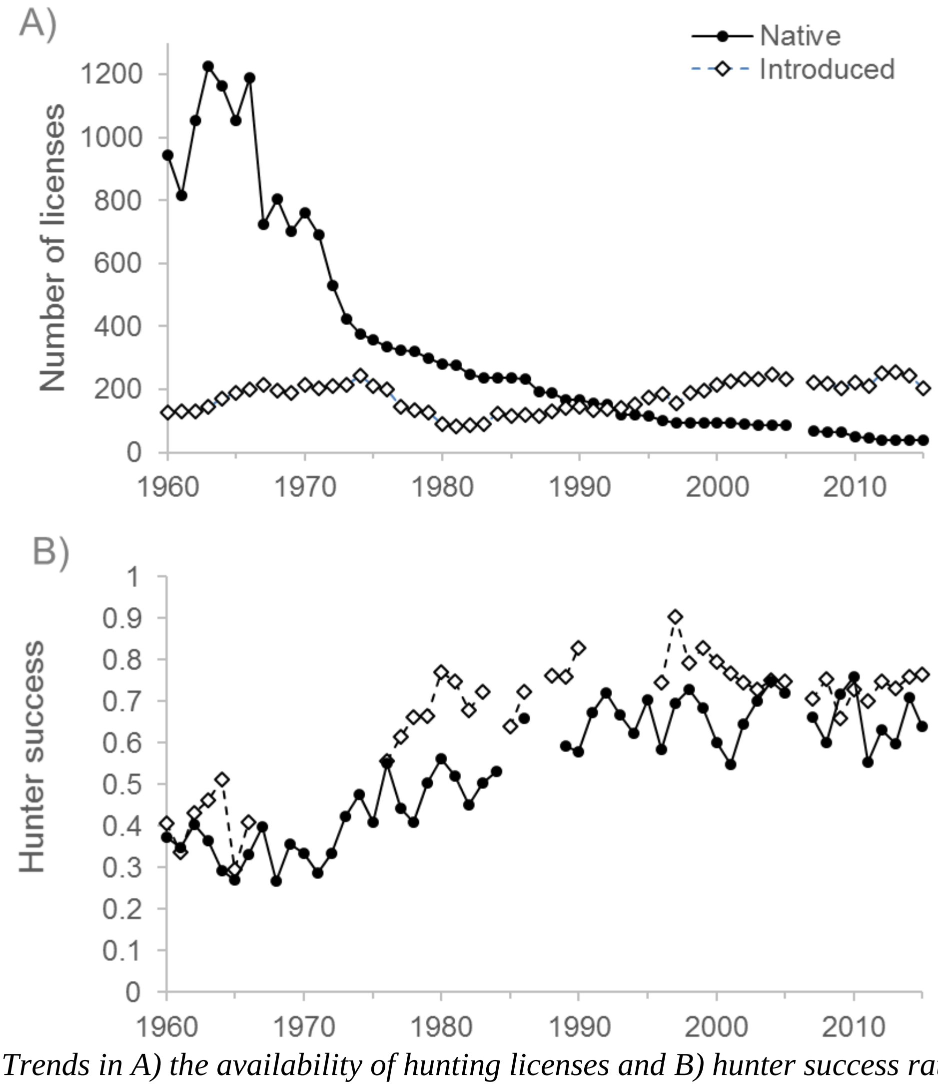


Figure 2. *Trends in A) the availability of hunting licenses and B) hunter success rates (kills per license) for native and introduced populations of mountain goats in Montana, 1960–2015.*

The success rates of hunters, measured as kills per license sold, were lowest during the beginning of this study period, averaging 34% for native populations and 41% for introduced populations during the 1960s (Figure 2b). During subsequent decades, as licenses were reduced in native ranges and increased in introduced ranges, success rates for both increased. Throughout this period, hunter success in introduced range has remained consistently higher than in native range. Thus far during the 21st century (2000–2015), success rates have averaged 65% for hunters of native populations and 74% for hunters of introduced populations. Hunter success rates are typically high and difficult to interpret for special big game species with low-odds license drawings. In such cases, we do not expect trends in hunter success to reflect those of abundance of mountain goats.

Mirroring trends in license availability, total harvest of mountain goats was highest during the early 1960s, peaking at 513 animals in 1963 (Figure 3). By the late 1970s and throughout the 1980s, total harvest became somewhat stable, averaging 216 goats per year during 1977–1989, and ranging from 170–242. Similar harvests have been achieved since, including during the 1990s (mean=212, range=197–228), the 2000s (mean=221, range=184–250), and most recently 2010–2015 (mean=198, range=174–214; Figure 3). Less visible during this 40-year period of stability in total harvest has been a dramatic shift in harvest from native to introduced populations (Figure 3). In the early 1960s, 87–88% of harvested animals were from native populations, averaging 377 native goats harvested per year compared to 55 introduced goats. Since that time, the proportionate harvest of native goats has declined substantially as a result of both reduced licenses in native populations and increased licenses in introduced populations (Figures 3, 4). In 2015, 25 goats were harvested from native ranges compared to 155 from introduced ranges.

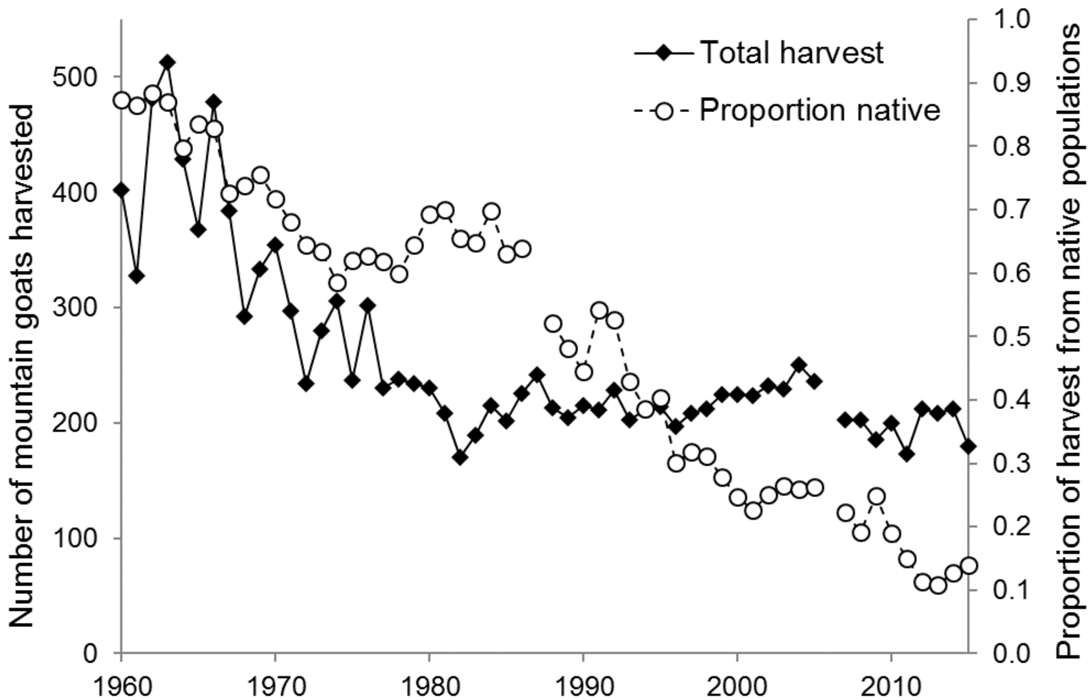


Figure 3. *Total harvest of mountain goats and the proportion of harvest coming from native populations in Montana, 1960–2015.*

When looking at trends in total harvest according to administrative region, large declines in native harvest are evident in Regions 1 and 2 of western Montana. To the contrary, substantial increases in harvest have occurred in introduced populations in Region 3 of southwestern Montana (Figure 4).

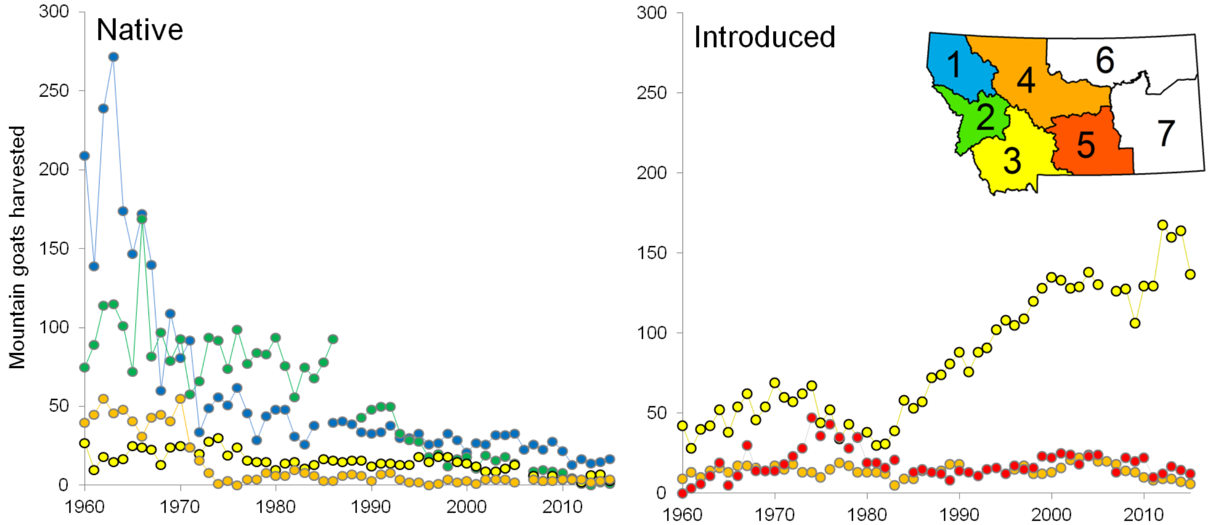


Figure 4. *Numbers of mountain goats harvested from native and introduced populations, by administrative region, in Montana, 1960–2015.*

Unlike other North American ungulates, mountain goats present a unique challenge to hunters and wildlife managers because the sexes are difficult to differentiate in the field. Male and female goats do, in fact, exhibit sexually dimorphic horn characteristics, but these and other subtle differences can be challenging for untrained observers to identify (Smith 1988a). Consequently, MFWP has consistently offered either-sex licenses that allow hunters to legally harvest either a male or female. Harvest of male goats is typically the goal for both wildlife managers (e.g., to harvest animals with lower reproductive value) and for hunters (e.g., to harvest animals with larger trophy scores). To support this goal, MFWP currently offers information and videos on their website as a voluntary educational opportunity for hunters. An exception to either-sex licenses was implemented in 2016 when 25 female-only licenses were issued in the Crazy Mountains HD313. Early indications are that hunters with these licenses were quite adept at successfully identifying and harvesting females during the 2016 season (e.g., preliminary data showed 14 of 14 harvested goats were females, K. Loveless, personal communication).

To assess how hunter education and/or selectivity may have changed in past years, we also summarized the proportion of females within the harvested sample of mountain goats during 1960–2015 (Figure 5). There was no statistical difference in proportionate harvest of females among native and introduced populations (t_{110}=0.543, P=0.588). A decreasing trend in the annual proportion of females in the harvest was evident among both native (β=-0.002, P=0.001) and introduced (β=-0.002, P=0.001) subsets of the statewide harvest, showing an average decrease of 0.2% per year. For example, an average of 42.2% of the annual harvest was females during the 1960s (excluding the outlier value of 18% from 1964), while an average of 30.7% of the harvest was females during 2010–2015.

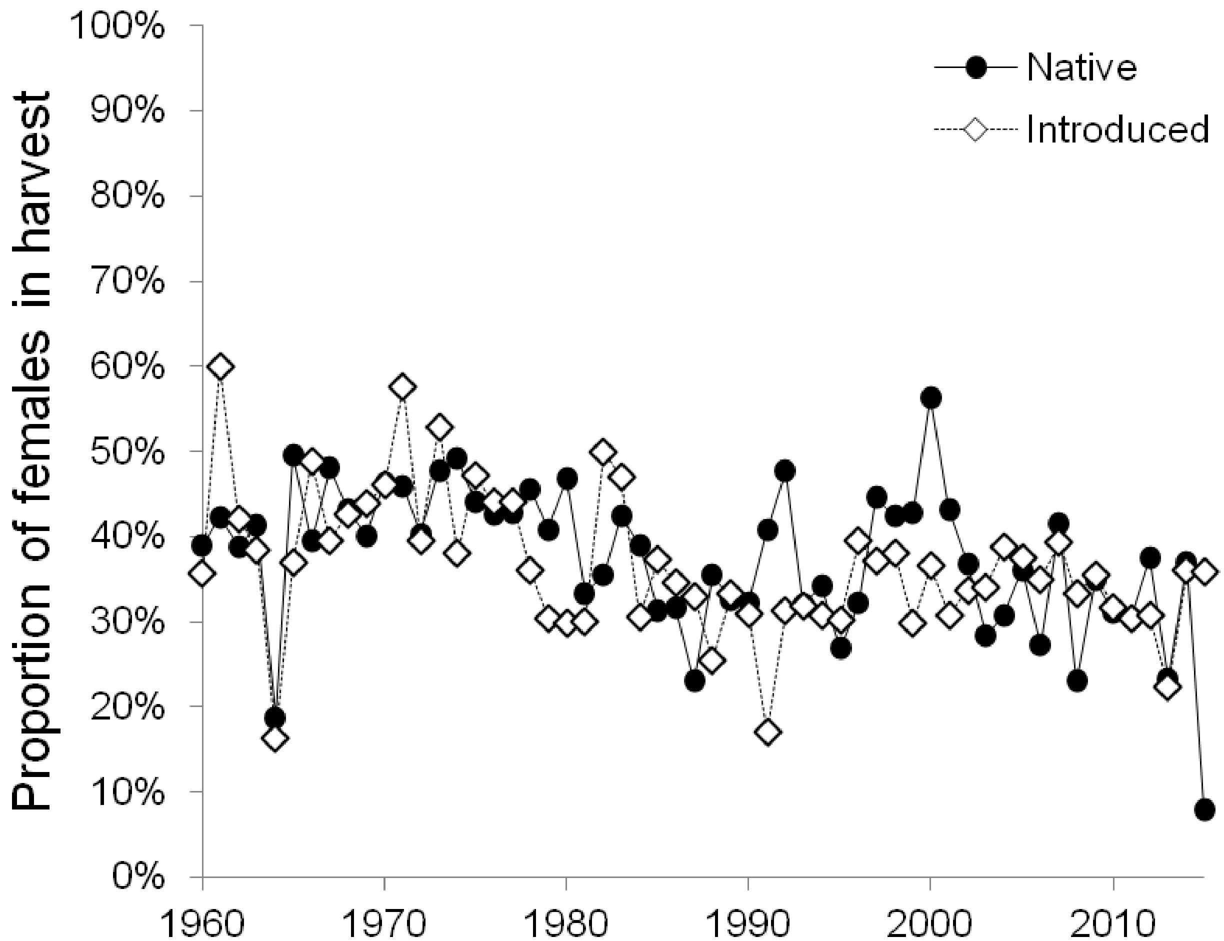


Figure 5. *Proportion of females within the annual harvest of mountain goats, among native and introduced populations, in Montana, 1960–2015.*

In order to compare trends in total harvest among regional populations, we grouped 69 different mountain goat HDs that have been used during various portions of the period 1960–2015 into 28 regional "populations" (Figure 6). The area and number of animals encompassed by each population were not consistent, although we attempted to delineate populations according to logical topographic or ecological boundaries. These groupings included 14 native populations and 14 introduced populations, and we plotted long-term trends in total mountain goat harvest for each (Figure 7). The native population in the Whitefish Range saw no harvest during this period and was eventually deemed as extirpated. Declines in harvest are evident for nearly all native populations (with the possible exception of the Cabinet Mountains) and some introduced populations, while other introduced populations show recent increases in harvest.

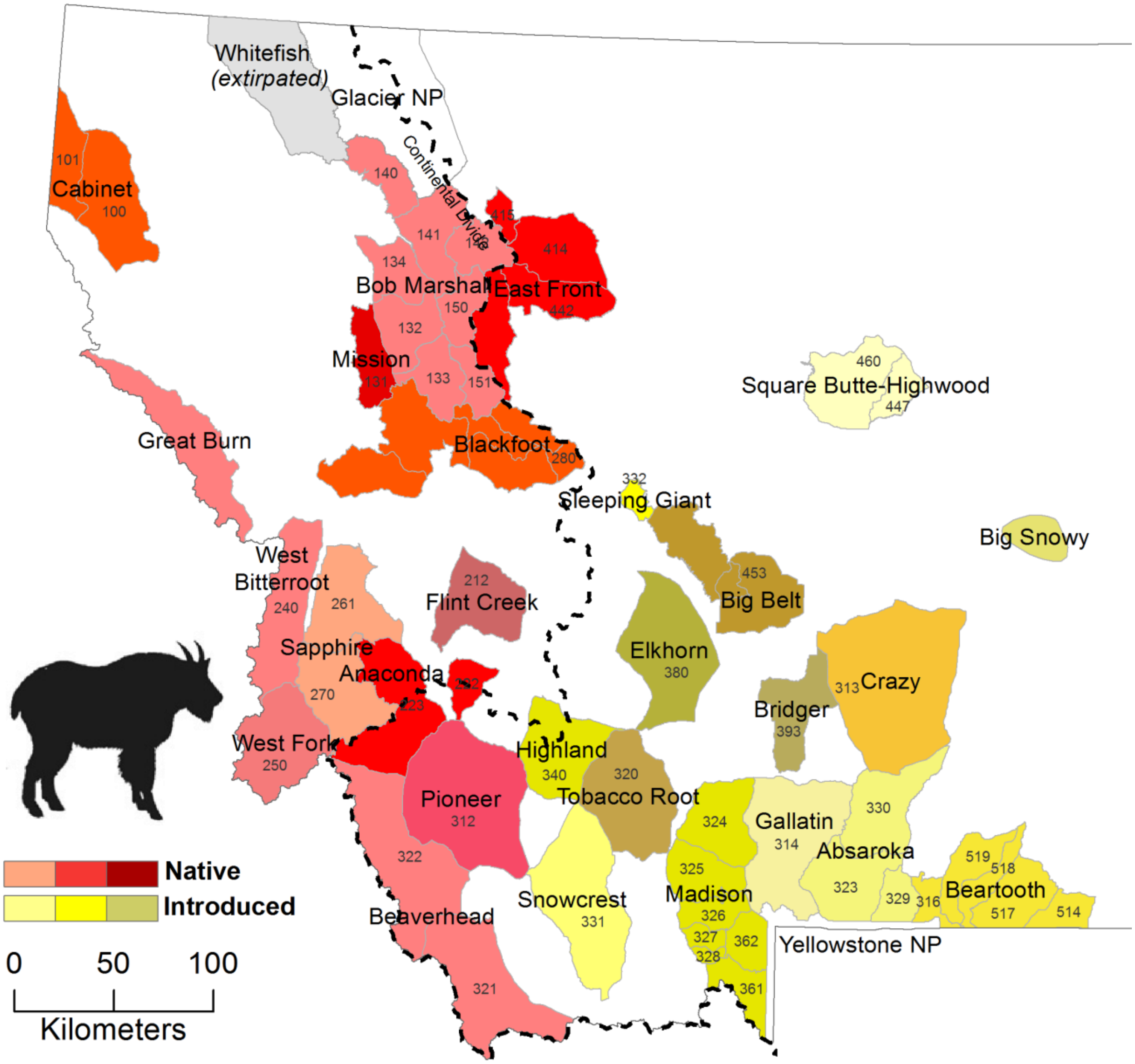


Figure 6. *Hunting districts and regional "populations" of mountain goats in Montana during 1960–2015, which were defined subjectively for purposes of summary within this report. Note: our summaries do not include popuations inside Glacier and Yellowstone National Parks.*

a) Total harvest: Native populations

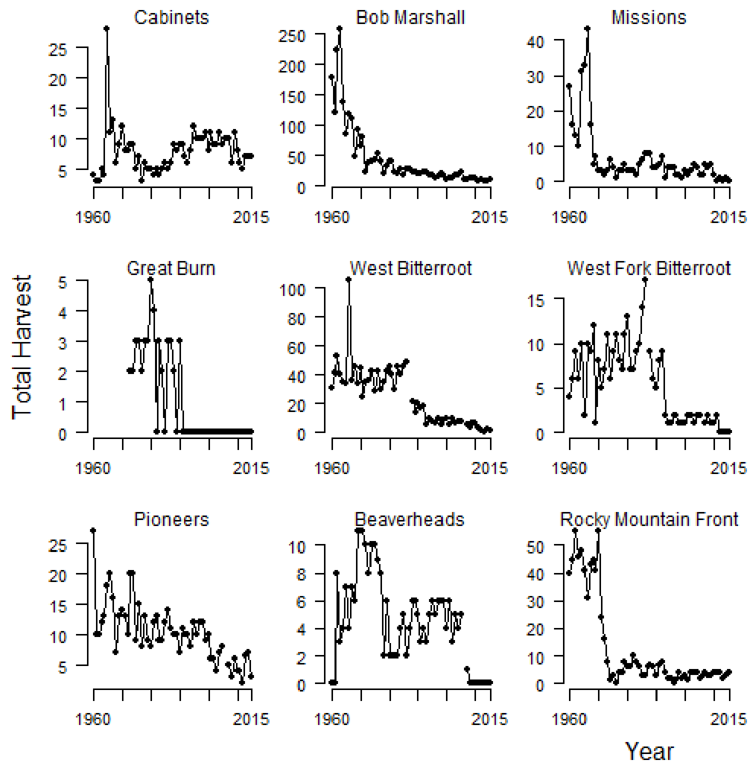
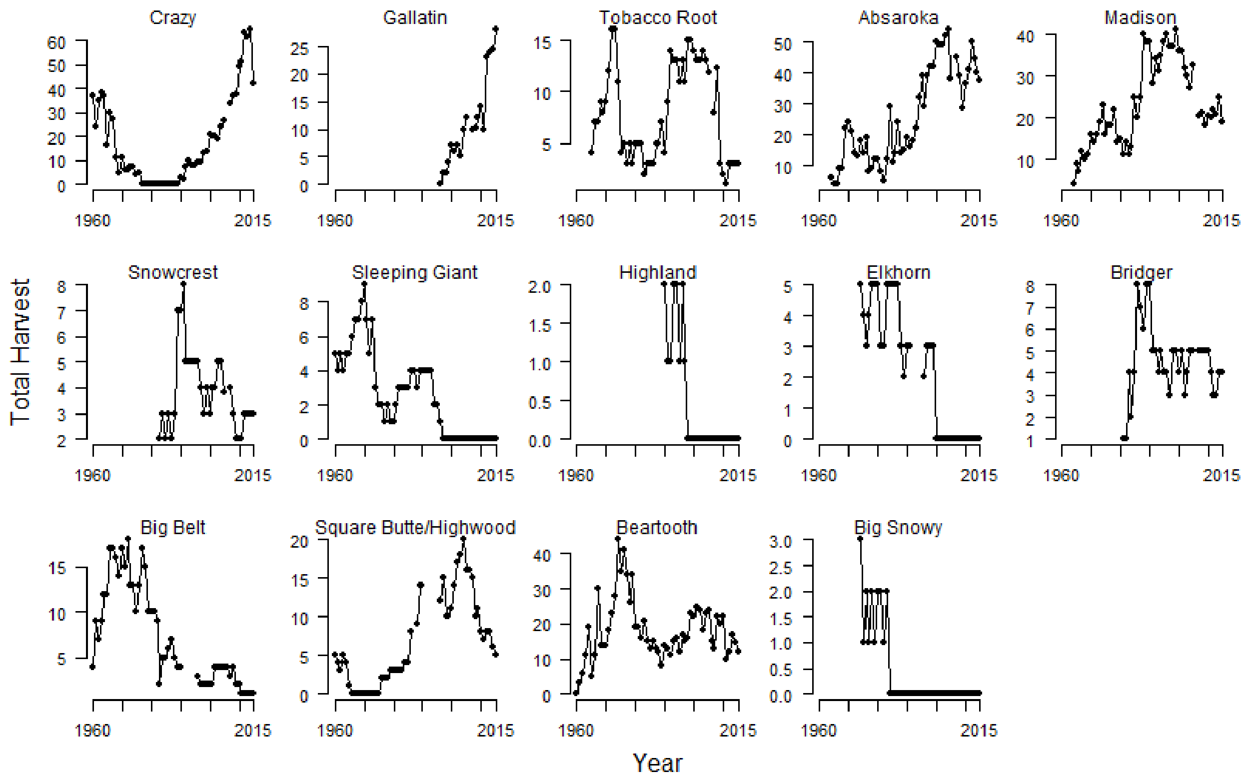


Figure 7. *Trends in total harvest of mountain goats within a) native populations and b) introduced populations (often combining results from multiple neighboring hunting districts, past and present) in Montana, 1960–2015.*

b) Total harvest: Introduced populations



Harvest rates

We estimated contemporary harvest rates of mountain goats by combining hunter harvest data presented here with population estimates developed below via questionnaires to FWP area biologists (see Objective 3). We estimated the "license rate" in 2015 as the number of licenses issued divided by the estimated population size of mountain goats within a given jurisdiction. We estimated the "harvest rate" as the 2015 estimated total harvest of mountain goats divided by the estimated population size (Table 1).

Table 1. *Population estimates, hunting licenses offered, total harvest, and estimated license rate (licenses/population size) and harvest rate (harvest/population size) of mountain goats among regional populations in Montana, 2015. See "Objective 3-Population estimates" below for more information about population estimates.*

	Regional population	Population estimate (Range)	Licenses	Total harvest	License rate	Harvest rate
Native populations	Cabinet	135 (125-155)	8	7	5.9%	5.2%
	Bob Marshall	360 (322-367)	13	10	3.6%	2.8%
	Mission	17 (16-18)	2	0	11.8%	0%
	Whitefish (extirpated)	0	0	0	--	--
	Anaconda	20 (0-40)	0	0	0%	0%
	Blackfoot	40 (20-55)	0	0	0%	0%
	Flint Creek	25 (0-70)	0	0	0%	0%
	Great Burn	23 (20-25)	0	0	0%	0%
	West Bitterroot	100 (80-120)	2	1	2.0%	1.0%
	Sapphire	10 (0-40)	0	0	0%	0%
	West Fork	30 (10-100)	0	0	0%	0%
	Beaverhead	51 (36-66)	0	0	0%	0%
	Pioneer	125 (75-150)	9	3	7.2%	2.4%
	East Front	223 (165-315)	5	4	2.2%	1.8%
Introduced populations	Absaroka	470 (355-538)	58	38	12.3%	8.0%
	Bridger	78 (56-98)	5	4	6.4%	5.1%
	Crazy	450 (330-550)	50	42	11.1%	9.4%
	Elkhorn	20 (9-30)	0	0	0%	0%
	Gallatin	250 (140-275)	30	28	12.0%	11.2%
	Highland	10 (10-15)	0	0	0%	0%
	Madison	617 (447-760)	24	19	3.9%	3.1%
	Sleeping Giant	0 (0-1)	0	0	0%	0%
	Snowcrest	48 (22-48)	3	3	6.3%	6.3%
	Tobacco Root	27 (11-44)	3	3	11.1%	11.1%
	Big Belt	105 (81-130)	2	1	1.9%	1.0%
	Square Butte-Highwood	105 (90-135)	6	5	5.7%	4.8%
	Big Snowy	1 (1-2)	0	0	0%	0%
	Beartooth	345 (290-422)	21	12	6.1%	3.5%

In 2015, MFWP issued a total of 241 mountain goat hunting licenses (39 for native populations, 202 for introduced populations). License holders harvested an estimated 180 mountain goats (25 from native populations, 155 from introduced populations). MFWP biologists estimated a total population of 3,685 mountain goats (1,159 in native populations and 2,526 in introduced populations) on MFWP-administered lands (excluding National Parks and Indian Reservations; see Objective 3). When summing estimates of harvest and goat populations statewide, the estimated statewide license rates in 2015 were 6.5% overall, or 3.4% from native populations and 8.0% from introduced populations. The estimated statewide harvest rates were 4.8% overall, or 2.1% from native populations and 6.1% from introduced populations.

We also estimated license and harvest rates specific to each regional population of mountain goats by grouping data among HDs into populations as described above for harvest trends. Among the 13 extant native populations, 7 were closed to hunting and 6 provided hunting opportunity in 2015. The average license rate among the hunted native populations was 5.5%, and the harvest rate averaged 2.0% (Table 1). Among the 14 introduced populations, 4 were closed to hunting and 10 provided hunting opportunity in 2015. The average license rate among the hunted introduced populations was 7.7%, and the harvest rate averaged 6.3% (Table 1).

Population survey data

We conducted pilot trend analyses of aerial survey data spanning 1960–2015 but found the results difficult to interpret. The availability of data varied substantially among areas and among time periods. The survey areas did not always appear consistent given small populations of goats and often challenging flying conditions, and the timing of surveys also varied in many cases. While consistent and rigorous data were available for several populations, there were many populations for which a consistent stream of data at reasonably high frequency of once per 1–5 years were unavailable within this period. For all of these reasons, we felt formal trend analyses of the survey data would be difficult to synthesize at a statewide scale in a meaningful way.

We instead focused our analysis on survey data collected during the 21st century (2000–2015), and identified 52 survey areas (typically HDs) with at least one survey during this period, for a total of 171 surveys (Table 2). To estimate annual population growth rates, λ, from survey count data, we used exponential growth state-space models developed by Humbert et al. (2009). These models have been shown to more rigorously measure uncertainty surrounding estimates of trend by accounting for process variance (i.e., biological variation) in annual growth rates as well as observation error that induces additional sampling noise around annual count data. Flesch et al. (2016) also used these methods in a recent analysis of mountain goat population trends from survey count data in the Greater Yellowstone Area. Our analysis includes some of the same populations as those studied by Flesch et al. (2016), although we focus only on a recent time period, 2000–2016. This statistical approach has been shown to perform well with a minimum of 5 data points spanning a ten-year survey period (Humbert et al. 2009, Flesch et al. 2016). For our analyses we identified a set of 21 survey areas for which at least 5 surveys for 5 unique years had been conducted. In our case, this spanned a 16-year study period.

We estimated survey-based population growth rates for 5 native populations and 16 introduced populations during 2000–2015 (Figure 8). Survey data were more limited for native than introduced populations. For native populations, point estimates of λ were <1 for 4 of 5 populations, although 95% confidence intervals of λ overlapped 1 for all but one of these (HD 101, West Cabinet Mountains). The estimated population growth rate for the 5th native population was λ=1.0. Among introduced populations, point estimates of λ were <1 for half (8 of 16) of populations and >1 for the other half. Confidence intervals of λ overlapped 1 for 14 of 16 introduced populations, while confidence intervals for the remaining 2 populations (HD 330, North Absaroka, and HD 514, Line Creek) indicated estimates of λ that were significantly <1.

Given the wide confidence intervals surrounding most estimates of λ, little can be said with statistical certainty about trends in survey data for many of these mountain goat populations using survey data alone. Plotting the precision of trend estimates relative to the number of individuals counted per survey area suggested a positive relationship between the magnitude of counts and precision (Figure 9). Thus, statistically rigorous estimates of trends are more difficult to attain under survey conditions of small populations and infrequent surveys.

Among all mountain goat survey areas, with at least one survey during 2000–2015, the average count was 39 animals. For the subset of 21 areas with >5 surveys the average count was 56 animals. When comparing the standard error of estimates of lambda by the magnitude of these counts per area, it appears that there is potential for a high amount of uncertainty (i.e., SE estimates >0.05 would lead to confidence intervals >0.2 units wide surrounding λ) when the average number of goats counted is <100 animals. This would apply to 48 of all 52 survey areas flown during 2000–2015, unless surveys were designed such that data could be pooled among multiple survey areas prior to interpretation. However, a formal power analysis of simulated mountain goat survey data would provide an improved depiction of the precision of trend estimates under various scenarios of monitoring goats with aerial surveys.

Table 2. Mountain goat survey areas and/or hunting districts (HD), the number of surveys conducted during 2000–2015, and the average total count per survey, Montana.

	Regional population	Survey area or HD	$N_{surveys}$	Average count
Native populations	Cabinet	100	7	80 (40-113)
		101	8	36 (7-57)
		121	9	8 (2-17)
		Montanore Mine	6	15 (3-43)
	Mission	131	1	38 (38-38)
	Mission – Bob Marshall	132	2	20 (15-24)
		133	3	27 (4-48)
		134	1	26 (26-26)
		140	1	47 (47-47)
		142	2	38 (20-56)
		150	2	39 (33-44)
		151	2	9 (2-16)
	Anaconda	222 223	2	25 (9-40)
	Blackfoot	283	2	10 (10-10)
		280 (Dunham)	3	27 (24-32)
		280 (Scapegoat)	4	31 (20-37)
	Flint Creek	212	2	19 (13-25)
		213	1	0 (0-0)
	Great Burn	220	2	4 (2-5)
	West Bitterroot	240	6	66 (19-119)
	West Fork Bitterroot	250 (portion)	2	41 (38-43)
	Beaverhead	321	1	7 (7-7)
		322	4	15 (10-19)
	Pioneer	312	4	11 (0-33)
	East Front	414	1	11 (11-11)
		415	3	26 (24-27)
		442 & Sun River Game Preserve	11	46 (22-71)
Introduced populations	Absaroka	323	7	167 (120-221)
		329	7	113 (75-147)
		330	7	27 (17-38)
	Bridger	393	5	54 (25-88)
	Crazy	313	8	288 (190-371)
	Elkhorn	380	2	5 (0-9)
	Gallatin	314	4	128 (34-180)
	Madison	324	3	60 (53-71)
		325	5	33 (25-41)
		326	4	20 (13-24)
		327	5	16 (6-22)
		328	3	4 (2-7)
		362	6	35 (6-74)
	Sleeping Giant	332	5	2 (0-4)
	Snowcrest	331	1	22 (22-22)
	Tobacco Root	320	3	49 (11-84)
	Big Belt	451	8	32 (17-53)
		453	10	30 (2-49)
	Square Butte-Highwood	447	3	53 (35-62)
		460	3	40 (26-50)
	Beartooth	316	10	43 (8-76)
		514 (winter trend area)	10	48 (12-94)
		517 (winter trend area)	10	24 (4-51)
		518 (winter trend area)	10	21 (2-49)
		519 (winter trend area)	5	8 (2-24)

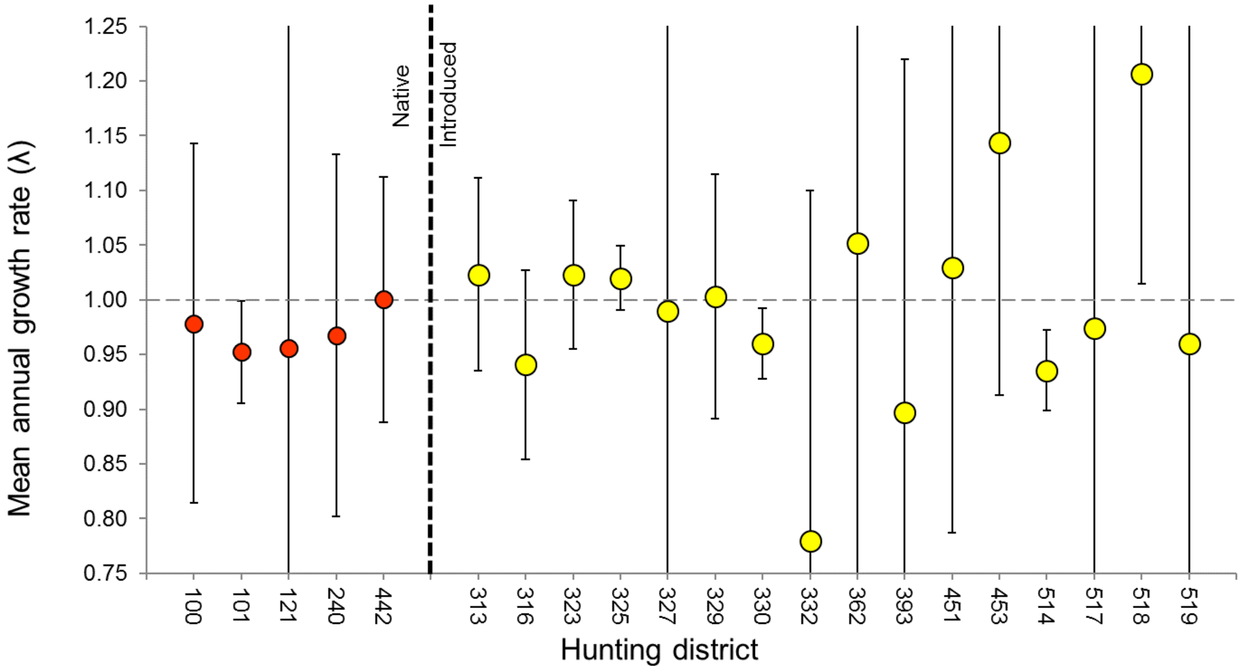


Figure 8. *Mean annual population growth rates and 95% confidence limits for 21 mountain goat survey areas in Montana, 2000–2016.*

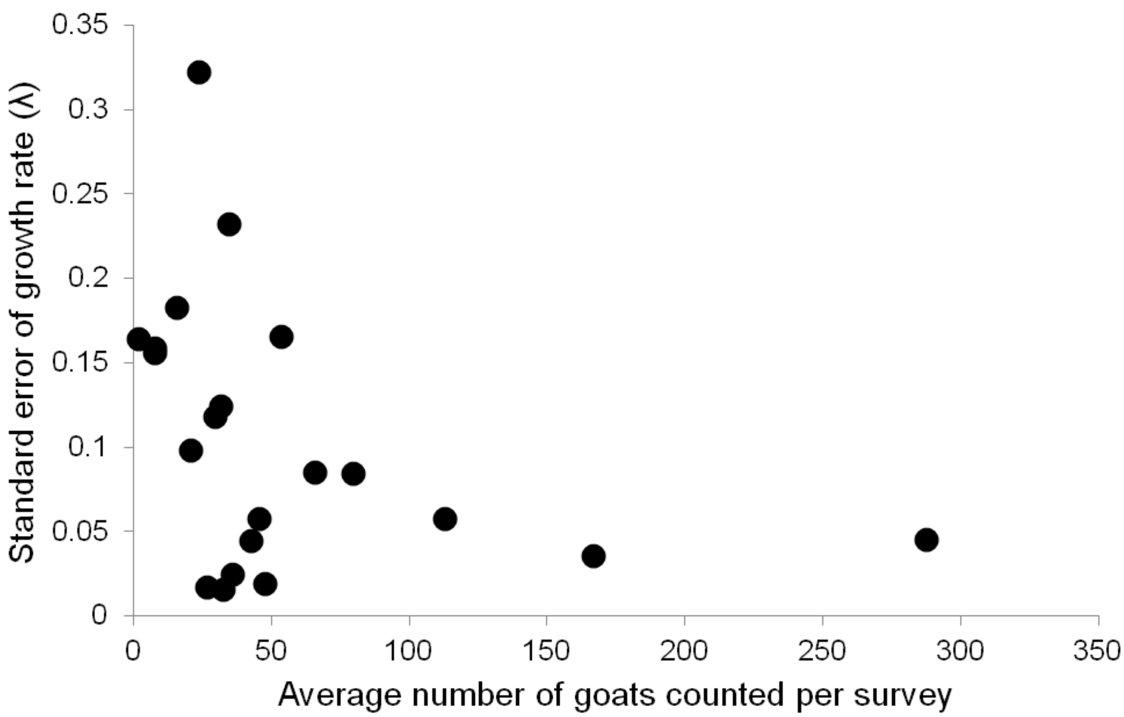


Figure 9. *The standard error of mountain goat population growth rate estimates as a function of the average number of individuals counted during trend surveys in 21 survey areas across Montana, 2000–2015.*

OBJECTIVE 3: SURVEY OF FIELD BIOLOGISTS

MFWP previously contracted a survey of population status, management practices, and research needs for another ungulate species, moose (*Alces alces*; Smucker et al. 2011). As in that project, we developed an original, standardized questionnaire for completion by MFWP area biologists whose jurisdictions include mountain goats. We emailed this 25-question survey to eighteen MFWP biologists in Regions 1–5 who have management responsibility for currently delineated mountain goat HDs. Responses were compiled and summarized separately for native and introduced mountain goat populations. We treated HDs as population sample units for summarizing results, and populations not currently within an administrative HD were included as independent samples. For a subset of questions (3, 7, 11, and 20), we asked respondents to rank a set of possible answers by their relative importance within each HD. In these cases, respondents were free to select and rank as many or as few options as were applicable, with their top choice receiving at rank of 1. We summarized answers to these questions in 2 ways: 1) first we recorded the number of times (the count) a given answer was selected, and 2) we scored rankings in reverse order such that ranks of 1 received the most points. For example, Question #3 included 7 possible answers, and a ranking of 1 received a score of 7, a ranking of 2 received a score of 6, and so on. Scores were then summed for each possible answer across all responses. Other questions were open-ended and received longer narrative responses. These responses are summarized in the following section, with complete details of responses from biologists presented in the Appendix.

Population estimates (Question 1)

We asked area biologists to provide population estimates for a total of 58 population units, including 26 HDs with native populations, 26 HDs with introduced populations, and 6 populations (4 native and 2 introduced) not currently within an HD (Appendix, Q1). These estimates were derived from the best available information from aerial and ground surveys, and applying sightability corrections and professional judgment. Several biologists provided narrative descriptions about individual HDs on their questionnaires. Along with population estimates, we also asked for a "range of confidence" of the estimate within each HD. This was not a statistical confidence interval. In some cases, a range of sightability values from the literature were used to estimate these ranges of confidence surrounding point estimates, and in other cases these were "best guesses" at the range of possible values of true abundance. When pooling estimates for summary purposes across multiple HDs, we used the sum of point estimates, low range of confidence boundaries, and high range of confidence boundaries to characterize total estimates and range of confidence boundaries for the pooled area.

The estimated total population (and range of confidence) of mountain goats in 2016 in native populations was 1,159 (885–1,537), and in introduced populations was 2,526 goats (1,842–2,958). The combined statewide population (excluding the 2 national parks) was 3,685 (2,727–4,495). An additional 2,000 (1,700–2,300) goats are estimated to live in native populations within Glacier National Park (Belt and Krausman 2012, J. Belt pers. comm.), and 225 (200–250) goats from introduced populations inhabiting northern Yellowstone National Park, either year-round or seasonally (Flesch et al. 2016). Including animals within national parks yields statewide estimates of 3,159 native goats and 2,751 introduced goats totaling 5,910 in all.

All introduced populations occur east of the Continental Divide in Regions 3–5. All native populations occur in Regions 1 and 2, west of the Continental Divide, plus three HDs in Region 3 and three HDs in Region 4 (Figure 1; Appendix Q1).

Past trends and limiting factors (Questions 2–5)

Area biologists estimated that 77% (23 of 30) of native mountain goat populations have declined over the past 50-year period of 1960–2010, including 1 extirpated population (Appendix, Q2). An additional 13% (4 of 30) were judged to be stable and 10% (3 of 30) had uncertain trends over this period. For introduced populations, biologists estimated that 43% (12 of 28) declined during this 50-year period, 11% (3 of 28) remained stable, and 43% (12 of 28) increased. Population trend was uncertain for the remaining herd of introduced goats.

The most commonly cited factors limiting goat numbers over the past 50 years were total hunter harvest followed by unknown reasons, harvest of female goats, habitat changes, and predation (Appendix, Qs 3, 4). That sequence was very similar for both native and introduced populations of goats, with ORV/snowmobile use a concern in several HDs of native goats, and predation a greater concern for introduced populations. Several respondents noted the uncertainty surrounding declines in native goat populations, sometimes as a consequence of insufficient population data needed to assess changes (Table 3).

Table 3. *Relative importance of factors limiting goat populations during past years (1960–2010) for native populations (26 HDs plus 3 populations not within current HDs) and for introduced populations (26 HDs plus 1 population not within a HD). Count data indicate the number of populations to which a limiting factor applies. Weighted scores reflect both the number of populations to which a factor applies and the relative rankings of that factor among others selected. See Appendix, Q3, 4 for detailed responses.*

		Disease	Predation	Hunter harvest (total # animals)	Hunter harvest (proportion of females)	Habitat changes (non-anthropogenic)	ORV/Snowmobile disturbance	Energy exploration	Logging and/or road construction	Non-motorized recreation	Climate change	Small population risks (inbreeding, …)	Other (describe in Q4)	Unknown
Native	Count	7	10	21	10	17	14			3	10	9		21
	Weighted score	23	49	126	70	78	79			15	13	52		123
Introduced	Count	4	12	11	10	10				1		4	5	8
	Weighted score	14	63	56	54	43				3		23	30	54
Total	Count	11	22	32	20	27	14			4	10	13	5	29
	Weighted score	37	112	182	124	121	79			18	33	75	30	177

From our compilation of hunting license records, we found that the total number of licenses issued to hunt native populations has declined over the study period (and 9 of 26 native HDs have been closed to hunting; Objective 2). When asked why licenses in their areas of management responsibility had declined, biologists most frequently indicated that licenses were reduced in response to observed declines in goat numbers (38%) and as precautionary actions until more reliable population data become available (25%; Appendix, Q5).

Current trends and liming factors (Questions 6–8)

We also asked about the status of goat populations in recent years: 2010–present. Biologists responded that 75% of native populations declined during this time or their status was uncertain; whereas introduced populations were largely stable (54%) with a few increasing and a few others decreasing (Appendix, Q6). The most commonly cited factors currently limiting goat numbers were habitat changes, followed by harvest of female goats, total goat harvest, predation, and ORV/snowmobile disturbance (Table 4, Appendix, Q7, 8).

There were marked differences between perceived factors limiting native versus introduced populations. For introduced populations, predation, harvest of females, total harvest, and habitat changes ranked similarly as most important. For native goats, habitat changes were most important, followed by ORV/snowmobile disturbance, small population risks, and climate change concerns.

Table 4. *Relative importance of factors limiting goat populations currently or into the future for native populations (26 HDs plus 3 populations not within current HDs) and for introduced populations (26 HDs plus 1 population not within a HD). Count data indicate the number of populations to which a limiting factor applies. Weighted scores reflect both the number of populations to which a factor applies and the relative rankings of that factor among others selected. See Appendix, Q7, 8 for detailed responses.*

		Disease	Predation	Hunter harvest (total # animals)	Hunter harvest (proportion of females)	Habitat changes (non-anthropogenic)	ORV/Snowmobile disturbance	Energy exploration	Logging and/or road construction	Non-motorized recreation	Climate change	Small population risks (inbreeding, …)	Other (please describe in Q4)	Ltd Available Habitat
Native	Count	10	14	14	13	18	21			4	20	16	4	
	Weighted score	50	66	74	81	101	95			20	91	99	15	
Introduced	Count	6	13	11	11	12	3			3		3		2
	Weighted score	41	69	62	67	60	17			11		17		14
Total	Count	16	27	25	24	30	24			7	20	19	4	2
	Weighted score	91	135	136	148	161	112			31	91	116	15	14

Compared to past limiting factors (1960–2010, see Table 3), there was less uncertainty about factors currently limiting populations. For introduced goat populations, concerns about effects of harvest levels on populations (total and females), habitat changes, and predation remained high. For native populations, there was a shift away from historical concerns about harvest levels to how populations are now being affected by habitat changes (see Habitat considerations section below), ORV/snowmobile disturbance, climate change, and small population risks. In part, this shift reflects a steep reduction in licenses issued for hunting of native populations over the years. As numbers of goats in native populations have decreased (see Questions 2–5 above), numbers of licenses and harvested goats have plummeted from an average 967 licenses and 329 harvested annually during the decade of the 1960s to an average of 50 licenses and 33 goats harvested during 2007–2015 (39 licenses and 25 goats harvested from native herds in 2015). Contrarily, introduced populations have generally prospered at most transplant sites since their introductions. Numbers of licenses and goats harvested from introduced populations have increased from an average 169 licenses and 71 goats harvested annually during the 1960s to an average of 225 licenses and 165 goats harvested during 2007–2015 (202 licenses and 155 goats harvested from introduced populations in 2015).

Regarding native goat populations, several biologists noted that the cumulative effects of specific factors listed in Table 4 may be perpetuating suppression of goat numbers that may have begun prior to 2010 (Appendix, Q8). Regarding introduced populations, biologists raised concerns about suspected predation on goats as well as the need for careful monitoring of harvest rates and potential overuse of available range by goats (Appendix, Q8).

Harvest and season setting (Questions 9–16)

Biologists managing HDs with native goats take an almost unanimously conservative approach to harvest, with the goal of minimizing impact on populations (Appendix, Q9). Nine of those 26 HDs are closed to hunting; and 8 of the 9 closed HDs are in Region 2. For HDs with introduced goats, objectives of harvest were more varied. Biologists have recommended harvest strategies to limit population growth in six HDs with introduced populations, whereas three of the 26 HDs with introduced populations have been closed to hunting.

Biologists varied in their assessment of the adequacy of survey and inventory information available to them for making management decisions (Table 5; Appendix Q10). The results suggest that, on average, more adequate survey data are collected in HDs with introduced goats. This corresponds to a greater proportion of statewide hunting opportunity being offered in HDs with introduced goats (84% in 2015), though there could be a variety of reasons for variations in survey frequency. When asked which factors were most limiting to population survey efforts, biologists identified aircraft/pilot unavailability, adverse weather conditions, and lack of funding as leading reasons (Appendix, Q12). Differences in population size may also play a role in the adequacy of information available, given our results show that larger populations yield more reliable, less variable, and thus more useful population survey data (Figure 9).

Survey minimum counts and survey recruitment ratios (e.g., kids per goat aged ≥1-year-old) are the two types of data on which biologists place the greatest reliance in setting harvest regulations (Table 6; Appendix Q11). This is true for both native and introduced populations, which underscores the importance of obtaining reliable population survey data to manage goat populations. The next two factors most relied on to set regulations were FWP harvest data (number of animals harvested relative to number of licenses issued) and hunter effort data (number of days/animal harvested). With mandatory reporting of mountain goat kills and consistent annual hunter harvest surveys, these may be the most consistently available data at biologists' disposal.

Table 5. *Tallied responses from 17 biologists regarding the quantity and quality of mountain goat survey and inventory information available for making management decisions, for those managing both native (N=10) and introduced (N=7) populations (see Appendix Question 10).*

	Adequate	Somewhat adequate	Somewhat inadequate	Inadequate
Native populations		2	4	4
Introduced populations	1	4	2	
Total	1	6	6	4

Table 6. *Relative importance of information that biologists use to set annual goat harvest regulations for native populations (26 HDs plus 3 populations not within current HDs) and for introduced populations (26 HDs plus 1 population not within a HD). Count data indicate the number of populations to which a limiting factor applies. Weighted scores reflect both the number of populations to which a factor applies and the relative rankings of that factor among others selected. See Appendix, Q11 for detailed responses.*

		FWP hunter harvest data	FWP hunter effort data (e.g., kills per unit effort)	FWP observations data (e.g., number seen/hunter)	Age and/or horn data	Anecdotal hunter reports (i.e., not in MRRE)	Survey minimum counts	Survey recruitment ratios	Other (please describe):
Native	Count	5	4	4	2	3	5	5	
	Weighted score	22	19	15	6	11	31	32	
Introduced	Count	7	6	4	6	5	8	10	1
	Weighted score	33	25	11	20	15	54	51	7
Total	Count	12	10	8	8	8	13	15	1
	Weighted Score	55	44	26	26	26	85	83	7

When asked if proposed quotas for other species, such as mountain lions, have been affected by population demographics of overlapping mountain goat populations, 16 of 17 respondents answered “no” (Appendix Q13).

We also asked biologists two questions regarding how considerations of the sex of animals entered into hunters’ decisions when targeting a mountain goat. Responses indicated that an average of 55% of hunters intend to harvest a male rather than a female (Appendix, Q14); and biologists estimated that an average of 52% of hunters can correctly identify a mountain goat’s sex under field hunting conditions (Appendix, Q15). These results suggest that over half of license-holders may be as likely to kill a female as a male, particularly with female-biased sex ratios being typical in the adult cohort of goat populations (Chadwick 1973, Rideout 1974, Gonzalez Voyer et al. 2003). In a simulated field test, 81% of attendees of a Northern Wild Sheep and Goat Council meeting accurately identified the sex of mountain goats after being shown a 20 minute presentation describing the diagnostic characteristics of each sex. However 77% of participants in that study had prior experience censusing or classifying goats (Smith 1988b). When asked if the educational information provided to license-holders was sufficient for hunters to make informed decisions about the age and sex of the animals they choose to harvest, three biologists answered yes, six no, and six were uncertain (Appendix, Q16).

Population surveys (Questions 17–19)
We asked biologists about the methodology used to conduct population trend counts, how often surveys are conducted and during which seasons. They reported using a combination of ground and aerial survey types during all seasons and at intervals ranging from annually to never (Appendix, Q17). When asked if standardized methods should be employed to monitor mountain goats across the state, the consensus was "no" (14 of 18 responses; Appendix, Q18).

When asked to compare native to introduced goat populations, 5 of 6 biologists who responded to this question felt that Montana's introduced populations were generally healthier or more productive with higher recruitment rates. The majority of biologists surveyed said they did not have enough experience or knowledge to make this assessment (Appendix, Q19).

Habitat considerations (Questions 20–21)
There was little consensus about which, if any, habitat management programs would benefit goat conservation or increase hunter opportunity (Table 7). Among the possible management scenarios suggested in the question, 3 recreational management categories had a combined weighted score (21), larger than any other category (Table 7; Appendix, Q20). Sixteen of 17 biologists had not completed any habitat-related projects alone or in cooperation with federal land managers to improve mountain goat habitat or conservation (Appendix, Q21).

Table 7. *Relative importance of habitat management programs that would promote mountain goat conservation and hunter opportunity. Count data indicate the number of populations to which a management program applies. Weighted scores reflect both the number of populations to which a factor applies and the relative rankings of that factor among others selected.*

	None; Habitat is not a limiting factor	More fire (natural or prescribed)	Less fire (suppression of wildfire)	Weed management	Road management (i.e., more restrictive)	ORV management	Snowmobile management	Non-motorized recreation management	Unknown	Other (please describe):
Count	3	5	2	1		2	4	3	6	
Weighted score	9	15	5	1		5	11	5	15	

Management and research needs (Questions 22–25)
Biologists expressed interest in translocating animals to sustain particular native and introduced mountain goat populations (Table 8). Several cautioned that introductions should be carefully evaluated on an area-by-area (herd-by-herd) basis (Appendix, Q22).

Biologists identified a wide array of research needs that would benefit their understanding and management of mountain goat populations (Appendix, Q23 details all topics). This question was open-ended (as was Question 24 about management needs) allowing respondents to offer any number of research topics that interested them. Of the 12 topics mentioned, 3 research themes or areas of study captured 62% of all topics respondents offered: assessments of habitat condition, use, and carrying capacity (9 responses); population demographics: productivity, recruitment, kid survival, and adult survival (7); and causes of mortality (5). The other 9 topics were each mentioned 3 times or less.

Biologists also identified 8 management or monitoring needs that would assist mountain goat management (Appendix Q24 details all topics). The 2 topics most often mentioned, and constituting 68% of all responses, were: better/more frequent monitoring of populations (10 responses); and sightability correction models and improved, standardized, survey methodology (5). Ten additional topics of relevance to mountain goat management and conservation in Montana were mentioned 1 or 2 times each by questionnaire respondents (Appendix, Q24–25).

Table 8. *Biologists' responses about whether there is a pressing need for translocation of mountain goats to sustain native and/or introduced populations.*

	Yes	No
Native	2	4
Introduced	3	7
Total	5	11

DISCUSSION

Population estimates and trends
The overall goals of this project were to synthesize population and harvest trends of mountain goats in Montana over the past 50–60 years and to summarize and evaluate their current status and management circumstances. Based on the responses of FWP biologists who manage Montana's goats, there were an estimated 2,526 animals (69% of total) in introduced populations and 1,159 animals (31%) in native populations in 2016 under MFWP jurisdiction. The combined statewide population managed by MFWP was 3,685 (2,726–4,493) mountain goats. Including another 2,225 goats estimated in the 2 national parks yielded an estimated 5,910 animals within Montana's borders.

To put current numbers in historical perspective, we reviewed previous statewide population estimates of native goats. In an early comprehensive study of Montana's mountain goat population, Casebeer et al. (1950) reviewed estimates of the statewide goat population during 1919–1942, as recorded by the US Forest Service, and during 1943–1948 from estimates made by the Montana Fish and Game Department (Rognrud and Lancaster 1947). Maximum annual estimates were from the years 1943 through 1946, when 5,000–5,200 goats were estimated statewide, of which about 940 occupied Glacier National Park. Although establishment of new herds in previously unoccupied mountain ranges began in 1941 (Picton and Lonner 2008), Casebeer et al. (1950) recorded an annual maximum of only 97 goats among all introduced populations during 1943–1946. From these records it appears that about 4,100 goats occupied native ranges across Montana during 1943–1946 (excluding national parks), a figure three to four times larger than the 1,159 goats estimated by Montana's biologists in 2016. Carlsen and Erickson (2008) estimated 2,719 mountain goats in Montana in 2007, based on population survey data. Of that total, 1,517 animals were in introduced populations and 1,202 were in native populations, based upon the raw data they provided to us from that analysis. While the potential for differences in estimation methods may confound direct comparisons across years, we estimated an additional 1,000 goats to exist in introduced populations compared to that estimated in 2007 (Carlsen and Erickson 2008). However, our native goat population estimate in 2016 (1,159) is only slightly lower than theirs from a decade earlier (1,202).

The disparity between native and introduced mountain goats evidenced by these changes in population estimates was also noted by area biologists' responses concerning population trends. Of the 30 native populations, at least 23 (77%) were judged to have declined or been extirpated since 1960, with trends for 3 additional populations labeled as unknown. To the contrary, 54% (15 of 28) of introduced populations were judged as stable or increasing, though some declines are also evident. In the Beartooth Mountains, for example, trend in recent summer aerial survey data suggests declines of >40% in this introduced population since the 1980s.

Survey responses suggested a variety of causes for declines in native populations over the years. During the 50 years prior to 2010, the limiting factors most often mentioned as responsible for influencing goat numbers were total hunter harvest, female harvest, and unknown reasons. Ranking of current and future threats to goat populations indicated a shift in factors influencing populations. As licenses were reduced in HDs with native populations, habitat changes, ORV/snowmobile disturbance, climate change, and small population risks were perceived as most affecting populations. For introduced goat populations, effects of harvest levels on populations (total and females), habitat changes, and predation ranked highest in importance.

Harvest management

For native goat populations, numbers of licenses and harvested goats have plummeted from an average of 967 licenses and 329 harvested annually during the decade of the 1960s to an average of 50 licenses and 33 goats harvested during 2007–2015 (39 licenses and 25 goats harvested in 2015). Contrarily, introduced populations have generally prospered at most transplant sites since their introductions. Numbers of licenses and goats harvested from introduced populations have increased from an average 169 licenses and 71 goats harvested annually during the 1960s to an average of 225 licenses and 165 goats harvested during 2007–2015 (202 licenses and 155 goats harvested from introduced populations in 2015).

Harvest management of mountain goats has been a topic of much interest and debate in the literature. Overharvest has been implicated as a source of population declines in native mountain goats in other parts of their range. Rice and Gay (2010) used population modeling to evaluate historical trends of mountain goats in Washington and found that population declines were primarily attributable to harvest. Goat populations, numbering less than 100 animals, are generally no longer hunted in Washington (Rice and Gay 2010). Hamel et al. (2006) modeled population dynamics of mountain goats in Alberta and showed high sensitivity of population dynamics to adult female survival and a subsequently detrimental role of female harvest in affecting population trends. As a result of these findings, the authors recommended closure of hunting in populations numbering <50 total individuals, and conservative harvest rates of 1–4% for larger populations depending on the population size and proportionate female harvest (Hamel et al. 2006, Rice and Gay 2010). In our study, the average license rates were 5.5% across hunted native populations and 7.7% across hunted introduced populations, while harvest rates averaged 2.0% for native and 6.3% for introduced populations. Twelve of the state's 52 currently delineated HDs have been closed to hunting, ostensibly due to populations too small to support harvest. Additionally, it's noteworthy that during the 55 years since 1960 about 38% of the mountain goats harvested in Montana were females.

Harvest rates of introduced populations have typically been higher, including cases of harvesting as many as 7.5–20% of the population in some cases (reviewed by Williams 1999 and Côté et al. 2001). Williams (1999) noted that introduced mountain goat populations likely occur in different stages of Caughley's (1970) 4 states of an ungulate irruption, as regulated by density-dependent quality of habitat. Thus, a single optimal harvest rate prescription may not apply to all populations after accounting for other limiting factors such as density dependence or predation rates. However, all authors have recommended caution with harvest of mountain goats in particular due to the difficulties of limiting harvest to males as well as their generally modest reproductive capacity.

Population monitoring

Current monitoring practices for mountain goats vary widely among local areas in Montana. Surveys are not frequently conducted in all HDs, and vary with respect to the platform, frequency, and season among HDs. Our results suggested that current monitoring practices using aerial surveys alone have not, for the most part, been adequate to reasonably distinguish increasing vs. decreasing population trends with statistical rigor over the most recent 15-year time period. Biologists offered that better and more frequent monitoring of populations was their top management need and suggested research leading to a better understanding population demographics of goats was a high priority.

Minimum counts documented during population surveys are a valid means of monitoring trend, as long as the average proportion of individuals seen relative to those in the entire population does not change over time (reviewed by DeCesare et al. 2016). In other words, an equal proportion of the population is assumed to be within the survey area and mean sightability of those within the area is assumed to be constant. While these counts provide a means of estimating trend, they cannot be used to estimate abundance without specific estimates of sightability. Measured sightability rates of marked goats have varied from ~40% to 80% in studies in British Columbia, Idaho, and Washington (Poole et al. 2000, Pauley and Crenshaw 2006, Rice et al. 2009). Sightability likely varies among goat populations and habitats in Montana, making it unlikely that a single sightability model would apply across the state (*sensu* Harris et al. 2015). Accounting for sightability bias across would Montana would likely require multiple studies and multiple models to fit varying conditions.

Managers of species that tend to occur in small populations commonly face an additional challenge of lacking statistical power when interpreting trend surveys. The precision of population estimates is known to decrease as the size of the population being monitored decreases (Taylor and Gerrodette 1993, Barnes 2002, DeCesare et al. 2016). For example, Barnes (2002) found that the confidence intervals for estimates for a West African elephant monitoring program were likely to be >100% of the point estimates when the population was below 600 animals. This threshold doesn't necessarily apply directly to mountain goat monitoring in Montana. Our results do suggest a positive relationship between the magnitude of counts and their precision (Figure 9). Thus, lumping subpopulations together into larger groups whether during surveys or during data analysis may increase our power to detect trends if done so consistently over time.

A formal power analysis of simulated and empirical mountain goat survey data would offer an improved depiction of how various survey sampling designs might affect the strength of results. Additionally, review of other survey techniques or monitoring practices (such as monitoring of trend via survival and reproductive rates of marked individuals or non-invasive DNA-based population estimation) may aid in evaluating current practices compared to those employed for mountain goats in other jurisdictions (Poole et al. 2011).

In addition to minimum counts, biologists indicated frequent use of recruitment ratios when monitoring mountain goat populations. These ratios are typically formulated as young/adult ratios, though the definition of the adult denominator appeared to vary across surveys depending on efforts to distinguish yearling or 2-year-old goats from older animals. Of significance to interpretation of these data is the important life history detail that the age of first reproduction for female mountain goats is 3 years of age (Rideout 1975) and primaparity can average >4 years-old for native populations (Festa-Bianchet and Côté 2008). It is likely that many of the adults counted in recruitment ratios are not in fact breeding-aged adults. Thus, variation in age structure of adults across years or populations should be expected to confound interpretation of recruitment ratio data.

Area biologists also indicated that other data, in addition to survey data, are used when managing mountain goats. These included hunter harvest data, hunter effort data, and data concerning the age and sex of harvested individuals. Statistical modeling of these forms of data is not typically employed, and it is currently unclear if catch-effort or age-at-harvest data would be sufficient to glean meaningful patterns statistically, whether as a stand-alone analysis or incorporated into an integrated population model (Skalski et al. 2007, Udevitz and Gogan 2012). Hunter success, in particular, may be of limited value in assessing the population status of mountain goats, particularly native goats in Montana. Over the past 60 years as harvest success has increased (Figure 2), we found that Montana's native goats have clearly been in decline as have the number of licenses issued annually. In HDs where only one or two licenses are issued annually, hunter success of 100% or 50% in a HD is dicey to interpret, and potentially misleading. Fidelity of goats to preferred areas of their ranges contributes to the ability of hunters to find and harvest goats, even when populations are small (Chadwick 1973, Smith 1976, Taylor et al. 2006, Festa-Bianchet and Côté 2008). This natural history trait may predispose hunted mountain goat populations to apparent "hyperstability" when monitored with hunter statistics alone (Hatter 2001). In such cases, hunter harvest statistics may convey a deceptively stable trend even for declining populations, because hunters continue to find and harvest goats in the same areas and with the same efficiency regardless of decreased numbers overall (Hatter 2001). Survey responses suggested that Montana's goat managers recognize the limited value of harvest success compared to biological data obtained from population surveys on which they place greater importance when establishing annual regulations (Table 6). Consequently, population monitoring ranked highest among management priorities (Appendix Q24).

Population identification

Defining and sampling populations is basic to wildlife management and conservation. For analytical purposes, we grouped mountain goat HDs into 28 "regional populations" (Figure 6), but the biological significance of these delineations is unknown.

Where goats occur on an isolated mountain range, for all practical purposes those animals can be considered a biological population. In mountain range complexes, however, geographically defining a population or subpopulations of a metapopulation can be problematic. This situation arises for a number of geographic areas of Montana's mountain goats, both native and introduced. In management practice, the definition of a population often necessitates imposing arbitrary boundaries on the landscape, which may not reflect population biology of the species on the landscape. Furthermore, if seasonal distributions of populations are not well understood, and population surveys do not reflect distributions during the hunting season, disproportionate harvests of individual populations or subpopulations could occur.

Concerns about small population effects raised by several biologists are justified, given the small and potentially isolated nature of many of Montana's goat populations. Biologists estimated that 25 of the state's 52 HDs may support fewer than 50 goats. Such populations risk heightened consequences of stochastic events and inbreeding depression, compared to large populations or metapopulations (Hebblewhite et al. 2010, Johnson et al. 2011). Effective conservation of mountain goats may require additional understanding of the extent to which populations face such risks. Research on movement and yearlong distributional patterns are needed for some of Montana's larger landscapes to determine where populations may now be reproductively isolated. For some native populations in Regions 1 and 2 this seems particularly germane.

Habitat changes

Of all Montana's large mammal species, the mountain goat's distribution is almost completely on federally or state-managed lands: national forest multiple-use lands, national forest wilderness areas, two national parks, plus state lands, and some tribal lands in the Mission Mountains. Steep, rugged terrain and snow are defining features of mountain goat ranges. For some populations, mineral licks are a seasonally important resource, such as the Walton Goat Lick in Glacier National Park (Singer 1978). These habitat features and associated, preferred, food sources largely dictate distributions and movement patterns of mountain goats.

Because of their high, rugged nature, mountain goat ranges tend to be less subject to human development and alteration than habitats of the state's other big game species. Yet, the biologists we surveyed offered a range of direct or indirect effects, both natural and anthropogenic, that are either suspected or known to be affecting mountain goats. Road construction into goat habitat to facilitate mining, energy and timber extraction, and motorized recreation can alter habitat with implications for goat distributions and demography (Fox et al. 1989, White and Gregovich 2017), and increased vulnerability of goats to harvest (Mountain Goat Management Team 2010). Numerous studies in Canada and the U.S. have demonstrated that mountain goats are particularly sensitive to helicopter disturbance (Foster and Rahs 1983, Côté, 1996, Gordon and Wilson, 2004). Mountain goat management plans for Alberta, British Columbia, and Washington review how habitat threats are being addressed.

In Montana, some of the most pertinent research conducted on habitat-mediated impacts on goats includes documentation of how helicopter over-flights associated with seismic testing affects population dynamics (Joslin 1986), and how road intrusion and timber harvest alter mountain goat behavior and distribution (Chadwick 1973). However, little is known about the effects of commercial and recreational activities on most mountain goat populations in the state, or about the condition and carrying capacity of most goat ranges and how that may relate to population performance. Likewise the effects of wildfire, or contrarily fire suppression, on goats through changes in habitat structure, plant succession, and forage are little known. These are noteworthy areas of research regarding the differing status and trends we identified of native versus introduced populations generally.

Mountain goats may also be among those species most sensitive to climate change because of their cold-adapted nature and because the climate is warming (and cascading environmental changes occurring) twice as rapidly at high elevations compared to the global mean rate of warming (Beever and Belant 2011).

FUTURE DIRECTIONS FOR RESEARCH AND MANAGEMENT

Montana is unique among the 8 U.S. and Canadian jurisdictions within the native range of the mountain goat. Montana supports more introduced populations in which numbers of goats collectively now exceed those in the state's native populations. Clearly one size fits all prescriptions for management would not serve the state's goat populations well. Management and conservation efforts require consideration of the wide range of habitats Montana's goats occupy with special attention to differences between native and introduced goats. However, statewide coordination of management planning and research prioritization may serve to leverage resources to address needs and answer questions for broad landscapes and multiple populations of goats.

From our findings, important topics deserving of future attention in comprehensive planning for Montana's mountain goats include:

- Recommendations for harvest of mountain goats: These may well differ for native and introduced populations. Not only population harvest rates, but sex-specific harvest prescriptions dependent on maintaining viable population size could be addressed. Given that mountain goats occupy habitats relatively secure from human impacts (with some exceptions) compared to other big game species, and that high natural mortality among juvenile cohorts is largely beyond managers' control, wildlife managers can influence mountain goat conservation largely through regulation of public harvest.

- Evaluation of monitoring practices: MFWP biologists rely heavily on population survey data to establish harvest levels of populations. Improved survey techniques, sightability modeling, and informed/optimal monitoring frequencies are all important management needs. Although biologists overwhelmingly felt that monitoring needed to be herd or hunting district specific because of local conditions, some consensus on data collected may be important for comparing populations and analyzing multi-year trends. The most difficult task in this study we conducted was to analyze population survey data due to inconsistencies in monitoring frequency and protocols. A formal power analysis of simulated and empirical mountain goat survey data would offer an improved depiction of how various survey sampling designs might affect the strength of results.

- Local monitoring protocols: We support area biologists' efforts to formally design, prescribe, and document monitoring protocols for mountain goats in their respective areas with the goal of detecting changes in population status that require management actions. These would greatly benefit future area biologists in their jurisdictions and synthesis efforts such as this one by ensuring comparable data streams over time.

- Species management plan: MFWP does not currently have a statewide management plan for mountain goats. Examples of such plans exist for other species in Montana, and for mountain goats in neighboring jurisdictions (e.g., Alberta, British Columbia, Idaho, Oregon, Utah, and Washington). Those state and provincial plans have brought together much of the pertinent literature and identified key planning elements, some unique to mountain goat conservation. Development of such a plan has been previously identified as a priority by MFWP, yet has not occurred in the face of limited time and resources. Relative to other ungulate species in Montana, a management plan for mountain goats may be particularly useful for a variety of reasons. First, various life history traits make them more sensitive to harvest management than other ungulates, which justifies a unique approach to harvest management of this species. Second, some of the variation in monitoring practices and/or harvest rates identified in this report might benefit from regional or statewide coordination or guidelines. Third, the reproductive isolation of many populations may render goats more vulnerable to natural and anthropogenic changes in their environment across broad areas of their distribution. Lastly, individual biologists have less funding and time to devote to gaining local experience and data with this species relative to other more abundant and/or controversial species, which might increase the value of a statewide resource for information and guidance.

- Ecological research: In addition to the monitoring-based research questions we identified above, our questionnaire indicated a variety of potential avenues for important research into mountain goat ecology. These included, but were not limited to, assessments of mountain goat foraging ecology and habitat condition, demographic vital rates and population dynamics, and causes of mortality.

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Appendix 1. Compiled Results from the Biologist Questionnaire

The following 18 MFWP biologists completed the questionnaire during May–September 2016:
Liz Bradley, R2-Missoula
Vanna Boccadori, R3-Butte
Tonya Chilton-Radandt, R1-Libby
Jessy Coltrane, R1-Kalispell
Julie Cunningham, R3-Bozeman
Scott Eggeman, R2-Blackfoot
Craig Fager, R3-Dillon
Adam Grove, R3-Townsend
Adam Grove, R4-White Sulphur Springs (on behalf of Jay Kolbe)
Cory Loecker, R4-Great Falls
Brent Lonner, R4-Fairfield
Karen Loveless, R3-Livingston
Rebecca Mowry, R2-Bitterroot
Ryan Rauscher, R4-Conrad
Jenny Sika, R3-Helena
Shawn Stewart, R5-Red Lodge
Mike Thompson, R2-Upper Clark Fork (on behalf of Julie Golla)
Dean Waltee, R3-Sheridan

Population Estimates

Q1. Based on available data and your professional opinion, please provide an estimate of the current total number of mountain goats (*N*) within each population that you manage (including 0's for extirpated populations), as of April, 2016. Please also provide an interval showing your confidence in the range of possible values for *N* ("Range of confidence"). If needed you can lump districts together and provide a single combined estimate.

HD	Bio	Native/ Introduced	*N*	Range of confidence
100	Chilton-Radandt	N	85	80 - 95
101	Chilton-Radandt	N	50	45 - 60
131	Coltrane	N	17	16 - 18
132	Coltrane	N	33	31 - 36
133	Coltrane	N	39	36 - 42
134	Coltrane	N	14	13 - 15
140	Coltrane	N	65	60 - 70
141	Coltrane	N	62	58 - 65
142	Coltrane	N	70	67 - 73
150	Coltrane	N	61	57 - 66
151	Coltrane	N	16	16 - 16
212	Golla	N	25	0 - 50
213	Golla	N	0	0 - 20
222	Golla	N	10	0 - 20
223	Golla	N	10	0 - 20
240	Mowry	N	100	80 - 120
250	Mowry	N	30	10 - 100
261	Mowry	N	0	0 - 10
270	Mowry	N	10	0 - 30
280	Eggeman	N	30	15 - 40
312	Fager	N	125	75 - 150
313	Loveless	I	450	330 - 550
314	Loveless	I	250	140 - 275
316	Loveless	I	55	40 - 62
320	Waltee	I	27	11 - 44
321	Fager	N	20	10 - 30
322	Boccadori	N	31	26 - 36
323	Loveless	I	295	221 - 338
324	Cunningham	I	210	156 - 252

HD	Bio	Native/ Introduced	*N*	Range of confidence
325	Cunningham	I	82	57 - 103
326	Cunningham	I	37	28 - 44
327	Cunningham	I	42	30 - 53
328	Cunningham	I	6	4 - 8
329	Loveless	I	150	115 - 170
330	Loveless	I	25	19 - 30
331	Waltee	I	48	22 - 48
332	Sika	I	0	0 - 1
340	Boccadori	I	10	10 - 15
361	Cunningham	I	92	66 - 115
362	Cunningham	I	148	106 - 185
380	Grove	I	20	9 - 30
393	Cunningham	I	78	56 - 98
414	Rauscher	N	40	20 - 60
415	Rauscher	N	75	50 - 125
442	Lonner	N	40	35 - 50
447	Loecker	I	60	50 - 75
453	Kolbe	I	55	45 - 70
460	Loecker	I	45	40 - 60
514	Stewart	I	75	60 - 100
517	Stewart	I	90	80 -100
518	Stewart	I	75	60 - 100
519	Stewart	I	50	50 - 60
Fill-in other populations **(**Sun River Preserve, Rattlesnake NRA, …)				
Bradley - Rattlesnake		N	10	5 - 15
Bradley – Great Burn		N	23	20 - 25
Lonner – Sun River Preserve		N	68	60 - 80
Grove – North Big Belts		I	50	36 - 60
Taylor – Big Snowy		I	1	1 - 2
Thier – Whitefish Range		N	0	-

For 26 Native HDs, plus the Great Burn, Rattlesnake, Sun River Preserve, and (extirpated) Whitefish Range herds, the estimated total population = 1,159 (885–1,537). For 26 Introduced HDs, plus the North Big Belt and Big Snowy Mountains, the estimated total population = 2,526 goats (1,842–2,958). Total statewide population (not including the two national parks) = 3,685 (2,727–4,495).

<u>PAST</u> trends and limiting factors

Q2. How have goat numbers in your area changed over the past 50 years (i.e., 1960-2010)?

Native Populations (HDs) **You can provide separate answers for individual or groups of HDs, or if answer is same across your area you can just put "ALL"	Increasing	Stable	Decreasing	Uncertain
100		X		
101		X		
131			X	
132			X	
133			X	
134			X	
140			X	
141			X	
142			X	
150			X	
151			X	
212			X	
213			X	
222			X	
223			X	
240			X	
250			X	
261			X	
270			X	
280				X
312				X
321			X	
322			X	
414			X	
415		X		
442			X	
Great Burn			X	
Rattlesnake				X
Sun River Game Preserve		X		
Whitefish Range (extirpated)			X	
NATIVE TOTAL		4	23	3

Introduced Populations (HDs) **You can provide separate answers for individual or groups of HDs, or if answer is same across your area you can just put "ALL"	**Increasing**	**Stable**	**Decreasing**	**Uncertain**
313	X			
314	X			
316	X			
320				X
323	X			
324	X			
325			X	
326			X	
327			X	
328			X	
329	X			
330	X			
331		X		
332			X	
340		X		
361	X			
362		X		
380			X	
393	X			
447	X			
453	X			
460	X			
514			X	
517			X	
518			X	
519			X	
North Big Belts (no HD)			X	
Big Snowy (formerly HD 516)			X	
INTRODUCED TOTAL	12	3	12	1

For 26 Native HDs, plus the Great Burn, Rattlesnake, Sun River, and Whitefish herds, goat numbers in 23 of 30 areas were judged to have decreased over the past 50 years with numbers in 4 others stable and 3 others uncertain.

For 26 Introduced HDs, plus the North Big Belt and Big Snowy Mountains, goat numbers in 12 increased, 3 were stable, and 12 decreased over the past 50 years.

Q3. Which limiting factors do you suspect may have affected goat numbers in your area of responsibility during the past (1960–2010)? Please numerically rank for each HD those that apply, with 1 being of highest importance. Leave blank those that don't apply. Compiled by hunting district (HD) as indicated by biologists (including Great Burn, Rattlesnake, Sun River Preserve, and North Big Belts). Weighted score accounts for relative rankings.

Count of HDs per category and ranking		Disease	Predation	Hunter harvest (total # animals)	Hunter harvest (proportion of females)	Habitat changes (non-anthropogenic)	ORV/Snowmobile disturbance	Energy exploration	Logging and/or road construction	Non-motorized recreation	Climate change	Small population risks (inbreeding, …)	Other (please describe in Q4)	Unknown
Native	Ranked 1st (7 points)		1	7	10	2	4					5		15
	Ranked 2nd (6 points)	1	3	10		1	1			1	1	1		1
	Ranked 3rd (5 points)	1	3	1		3	9			1		1		
	Ranked 4th (4 points)	1		3		10				1		1		2
	Ranked 5th (3 points)		3			1					9			
	Ranked 6th (2 points)	4										1		1
	Ranked 7th (1 point)													2
	Count of HDs	**7**	**10**	**21**	**10**	**17**	**14**			**3**	**10**	**9**		**21**
	Weighted score	**23**	**49**	**126**	**70**	**78**	**79**			**15**	**33**	**52**		**123**
Introduced	Ranked 1st (7 points)		3	2	2								4[a]	7
	Ranked 2nd (6 points)	2	3	2	4	2						3		
	Ranked 3rd (5 points)		2	5	2	2						1		1
	Ranked 4th (4 points)		2	1	1	5								
	Ranked 5th (3 points)		2							1				
	Ranked 6th (2 points)				1								1	
	Ranked 7th (1 point)	2		1		1								
	Count of HDs	**4**	**12**	**11**	**10**	**10**				**1**		**4**	**5**	**8**
	Weighted score	**14**	**63**	**56**	**54**	**43**				**3**		**23**	**30**	**54**
Pooled	**Count of HDs**	**11**	**2**	**32**	**20**	**27**	**14**			**4**	**10**	**13**	**5**	**29**
	Weighted Score	**37**	**112**	**182**	**124**	**121**	**79**			**18**	**33**	**75**		**177**

[a] Other factors were ranked 1st and described in Q4 below for 4 introduced populations (HDs 313, 331, 332, 340)

[b] Other factors were ranked 6th and described in Q4 below for 1 introduced population (HDs 320)

The most commonly cited factors limiting goat numbers over the past 50 years (through 2010) were total hunter harvest followed by unknown reasons, harvest of female goats, habitat changes, and predation. That sequence was very similar for both native and introduced populations of goats, with ORV/snowmobile use a concern in several HDs of native goats, and predation a greater concern for introduced populations.

Q4. Please elaborate here on the limiting factors you marked in Question 3. For example, if you selected predators, disease, hunter harvest of females or climate change, please explain.

Following are some specific comments reported by respondents:

- "I marked "UNKNOWN" as a top factor in my areas, as I think the bottom line is that we really don't know what has been driving declining goat numbers [native herds] and therefore research is crucial."
- The percent of adult females in the harvest is disturbing/a concern, and overall harvest was probably excessive in the past (several respondents).
- Small isolated populations are potentially affected by inbreeding depression.
- Others mentioned that they suspect climate change effects on goats (or their habitats) but have no direct information.
- Too little population data to assess changes.
- There has been pneumonia complex disease in sheep which may have affected goat production.
- "We have the full complement of predators and I would very much like to know how they influence survival."
- Cumulative effects (hunting + natural mortality) may have caused declines and kept some native populations low.
- Several hunting districts have unique circumstances where trapping and removal of goats may have contributed to declines (HD442); struggling native herds were supplemented with transplanted goats (HD101 and also the Rattlesnake); bighorn sheep were reintroduced on top of a small goat population and may have competed with goats (HD332); habitat was limited where goats were introduced (HD331 and 340); a population crashed possibly due to density-dependent factors and/or disease but has subsequently recovered (HD313).

Q5. In your area of responsibility, why have licenses for <u>native</u> goats been reduced in recent decades (check all that apply)? One response per biologist with responsibilities for native herds.

- Reduced licenses in response to observed declines in goat numbers based on monitoring data (6)
- Reduced licenses as precautionary action until more reliable population data are available (4)
- Reduced licenses in response to change in the objectives or science behind harvest management (2)
- Reduced licenses to maintain higher numbers for other users (e.g., non-consumptive recreationists) (1)
- Other (3) Please describe:

Note that for 2 biologists who indicated "Other," licenses had not been reduced in recent years, and in the third case, permits have been increased.

<u>CURRENT</u> trends and limiting factors

Q6. How do you feel those same populations are doing now (i.e., 2010-present)? Some biologists indicated more than one category for a HD.

Populations (HDs) **You can provide separate answers for individual or groups of HDs, or if answer is same across your area you can just put "ALL"	**Increasing**	**Stable**	**Decreasing**	**Uncertain**
100			x	
101		x		
131			x	
132			x	
133			x	
134			x	
140			x	
141			x	
142			x	
150			x	
151			x	
212			x	
213?			x	
222			x	
223			x	
240		x		x
250				x
261				x
270	x			x
280		x		x
312		x		
321				x
322	x			
414				x
415				x
442		x		
Great Burn				x
Rattlesnake				x
Sun River Preserve		x		
NATIVE TOTAL	2	6	14	10

Introduced Populations (HDs) **You can provide separate answers for individual or groups of HDs, or if answer is same across your area you can just put "ALL"	**Increasing**	**Stable**	**Decreasing**	**Uncertain**
313		x		
314		x		
316	x	x		
320				x
323		x		
324	x			
325		x		
326				x
327/362	x			
328			x	
329		x		
330		x		
331		x		
332		No goats		
340		x		
361	x			
380		x		
393		x		
447	x	x		
453				x
460		x	x	
514			x	
517			x	
518		x		
519		x		
North Big Belts (no HD)	x			
INTRODUCED TOTAL	6	15	4	3

Goats in HDs with native populations are mostly decreasing in recent years (2011–present) or their status is uncertain; whereas introduced populations are generally considered stable with a few increasing and a few others decreasing.

Q7. What are your thoughts as to the current and future threats to sustaining goat numbers? Please numerically rank for each HD those that apply, with 1 being of highest importance. Leave blank those that don't apply. Compiled by hunting district as indicated by biologists (including Big Burn, Rattlesnake, Sun River, and North Big Belts).

	Count of HDs per category and ranking	Disease	Predation	Hunter harvest (total # animals)	Hunter harvest (proportion of females)	Habitat changes (non-anthropogenic)	ORV/Snowmobile disturbance	Energy exploration	Logging and/or road construction	Non-motorized recreation	Climate change	Small population risks (inbreeding, …)	Other (please describe in Q4)	Ltd Available Habitat
Native	Ranked 1st (7 points)	2	3	1	9	4	1				1	9		
	Ranked 2nd (6 points)	2	2	9		3				1	8	3		
	Ranked 3rd (5 points)	4	1		3	11	9			2	1	2		
	Ranked 4th (4 points)		5	2			10			1	1	2	3[a]	
	Ranked 5th (3 points)		2	1	1		1				9		1[b]	
	Ranked 6th (2 points)	2	1	1										
	Ranked 7th (1 point)													
	Count of HDs	**10**	**14**	**14**	**13**	**18**	**21**			**4**	**20**	**16**	**4**	
	Weighted score	**50**	**66**	**74**	**81**	**101**	**95**			**20**	**91**	**99**	**15**	
Introduced	Ranked 1st (7 points)	5	3	2	4	3	1			1		1		2
	Ranked 2nd (6 points)	1	4	4	5		1							
	Ranked 3rd (5 points)		2	4	1	4						2		
	Ranked 4th (4 points)		3	1	1	4	1							
	Ranked 5th (3 points)					1				1				
	Ranked 6th (2 points)		1											
	Ranked 7th (1 point)									1				
	Count of HDs	**6**	**13**	**11**	**11**	**12**	**3**			**3**		**3**		**2**
	Weighted score	**41**	**69**	**62**	**67**	**60**	**17**			**11**		**17**		**14**
Pooled	**Count of HDs**	**16**	**27**	**25**	**24**	**30**	**24**			**7**	**20**	**19**	**4**	**2**
	Weighted score	**91**	**135**	**136**	**147**	**161**	**112**			**31**	**91**	**116**	**15**	**14**

[a] Other factors were ranked 4th and described below in Q8 for native populations (HDs 312, 321)
[b] Other factors were ranked 5th and described below in Q8 for 1 native populations (HD 442)

The most commonly cited factors currently limiting goat numbers were habitat changes, followed by harvest of female goats, total goat harvest, predation, and ORV/snowmobile disturbance. But there were marked differences between perceived factors limiting native versus introduced populations. For introduced populations, predation, harvest of females, total harvest, and habitat changes ranked nearly equally as most important. For native goats, habitat changes were most important, followed by ORV/snowmobile disturbance, small population risks, and climate change concerns.

Compared to historical limiting factors (Question 3), there was less uncertainty about perceived limiting effects on populations. For introduced goat populations, effects of harvest levels on populations (total and females), habitat changes, and predation remain high.

For native populations, there is a shift away from concerns about harvest levels, to how impacts of habitat changes, ORV/snowmobile disturbance, climate change, and small population risks are affecting populations. In part this is because harvest levels of native populations have been slashed over the years (9 HDs with native goats are now closed to hunting). Only 38 permits were issued to hunt goats in the 26 HDs with native populations in 2015. Thus other risks to population viability have replaced earlier concerns with harvest levels.

Q8. Please elaborate here on the limiting factors you marked in Q7. For example, if you selected predators, disease, hunter harvest of females, or climate change, please explain.

Native Populations:

- Several biologists wrote that the concerns they identified in Question 7 were cumulative, perpetuating suppression of goat numbers that may have begun prior to 2010.
- Where populations are now small and isolated, inbreeding depression is a concern.
- For several populations, habitat is limited. "Forest encroachment, due to fire suppression, on some of these higher elevation ranges may be limiting available winter forage." Also noted were concerns that fire suppression has exacerbated forage competition with elk, bighorns, moose, or deer populations in places.
- Concern was expressed that hunter harvest success and effort are not good measures of how a herd is doing.
- Disease impacts (both introduced and native goat herds) are surmised, but not documented. These concerns were expressed for HDs where bighorns have experienced pneumonia die-offs, although the same has not been documented in goats. A disease die-off is circumstantially implicated in HD313 in the past.
- Harvest of adult female goats (roughly 38% of the total harvest historically) is a concern in some populations of native and introduced herds.
- Increased recreation (both motorized and non-motorized) are suspected of impacting growth of goat populations. This could result from displacement and/or physiological stress, but neither has been studied to confirm.
- Through changing plant phenology, dwindling snow in summer, and late-winter snow events, climate change probably contributes to declining viability of some herds.

Introduced Populations:

- More concerns were expressed about predation on goats in introduced than native populations, with lions stated to be of greatest concern. However, several biologists noted that predation on goats was not well documented, or only suspected (in some introduced and native HDs).
- In HDs in the Madison, Gallatin, and Crazy Mountains, harvest objectives and rates that are higher than are sustainable in native herds are being monitored to insure overharvest doesn't occur.
- Concern expressed that for herds with limited habitat, insufficient harvest could lead to overuse of available range. And transplanting bighorns into HD332 may have not only disadvantaged a small goat population but contributed to an increase in lion predation on goats.

Harvest and Season-setting

Q9. What best describes your objectives when allocating mountain goat licenses (select one)? One response per HD only for those HDs open to hunting now.

Native Populations (HDs) **You can provide separate answers for individual or groups of HDs, or if answer is same across your area you can just put "ALL"	Provide conservative number of licenses to allow opportunity with **minimal impact**	Provide **maximum sustainable** number of licenses that still maintains current population size	Provide enough licenses to **limit or decrease the current population** size	Other (please describe):
100	x			
101	x			
131	x			
132	x			
133	x			
134	x			
140	x			
141	x			
142	x			
150	x			
151	x			
212				No licenses
213				No licenses
222				No licenses
223				No licenses
240	x			
250				No licenses
261				No licenses
270				No licenses
280				No licenses
312		x		
321				No licenses
322	x			
414	x			
415	x			
442	x			
Great Burn (No HD)				No licenses
Rattlesnake (No HD)				No licenses
Sun River Preserve (No HD)				No licenses
NATIVE TOTAL	16	1		

Introduced Populations (HDs) **You can provide separate answers for individual or groups of HDs, or if answer is same across your area you can just put "ALL"	Provide conservative number of licenses to allow opportunity with **minimal impact**	Provide **maximum sustainable** number of licenses that still maintains current population size	Provide enough licenses to **limit or decrease the current population** size	Other (please describe):
313			X	
314				x
316				x
320	x			
323				x
324		x		
325		x		
326				x
327		x		
328		x		
329				x
330				x
331	x			
332				No licenses
340				No licenses
361	x			
362		x		
380				No licenses
393		x		
447	x			
453		x		
460	x			
514	x			
517	x			
518	x			
519	x			
North Big Belts (no HD)				No licenses
INTRODUCED TOTAL	9	7	1	6

Biologists managing native HDs take an almost unanimously conservative approach to harvest. For HDs with introduced goats, objectives are more varied with the "Other" responses aimed at limiting population growth.

Q10. Which of the following describes the quantity and quality of your goat survey and inventory information with respect to making management decisions (select one)? One response per biologist.

	Adequate	Somewhat adequate	Somewhat inadequate	Inadequate
Native Populations (HDs)		2	4	4
Introduced Populations (HDs)	1	4	2	
Pooled	1	6	6	4

These results suggest that more adequate survey data are collected in HDs with introduced goats. This may be because most goat permits (84% in 2015) are issued in HDs with introduced goats and therefore these goat populations are surveyed more often or thoroughly.

Q11. What information do you currently use to set annual goat harvest regulations? Please numerically rank those that apply with 1 being of highest importance, leaving blank those that don't apply. Compiled by hunting district as indicated by biologists.

	Count of HDs per category and ranking	**FWP hunter harvest data**	**FWP hunter effort data** (e.g., kills per effort)	**FWP observations data** (e.g., number seen/hunter)	**Age and/or horn data**	**Anecdotal hunter reports** (i.e., not in MRRE)	**Survey minimum counts**	**Survey recruitment ratios**	Other (please describe):
Native	Ranked 1st (7 points)	1					3	2	
	Ranked 2nd (6 points)		1			1	1	3	
	Ranked 3rd (5 points)	2	1	1	1				
	Ranked 4th (4 points)		2	1			1		
	Ranked 5th (3 points)	1		2		1			
	Ranked 6th (2 points)	1				1			
	Ranked 7th (1 point)				1				
	Count of HDs	**5**	**4**	**4**	**2**	**3**	**5**	**5**	
	Weighted score	**22**	**19**	**15**	**6**	**11**	**31**	**32**	
Introduced	Ranked 1st (7 points)	2					6		1
	Ranked 2nd (6 points)		1				2	5	
	Ranked 3rd (5 points)	2	1		2	1		1	
	Ranked 4th (4 points)	1	3	2 /8				4	
	Ranked 5th (3 points)	1			3	2			
	Ranked 6th (2 points)	1	1	1		2			
	Ranked 7th (1 point)			1	1				
	Count of HDs	**7**	**6**	**4**	**6**	**5**	**8**	**10**	**1**
	Weighted score	**33**	**25**	**11**	**20**	**15**	**54**	**51**	**7**
Pooled	**Count of HDs**	**12**	**10**	**8**	**8**	**8**	**13**	**15**	**1**
	Weighted score	**55**	**44**	**26**	**26**	**26**	**85**	**83**	**7**

Survey minimum counts and survey recruitment ratios are the two types of data on which biologists place the greatest reliance in setting harvest regulations. This is true for both native and introduced populations. This emphasizes the importance of obtaining reliable population survey data.

The next two factors most relied on to set regulations were FWP harvest data (number of animals harvested relative to number of permits issued) and hunter effort data (number of days/animal harvested). With mandatory reporting of mountain goat kills, these may be the most consistently available data at biologists' disposal.

Q12. If better or more frequent survey data would help you set harvest quotas, what factors are *__most__* limiting to survey efforts (e.g., funding, time, aircraft availability, weather, other logistics, etc…)? Compiled by responses from each biologist (multiple factors listed by biologists are included).

The factors most frequently reported were:

- Aircraft/pilot availability (11)
- Weather (11)
- Funding (10)
- Time (6)
- Sightability Correction Model needed (1)
- Cooperation with Idaho on the border goat herd in HD322 (1)

Several biologists listed all of the top 4 factors in their responses.

Q13. Have any of your proposed quotas for other species, such as mountain lions, been affected by numbers or recruitment ratios of overlapping mountain goat populations? If so, please explain. One response per biologist.

	Yes	No
Native populations		9
Introduced populations	1	7
Pooled	**1**	**16**

Q14. Based on your conversations with hunters, what % of hunters in your area take into consideration the animal's sex (i.e., deliberately target males) when choosing to harvest a given mountain goat (circle one)? One response per biologist for those with licensed HDs.

0	10	20	30	40	50	60	70	80	90	100	Uncertain
1			2		5		2	1	2		2

The weighted average of the responses was 55%.

Q15. Based on your conversations with hunters, what % of hunters in your area would you expect correctly identify the animal's sex when choosing to harvest a given mountain goat (circle one)? One response per biologist for those with licensed HDs.

0	10	20	30	40	50	60	70	80	90	100	Uncertain
1			2		6	1		1	2		2

The weighted average of the responses was 52%. This suggests that half of permittees are as likely to kill a nanny as a billy, all other factors being equal (goat population demographics, sex-biased distribution, etc.).

Q16. Is the educational information provided to license-holders sufficient for hunters to make informed decisions about the age and sex of the animals they choose to harvest? If not, what more could be done? One response per biologist for those with licensed HDs.
[3] Yes
[6] No
[6] Uncertain

Comments offered:

- Work with other states to improve educational materials (3)
- Use Alaska education information or something similar (2)
- FWP used to send out informational letters (1)
- Mandate billy only seasons (1)
- Send hunters the brochure developed by Gayle Joslin (1)
- Hunters could be required to take in-person mandatory training (1)

Population surveys

Q17. What survey methodology do you use to assess mountain goat population size and trend? Please check all that apply. Compiled by responses from each biologist.

Populations (HDs)	Methodology			Season				Frequency				
	Fixed-wing	Heli-copter	Ground	Winter/ Early Spring	Jul-Aug	Aug-Sept	Early Fall	Annual	Every other year	Every few years	Rarely	Never
Fixed-wing	6				1	3	2	2		1	2	1
Helicopter		20		6	4	5	3	5	5	4	7	1
Ground			8		1	7		3	2	2	1	

Some respondents indicated they use multiple survey methods at differing times of the year.

Q18. Do you feel it is important that FWP monitors mountain goats using similar methods across regions of the state (e.g., timing and frequency of surveys, choice of aircraft, etc.)? One response per biologist.

Yes	No	Uncertain
2	14	2

Q19. Do you see a difference between native vs. introduced goat populations in terms of general health or productivity/recruitment? If so, please describe. One response per biologist.

Yes	No
5	1

Several biologists noted they did not have enough information to answer this question or that they only had either native or introduced goats in their area of responsibility and therefore could not judge. Several others did not respond.

Comment: "We have a health baseline for the Crazies. Maybe it would be prudent to do some health captures in other areas to compare, or at a minimum, get a hunter sampling protocol going similar to bighorns."

Habitat Considerations

Q20. What habitat management programs would promote mountain goat conservation and hunter opportunity in your area of responsibility? Please numerically rank those that apply for each population or group of populations with 1 being of highest importance. One response per biologist.

Count of HDs per category and ranking	**None;** Habitat isn't a limiting factor	**More fire** (natural or prescribed)	**Less fire** (suppression of wildfire)	**Weed management**	**Road management** (ie., more restrictive)	**ORV management**	**Snowmobile management**	**Non-motorized recreation management**	**Unknown**	**Other** (please describe):
Ranked 1st (3 points)	3	5	1			1	3	1	4	
Ranked 2nd (2 points)			1			1	1	2	1	
Ranked 3rd (1 point)				1					1	
Count	**3**	**5**	**2**	**1**		**2**	**4**	**3**	**6**	
Weighted score	**9**	**15**	**5**	**1**		**5**	**11**	**5**	**15**	

There was little consensus about which, if any, habitat management programs would benefit goat conservation or increase hunter opportunity. The three recreational management categories had a combined weighted score (21) larger than any other category.

Q21. Have you completed any habitat-related projects alone or with federal or other land managers related to the subjects in Question 20 that were geared to improve mountain goat habitat or conservation? Please explain, listing HDs for which the projects were completed. One response per biologist.

Yes	No
1	16

Comments offered:

- Would like to support more burning on USFS lands
- Have worked with BLM and USFS to remedy conifer encroachment but no projects yet
- Yes response is for comments on USFS motorized travel restrictions in goat habitat

Research & Management Needs

Q22. Is there a pressing need for translocation of mountain goats into a portion(s) of your area to sustain native and/or introduced populations? If so, would this be to reintroduce an extirpated population or augment an extant population? Please explain. One response per biologist.

	Yes	No
Native	2	4
Introduced	3	7
Pooled	**5**	**11**

Introductions need to be carefully evaluated on an area-by-area (herd-by-herd) basis, as indicated by the comments below.

Comments offered:

Native Herds:

- Need better population data to determine any needs for augmentation. (2)
- We would need to first understand what is driving population declines and get a better idea of the actual number of goats in the area. If it is disease or habitat driven, then why dump more goats into areas? (2)
- Yes, HD240 and possibly 250 to augment struggling populations.
- Yes, for augmentation in 212, and 222-223. However, the disease issue (bighorn pneumonia) is a huge unknown.

Introduced Herds:

- Yes, HD380 and North Big Belts: To augment small, extant populations.
- Yes, Boulder Baldy area and Big Baldy area of Little Belts
- Yes, Highwoods and Square Butte to improve genetic diversity of isolated populations.
- No, all habitat is occupied and goats are self-sustaining.

Q23. What are the most urgent *research* needs that would help you manage mountain goats in your area of responsibility?

- Habitat condition and use and carrying capacity (9)
- Population demographics: productivity, recruitment, kid survival, and adult survival (7)
- Causes of mortality (5)
- Animal health (3)
- Sightability correction model for survey data (2)
- Improved survey methodology (2)
- Effects of recreation on populations (1)
- Effects of climate change on populations (1)
- Better information on dispersal of introduced herds (1)
- Impacts on populations of female harvest (1)
- Competition and disease transmission of sympatric bighorns and goats (1)
- Do we know if population augmentation can overcome small population effects? (1)

Q24. What are the most urgent *management or monitoring* needs that would help you manage mountain goats in your area of responsibility?

- Better/more frequent monitoring of populations (10)
- Sightability correction model and improved, standardized survey methodology (5)
- Monitoring of health (2)
- Coordinated and cooperative management with Idaho of boundary herds (1)
- Field work to determine movements of goats between adjacent HDs (1)
- More time to devote to learning about goats to improve management (1)
- Transplant augmentation (1)
- Continue to collect harvest data and ages of harvested goats (1)

Q25. What other topics of relevance did we miss with these questions?

- Focus on predation of goats (1)
- Potential effects of goats on bighorns in the GYE, i.e. Bob Garrott's research (1)
- More FWP effort should be shifted to species that may be at risk, like goats (1)
- Need extended field studies of small goat populations to develop an understanding of how remnant native populations survive. This could help develop bigger research questions and conservation priorities. Need more first-hand familiarity via field studies (e.g. grad students) (1)

Appendix I

Kylie Paul Environmental Consulting

4/28/17

POTENTIAL CONFLICTS BETWEEN WILDLIFE AND OVER-SNOW RECREATION IN THE SCOTCHMAN PEAKS/SAVAGE PEAK AREA

SUMMARY

The Scotchman Peaks, including Savage Peak and Savage Basin, contain valuable winter range habitat for mountain goats and important habitat for other species such as wolverines, grizzly bears, and Canada lynx. Winter is a difficult time for wildlife survival, with marginal food resources and higher physiological stress. For mountain goats in particular, winter range is a highly restricted and thus critical area for them, as they require both protection from predators and proximity to limited food sources in mountainous areas. In addition to these wintery challenges, mountain goats are also highly sensitive to human disturbances such as snowmobiles. Their responses to disturbance can change mountain goat population dynamics. Restricting motorized recreational use from mountain goat winter range helps minimize impacts during this difficult season.

Land and wildlife management agencies (Montana Fish Wildlife & Parks and United States Forest Service) have been concerned about snowmobiling in mountain goat habitat in the Scotchman Peaks area, particularly into Savage Peak/Mountain region for many years. Those agencies support the continuation of non-motorized activities and wilderness designation in the Scotchman Peaks and Savage Peak area. Preserving the year-round closure to motorized activity across the Scotchman Peaks including the Savage Peak area, regardless of wilderness designation, will continue to protect the wildlife and wildlife habitat in this unique setting.

THE SCOTCHMAN PEAKS CONTAIN HIGH-QUALITY WILDLIFE VALUES

The Scotchman Peaks Recommended Wilderness Area (Scotchman Peaks) is within the Cabinet Mountains on the border of Montana and Idaho. The Scotchman Peaks sit within both the Kootenai National Forest and the Idaho Panhandle National Forest. Savage Peak (also known as Savage Mountain) and Savage Basin, the basin northeast of Savage Peak, is an important area within the Scotchman Peaks on the Montana side in the Kootenai National Forest. This area contains valuable habitat and supports a variety of important wildlife species such as mountain goats, wolverine, and grizzly bears.

MOUNTAIN GOAT HABITAT IN THE SCOTCHMAN PEAKS

Mountain goats are native to most of the mountain ranges of western Montana (Rideout 1977). They occupy the highest, coldest, most rugged regions of any ungulate in North America (Chadwick 1983). Mountain goats display seasonal altitudinal migrations over short distances (White 2006; Rice 2008), with all mountain goat habitat generally characterized as areas close to escape terrain (steep slopes, usually ≥40°) such as cliffs and away from valleys (Festa-Bianchet and Côté 2008; Shafer et al. 2012). Mountain goats thus are limited to relatively small areas of suitable habitat (Canfield et al. 1999).

Winter is an important season for mountain goats and is characterized by high juvenile mortality (Poole et al. 2009) and restricted, shorter movements (Chadwick 1983; White 2006) that are influenced by snow depth and snowpack (Richard et al. 2014). Winter range is considered critical habitat for mountain goats (Côté and Festa-Bianchet 2003), and their winter ranges are much smaller than summer ranges, ranging from 2%–50% of the size of summer ranges (Taylor et al. 2006; Poole et al. 2009).

Generally, mountain goats winter range occurs in rugged habitat at upper mid-elevations and on warmer aspects, close to escape terrain (Poole et al. 2009). They spend most their time near escape terrain to avoid and escape predation (Chadwick 1983; Gross et al. 2002; Hamel and Côté 2007; Poole et al. 2009) and for shelter from harsh weather (von Elsner-Schack 1986). They also require easy access to summer range and kidding areas. As early as late April, nannies select the most isolated and forbidding terrain to give birth (MFWP 2016).

There are some winter habitat use differences between populations in western North America, with two wintering strategies that occur: (1) populations from interior regions (e.g., the Rockies) spend winter above treeline on windswept ridges and ledges found in steep rugged terrain (Hebert and Turnbull 1977; Côté and Festa-Bianchet 2003; Poole et al. 2009), while (2) coastal populations living in areas of greater snowfall migrate downhill to spend winters in low-elevation forested areas (Hebert and Turnbull 1977; Poole and Heard 2003; Taylor et al. 2006; Poole et al. 2009). There also appear to be different strategies to avoid deep snow within the populations of the interior mountainous regions, with animals wintering either: (1) on high-elevation wind-swept slopes or (2) inhabiting rocky bluffs at treeline in areas of higher snowfall where wind-swept slopes are unavailable (Hebert and Turnbull 1977; Rideout 1977; Chadwick 1983; Poole and Heard 2003). There are also differences of fine-scale habitat use in the winter depending on sex and individual, with some level of differing habitat preferences between the sexes (Festa-Bianchet and Côté 2008; Shafer et al. 2012) and with differences in movement patterns accounting for differences in home range sizes among individuals (Poole and Heard 2003).

Throughout the entire Kootenai National Forest, only the West Cabinet and Cabinet Mountains, within which the Scotchman Peaks is situated, offer mountain goat habitat (KNF 2015a). The Scotchman Peaks, including Savage Peak, contain high-quality mountain goat winter range (Figure 1) and have long had a population of mountain goats (Joslin 1980). Savage Peak and surrounding smaller summits are characterized by very steep slopes with cliffs, offering escape terrain. The Savage Peak area contains both important winter range and summer transitional range, between and within which mountain goats need to move easily to prosper (Joslin 1980; Joslin, G. personal communication, April 6, 2017).

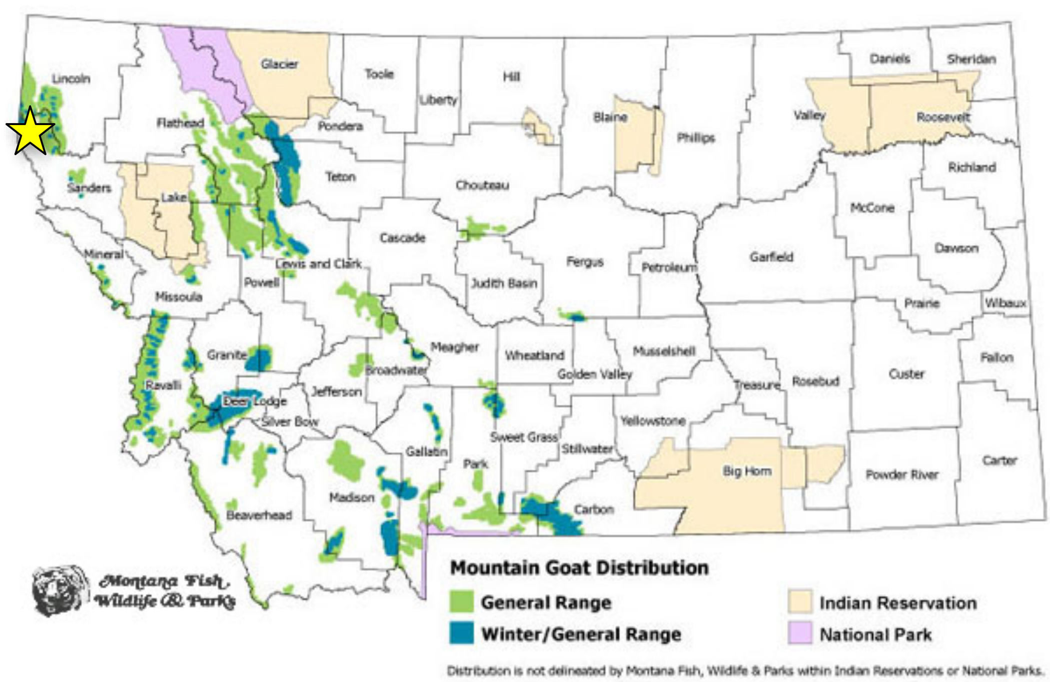

Figure 1 Mountain goat general range and winter range in Montana. Star is Savage Peak area. Data available at Montana Field Guide.

POPULATION AND STATUS OF MOUNTAIN GOAT IN MONTANA

Mountain goats are currently ranked as a Montana Species Ranking Code S4, so they are considered "apparently secure, though it may be quite rare in parts of its range, and/or suspected to be declining."[1] Similarly to other nearby regions such as Alberta, the overall population declined in the past and now contains some smaller populations that are stable, some that are increasing, and others that are continuing to decline (Gonzalez-Voyer et al. 2003; Koeth 2008).

Montana Fish Wildlife & Parks has documented mountain goats in the Scotchman Peaks area for decades (MFWP 2016). Goat numbers peaked in the late 1930s at 110 animals and steadily declined to 20-25 goats in the 1970s (Burleigh 1978). In the late 1970s, due to concerns over these decreasing mountain goat numbers, Montana Fish Wildlife & Parks closed goat hunting in mountain goat Hunting District 101, which includes the Scotchman Peaks. Montana Fish Wildlife & Parks and the Kootenai National Forest then performed research that led to the development of a goat management plan, a joint memorandum of understanding, and a population augmentation project for mountain goats (Joslin 1980). Montana Fish Wildlife & Parks slowly reinstated harvest in the late 1980s, but because mountain goat numbers did not increase to the degree expected and because of concerns over decreasing goat numbers in this area and across their range, the agency again reduced goat harvest quotas in mountain goat Hunting District 101 in 2010 (MFWP 2016). Currently, Montana Fish Wildlife & Parks continues to monitor goat numbers and other game species using aerial surveys and hunter harvest information.

GRIZZLY BEAR HABITAT IN THE SCOTCHMAN PEAKS

Grizzly bears are listed as a threatened species under the Endangered Species Act. Grizzly bear distribution has been reduced to five areas in the western United States, and there are six individual recovery zones delineated in the lower-48 states to include "adequate space and suitable habitat for securing and restoring viable self-sustaining grizzly bear populations in perpetuity" (USFWS 1993). These six recovery zones include the Greater Yellowstone, Northern Continental Divide, Cabinet-Yaak, North Cascades, Selkirk, and Selway-Bitterroot grizzly bear ecosystem.

The Cabinet-Yaak Recovery Zone includes the Scotchman Peaks, which contain core grizzly habitat (Figure 2)(Proctor et al. 2015). The grizzly bear population in the Cabinet-Yaak Recovery Zone was estimated at 48-50 bears in 2012, with 22-24 of those occurring in the Cabinets area (including Scotchman Peaks) (Kendall et al. 2016). To improve genetic diversity and increase the population, population augmentation has been successfully accomplished on several occasions in the Cabinet Mountains since 1979, with the most recent grizzly bear released in 2016 at Spar Lake, near the Savage Peak area (IGBC 2016). Given its small population size and the slow reproductive rate of the species, the Cabinet-Yaak population is highly sensitive to mortality and disturbance.

[1] Montana Fish Wildlife & Parks. Montana Field Guide: Mountain Goat. http://fieldguide.mt.gov/speciesDetail.aspx?elcode=AMALE02010

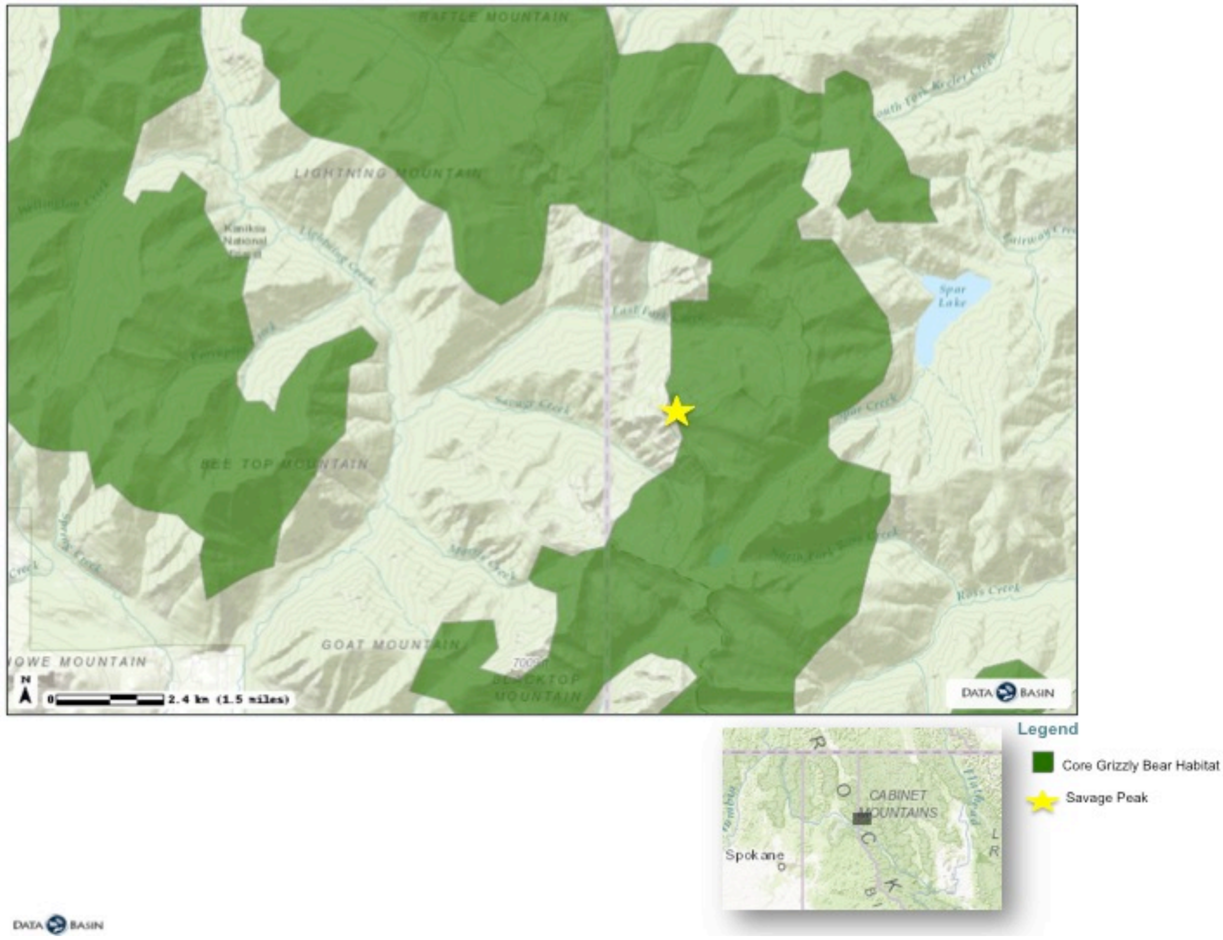

Figure 2. Grizzly bear core habitat in the Scotchman Peaks area. Data from Proctor et al. 2015 on Databasin.

WOLVERINE HABITAT IN THE SCOTCHMAN PEAKS

Wolverines are again under consideration for listing under the Endangered Species Act. Population number and trend in the contiguous United States are unknown, though the population is generally estimated at 250-300 individuals (USFWS 2013).

Wolverines in the northern Rockies live primarily in high-elevation environments that maintain colder temperatures and reduce competition with other carnivores (Copeland et al. 2010; McKelvey et al. 2011; Inman et al. 2013). The Scotchman Peaks contain both primary and maternal wolverine habitat, with the Savage Peak area containing maternal denning habitat, the most limiting and thus valuable habitat type for wolverines (Figure 3).

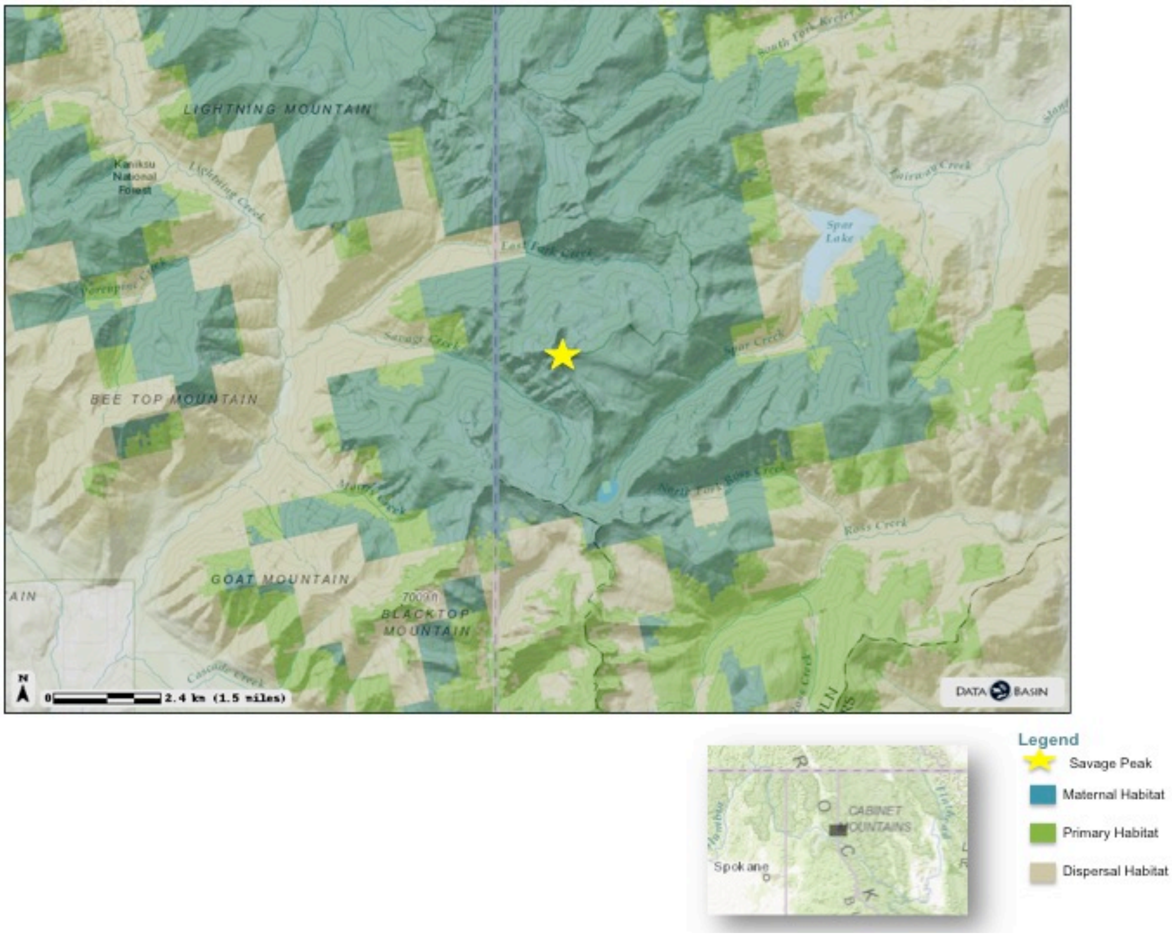

Figure 3. Wolverine primary habitat, maternal habitat, and dispersal habitat in the Scotchman Peaks area. Data from Inman et al. 2013 available on Databasin.org. Primary wolverine habitat is defined as the area within the climactic limits of wolverines that resident adult wolverines are expected to occupy, and maternal habitat is defined as areas that contain attributes consistent with those measured around the known wolverine dens used in the Inman et al. (2013) study.

CANADA LYNX HABITAT IN THE SCOTCHMAN PEAKS

Canada lynx are listed as threatened under the Endangered Species Act. Population number and trend in the contiguous United States are unknown.

Lynx habitat is characterized by moist boreal forests that have cold, snowy winters and a high-density snowshoe hare prey base (Interagency Lynx Biology Team 2013). The range of lynx in the West has diminished over the last century, suggesting that lynx may be negatively impacted by human activities (Koehler and Aubry 1994).

The Kootenai National Forest is home to one of just a few known resident lynx populations in the lower 48 states. Critical habitat has been designated within the Kootenai National Forest, and the Forest is designated “occupied lynx habitat” (Figure 4). The entire Kootenai National Forest is in “core area” as described in the Lynx Recovery Outline (USFWS 2005). The Scotchman Peaks are considered occupied and core habitat, though they are not included within Critical Habitat.

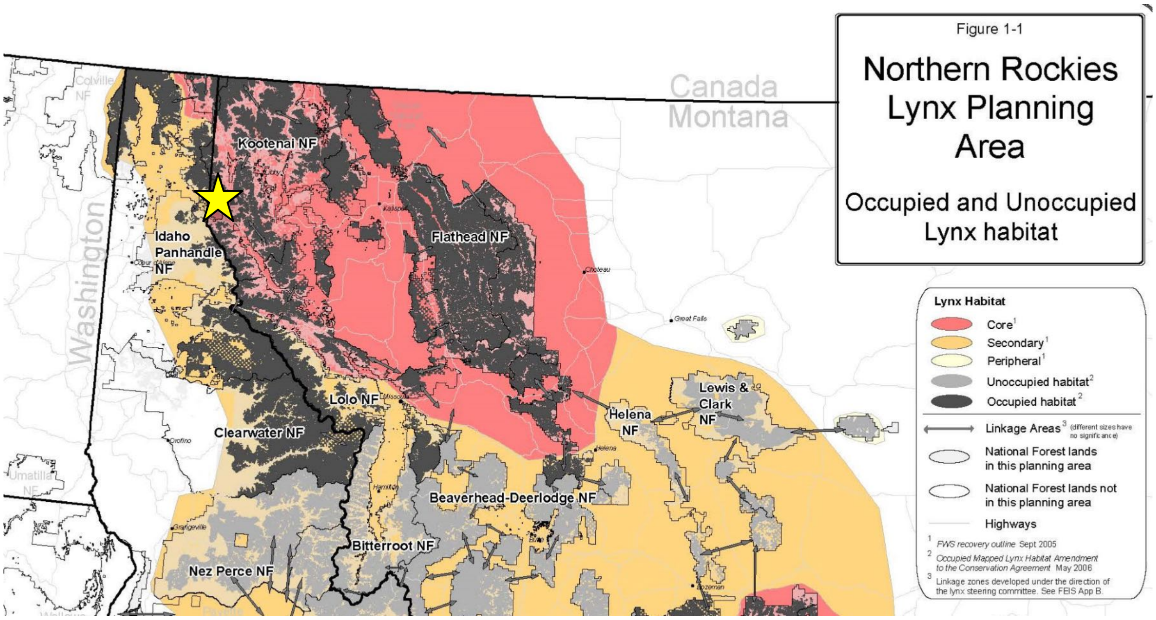

Figure 4 Canada lynx habitat in the Northern Rockies, including Kootenai National Forest and Scotchman Peaks area. Star is Savage Peak area. Map from USFS at www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5446686.pdf

WILDLIFE ARE IMPACTED BY SNOWMOBILES

Motorized winter backcountry recreation is one of the fastest growing recreational activities in the United States (Cook and O'Laughlin 2008). In 1982-83, government surveys put the number of snowmobile participants in the United States at 5.3 million (Cordell et al. 1999). The most recent survey, conducted in 2010, estimates that in the United States, 10.7 million people now snowmobile annually (Cordell 2012). Due to advanced technology with more powerful machines, snowmobiles and new "snow bikes" (modified motorcycles with tracks instead of wheels) are now better able to reach areas that were previously inaccessible.

While snowmobiling continues to grow in popularity, snowpack continues to decline due to climate change. Recent warming has already led to substantial reductions in spring snow cover in the mountains of western North America (Mote et al. 2005; Pederson et al. 2010). This continues to further concentrate motorized winter recreation into the smaller amounts of available, sufficiently snowy areas. Wildlife that require snowy habitats will also have reduced amounts of available habitat and will essentially need to compete for the same remaining snowy habitat as snowmobilers. For instance, numerous studies indicate that global climate change is likely to negatively affect wolverine habitat (Gonzales et al. 2008; Copeland et al. 2010; McKelvey et al. 2011; Peacock 2011; Johnston et al. 2012). Additionally, climate modeling suggests that snow accumulation and duration are expected to decline and that lynx habitat and populations are anticipated to decline accordingly (Carroll 2007) and may disappear completely from parts of their range by the end of this century (Johnston et al. 2012). This leads to increasing concern for wildlife and their ability to find secure winter habitat.

Any disturbance, such as that from snowmobiles, during this important winter period can negatively affect productivity and other vital rates (May et al. 2006; Krebs et al. 2007). Snowmobiles can cause harassment, habitat loss, and mortality of wildlife such as ungulates (Dorrance et al. 1975; McLaren and Green 1985;

Freddy et al. 1986; Tyler 1991; Olliff et al. 1999a; Olliff et al. 1999b; Seip et al. 2007; Harris et al. 2014; Switalski 2016).

OTHER SPECIES ARE IMPACTED BY SNOWMOBILES

This document focuses on snowmobiles and mountain goats – yet other species of concern within this region are also impacted by snowmobiles including wolverines, grizzly bears, and Canada lynx.

Wolverine researchers and natural resource managers have long expressed concerns about effects of winter recreation on wolverine populations, as motorized winter recreation can negatively impact wolverine particularly by disrupting natal denning areas (Hornocker and Hash 1981; Copeland 1996; Carroll et al. 2001; Rowland et al. 2003; May et al. 2006; Copeland et al. 2007; Inman et al. 2007; Krebs et al. 2007; Lofroth and Krebs 2007; Ruggiero et al. 2007; Heinemeyer and Squires 2013). Female wolverines select and enter dens and give birth in February to mid-March (Magoun and Copeland 1998) and the overlap of winter recreation with this energetically taxing period is highly concerning.

Grizzly bears denning habitat often overlaps with winter recreation areas, making them susceptible to disturbance at their den sites and upon emergence (Linnell et al. 2000). Potential effects of disturbance to denning bears include elevated energy use from increased movements in the den (Reynolds et al. 1986; Schoen et al. 1987), den abandonment (Craighead and Craighead 1972; Reynolds et al. 1976; Harding and Nagy 1980; Schoen et al. 1987), potential loss of cubs (Schoen et al. 1987), and displacement from denning areas (Craighead and Craighead 1972; Schoen et al. 1987). Females with cubs may be more vulnerable to snowmobile disturbance following den emergence than during the denning period (Mace and Waller 1997).

Snow-packed trails created by snowmobiles have been considered as possibly serving as travel routes for potential competitors and predators of Canada lynx, especially coyotes (Ozoga and Harger 1966; Murray and Boutin 1991; Koehler and Aubry 1994; Murray et al. 1995; Buskirk et al. 2000) though the causal relationship is not entirely clear (Bunnell et al. 2006; Kolbe et al. 2007; Burghardt-Dowd 2010). As snow levels diminish with climate change, lynx habitat will shrink and winter recreation will continually become a more serious threat to the persistence of lynx.

MOUNTAIN GOATS ARE IMPACTED BY SNOWMOBILES

Research has firmly established that undisturbed winter range is essential for ungulate survival (Canfield et al. 1999). Snowmobile activity disturbs wintering ungulates through physiological stress (Canfield et. al 1999; Creel et al. 2002) from increased movements and higher energy expenditures (Dorrance et. al 1975; Freddy et. al 1986; Tyler 1991; Colescott and Gillingham 1998; Borkowski et al. 2006).

Predation appears to be the main cause of mortality for mountain goats (Festa-Bianchet and Côté 2008). As such, predation risk appears to be the main factor influencing mountain goat space use, as they are strongly associated with escape terrain and aggregate in groups (Hamel and Côté 2007; Gross et al. 2002; Festa-Bianchet and Côté 2008; Richard et.al 2014). To avoid predators, mountain goats rely on detecting them by sight or sound from distance and then moving into escape terrain where predators are unable to follow (Festa-Bianchet and Cote 2008). Mountain goats are particularly sensitive to human disturbances (Festa-Bianchet and Côté 2008; St-Louis et al. 2013; Richard and Côté 2016), using the same anti-predator strategy. They change their behavior (e.g. increased alertness and reduced time foraging) and their spatial distribution (e.g. moving or running) when facing various human-caused activities (Singer 1978; Foster and Rahs 1983; Joslin 1986; Côté 1996; Gordon and Reynolds 2000; Côté et al. 2013; St-Louis et al. 2013; Richard and Côté 2016). These short-term impacts on behavior could translate to consequences to movement rates, range use, and ultimately, survival and population productivity (Festa-Bianchet and Côté 2008).

The trigger for behavioral responses to human disturbances can be quite distant; in one study in Alberta, goats were highly disturbed and increased their alertness behaviors when helicopters flew nearby, with no habituation seen across numerous years of helicopter traffic (Côté et al. 2013). Researchers subsequently recommended helicopter flights should not approach closer than 1,500 m (4,920 ft) from mountain goat groups (Cadsand 2012; Côté et al. 2013).

Mountain goats' struggle to move away from disturbance can be energetically taxing. Living in harsh winter habitat, mountain goats have a low margin for unnecessary energy costs without impacts on survival and reproduction (Harris et al. 2014). As Montana Fish Wildlife & Parks has noted, at winter's end, goats have nearly depleted all their fat reserves, and "goats are right on the survival line in late winter and early spring…That's also when snow is hardest and snowmobilers like to 'high-mark' [climb snow-covered mountainsides]" (Koeth 2008).

These responses to disturbance can change mountain goat population dynamics. One of the factors thought to contribute to declines in mountain goat populations is repeated disturbance (Joslin 1986; Festa-Bianchet and Côté 2008). For instance, the cumulative effects of stress caused by a high amount of motorized human disturbance in one Montana population may have been responsible for reduced kid production, reduced numbers of female goats, and a declining goat population (Joslin 1986). All-terrain vehicle use on trails in mountain goat summer range in Alberta caused moderate to strong disturbance reactions by goats 44% of the time, with potential detrimental effects on fitness-related behaviors such as feeding and parental care (St-Louis et al. 2013).

For mountain goats, winter range is a highly restricted area, as they spend most of their time close to escape terrain (Poole et al. 2009). While these small areas of winter range are often less accessible to humans, the advancing technology of over-snow vehicles offers increased human access to areas of mountain goat winter habitat (Koeth 2008). In general, mountain goats are at risk from snowmobile activity, with their high sensitivity to disturbance and with the ensuing behavioral responses and energetic costs that can negatively impact population dynamics.

POTENTIAL CONFLICTS BETWEEN MOUNTAIN GOATS AND SNOWMOBILES IN THE SCOTCHMAN PEAKS AND SAVAGE PEAK AREA

SNOWMOBILE USE IN THE SCOTCHMAN PEAKS AND SAVAGE PEAK AREA

The Kootenai National Forest has long recognized the ecological importance of the Scotchman Peaks area and has supported congressional action for wilderness designation of the Scotchman Peaks. In 1987 and 2015 the Kootenai National Forest recommended the Scotchman Peaks area for wilderness (KNF 1987). Motorized restrictions proposed in the 1987 Forest Plan in recommended wilderness were formalized in 2001, when the Kootenai National Forest created a Special Order (#F14-064S01) that restricted all motorized access year-round in the Scotchman Peaks (KNF 2013a). In the 2015 forest plan revision, the Kootenai National Forest re-evaluated the area and concluded it continues to merit for a recommended wilderness designation (KNF 2015b). The Scotchman Peaks thus remain closed to over-snow vehicles (KNF 2015b).

Over-snow motorized access was legal on the Idaho Panhandle National Forest until 2015, when the Forest signed an Order (#01-04-00-15-001) prohibiting winter motorized access on the Idaho Panhandle National Forest side of the Scotchman Peaks within the Sandpoint Ranger District. This preserved the motorized access restrictions on the Montana side, especially in the areas around Savage Peak, and helped maintain consistency of existing conditions from Idaho into Montana.

Some snowmobilers would prefer to have access into the Savage Peak/Basin area.[2] However, this is not feasible nor in line with Kootenai National Forest goals, as clarified by the Kootenai National Forest:

> The Savage Peak…area [is an] important part of the Scotchman Peaks recommended wilderness area…The Savage Peak area has been closed to over-snow vehicle use since the 1987 Forest Plan was adopted… Under the revised Forest Plan, the boundary for the Scotchman Peaks recommended wilderness area was drawn to be identifiable on the ground and manageable. (KNF 2013b).

CONCERNS WITH MOUNTAIN GOATS AND SNOWMOBILES IN SAVAGE PEAK AREA

The mountain goat population in the Scotchman Peaks has concerned Montana Fish Wildlife & Parks and Kootenai National Forest due to its declining population for decades (MFWP 2016). Research indicates that small mountain goat herds (<50 animals) have a high extinction risk (18%-82% over 40 years) even in the absence of harvest (Hamel et al. 2006), so managing for the factors underlying these population declines is critical.

Scotchman Peaks and Savage Peak/Basin area contain important high quality winter range for mountain goats, and there is concern with human disturbance to mountain goats in the area. The Savage Peak area contains "management situation 1" lands in Joslin (1980), which are areas that provide critical mountain goat range during summer and/or winter. Joslin (1980) states: "Mechanized human activities should not occur in these areas. Human activities on adjacent areas should be kept to a minimum during the seasons when these areas are used by goats."

For over a decade, Montana Fish Wildlife & Parks has shared concerns of snowmobiling in mountain goat habitat in the Scotchman Peaks area, particularly into Savage Peak/Mountain region:

> The need to maintain mountain goat habitat security in the Scotchman Peaks Area is no less relevant today than it was 25 years ago. It is unfortunate that snowmobiling activity into Savage Mountain, right in the heart of the Scotchman Peaks goat range, has been allowed to continue unchecked over the past several years, despite the illegality of motorized access into this area as identified in the MA guidelines for this area. (MFWP 2004).
>
> FWP concurs with the proposed Scotchman Peaks #662 proposed Wilderness area as identified due to its value as critical native mountain goat habitat, elk and mule deer habitat, and important grizzly bear season-long habitats. FWP will soon be initiating grizzly bear augmentation efforts in the vicinity of this area. This area also satisfies a national demand for a backcountry hunting experience. FWP also concurs with the 5A designations for areas surrounding this proposed wilderness area (MFWP 2005).
>
> There is a mountain known as Savage Peak…that, despite precipitous elevations and forested areas, shows snowmobile tracks nearly to the top of the 6900' peak into March of most years. Despite steep terrain and high tree lines, snowmobilers continue to make advancements into sensitive terrain, areas particularly important to mountain goats such as that on Savage Peak. Because of this, MFWP sees many of the wilderness recommendations, including increased wilderness and backcountry areas presented in Alt. B…as positive (MFWP 2012).

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[2] http://missoulian.com/lifestyles/recreation/scotchman-peaks-straddle-weird-winter-patchy-politics/article_69ebd027-600e-5597-a083-a4f671d3fd0d.html

> This closure has "helped maintain habitat security for a number of species including mountain goats, wolverine, elk, and mule deer, to name a few." If this area were open to snowmobiling, it would "contradict what FWP recognizes as important and what literature suggests is tolerated by species like goats, lynx, wolverine, elk, and mule deer" (MFWP 2015).
>
> Jerry Brown, the now-retired FWP Biologist whose experience dates back to the 70s, created maps of the areas that he recommended remain restricted to motorized access year-round... His map included the area north of Drift Peak and south, through Star Peak - including both the Savage Mountain and Dry Creek areas - and even extended into Idaho, the entire area of which we have had concerns over potential impacts to wildlife security since the 70s. (MFWP 2016).
>
> We would like to reiterate that the entire Scotchman Peaks area contains important winter range habitat. As winter is a time of restricted ranges, limited food resources, and stress for many species, winter range is known as a limiting factor for big game. Containing and/or limiting motorized recreational use on unique habitat like winter ranges can help minimize direct impacts to wildlife (e.g., mortality due to increased human pressure, which results in higher activity rates, increased energy use, and stress). In general, current wildlife literature recommends routing human activities – especially motorized – away from goat winter range when possible… This act [closure to snowmobiles] has helped maintain habitat security for a number of species in this area, including goats, and we at FWP wish to maintain this important status quo. This existing condition of use is especially important because we know that current literature recommends minimizing the impacts of human disturbance on wildlife with standards such as: 1. Minimizing activities outside of currently used sites (Canfield 1999), 2. Concentrating activities within existing and designated sites (Canfield 1999), and 3. Limiting human intrusion into critical area such as winter range (Canfield 1999, USFS and BLM 2007, and Olliff et al. 1999) (MFWP 2016).

The Kootenai National Forest acknowledges the issue of mountain goats, snowmobiles, and the partnership with Montana Fish Wildlife & Parks in the Scotchman Peaks and Savage Peak area:

> Even if over-snow motorized recreation does not occur on the exact spot where mountain goats winter, the presence of over-snow motorized recreation near to those mountain goat winter ranges may cause enough disturbance to apply the aforementioned revised Forest Plan. Additionally, if through coordination with the State, and review of the best available information, it is determined that an area was winter range for mountain goats historically but they may no longer be present, it may be desirable to keep those areas available for re-colonization by mountain goats in the future. Again, FW-DC-WL-16 states that the KNF would coordinate native ungulate habitat management with the State. During that coordination the State may help the KNF identify areas of historic mountain goat winter range that are important for future re-colonization by mountain goats. Montana Fish, Wildlife and Parks has repeatedly noted their concern over potential snowmobiling impacts to mountain goats on winter range in the Savage Peak area, including during the public comment period in 2012 on the draft Forest Plan and DEIS. (KNF 2013b).

CONCLUSION

The Scotchman Peaks, including the Savage Peak region, is a special area, containing critical winter range habitat for mountain goats and important habitat for other species. Winter range is a limiting factor for mountain goats, and winter is a time of restricted ranges, limited food resources, and higher stress. As mountain goats are highly sensitive to human disturbances such as snowmobiles, restricting motorized recreational use from mountain goat winter range helps minimize impacts during this difficult season. Land and wildlife management agencies (Montana Fish Wildlife & Parks and United States Forest Service) support the continuation of non-motorized activities and wilderness designation in the Scotchman Peaks and Savage Peak area. Preserving the year-round closure to motorized activity across the Scotchman Peaks including the Savage Peak area, regardless of wilderness designation, will continue to protect the wildlife and wildlife habitat in this unique setting.

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Appendix J

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MOUNTAIN GOAT POPULATION CHANGES IN RELATION TO ENERGY EXPLORATION ALONG MONTANA'S ROCKY MOUNTAIN FRONT

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Abstract: A mountain goat (Oreamnos americanus) study was initiated in 1981 along the east slope of Montana's Rocky Mountains (823 km^2) to quantify population parameters and monitor energy exploration activity. Twenty-four radio-marked goats provided seasonal home range information. Observations of the radio-marked and 8 neckbanded goats provided reproductive histories for adult females, and annual survey efficiency. The adult female population trend was stable in the Birch-Badger segment but declined significantly in the Teton-Dupuyer segment. Kid:adult female (K:ADF) ratios in the Birch-Badger segment dropped 81% from 1983 to 1984, and 62% in the Teton-Dupuyer segment from 1982 to 1983. Beginning in 1981, energy exploration dramatically increased. From 1981 to 1985, about 579 km of seismic lines were shot within mountain goat habitat. This activity peaked during 1983 and 1984. Radio-telemetry information did not indicate abandonment of home range, however the peak in seismic activity did coincide with declining adult female numbers, kid numbers, and productivity in the Teton-Dupuyer segment. Differences in population characteristics in the Birch-Badger and Teton-Dupuyer segment appear to be attributable to differences in levels of human disturbance within each area. Other factors were addressed which may have influenced mountain goat population characteristics, including weather, hunter harvest, livestock grazing, timber harvest, and disease. The added impact of seismic activity, over and above other human activities in the Teton-Dupuyer segment, appeared to be the primary cause of changing population characteristics.

Native mountain goats of Montana's Rocky Mountain Front (RMF) occur along the theoretically petroleum-rich Overthrust Belt. Industrial and recreational projects have been implicated in declines of native mountain goat populations throughout North America (Chadwick 1973, Hebert and Turnbull 1977, Kuck 1977, Pendergast and Bindernagle 1977, Foster and Rahs 1983, Rice and Benzon 1985). Therefore, concern about human impacts from energy exploration has focused upon mountain goats along the RMF as the pace of exploration accelerates and gas/oil field development begins. Research on mountain goats from 1981 through 1986 was conducted to describe the mountain goat population along the RMF, document changes in population parameters, and describe the upsurge of human activity within the area and the possible consequences of human-induced stress upon the population.

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STUDY AREA

The RMF study area (Fig. 1) occured in the Sawtooth Mountains of northcentral Montana. Lying along the east slope of the Continental Divide, the study area extended some 82 km south of Glacier National Park to the main Sun River and was bordered on the east by the prairie. The study area was divided into 3 segments (Fig. 1) based on relatively autonomous mountain goat population segments. The Deep-Sun segment is not considered in this analysis because it was not intensively surveyed and was therefore not comparable.

Geological forces shaped the magnificent reefs of the RMF. The awesome cliffs and ridges of the RMF are composed primarily of Madison limestone from the Cambrian era, although the bulwark of the mountains is Precambrian sedimentary rocks (Alt 1985).

Gale-force chinook winds, often blowing over 100 km per hour, melts and blows away snow on the eastern slopes and exposes forage. The coldest average winter temperatures (January) range from -8.9°C at East Glacier to -6.0°C at the Sun River's Gibson Dam. The warmest average summer temperatures (August) range from 15.9°C to 16.7°C, respectively. Yearly precipitation averages 59.7 cm at East Glacier and 47.0 cm at Gibson Dam (Nat. Oceanic and Atmos. Admin. 1980-1985). Maximum snow pack occurs in April with depths (from north to south) averaging 252.5 cm at Badger Pass (2103 m), 168.1 cm at Mount Lockhart (1951 m), and 148.3 cm at Wrong Ridge (2073 m) (U.S.D.A. SCS 1922-1985) (Fig. 1). Meteorological data indicate a subtle gradient toward warm and dry, moving from north to south along the RMF. Detailed descriptions of vegetation, habitat types and landtypes are described in Harvey (1980), Thompson (1980), Holdorf et al. (1980) and Holdorf (1981).

METHODS

Repeated, systematic helicopter surveys were conducted on that portion of the population north of the Middle Fork Teton River (823 km², Fig. 1). Surveys were flown during July, from 1981 through 1986, during morning and evening hours, by the author in a G3 47 Bell helicopter. Subsequent to each helicopter survey, a radio-relocation flight was made to determine the presence or

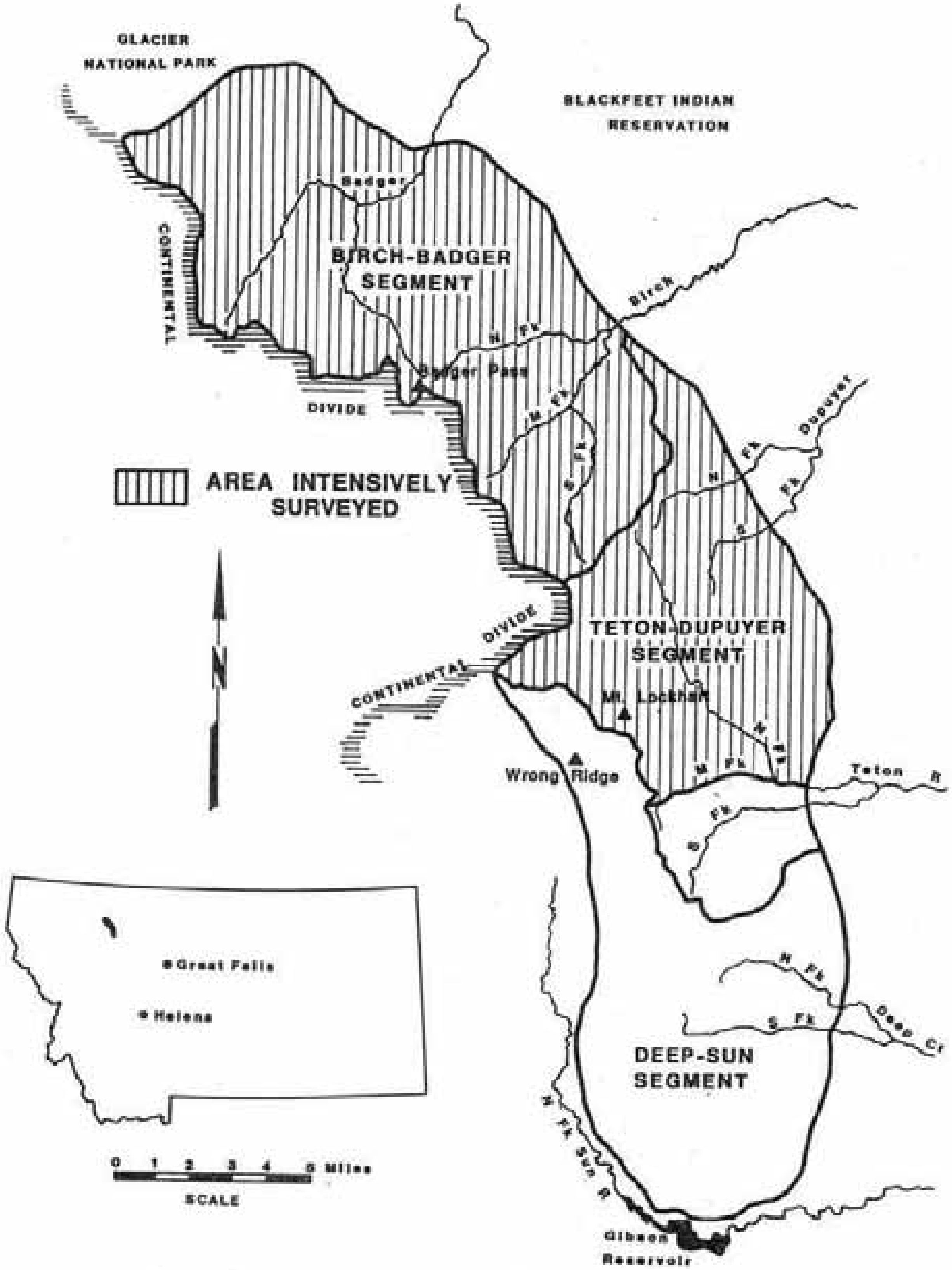

Figure 1. Rocky Mountain Front Study Area.

absence of radio-marked goats within the Teton-Dupuyer area. The percentage of radio-marked and neckbanded animals observed during annual surveys provided the basis for establishing survey efficiency.

Thirty-five mountain goats in the Teton-Dupuyer segment were fitted with radio collars (23), neckbands (8), and eartags (4), from 1979 through 1982 (Joslin 1986). Nine and 6 adult males, 8 and 2 adult females, and 6 (4 females and 2 males) and 0 subadults were marked with radio-collars and neckbands, respectively. Four male kids were eartagged. All radio-collars placed on subadults were expandable elastic collars which were not observable from the air. During aerial telemetry, observations of radio-marked animals were obtained when possible. Both air and ground observations provided data on the reproductive histories of 11 radio-marked and 2 neckbanded females. Mountain goats were classified as adults (male or female), 2-year-olds, yearlings, and kids based on morphological features, molting patterns, and group association.

Seasonal home range sizes (convex polygons) of 24 radio-marked mountain goats were calculated based on bi-monthly radio-relocation flights. Average number of fixes used in calculating home ranges for adult animals was 56 (range 25-120).

Snow depth information was collected from 3 snow survey sites which occur in the Birch-Badger, Teton-Dupuyer, and Deep-Sun segments of the study area, respectively (Fig. 1). These sites occur either within mountain goat winter range, or in the case of Badger Pass, which is at the edge of the study area, at an elevation which coincides with mountain goat winter range.

Information concerning energy exploration activities was provided by the Rocky Mountain Ranger District, Lewis and Clark National Forest. The term seismic activity, as used here, includes all ground and air activity associated with seismic line set up, shooting, and clean up.

RESULTS

Population Characteristics

Home range information was collected for 9 adult females, 9 adult males, and 6 subadults in the Teton-Dupuyer segment (Joslin 1986). Comparing adults for which at least 2 years of information was available, the largest yearlong home range was 181.5 km^2 for a male while the smallest was 16.0 km^2 for a female. Only 1 male had a yearlong home range (22.9 km^2) smaller than the average for females (34.9 km^2), while all female ranges were smaller than the average for males (89.4 km^2). The average female summer range (19.2 km^2) was slightly smaller than the average winter range (22.2 km^2), but the reverse was true for males (48.5 and 46.4 km^2). Although goats tended to adjust their movements over the course of the study, none were known to abandon their established home range.

All marked goats generally confined themselves to the Teton-Dupuyer segment. Over the course of the study, observability of marked adult females was higher (80%, SD=13) than marked adult males (30%, SD=18) (Table 1). Because observability of adult females was consistently high, population trends were based on actual number of females and kids observed in both the Teton-Dupuyer and Birch-Badger population segments.

Table 1. Observability of marked adult mountain goats, July 1981 - 1986.

YEAR	MARKED FEMALES	NO. OBSERVED	% OBSERVED	MARKED MALES	NO. OBSERVED	% OBSERVED
1981	3	2	67	3	0	0
1982	7	6	86	11	4	36
1983	7	5	71	8	1	12
1984	7	7	100	10	5	50
1985	4	3	75	8	4	50
Total	28	23	399	40	14	148
Average			79.8			29.6

Population trend of female goats in the Teton-Dupuyer segment from 1981 through 1986 is presented in Table 2. The decline in adult females in this segment (Fig. 2) was significant ($R=-0.851$, $p < .05$). The trend in the Birch-Badger segment on the other hand was not significant ($R=-0.833$, $p > .1$) (Table 3 and Fig. 3). At the beginning of the study, numbers of adult females in both population segments were similar, but by 1986, adult females in the Teton-Dupuyer segment had dropped about 50%.

Table 2. Summer helicopter surveys of mountain goats in the Teton-Dupuyer segment, 1981-1986.

YEAR	TOTAL	ADM	ADF	SA	KID	K:100ADF
1981	75	13	33	17	12	36.3
1982	60	16	25	10	9	36.0
1983	43	13	22	5	3	13.6
1984	58	15	28	9	6	21.4
1985	37	12	18	3	4	22.0
1986	32	9	15	6	2	13.3

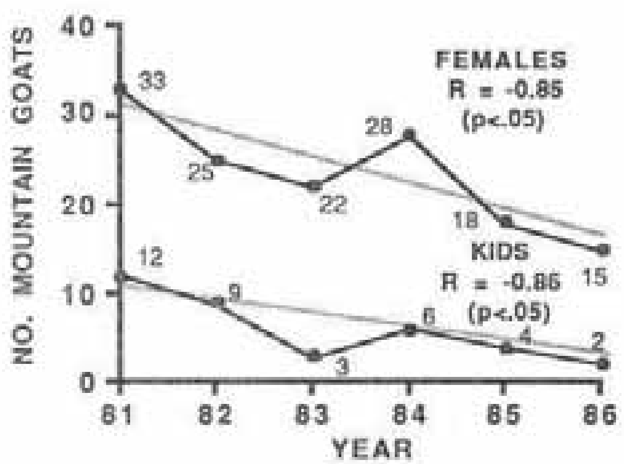

Figure 2. Adult female and kid mountain goats observed during annual surveys of the Teton-Dupuyer segment.

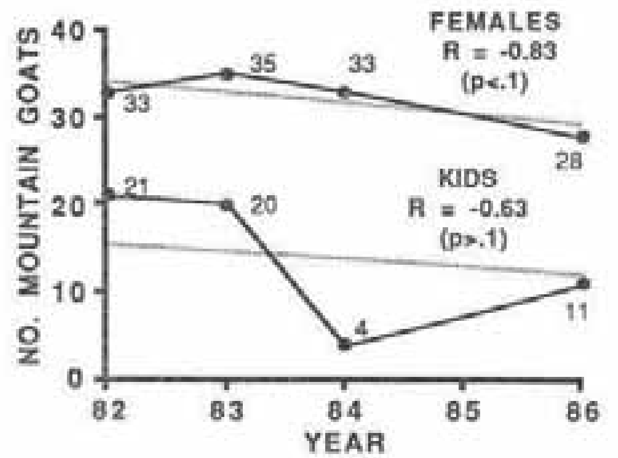

Figure 3. Adult female and kid mountain goats observed during annual surveys of the Birch-Badger segment.

Table 3. Summer helicopter surveys of mountain goats in the Birch-Badger segment, 1982-1986.

YEAR	TOTAL	ADM	ADF	SA	KID	K:100ADF
1982	77	15	33	8	21	63.6
1983	80	9	35	16	20	57.1
1984	56	10	33	9	4	12.1
1986	72	21	28	12	11	39.3

Even though population levels were similar at the onset of this study, kid production levels were not. In the Teton-Dupuyer segment, kid:adult female (K:ADF) ratios in 1982 were over 40% lower than in the Birch-Badger segment. By 1983 and 1984, kid production dropped 62% and 81% in the Teton-Dupuyer and Birch-Badger segments, respectively. By 1986, kid production in the Birch-Badger segment appeared to be recovering and had more than tripled from a low of 12 K:100ADF (Fig. 3). But kid production in the Teton-Dupuyer segment improved only slightly in 1984 and 1985, then dropped back to the low of 13K:100ADF in 1986 (Fig. 2).

Reproductive histories of 11 marked adult female goats indicates the possible cause of decline in both females and kids in the Teton-Dupuyer segment. From 1 to 6 years of reproduction information was documented for each marked adult female (Table 4). Potentially 42 young could have been born to these females over the course of the study, assuming 1 kid born per female per year. Six of 18 kids that were born died, while the fates of 4 others were undetermined. No twins were produced. Sixty percent of the kids that died did so between July and September. Production ranged from a maximum of 100% (N=3) in 1979 to 0 (N=5) in 1984. Recruitment was highest prior to initiation of this study (Thompson 1980), then it dropped to 0 (1984-86). Apparently, the consistently low kid production and poor recruitment resulted in a lack of reproductive females being recruited into the population, and therefore, the population continued to decline.

Table 4. Reproductive history of 11 marked female mountain goats.

Radio #	Age Marked (Yrs.)	1979	1980	1981	1982	1983	1984	1985	1986
1172	4	K	Y, K	Y, K-died	K-died	0	0	Trans[a]	
1082	4	K	Y, ?	?	?	0 Trans			
1052	3	K-died	K	Y, K-died	K Trans				
1222	3		K	Y, K	Y, K	Y, Trans			
1290	AD				K-died	K	Dead[b]		
1230	AD				K	Y, 0	0	0	0
42[c]	5				0	0	0	0	0
32[c]	AD				0	0	0	K	0
1240	2				---	0	0	0	0
1814	4				K-died	K/Dead[b]			
492	3					0			

[a] = transmitter failed
[b] = adult goat died
[c] = neckband

Energy Exploration

Seismic exploration activity along the RMF has increased 37 fold from the 1960-1980 period when an average of 9.5 km of line was shot per year, to 1981-1984 when an average of 351.0 km of line was shot per year (Fig. 4). Wildcat drilling in the 59 years between 1921 and 1980 amounted to an average of 1 well drilled every 2.7 years. From 1981-1984 an average of 1 well per year was drilled. Although only a portion of this seismic and drilling activity occurred within the study area, the trend is clear. Nearly all of the 579 km of seismic lines which were shot in the mountain goat study area since 1981 were helicopter supported. An estimated 21.7 man days and 6 to 8 helicopter km are associated with each km of helicopter based seismic line shot

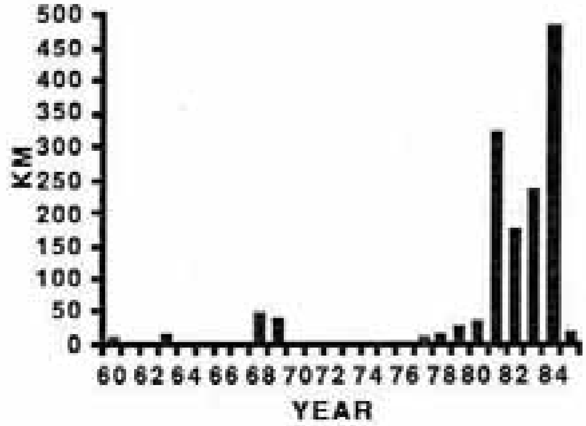

Figure 4. Rocky Mountain Front seismic exploration.

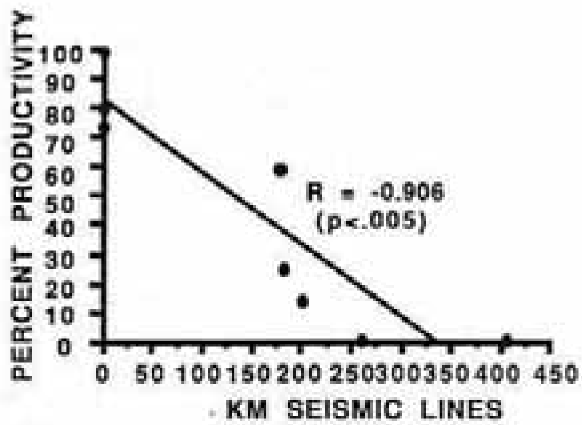

Figure 5. Productivity of radio-marked adult females in relation to cumulative seismic activity (2 previous years).

within mountainous terrain along the RMF. This equates with 12,564 man days and about 4,053 km of helicopter activity within the study area since 1981.

Declines in adult females, kids, and productivity of marked adult females in the Teton-Dupuyer segment were negatively correlated with the amount of seismic activity occurring in mountain goat habitat within the study area from 1979 to 1986. A regression of the amount of seismic activity occurring 1 year prior to the July annual population survey, and the number of females observed during that survey, accounted for 71% (R^2) of the variability in adult females ($R=-0.846$, $p < .05$) over the course of the study. The number of kids present in the population segment was inversely correlated ($R=-0.875$, $p < .05$) with the sum of seismic activity occurring 2 years previous to the year in which the population was surveyed, i.e. km of seismic line in 1979 plus 1980 were compared to the number of kids present in 1981. As might be expected, number of kids in the segment was positively correlated with number of females in the population segment ($R=0.874$, $p < .05$). As the number of adult females declined so did numbers of kids, indicating that compensatory reproduction was not occurring. This was also illustrated by reproductive information from marked adult females. Although the correlation between productivity and seismic activity that year, or the previous year was weak, the correlation of productivity and seismic activity for two years prior to the birth year was highly significant ($R=-0.906, p < .001$) (Fig. 5). Thus, the amount of seismic activity during 2 years explained 82% (R^2) of the variation in productivity during the birth year.

DISCUSSION

The objective of this study was to evaluate changes in the RMF mountain goat population in relation to energy exploration. At the beginning of this study, kid production was higher in the Birch-Badger segment than the Teton-Dupuyer segment. Reasons for this difference are not fully understood, although it may be related to the degree of human activity historically occurring in these areas. The Birch-Badger segment is relatively inaccessible, and has not been greatly influenced by human activities. In contrast, the Teton-Dupuyer segment has had a much higher level of human activity, including a seasonal ranger station, timber harvest, and developed recreation involving a downhill ski resort, a guest ranch, groomed snowmobile trails, developed campgrounds, and major trail head parking facilities. This has resulted in traditionally more motorized access and use.

Prior to the increase in helicopter based seismic activity along the RMF in 1981, it appeared that the Birch-Badger segment contained an undisturbed mountain goat population that had good reproductive performance, while the Teton-Dupuyer segment was comparatively more heavily utilized by people and contained a mountain goat population that had relatively low reproductive performance. The number of adult females in the Birch-Badger segment remained relatively stable from 1981 to 1986, but declined in the Teton-Dupuyer segment. During this period, the total number of kids in the Teton-Dupuyer segment declined, while the number of kids in the Birch-Badger segment showed a sharp decline in 1984, and then some increase in 1986 (although still substantially below 1982-1983 levels). The reproductive decline appeared coincident with the peak in seismic activity along the RMF from 1981 through 1984. The recovery of kids in 1986 in the Birch-Badger segment also appears coincident with the cessation of seismic activity in 1985. However, kid production, number of kids, and number of adult females in the Teton-Dupuyer segment continued to decline through 1986. The Teton-Dupuyer population did not respond as the Birch-Badger population had, once seismic activity ceased, possibly due to the long-term additive effects of several stressors upon the Teton-Dupuyer segment.

Other factors which may have influenced mountain goat population characteristics include weather, hunter harvest, livestock grazing, timber harvest, and disease. Several authors have documented an inverse correlation between winter snow depths and kid survival (Chadwick 1973, Rideout 1974, Smith 1976, Bailey and Johnson 1977, Thompson 1980, Johnson 1983, Swenson 1985). The average of March, April and May snow depths from 3 snow survey sites in or near mountain goat winter range along the RMF from 1980 through 1985 indicated that winter snow pack has been at or below normal during 5 of 6 years. A regression of Teton-Dupuyer kid:non-kid ratios upon snow depth showed little correlation ($r=0.5$, $df=5$, $p > 0.1$). The relatively low snow depths apparently did not have an influence on kid survival.

Cool wet summers are also known to affect survival of newborn young (Brandborg 1955, Johnson 1983). Temperature and precipitation data during June, July and August from 1980 to 1985 indicated that summers have generally been slightly warmer and dryer than average. The month of June in particular, is critical, but 1981 was the only year slightly cooler than normal (-0.4 C below average), and 1983 and 1984 were the only years slightly wetter than normal (1.0 cm and 0.1 cm above the average, respectively). A combination of abnormally cold and wet conditions did not occur during the course of the study.

Numbers of hunting permits issued for the 2 districts along the RMF have varied from 15 to 8 between 1978 and 1985, and currently stand at 8. An average of 4 to 5 mountain goats have been harvested per year. Since the beginning of this study in 1981, when population levels were first estimated (Joslin 1986), mountain goat harvest has not exceeded 4% of the population, and has averaged 3%. Females have composed 46% of the harvest, or between 2 and 3 per year. Analysis of harvest indicated that adult females in the Birch-Badger segment were neither negatively nor positively influenced by existing harvest levels ($R=0.321$, $p < .5$). A positive correlation between number of females harvested and number of females in the Teton-Dupuyer segment indicated that females were harvested in proportion to their abundance ($R=0.876$, $p < .05$). Hunting, therefore, does not appear to be responsible for the observed kid decline, although it is possibly a contributing stressor to the Teton-Dupuyer population segment.

Livestock grazing peaked along the RMF in the early 1900's. Changes in allotment size, duration of use, species use, and management systems have helped reduce livestock competition with wildlife. Generally, livestock use is now at its lowest level in 50 to 80 years, and the current level of use has been maintained since the most sweeping changes were instituted in allotment use 10 to 20 years ago. Although livestock grazing continues to be a significant land-use activity along the RMF, the extent and degree of use has not appreciably changed in the last 2 decades, and would not appear to have created a new or increased stress upon the mountain goat population.

Approximately 7,600 cubic meters (1 million board feet) per year of timber has been removed from the RMF over the past decade (USDA Lewis and Clark National Forest Plan 1986). All of the few logging roads that created access into mountain goat range occurred in the Teton-Dupuyer segment, but timber sales have generally not been located in critical mountain goat range. The decline in kid production in the Birch-Badger segment was not related to timber harvest since none occurred in this area. However, limited harvest in the Teton-Dupuyer segment may have indirectly affected that area's population.

The drop in mountain goat kid production in 1983 and 1984 tends to parallel the pattern of bighorn sheep decline due to a pneumonia die-off along the RMF in 1983 and 1984 (Hook 1986).

Onderka and Wishart (1984) indicate that in bighorns, some adults initially succumb of respiratory disease complex, while others may survive to become carriers and thus bare a poor lamb crop.

Mountain goat deaths as a result of respiratory disease complex were not documented, although 2 possible cases were discovered. In 1983, 2 mountain goats were found dead along the RMF, one along the South Fork Teton River in Green Gulch (T. Bivins 1983, pers. commun.) and the other near Many Glacier in Glacier National Park (K. Keating 1985, pers. commun.). The Teton goat, a 5-6 year old male, was diagnosed as having fibrinopurulent bronchopneumonia (Corynebacterium sp.) (J. Rhyan, DVM, Lab Rpt. No. 8-470 Veterinary Lab, Bozeman, MT). The Glacier goat also suffered from severe suppurative pneumonia (M. Harries DVM, Path. No. L83-3228-H Alberta Animal Health Division, via Glacier National Park, MT). Harries noted that the pathological changes in the Glacier goat were essentially similar to those of the bighorn sheep which had been suffering from a Pasteurella hemolytica biotype T pneumonia in areas of Alberta, British Columbia, and the RMF of Montana. Although significant bacteria could not be isolated, similar organisms to those of the sheep had very likely been present (op. cit.).

Limited evidence indicates that broncho-pneumonia might have been a contributing factor in the observed mountain goat population changes. Respiratory disease complex is often stress-induced (Onderka and Wishart 1984). If this stress-related pneumonia was latent in mountain goats along the RMF, it is possible that this disease, and the stress-inducing effects of seismic disturbance could have acted in concert to cause the observed decline in females and kid production.

The correlation between mountain goat productivity and seismic activity in previous years, suggest that the stress induced by this seismic activity is cumulative over the years. The 4 year period of very intensive activity probably created more stress than it would have, had the individual years of activity not been consecutive. Stemp (1983) indicated that pre-natal stress is of particular concern because the extreme sensitivity of the young is related to their rapid development, and the most rapid development of a mammal takes place as a fetus. Particular organs and behaviours are especially susceptible during the critical periods in which they are maturing most rapidly (Scott 1962). Changes in development can be so pronounced that the individual's emotionality and behavior, phenotype, and even viability can be profoundly and permanently altered (Stemp 1983).

A number of researchers have reported upon behavioral and physiological response of wildlife to helicopter harassment. Helicopters, sonic booms, gunshots, people on foot, stopped occupied vehicles, and domestic dogs elicit strong behavioral and/or heart rate responses from a variety of wildlife (Horejsi 1976, Ward and Cupal 1979, Gunn et al. 1983, Stemp 1983, Irwin and Gillin 1985). MacArthur et al. (1982), Stemp (1983), Ward and Cupal (1979) and others have indicated that behavioral response does not necessarily reflect physiological response to harassment. Despite the animals outward behavior, Stemp (1983) reported that heart rate of bighorn sheep was significantly elevated and remained elevated as long as helicopter activity occurred in their vicinity. Up to 45 minutes was required after the disturbance was gone for heart rate to return to normal. He indicated that "behavioral response can be extremely misleading: helicopters can sensitize bighorns and can produce marked and prolonged physiological responses in the absence of pronounced--or even any--behavioral reaction".

To avoid stress-inducing disturbance, an animal may withdraw, but withdrawal is also costly because exertion demands increased energy expenditure. Also, injury is a risk during escape attempts, and the opportunity to feed at that location is removed. If animals withdraw from key habitat areas, carrying capacity can be reduced (Geist 1975). During this study, mountain goats generally moved away from human activity, and used topographic relief to screen themselves from line-of-sight disturbance. Although they did redistribute themselves within their home ranges to avoid disturbance, fidelity to familiar terrain was strong and none abandoned their traditional home ranges.

Abundant literature exists detailing the maladaptations that may result from repeated or prolonged stress stimulation, including decreased resistance to infection and disease (Hudson 1973, Stein et al. 1976), and impaired or complete failure of reproductive function (Geber 1962, 1970; Petropoulos et al. 1972, Sontag 1970 - in Stemp 1983). Stemp (1983) indicated that "any stressor sensitizes the individual to other stressors. Moreover, a prolonged stress response decreases an individuals ability to cope psychologically (Shanan et al. 1976) which is likely to make them more susceptible to subsequent stressors". This is particularly evident in the Teton-Dupuyer segment where seismic activity appeared to be additive to other long-term, human-induced stressors. A constant level of stress may explain why kid production in this area did not improve, even though seismic activity ceased in 1985. Although the Birch-Badger segment was relatively immune from long-term stressors, it appeared that seismic activity did cause a decline in kid production in 1984. Once seismic work ended, productivity improved. In comparison, mountain goat populations in adjacent Glacier National Park (where no seismic activity occurred) appeared to be stable (K. Keating 1985, pers. commun.)

This study indicates that efforts should be made to reduce human activities in the Teton-Dupuyer segment in order to allow mountain goat population recovery. If the Birch-Badger segment remains inaccessible and relatively free of human activity, it appears that it might be able to recover from temporary, short-term disturbance. Monitoring will be necessary to determine whether mountain goat production returns to pre-1981 levels along the RMF.

Detailed multi-agency guidelines were developed to ameliorate the effects of energy exploration activities upon wildlife along the RMF (Interagency Rocky Mountain Front Wildlife Guidelines 1984). One objective of the guidelines was "to avoid or minimize human related activities which may adversely impact selected species of wildlife". The mountain goat population declines suggest that the guidelines were insufficient to maintain pre-development kid production and female population levels in the face of intense human activity. Application of the guidelines was particularly deficient in controlling distribution of seismic activity. If the mountain goat population is to increase and be maintained at or above the 1980 level, managing the timing and intensity of human activity within mountain goat habitat will be critical.

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QUESTIONS AND ANSWERS

Bruce Smith, Wyoming: Gayle, I was interested in your juvenile mortality rates and wonder if you could tell us how you determine those for kids and yearlings. Yours were fairly high.

Gayle Joslin: They were fairly high. They were determined from five consecutive years of actual survey information.

Smith: So it was actual counts of kids during consecutive years and comparing them from year to year?

Joslin: yes and over the course of the year by watching kids of radio-marked adult females from week to week.

Smith: Were you surveying during the winter?

Joslin: No, I surveyed in July.

Smith: You had an inverse correlation between productivity of the goats and the seismic activity, is that right?

Joslin: Right

Smith: Was there any correlation between the mortality rates of kids and yearlings and the activity in the areas you looked at. For example, before the seismic activity increased, did you have lower mortality of kids and yearlings than you did after you had higher levels of activity in the area.

Joslin: Yes. Using both yearly survey data and monthly observation data from marked females which had kids at side, I could see that as consecutive years of seismic activity elapsed, either productivity or survival or both decreased.

Smith: So then there was a correlation?

Joslin: Yes.

Nicki Goodson, Colorado: Do you have any information on distribution activity, or movement patterns of goats relative to disturbance?

Joslin: Regarding distribution, radio-marked goats did not leave established home ranges when disturbance occurred, but they would use topography to screen themselves from human activity by moving out of a drainage where the activity was occurring. I observed four marked goats do this. When disturbed, goats would often watch the activity for quite some time. They would be alert, not feed, and they seemed to chew their cud less, but this was not a behavior study per se, so I don't have quantitative ethological data. Doug Chadwick's thesis gives excellent descriptions of mountain goat response to disturbance. I did not observe anything during this work which was counter to what Chadwick reported.

Jim Bailey, Colorado: How far did they move?

Joslin: They always stayed within their home range, but males would move the greatest distances. It was common for three radio-marked males to travel about nine air miles over the Continental Divide into the Flathead drainage to summer, but that was more of a seasonal movement. Males had the opportunity to avoid the activity more because their home ranges included more wilderness area and were, therefore, more isolated.

Bailey: You said that was a seasonal thing. I was asking in addition to Nicki's question, how far do these animals move in response to seismic.

Joslin: Let me explain a complicating aspect of this work. Several of us who were collecting data along the Rocky Mountain Front at that time will attest that we often did not know where or when a seismic line would be shot. Communication from the Rocky Mountain Ranger District to field biologists regarding seismic activity was limited. So, we were often not aware that a particular line had been shot until it was too late to monitor an animal's response. Relative to those goats which I did observe subsequent to disturbance when I had the opportunity, they would often move no more than a half mile. They would go over a ridge to get away from direct line-of-sight disturbance. It is this kind of limited reaction by goats which elicits a judgment that goats are not affected by disturbance. However, in the case of mountain goats at least, I believe the population data is more revealing than behavioral reaction.

Wayne Heimer, Alaska: Is this exploration over at this point or ongoing?

Joslin: Good question, we would like to know, too. When the price of oil goes back up, I suppose we'll see more exploration. They do have three exploratory wells that are pending right now.

Heimer: The other question is, presuming with the price of oil being low like it is, there won't be anything going on right away, are you going to continue watching to see if the stress goes away if things get better?

Joslin: There's no more funding. No.

Heimer: There ought to be.

Peter Davidson, BC: There was obviously a lot of activity by helicopters in that area. I assume that most of this is done from helicopters judging by the terrain. You mentioned Raymond Stemp's work, University of Calgary, was there any attempt to collect that type of information from these goats?

Joslin: No heart rate response or fecal cortisol information was gathered so there was no direct measure of stress effects.

Davidson: I'm suggesting that the helicopters might have had as much effect as the blast itself or more.

Joslin: I think it had more. That's my personal opinion, but I think it had more. There was a great deal of helicopter activity.

Kirby Smith, Alberta: You said you lost 60% of your kids by September, did I miss the explanation or do you have any explanation? Were you watching kids on cliffs during helicopter passes or anything like that? Do you think it was because of falls or what?

Joslin: I don't know what the exact mechanism of the kid decline was. Of the kids born to marked female goats, six were known to have died. Four of these six disappeared between July and October.

Ted Benzon, South Dakota: After the seismic activity was gone, how long did it take before the goats went back into that area?

Joslin: Some times by evening they would come back. Especially when they were on salt licks. Goats will take big risks when their drive for salt is high. They almost seem suicidal.

Appendix K

Bienn. Symp. North. Wild Sheep
and Goat Counc. 14:131-136

NWSGC POSITION STATEMENT ON HELICOPTER-SUPPORTED RECREATION AND MOUNTAIN GOATS

Kevin Hurley, NWSGC Executive Director

July 9, 2004

Introduction:

Less is known about mountain goats than other North American ungulates, due primarily to their relative scarcity and the inaccessible terrain they inhabit (Smith 1982, Festa-Bianchet et al. 1994, Wilson and Shackleton 2001). Disturbance of ungulates by helicopters can result in a variety of negative effects, including habitat abandonment significant enough to affect population status and herd viability, dramatic changes in seasonal habitat use, increased vulnerability to predation, alarm responses, decreased bouts of foraging and resting, increased animal movement and energy expenditure, and reduced productivity (Pendergast and Bindernagel 1976, MacArthur et al. 1979, Foster and Rahs 1981, Foster and Rahs 1983, Hook 1986, Joslin 1986, Pedevillano and Wright 1987, Dailey and Hobbs 1989, Côté 1996, Frid 1999, Denton 2000, Duchense et al. 2000, Gordon and Reynolds 2000, Phillips and Alldredge 2000, Dyer et al. 2001, Frid 2003, Gordon 2003, Keim and Jerde 2004).

Population and/or fitness-enhancing behaviors such as feeding, parental care, and mating may be detrimentally impacted in response to repeated helicopter disturbance, even when overt reactions to disturbance are not visible (Bunnell and Harestad 1989, Gill and Sutherland 2000, Frid and Dill 2002). Significant effects on reproduction, survival, and population persistence may occur. Increased vigilance resulting from disturbance may reduce the physiological fitness of disturbed animals by increasing stress, increasing locomotion costs (particularly during winters with severe snow conditions), and by reducing time spent in necessary behavior such as foraging or ruminating (Frid 2002). Physiological responses (e.g., elevated heart rates) to disturbance may not be directly reflected in overt behaviors, (Macarthur et al. 1982, Stemp et al. 1983, Harlow et al. 1986, Chabot 1991), but are nonetheless costly to individual animals, and ultimately, to populations.

Although the short-term behavioral responses of mountain goats to helicopter activity have been documented, longer-term habitat use and demographic consequences of disturbance remain poorly understood. Our recommendations are aimed at minimizing short-term behavioral disruptions that we believe are correlated with longer-term impacts. Research to date has not clearly identified thresholds of disturbance that trigger unacceptable responses; as a result, approach distances and other specific mitigation measures are precautionary recommendations.

Management recommendations:

Exclusion zones/avoidance:

Habitat segregation is typical of many ungulate species (Main et al. 1996), including mountain goats. During spring/summer/fall periods, adult male goats occupy habitats other than those occupied by nanny-juvenile ("nursery") groups (Geist 1964, Foster 1982, Risenhoover and Bailey 1982), with nursery groups typically occupying habitats more favorable for survival and reproduction (Fournier and Festa-Bianchet 1995). Adult female mountain goats have heightened sensitivity to disturbances during kidding and post-kidding periods (Penner 1988). Mountain goats are known to have a lower recruitment rate, compared to other ungulates (Bailey 1991, Festa-Bianchet et al. 1993). The health of mountain goat nursery groups provides obvious contributions to the reproductive success and survivorship of goat populations. Due to the sensitivity of adult female mountain goats to disturbance, and the importance of this age/sex class to the persistence of local goat populations, restrictions on late spring and early summer helicopter activities should focus on areas occupied or likely to be occupied by nursery groups. The very activities that serve to document use are, in themselves, disruptive to mountain goats. However, documentation of crucial winter habitat use by mountain goats is essential to identify and conserve those important winter ranges, particularly in coastal mountain ranges where deep snows are typical.

Recommendation:
Helicopter avoidance should focus on those areas identified as crucial winter range, and those areas occupied or highly suspected as used by nursery groups. Particular attention should be given to helicopter activities during identified pre-kidding, kidding, and post-kidding periods; such restrictions require identification and mapping of mountain goat habitats and identifying exclusion zones prior to the issuance of annual or multi-year heli-recreation special use permits.

Distance from occupied habitats:
Behavioral responses to helicopter activity have been documented at distances of up to 2 km for mountain goats and other ungulate species (Côté 1996, Frid 2003, Gordon 2003). Recent studies have shown that short-term behavioral responses of mountain goats increase as helicopters approach within approximately 1.5 km of mountain goats. It must be noted, however, that minimum distance needed is modified strongly by topography and the amount of cliff cover/escape terrain available; increased buffer distances may be needed in more rolling terrain with less cliff cover, or in very narrow canyons/valleys.

Recommendation:
Helicopter activity should not occur within 1.5 km of occupied/suspected nursery group or crucial winter range habitats during critical periods.

Timing of activities:
Winter is of particular concern for management of disturbance stimuli. Winter is a period of severe nutritional deprivation for mountain goats (Chadwick 1983, Fox et al.1989, Shackleton 1999). Periods of deep snow can reduce food availability and dramatically increase locomotion costs (Dailey and Hobbs 1989). In winter, mountain goats are known to be relatively immobile (i.e., movements not exceeding 50m/hour) (Keim 2003), to occupy small ($<4km^2$) and specific habitat areas (Keim 2003, Schoen and Kirkoff 1982, Smith 1982), and to have high rates (>0.66) of winter home range fidelity (Keim 2003. Schoen and Kirkoff 1982). Selection of small, isolated winter habitats by goats may become compromised if management of helicopter-recreation activity neglects to consider winter mountain goat habitats and the needs of wintering goats. It is imperative that management of activities such as helicopter-skiing address and acknowledge the potential effects on mountain goat populations, through development of enforceable mitigation strategies.

Recommendation:
Helicopter activity should not occur on or near occupied winter ranges between November 15-April 30 each year. Helicopter activity should not occur on or near occupied or suspected nursery group habitats between May 1-June 15 each year. Mountain goat winter and kidding distribution and habitat selection should be known and mapped prior to issuance of annual or multi-year heli-recreation special use permits.

Helicopter approach vectors:
The rate and horizontal distance of helicopter approach vectors affect the degree of overt disturbance to ungulates. The degree of overt disturbance also varies, according to the availability of escape terrain and topography (Frid 2003, Wilson and Shackleton 2000). Additional research should be directed at identifying and documenting best management practices for mitigating approach vectors.

Recommendation:

Vertical and horizontal approach vectors should be considered when developing mitigation strategies. Strategies should also consider local conditions including refuge availability, topography, and amount and distribution of cliff cover suitable as escape terrain.

Habituation/Sensitization:
Animals may not be able to habituate to disturbance stress when disturbance is irregular and unpredictable (Bergerud 1978, Risenhoover and Bailey 1982, Penner 1988). Frid (2003) found that the proportion of Dall's sheep fleeing did not decrease with the number of cumulative weeks of disturbance. Habituation to disturbance stimuli often is partial or negligible, and habituation to strong disturbance stimuli may only partially occur (Bleich et al. 1994, Steidl and Anthony 2000, Frid 2003). Flight-initiation distance or vigilance might actually increase with repeated exposure to non-lethal stimulus if the stimulus is sufficiently adverse, resulting in sensitization to disturbance stimuli, the opposite of a habituation response (Frid and Dill 2002).

Recommendation:
It is inappropriate to assume that habituation of mountain goats to helicopter disturbance will occur over time. Reluctance to flee should not be perceived as habituation; numerous physiological responses occur, even in the absence of overt behavioral responses. All helicopter flights over or near crucial mountain goat habitat should be considered harmful to mountain goats populations, based on current knowledge. Additional research on the long-term behavioral effects of helicopters on mountain goats should be undertaken. Establishment of a cross-jurisdictional Research Steering Committee comprised of state and provincial government and non-government/academic experts is recommended. To enable such behavioral research to occur, spatially explicit control areas should be designated in which no helicopter-supported recreation term permits are issued.

Monitoring/Enforcement
Additional monitoring of the medium and long-term effects of helicopter activity on mountain goats is needed (Wilson and Shackleton 2000). Comprehensive, long-term land use and resource management plans, as well as project-specific activity plans, need to incorporate strategies and mitigation to protect and conserve critical mountain goat habitats, while still allowing commercial activities to occur, where appropriate. These plans need to thoroughly address helicopter-supported recreation effects on wildlife populations, both short and long term. These plans should identify research needed, cite pertinent existing research from other areas, and base helicopter-activity management on the best available scientific information. Enforcement of existing terms and conditions in special use permits should occur. If lacking, those terms and conditions, along with appropriate sanctions, should be developed for inclusion in activity/operating plans.

Recommendation:
Long-term monitoring is essential. If baseline data on mountain goat numbers, distribution, and seasonal habitat selection are lacking, steps should be taken to obtain those data. Monitoring should include both compliance with, and evaluation of the effectiveness of, mitigation strategies and exclusion zones. Long-term monitoring of mountain goat population performance is needed. Control areas to facilitate future behavioral research should be maintained, in which commercial helicopter activity is not permitted. Term permits should include enforceable provisions to address cases of non-compliance. Provisions should be included to modify permitted areas or conditions, based on new information, in an adaptive management approach. Permit fees should be adequate enough and used to conduct the monitoring and baseline data collection to manage these activities. Permitting of helicopter-supported recreation, especially in new areas, should not occur until managers have the ability, funding, and mechanism to collect adequate population demographic and habitat use data, to properly manage, mitigate, and monitor this activity.

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Appendix L

Seeing the Forest and the Trees

Assessing Snowmobile Tree Damage in National Forests

A Report by Winter Wildlands Alliance
November 2009

Typically, when land management plans address the environmental impacts of snowmobiles, the focus is on air quality, noise and wildlife impacts. Little has been documented regarding the impacts of snowmobiles on vegetation.

Recently, Winter Wildlands Alliance, a national nonprofit organization that promotes human-powered winter recreation, learned that the US Forest Service, as part of forest re-vegetation surveys, has gathered data documenting tree damage caused by snowmobiles in the Gallatin National Forest near West Yellowstone, Montana. The tree damage data show that in addition to well-documented impacts on air quality and endangered lynx, caribou and other animals, snowmobiles may be more directly and immediately impacting the health of forests. Simply put, USFS data demonstrate snowmobiles are chopping the tops off of trees, possibly in significant numbers.

As part of ongoing efforts to evaluate regeneration and thinning needs, the Gallatin National Forest (GNF) conducted regeneration transect surveys of previously logged timber stands. These surveys are required by NFMA (the National Forest Management Act), and look for a variety of damage types and causes, including insect-, disease- and human-caused damage. Through a Freedom of Information Act (FOIA) request, Winter Wildlands Alliance acquired and analyzed the Gallatin National Forest regeneration survey data collected through 1996, when funding cuts curtailed regular survey efforts.

Forest Service surveyors were asked to identify and quantify tree damage observed. Snowmobile damage wasn't difficult to identify—surveys often include notes such as "Broken tops from snow machines."

Gallatin National Forest surveys show that between 1983 and 1995, snowmobiles damaged between 12 and 720 trees per acre in the approximately 72,393 acres of harvested areas studied on the 1.8 million-acre Gallatin National Forest. Tree damage caused by snowmobiles was specifically noted on 366 acres, or 0.5% of areas surveyed.

The rate of tree damage throughout unsurveyed areas of forest may be even higher. The Gallatin's surveyed only areas that had been logged, which is a small portion of the overall acres used by winter recreationists. Surveyed sections were not necessarily heavily used by snowmobiles, though three mentioned the presence of snowmobile trails in the stand. Given that GNF snowmobile use has increased since surveys stopped in 1996, it's almost certain that additional surveys focusing on tracts used by snowmobiles would demonstrate even greater impacts. The three stands surveyed with the highest rates of tree damage had snowmobile trails within the tracts (see chart below).

Tree damage not only hurts the environment, it wastes taxpayer money. The areas surveyed by the GNF were re-planted by the Forest Service after logging. Allowing damage to continue unchecked disregards the investment we taxpayers have made into our natural resources. USFS policy should protect its investment in renewable forest products, not allow it to be destroyed by careless recreationists.

While this Forest Service data covers only one national forest, it clearly shows that the potential for tree damage from snowmobiles is significant across all Snowbelt forests and points to the need for better management of over-snow vehicles. Given the potential for snowmobiles to cause damage over many acres and miles of forest per day, prudent management policy would prohibit unmanaged and off-trail over-snow travel in forested areas to reduce or eliminate future tree damage, and protect important natural resources and taxpayer investment.

Summary of tree Survey Data Provided by USFS

Timber Stand Number	Area name	Year logged	Year inventoried	Acres	Avg # damaged trees per acre	Total number of trees damaged
07-01-04-005	Little Teepee Creek Drainage	1969	1995	122	140	17,080
07-03-02-062*	Horse Butte Road*	1992	1995	15	514*	7710*
7-04-05-063	Madison Arm	1991	1995	12	5	60
7-07-02-037	Unknown	1960s	1983	68	23	1564
7-07-02-038*	Unknown*	1960s	1983	100	652*	65,200*
7-08-03-038*	Cream Creek*	1986	1995	60	725*	43,500*
	**surveys note the presence of a snowmobile trail in this stand*				**Total damaged trees**	**135,114**

Appendix M

A general model to quantify ecological integrity for landscape assessments and US application

David M. Theobald

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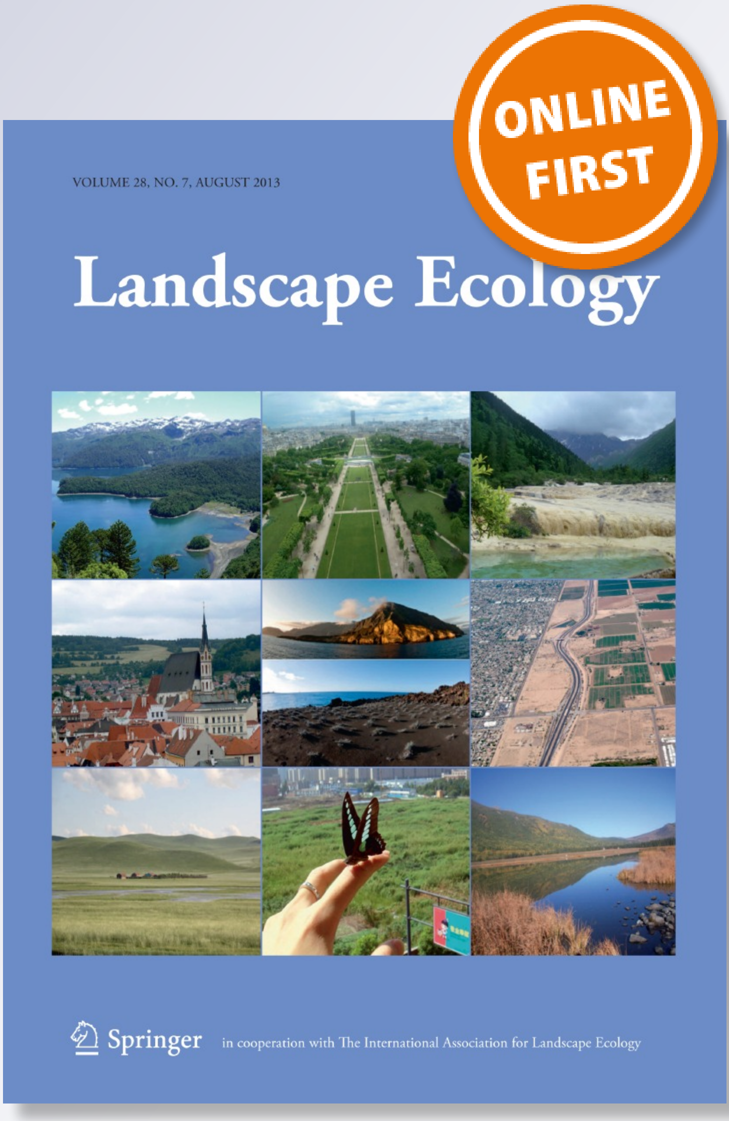

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RESEARCH ARTICLE

A general model to quantify ecological integrity for landscape assessments and US application

David M. Theobald

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Abstract Increasingly, natural resources agencies and organizations are using measures of ecological integrity to monitor and evaluate the status and condition of their landscapes, and numerous methods have been developed to map the pattern of human activities. In this paper I apply formal methods from decision theory to develop a transparent ecological indicator of landscape integrity. I developed a parsimonious set of stressors using an existing framework to minimize redundancy and overlap, mapping each variable as an individual data layer with values from 0 to 1.0, and then combined them using an "increasive" function called fuzzy sum. A novel detailed land use dataset is used to generate empirical measures of the degree of human modification to map important stressors such as land use, land cover, and presence, use, and distance from roads. I applied this general framework to the US and found that the overall average degree of human modification was 0.375. Regional variation was fairly predictable, but aggregation of these raw values into terrestrial or watershed units resulted in large differences at local to regional scales. I discuss three uses of these data by land managers to manage protected areas within a dynamic landscape context. This approach generates an internally-valid model that has a direct, empirical, and physical basis to estimate the degree of human modification.

Keywords Landscape assessments · Ecological integrity · Land use · Degree of human modification · Fuzzy sum

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Introduction

Landscape ecologists and conservation scientists have often characterized landscape and ecological systems in terms of composition, structure, and function (Noss 1990). Building on this framework, Parrish et al. (2003) defined ecological integrity of a landscape as the ability of an ecological system to support and maintain a community of organisms that has species composition, diversity, and functional organization comparable to those of natural habitats within a region. High integrity refers to a system with natural evolutionary and ecological processes, and minimal or no influence from human activities (Angermeier and Karr 1994; Parrish et al. 2003). Species-specific approaches typically develop ecological indicators that attempt to measure attributes of a species or community, such as population size or species diversity. A complementary, and more general, approach is to develop indicators of the absence of human modification of habitat and alteration of ecological

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processes. An *ecological indicator* is a measurable attribute that provides insights into the state of the environment and provides information beyond its own measurement (Noon 2003). Indicators are usually surrogates for properties or system responses that are too difficult or costly to measure directly (Leibowitz et al. 1999).

Increasingly, natural resources agencies and organizations are monitoring and evaluating the status and condition of their lands and waters by measuring the ecological integrity of landscapes (e.g., Canada National Parks Act, Lindenmayer et al. 2000, IUCN 2006; Fancy et al. 2008; Borja et al. 2008; the 2012 US Forest Service Forest Planning Rule). For example, some measure of ecological integrity is typically used when assessing the current status and likely future condition of coarse-filter conservation elements that are key to the Bureau of Land Management's (BLM) Rapid Ecological Assessments (REAs) "landscape approach." Additional examples include the National Park Service's Natural Resource Condition Assessments, the Western Governor's Association (WGA) initiative on Wildlife Corridors and Crucial Habitat (www.westgov.org/initiatives/wildlife) and the US Fish & Wildlife Service's Landscape Conservation Collaboratives (LCCs; www.fws.gov/science/shc/lcc.html).

Many composite scoring systems have been used as an indicator of ecological integrity by mapping the influence of human activities on natural landscapes, including wildness (Aplet et al. 2000) and the human footprint (Hannah et al. 1995; Sanderson et al. 2002; and Leu et al. 2008; Woolmer et al. 2008). These provide general maps of human influence and have been useful, but two improvements have been offered recently (Gardner and Urban 2007; Riitters et al. 2009; Theobald 2010). First, landscape ecologists have established that proportion of cover is a fundamental metric (Gardner et al. 1987; Gardner and Urban 2007) because no other landscape metric can be interpreted independently of it (Neel et al. 2004; Wickham et al. 2008), and it provides the basis for unambiguous interpretation needed to assess landscape change (Riitters et al. 2009). Second, ad hoc scoring systems such as the human footprint are limited because the final score typically has no direct physical basis, conversion of quantitative values to ordinal categories can violate mathematical axioms, and colinearity of individual factors leads to difficulty when interpreting results (Schultz 2001). Formal methods are available from decision theory to provide transparent accountable indicators, such as multiple criteria analysis (Hajkowicz and Collins 2007).

My goal in this paper is to describe the development and applications of a quantitative, empirically-based measure of ecological integrity that is suitable for landscape-level assessments. To achieve this goal, I extend previous work (Theobald 2010; Theobald et al. 2012) and provide a formal analytical method that allows compensatory or additive effects when considering multiple stressors to: (a) describe common human modification stressors to landscapes in the US and their data sources; (b) estimate the degree of human modification that can be attributed to each stressor; (c) combine the stressors into an overall estimate of human modification; (d) incorporate spatial and landscape context into the measure; (e) validate the estimates using a national dataset of watershed condition; (f) examine the consequence of three common methods to aggregate landscape data into management-relevant decision-making units; and (g) describe general results and initial applications of this dataset. I develop a comprehensive list of common stressors and datasets used to represent them in the "Methods" section, provide basic summaries and comparison to validation data in the "Results" section, and describe some uses and ways ecological integrity maps are commonly applied by land management agencies in the "Discussion" section.

Methods

To calculate the degree of human modification I conducted three major analysis steps. First, I distinguished the magnitude (or intensity) of impact from the spatial extent (or footprint) of a given activity at a given location. Values for both the intensity and footprint range from 0.0 (low) to 1.0 (high). Second, I used an existing framework that catalogues and organizes multiple stressors into a comprehensive but parsimonious list of stressors and the spatial databases used to represent them. I generated a data layer for each stressor for which both spatial data and estimates of intensity and footprint were readily available or made. Finally, I combined the multiple stressor layers into a single, overall metric of the degree of human modification that ranges in values of 0.0 (low modification) to 1.0 (high modification).

Similar to existing approaches to map the effects of human activities on ecological integrity, for many of the stressors I relied on impacts estimated by experts documented in the literature and/or considered to be standard. However, a critical advance in this paper is that I developed empirical estimates of the degree of human modification for the key stressors on land cover, roads, and road use (based on findings of Woolmer et al. 2008) using a detailed land use dataset generated from interpretation of aerial photography. After detailing the methods used in each of the three steps to calculate the degree of human modification, I describe how I evaluated the model and some applications of the resulting data layer. The spatial datasets for each stressor were processed at 30 m resolution unless otherwise noted, and the final human modification dataset and applications of it were produced at 90 m resolution.

Estimating human modification

When measuring the degree of human modification *h*, I distinguished two factors of an activity at a given place: magnitude and footprint. The intensity *I* (or magnitude) is the degree to which an activity at a location modifies an ecological system. This helps to differentiate effects of different types of land uses—for example, using a patch of land as pasture is likely to have a lower overall effect on the ecological integrity than conversion to a parking lot. The second factor in measuring the degree of human modification is the footprint *F*, or the areal extent of a given human activity. In practice, the footprint is measured as the proportion of a raster cell that is occupied by a given land use. Thus, the overall effect at a location is $h = IF$, where a value of 0.0 has no human modification and a value of 1.0 has high modification. Although somewhat simplified, this equation is critical because *h* has a direct physical interpretation, and its value remains a ratio data type so that differences within the range are meaningful (i.e. a value of 0.8 is twice the effect of 0.4).

Estimates of *I* and *F* were made from two different sources: expert opinion or empirical datasets. Table 1 details the data sources used to represent each stressor, as well as the source of the estimates of *I* and *F*. For about half of the stressors reasonable parameters were estimated using common expert-based values, but to the extent possible, *I* and *F* were quantified using empirical estimates of modification.

For the empirically-based stressors, I estimated *I* as a value from 0.0 to 1.0 based on the relative amount of energy required to maintain a particular land use type (Table 2; Brown and Vivas 2005). The footprint *F* was calculated as the magnitude-weighted proportion of cells of land cover type *c* that overlap with polygons from a detailed land use dataset, which was generated interpreting land uses from recent high-resolution (<1 m) aerial photography sampled at ~6,000 random locations across mainland US. For each sample location or "chip" (roughly 600 m × 600 m), a trained photo interpreter mapped polygons of each land use type following an established protocol (Leinwand et al. 2010). To quantify *F* for the roughly 577 ecological system classes in the USGS Gap land cover dataset, I intersected the centers of the cells that overlap polygons found within each chip, resulting in ~400 data points in each chip. I then combined each of the natural ecological system classes into their level 3 "formation" level (Grossman et al. 1998). For human-dominated formations (Developed and Urban and Agricultural Vegetation), I maintained the detailed ecological land type. To account for bioregional variability in these broad formations and human-dominated land cover classes, the 41 formation groups were intersected with eight eco-division-groups generated based on ecodivisions that characterize both climate and biogeographic history at a sub-continental scale (Grossman et al. 1998). I then calculated the mean and standard deviation of *h* for each of the resulting 86 formation/ecodivision-group classes (Table 3). For formation/ecodivision group classes for which there were less than 100 data points coming from a minimum of 10 chips, I manually re-grouped these types into most similar class, first grouping across similar ecodivision groups, then formation. The final dataset had 241 unique combinations of land cover and ecodivisional classes. Note that not all formations were found in all ecodivision classes.

The detailed land use dataset was also used to derive a empirical estimates of human modification as a function of distance from interstates and highways, in 150 m increments. *h* was set to 0 at a distance of ≥20 km because there were fewer than 30 chips that contributed data to the calculation. Figure 1 shows a strong relationship ($r^2 = 0.98$) between the impact to the distance from major roads.

Table 1 The list of stressors incorporated in the development of the degree of human modification dataset, based on the Conservation Measures Partnership framework (Salafsky et al. 2008)

Human modification activities framework		Stressor in human modification		Parameters*		
Level 1	Level 2	Name	Data source and scale	Source	Intensity value (I)	Footprint value (F)
Residential and comm. development	Commercial and industrial areas	Urban	Fry et al.[a] (2011); Table 3, 90 m	E	Table 3	1.0
	Housing and urban areas	Residential density	Bierwagen et al. (2010); Theobald (2005) and Leinwand et al. (2010); 90 m	E	Leinwand et al. (2010)	1.0
	Tourism and recreation areas[b]	Urban	Fry et al.[a] (2011); 90 m	E	Table 3	1.0
Agriculture	Annual and perennial non-timber crops	Croplands	Fry et al. (2011); 90 m	E	Table 3	1.0
	Wood and pulp plantations	Plantations	US Geological Survey Gap Analysis Program (2011); 90 m	E	Table 3	1.0
	Livestock farming and ranching[c]					
	Marine and freshwater aqua-culture[d]					
Energy production and mining	Oil and gas drilling	Oil and gas wells	Copeland et al. (2009)	X	0.5	KD = 1.0
	Mining and quarrying	Mining	USGS Mineral Resources Data System (http://tin.er.usgs.gov/mrds/); USGS Topographic Change (http://topochange.cr.usgs.gov/)	X	0.25	KD = 0.5
	Renewable energy	Wind turbines	Federal Aviation Administration (https://oeaaa.faa.gov/oeaaa/external/portal.jsp)	X	0.17	KD = 0.5
Transportation and service corridors	Roads and railroads	Road type	TIGER (http://www.census.gov/geo/maps-data/data/tiger.html); Highways = 30 m, Secondary = 15 m, Local = 10 m 4WD/dirt = 3 m; Railways = 30 m; 30 m cell	E	1.0	Hiwy = 1.0 Sec. = 0.5 Local = 0.3 4WD = 0.1 Rlwy = 1.0
		Highway traffic	FHA (2010)—AADT (cars per day); Estimated at 50 % max for AADT = 100,000 (Theobald 2010);	X	0.5	KD = 1.0
	Utility and service lines	Utility powerlines	Electric power from Platts (http://www.platts.com/products/gis-data);	X	0.17	KD = 0.5
		Towers	FCC (2012)	X	0.25	KD = 0.25
	Shipping lanes[d]					
	Flight paths[c]					
Biological resource use	Hunting and collecting terrestrial animals[e]					
	Gathering terrestrial plants[e]					
	Logging and wood harvesting[f]					
	Fishing and harvesting aquatic resources[d]					

Table 1 continued

Human modification activities framework		Stressor in human modification		Parameters*		
Level 1	Level 2	Name	Data source and scale	Source	Intensity value (I)	Footprint value (F)
Human intrusions and disturbance	Recreational activities	Distance to major roads	FHA (2010); Theobald (2010); 90 m	E	1.0	Figure 1
	War, civil unrest and military exercises[c]					
	Work and other activities[e]					
Natural system modifications	Fire and fire suppression[c]					
	Dams and water use[d]					
Invasive species and disease	Invasive non-native species	Cover dominated by introduced species	USGS Gap Analysis Program (2011); (five classes include introduced vegetation types)	E	Table 3	1.0
	Problematic native species[c]					
	Introduced genetic material[c]					
Pollution	Household sewage and urban waste water[g]					
	Industrial and military effluents[e,g]					
	Agricultural and forestry effluents[c]					
	Garbage and solid waste[c]					
	Air-borne pollutants[e,g]					

Note that a comprehensive list of the first and second levels of classes in the framework is provided, and data gaps exist when no readily-available dataset was identified to map a stressor

* *Source of estimation E* empirical, *X* expert-opinion. *KD* kernel density with radius in kilometers. Values used to parameterize the calculation of *h*

[a] NLCD was filtered to remove the built-up land cover types that were "burned-in" on the basis of rural highways

[b] Some intense tourist and recreation uses are mapped in land cover datasets, but lower intensity tourism and recreation such as ski areas often are not included

[c] No nationally consistent, readily-available dataset is known

[d] Not applicable for terrestrial-based ecosystems

[e] Partially addressed by distance to major roads

[f] Partially addressed by plantations

[g] Partially addressed by residential and commercial development

Table 2 Estimated magnitude (*I*) values (0 → 1.0) for different land use types, from cross-walking categories to Brown and Vivas (2005)

Description	Magnitude
Undeveloped	0.0
Residential	0.7
Mixed use developed	0.9
Agriculture	0.5
Resource extraction	0.8
Industrial	1.0
Recreation	0.2
Transportation	1.0
Unknown[a]	0.3

[a] But human modified—estimated to be 0.3 because it reflects clear signs of human modification but from miscellaneous and unknown types of activities

Stressors framework and spatial datasets

The Conservation Measures Partnership (CMP) framework catalogues and organizes multiple sources of stressors or threats associated with different human activities (Salafsky et al. 2008). Organizing the multiple stressors that can influence a landscape using this existing framework helps to minimize redundancy and potential overlap. It also results in a comprehensive but parsimonious list of roughly a dozen different major threats that are further broken down into classes (or stressors) that I mapped as variables (Table 1). Each variable is represented as an individual data layer, with values that range from 0 to 1.0 (no to complete impact).

I mapped residential and commercial development stressors from the National Land Cover Dataset 2006 (NLCD; Fry et al. 2011; www.mrlc.gov) using the developed cover classes that include commercial, industrial, and residential land uses. Housing density data from Bierwagen et al. (2010) were used to map residential areas, particularly because low-density residential areas (<1 dwelling unit per acre; dua) are largely unmapped in NLCD. Agricultural stressors were mapped from NLCD classes of cropland and pastureland. I was unable to locate a consistent, reliable, and readily-available dataset on livestock farming and ranching (i.e. grazing). Energy development stressors were mapped using a kernel density (KD) function applied to oil and gas well locations (Copeland et al. 2009) with a 1 km radius and maximum impact estimated to be 0.5. State natural resource experts (WGA Landscape Integrity working group) estimated a maximum impact of 0.25 for effects associated with active mines and quarries and 0.17 for wind tower/turbine locations (https://oeaaa.faa.gov), both with a 0.5 km radius. Transportation stressors (Forman et al. 2003; Fahrig and Rytwinski 2009) were mapped using several datasets. The physical footprint of roads and railroads was mapped using TIGER 2010 data (www.census.gov/geo/www/tiger), with average widths estimated empirically from aerial photography by road type. Road use was measured by highway traffic or average annual daily traffic (AADT; number of vehicles per day) from the National Transportation Atlas Database 2012 (www.bts.gov) by applying a KD with 1 km radius and an estimated maximum impact of 0.5 for AADT ≥100,000 (Theobald 2010). Utility power lines were mapped to current power line infrastructure locations with a KD of 0.5 km and maximum impact of 0.17. I mapped communication towers and antennae from the Federal Communications Commission's Antenna Structure Registration dataset (FCC 2012) by applying a KD of 0.25 km, assuming a maximum impact of 0.25. Potential stressors associated with airplane flight paths were not mapped, due to a lack of readily-available data and limited knowledge about their impacts to biodiversity.

I was able to only partially address effects associated with biological use stressors such as hunting, fishing, plant gathering, and timber logging. These resource extraction activities tend to be quite dispersed and because they are limited by accessibility to locate a resource and to transport materials back to process, I used a measure of impact as a function of the distance from major roads (state and county highways) as a proxy (Gelbard and Belnap 2003; Coffin 2007; Fahrig and Rytwinski 2009). I did not include maps associated with fire because spatial data are limited about the degree of human modifications to these natural processes. Data on dam (and reservoir) locations are readily available, but mapping their effects is challenging, in part because much of their ecological impact manifests in an indirect way at some distance from the dam, the data required to calculate the hydraulic residency time are limited (Poff and Hart 2002) and because mapping them requires processing complex hydrologic networks. I mapped land cover that was dominated by introduced species (i.e. invasive), as mapped by the five classes in the USGS Gap land cover v2 (USGS 2011) dataset. The importance of

Table 3 Empirical estimates (average values) of human modification derived by combining detailed land use dataset with the USGS Gap land cover v2 dataset

Formation class[a]	Pacific 1	IM basin 2	Rocky mtns. 3	Deserts 4	Great plains 5	Coastal 6	Appalachian 7	Northern/boreal forests 8	
Polar and alpine cliff, scree and rock vegetation			0.0023	0.0023	0.0023				0.0023
Alpine scrub, forb meadow and grassland	0.0007	0.0007	0.0007					0.0007	0.0007
Temperate and boreal cliff, scree and rock vegetation			0.0185	0.0185	0.0165	0.0165	0.0165	0.0165	0.0172
Cool semi-desert cliff, scree and rock vegetation	0.4	0.0046	0.0021	0.0021	0.0021	0.0021			0.0688
Warm semi-desert cliff, scree and rock vegetation	0.0087	0.0087	0.0087	0.0061	0.0061	0.0061		0.0061	0.0072
Mediterranean cliff, scree and rock vegetation	0			0		0	0	0	0.0000
Cool temperate forest	0.0436	0.0074	0.0178	0.0119	0.1171	0.215	0.1289	0.0911	0.0791
Lowland and montane boreal forest					0.053		0.053	0.053	0.0530
Warm temperate forest	0.0328	0.1024	0.0114	0.0071	0.0683	0.2203	0.2141	0.2141	0.1088
Tropical (semi-) deciduous forest						0.1468			0.1468
Boreal grassland, meadow and shrubland								0.1427	0.1427
Temperate grassland, meadow and shrubland	0.0687	0.0232	0.0413	0.0098	0.1454	0.3129	0.1961	0.1427	0.1175
Mediterranean scrub	0.0384	0.0384		0.0384		0.0384		0.0384	0.0384
Mediterranean grassland and forb meadow	0.0962	0.0962		0.0962		0.0962		0.0962	0.0962
Cool semi-desert scrub and grassland	0.0037	0.0122	0.0103	0.0203	0.0236	0.0236			0.0156
Warm semi-desert scrub and grassland	0.0222	0.0222	0.0222	0.0236	0.0853	0.1204	0.1021	0.0613	0.0574
Temperate and boreal scrub and herb coastal vegetation	0.1859	0.1859	0.1859	0.1859	0.1859	0.1859	0.1859	0.1859	0.1859
Tropical wooded, scrub and herb coastal vegetation						0.1859			0.1859
Boreal flooded and swamp forest					0.0985		0.0985	0.0985	0.0985
Mangrove						0.1468			0.1468
Marine and estuarine saltwater aquatic vegetation	0.0248		0.0248			0.0364		0.0364	0.0306
Salt marsh	0.0248	0.0248	0.0248	0.0248	0.0231	0.0364	0.0364	0.0364	0.0289
Temperate and boreal bog and fen	0.1214	0.1214	0.1214		0.1214	0.1214	0.1214		0.1214
Temperate and boreal scrub and herb coastal vegetation	0.1859	0.1859	0.1859	0.1859	0.1859	0.1859	0.1859	0.1859	0.1859
Temperate flooded and swamp forest	0.0862	0.0439	0.0512	0.0861	0.1643	0.1468	0.1374	0.0901	0.1008
Tropical flooded and swamp forest						0.1468			0.1468
Tropical Freshwater Marsh						0.0096			0.0096
Freshwater aquatic vegetation	0.0096					0.0096	0.0096		0.0096
Open water	0.0323	0.012	0.0013	0.0013	0.1088	0.0614	0.08	0.0341	0.0414
Barren	0.0541	0.0541	0.0541	0.0541		0.0541	0.0541	0.0541	0.0541
Current and historic mining activity	0.5887	0.3765	0.3469	0.4682	0.5328	0.4721	0.5474	0.4338	0.4708

Table 3 continued

Formation class[a]	Pacific 1	IM basin 2	Rocky mtns. 3	Deserts 4	Great plains 5	Coastal 6	Appalachian 7	Northern/boreal forests 8	
Introduced and semi natural vegetation	0.0847	0.0556	0.09	0.09	0.2628	0.1649	0.1649	0.1649	0.1347
Woody agricultural vegetation	0.419		0.419						0.4190
Herbaceous agricultural vegetation pasture	0.421	0.2315	0.2986	0.2806	0.3845	0.3475	0.3919	0.387	0.3428
Herbaceous agricultural vegetation cropland	0.4764	0.4201	0.351	0.4248	0.4574	0.4557	0.4335	0.3917	0.4263
Recently disturbed or modified	0.2893	0.0029	0.1297	0.069	0.2885	0.3524	0.2855	0.3466	0.2205
Developed, Open Space	0.2965	0.4064	0.2554	0.1379	0.4187	0.3953	0.397	0.3285	0.3295
Developed, low intensity	0.5887	0.3765	0.3469	0.4682	0.5328	0.4721	0.5474	0.4338	0.4708
Developed, medium intensity	0.6505	0.6505	0.6505	0.6505	0.6341	0.5329	0.6851	0.6389	0.6366
Developed, high intensity	0.6033	0.6033	0.6033	0.6033	0.7303	0.6909	0.7621	0.7621	0.6698
Average	0.20	0.16	0.15	0.15	0.22	0.19	0.23	0.19	

Note these are calculated independently for each ecodivision grouping, ensuring that there were at least 100 data points that the estimates were based on, following methods described in Leinwand et al. (2010) and Theobald et al. (2012)

[a] This is the formation level for natural cover types from the National Vegetation Classification (Grossman et al. 1998)

invasive species and problematic native species in altering the condition of ecological systems is widely recognized, but a detailed, readily-available dataset on the location or proportion of these species is not available (Bradley and Marvin 2011). Also, note that stressors related to pollution were not directly included—although effects from these are partially included in our overall model because I directly map roads, urban areas, residential housing, and croplands.

Combining stress layers to overall degree of human modification

I used a method that minimizes bias associated with non-independence among multiple stressor/threats layers. That is, I assumed that locations with multiple threats have a higher degree of human modification than locations with just a single threat (assuming the same value), but the cumulative human modification score converges to 1.0 with multiple stressors. The specific algorithm is called a "fuzzy algebraic sum" (Bonham-Carter 1994) and the result is always at least as great as the largest contributing factor, so the effect is "increasive", but never exceeds 1.0 (Theobald 2013). The overall degree of human modification H_i at each cell i, with values that range from 0.0 (no modification, natural) to 1.0 (highly modified, un-natural) and is calculated as:

$$H_i = 1.0 - \prod_{j=1}^{k} \left(1 - h_j\right)$$

and let h = human modification score for individual stressor, with values ranging from 0.0 (no human modification, natural) to 1.0 (high degree of modification, un-natural), for $j = 1 \ldots k$ data layers. For example, H_i for three layers of 0.6, 0.5, and 0.4, the computation would be: $H_i = 1.0 - ((1 - 0.6) \times (1 - 0.5) \times (1 - 0.4))$, or 0.88. Note that the final human modification layer where each raster cell value equals H_i is denoted as H. I also identified the stressor h_j that contributed the highest level impact at a given location, which I called "dominant."

Model evaluation and application

Because measures of ecological integrity commonly are used in spatially-explicit models, such as the

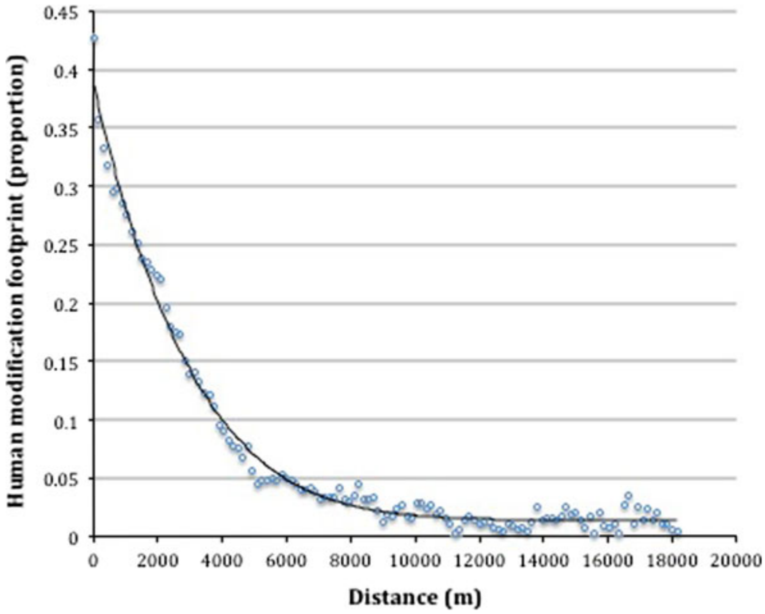

Fig. 1 The relationship of human modification to distance from major roads, fit using a 4th order polynomial trend line: $y = -5E - 22x^5 + 4E - 17x^4 - 1E - 12x^3 + 2E - 08x^2 - 0.0001x + 0.387$ ($R^2 = 0.98$)

resistance layer for connectivity mapping (e.g., Carroll et al. 2012; McRae et al. 2012; Theobald et al. 2012), it is important to understand and evaluate the degree to which spatial processes are integrated into a measure and the spatial patterns that emerge, so that reasonable interpretations can be made. That is, most landscape integrity maps account for local or very fine scale (e.g., within a cell or nearby such as 500 m), but for some purposes are aggregated to watersheds (e.g., Esselman et al. 2011). Commonly in landscape ecology two dominant ecological processes have been discussed (Wiens 2002): those dominated by terrestrial processes (animal movement, wind dispersal, etc.) and those that are dominated by freshwater processes (i.e. hydrologic and riverine).

To evaluate the role of a presumed dominant ecological process in forming spatial pattern, I calculated and compared three ways to process the raw values in the human modification dataset. To represent *local* or in situ processes, I calculated the mean value of H from the 90 m dataset for each 12-digit HUCs, denoted as H_l. To represent a *watershed* perspective where hydrologic connectivity dominates but is not limited to downstream-only flows (and therefore this is not freshwater in the strict sense), I calculated a hierarchical watershed average value, denoted as H_w. That is, the mean H value within each HUC found within each 12, 10, 8, 6, and 4-digit layer was calculated, and then the mean H value at each raster cell across the 5 layers was calculated. An important distinction here is that this approach does not assume that a given process can be adequately captured at a single scale (or even known adequately), but rather it makes use of a multi-scale averaging process that is more appropriate for general representation of landscape-level processes (Riitters et al. 2009; Theobald 2010). To represent a *terrestrial* perspective, I applied the multi-scale averaging approach and assumed that the dominant ecological processes were isotropic and therefore were represented by a moving *circular* windows, scaled in size equal to the average HUC area: 101, 545, 3981, 25426, and 42168 km^2 for HUC 12-4, denoted as H_t.

To compare the process perspectives, I calculated a Z-score by standardizing the H_l values in each HUC12 against the values from the local process layer. Locations with a large negative Z-score signify that the local scores are significantly higher and over-represent the impact compared to when areas are integrated according to either a watershed or terrestrial perspective. Locations with a large positive Z-score signify that the local scores (H_l) are significantly lower and under-represent the impact compared to the watershed (H_w) or terrestrial (H_t) maps.

I assessed how well the degree of human modification predicts a general indicator of field-level conditions from the EPA's Wadeable Stream Assessment (WSA) following the approach of Falcone et al.

(2010). I classified sites into two levels of disturbance: *reference* sites that were considered to be natural or least-disturbed conditions in their ecoregions (n = 1,699) and *disturbed* which were considered to be most heavily-modified by human activities (n = 440). I expected that there would be a significant difference between the human modification values within the reference sites versus the disturbed sites. I expected that the watershed characterization would have the best fit with the WSA sites, followed by terrestrial (because of spatial process), and the poorest fit with the local process (HUC12). Finally, I summarized findings by protection status level from the Protected Areas Database (http://gapanalysis.usgs.gov/padus/).

Results

For the conterminous US, I found the overall average degree of human modification H value was 0.3756 (SD = 0.243). Of course this varies regionally (Fig. 2; Table 4), and not surprisingly the intermountain west was least modified ($H = 0.2216$, SD = 0.193), while the Great Lakes region was most heavily modified by human activities ($H = 0.5349$, SD = 0.211). The general pattern of human modification also increases predictably as a function of decreasing protection level, so that H in status 1 = 0.1556 (SD = 0.141), 2 = 0.2004 (SD = 0.176), 3 = 0.2021 (SD = 0.162), and 4 = 0.4349 (SD = 0.236).

Table 4 Results of the degree of human modification within census regions (http://www.census.gov/geo/maps-data/maps/pdfs/reference/us_regdiv.pdf)

Region	Mean	SD
Pacific	0.2860	0.237
Intermountain West	0.2216	0.193
North Central	0.4715	0.185
South Central	0.4206	0.215
Great Lakes	0.5349	0.211
Northeast	0.4805	0.248
Southeast	0.5187	0.213

Figure 3a–c show the degree of human modification mapped to examine results from different spatial processes: local (HUC12), watershed, and terrestrial. Figures 4a–c show the same data but zoomed into the Austin, Texas area as an example of the detailed patterns. At a continental extent, all three patterns are generally similar, but Fig. 5a and b show the departure from local values for both the watershed and terrestrial maps. Zooming into a narrower region (for example, Austin Texas; Fig. 5c, d) shows the fine-grained heterogeneity of these differences, including a difference in direction

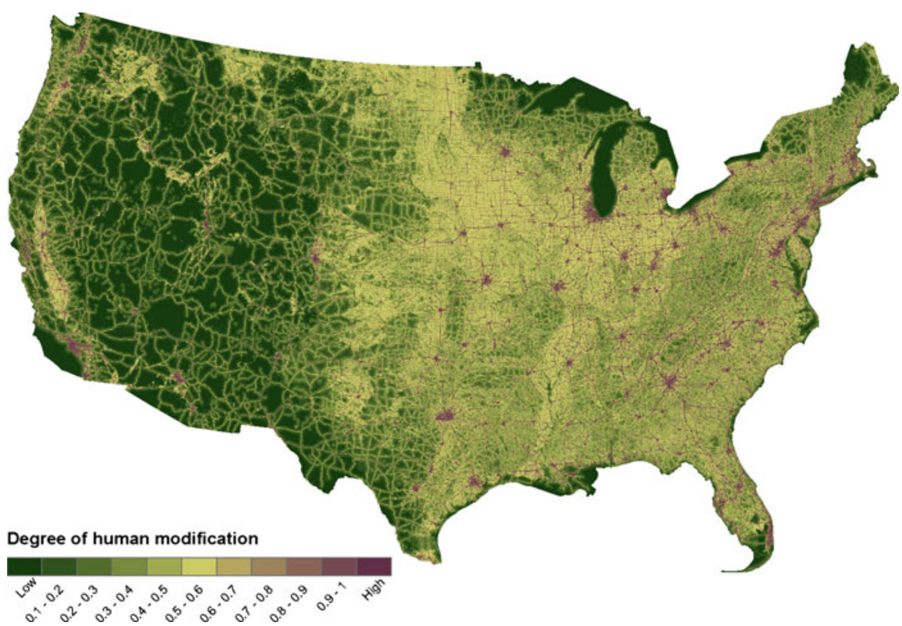

Fig. 2 The degree of human modification (H) for the conterminous US at 90 m resolution, showing low levels of human activities in *green*, moderate levels in *yellow*, and high levels of human activities in *red*. *Note* major water bodies are included for reference, but water-based stressors are not included in a primary way. (Color figure online)

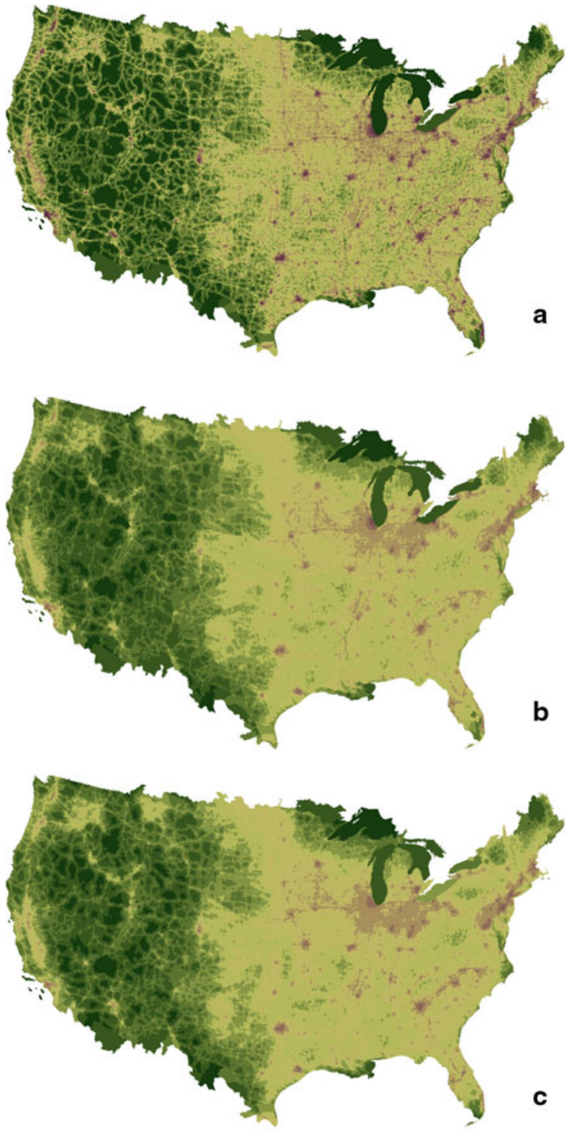

Fig. 3 Maps showing the degree of human modification (see Fig. 2 for legend), for different assumed ecological processes: **a** the "local" shown at a 12-digit hydrologic unit code; **b** "watershed" perspective by hierarchical averaging across HUC units 12, 10, 8, 6, and 4; **c** "terrestrial" using five moving windows sized equal to the average HUC units at the various scales

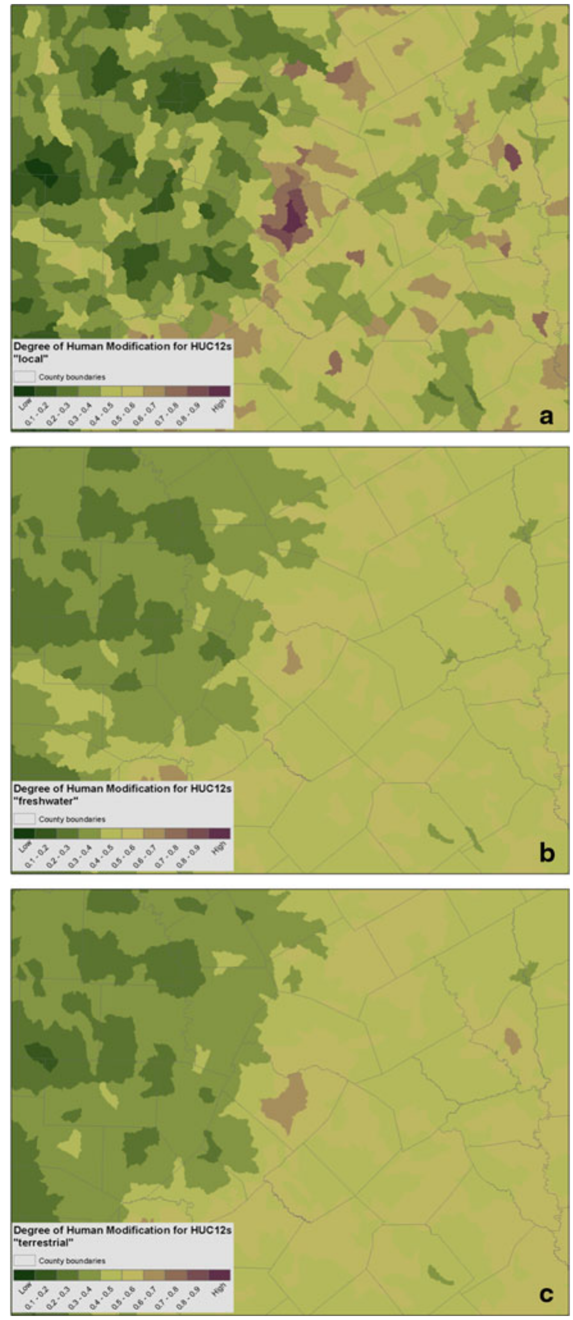

Fig. 4 A zoom-in map around Austin, Texas showing the degree of human modification, for different assumed ecological processes: **a** the "local"; **b** "watershed"; and **c** "terrestrial"

(under- vs. over-estimation) for some locations between watershed and terrestrial results.

Not surprisingly, I found that stressors associated with land uses that resulted in conversion to developed lands were dominant. Urban and residential density and agricultural activities were dominant for 44 % of the US, while impacts associated with distance from major roads dominated 51 %—particularly in the western US. For 2 % of the US, the road footprint was dominant, while effects associated with housing density was dominant in 0.3 %. Recall that the road footprint represents only the physical extent up to

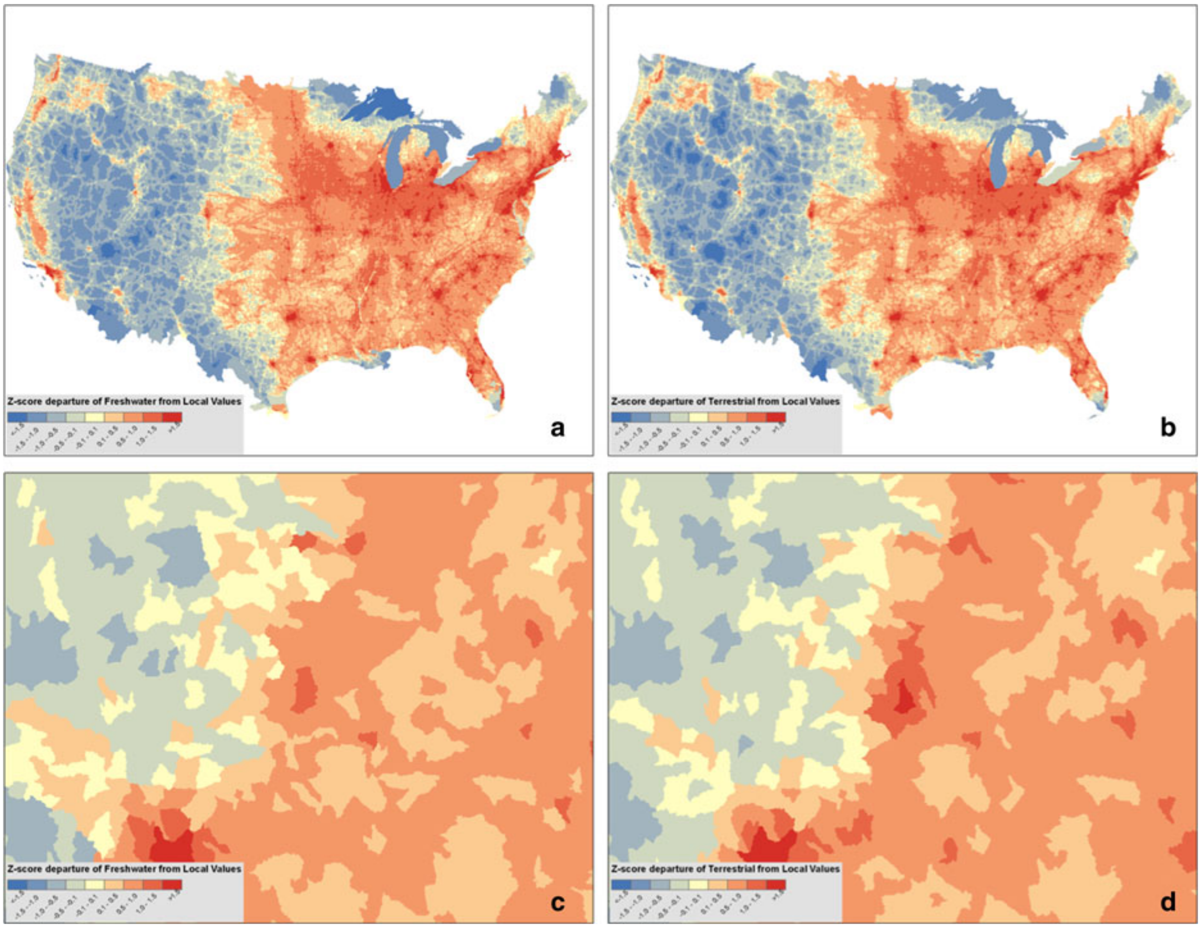

Fig. 5 A map showing the departure from local values for both **a** watershed and **b** terrestrial maps, as compared to local 12-digit HUC scores; **c** is freshwater near Austin, TX; and **d** is terrestrial near Austin, TX. That is, a *Z*-score was calculated by standardizing the *h* values in each HUC12 against the local values

30 m, and note that for most locations multiple stressors occurred together.

I compared results of *H* values (90 m resolution) from 2,139 WSA sites, and found that the mean value of *H* is less in reference sites (mean $= 0.351$, SD $=$ 0.173) than disturbed sites (mean $= 0.432$, SD $=$ 0.197). The distributions of reference to disturbed sites were significantly different using a Cramer–von Mises two-tailed test ($p = 0.005$) for all three forms: local ($W^2 = 6.558$), watershed ($W^2 = 3.495$), and terrestrial ($W^2 = 3.907$). Also, there is less variability in the watershed values for reference and disturbed sites (SD $= 0.147$, 0.161) as compared to the terrestrial (SD $= 0.152$, 0.164) and the local (0.173, 0.197) datasets, one indication that the watershed-process layer had the best fit with the validation dataset.

Discussion and application

The finding that about 38 % overall degree of human modification is roughly comparable with past estimates of human footprint and naturalness (34–35 %; Theobald 2010), though the variability of values in the current results has been reduced roughly in half. This is likely due to a tighter estimation of the degree of human modification.

Landscape integrity values changed substantially depending on what ecological process was assumed to be dominant. That is, for most urban and highly-modified locations (particularly in the eastern US), a map of local values tends to underestimate impacts because it does not consider any spill-over or influence from adjacent or nearby HUC12s. This assumption may be justified for some situations where local-scale processes dominate. For other situations, such as potential effects of human activities on river water quality, clearly nearby (and especially upstream) impacts can strongly influence nearby (especially downstream) conditions. Note that even a simple isotropic assumption of spatial process can result in estimated values that are quite different from local conditions. Very fine-grained differences can occur—including a difference in direction (under- vs. over-estimation) for some locations between watershed and terrestrial results (e.g., Fig. 5d). The main point from this process comparison is that strongly different results can be obtained depending on the assumed ecological process and neighborhood or scale of analysis (Wiens 1989).

These results could be applied in three main ways by land management agencies. First, many programs directly use a measure of ecological integrity as a key variable in landscape assessments. For example, the results here could be used to update the BLM's REAs to provide a more consistent basis for their results. That is, using a comprehensive and empirically-based estimate of human modification would strengthen the findings of existing REAs and would enable consistency across the roughly dozen assessments. The degree of human modification results found here could also be used directly in the ongoing ecoregional landscape assessments conducted by the 16 LCCs, or to identify the large intact landscapes that is a primary data layer in the WGA Crucial Habitat Assessment Tool.

Second, the data layer here can be summarized to provide a measure of landscape context to inform management within a specific protected area (e.g., Hansen et al. 2011). As described earlier, Table 5 provides a summary of the degree of human modification averaged across each LCC, ranging from a low of 0.1835 for Great Basin and a high of 0.5797 in the Eastern Tallgrass Prairie and Big Rivers LCC. From a continental or national perspective, analyzing these scores in this way provides a robust and consistent measure of landscape integrity that can be used to roughly compare among broad units. Similar measures can be easily developed, for example for the 17 states in the WGA CHAT, the 32 networks of the National Park Service and the 14 ecoregions of the BLM's REAs.

A third type of use is to characterize the ecological context outside of existing protected areas to provide more locally-relevant and meaningful measures that can be used to inform the selection of conservation targets and/or help to prioritize specific locations of conservation action within each administrative unit—at the local, state or federal managerial unit. For example, Fig. 6 provides a depiction of areas of potential conservation opportunity that combines a regionalized landscape integrity score with a protection status score to help distinguish potential audiences and actions. That is, the H values at each location were standardized to the LCC so that importance is expressed relative to each LCC. Locations (in this case HUC12 s) with each LCC were then ranked to identify the 90th-percentile, the 75th, and the 50th (i.e. the median) as a rough classification of importance. These are portrayed in different colors for conservation status (Gap status level) 1&2 (highest protection level for biodiversity (i.e. biodiversity reserves), 3 (protected with some extractive activities), and 4 (unprotected, mostly privately-owned). Opportunities and actions differ with each status category (Wade et al. 2011); indeed, for each land owner and management unit as well, but those are beyond the scope of this paper. For example, status 1&2 will likely be focused on management of currently protected lands, rather than targeting specific locations to change management of status 3 lands to be more compatible with biodiversity protection—particularly those with high

Table 5 Results of the degree of human modification within US Fish & Wildlife Service Landscape Conservation Collaboratives for the conterminous US

Name	Mean	SD
North Pacific	0.3143	0.230
California	0.3901	0.261
Great Northern	0.2150	0.193
Great Basin	0.1835	0.183
Southern Rockies	0.1962	0.176
Desert	0.1952	0.188
Plains and Prairie Potholes	0.4028	0.198
Great Plains	0.4269	0.184
Gulf Coast Prairie	0.4176	0.227
Upper Midwest and Great Lakes	0.3754	0.260
Eastern Tallgrass Prairie and Big Rivers	0.5797	0.173
Gulf Coastal Plains and Ozarks	0.4888	0.189
North Atlantic	0.4763	0.265
Appalachian	0.5014	0.211
South Atlantic	0.5406	0.210
Peninsular Florida	0.5150	0.259

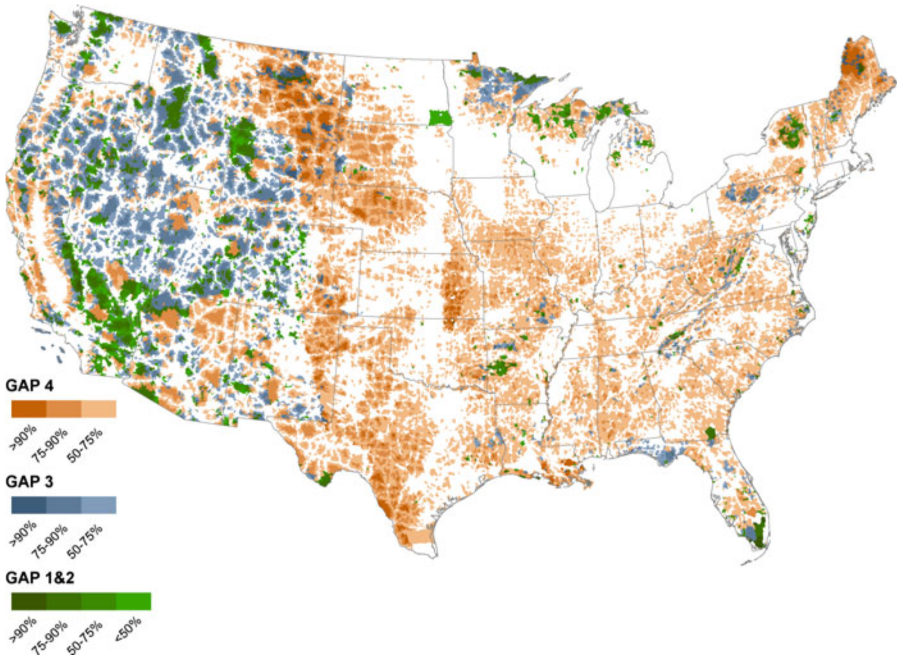

Fig. 6 Potential conservation opportunities to conserve large, intact landscapes. Results are shown for three protection level status codes: parks and wilderness areas in Gap level 1&2 (*green*), multi-use public lands in Gap 3 (*blue*), and privately-owned lands without formal conservation protection in Gap 4 (*orange*). Deeper hues signify 12-digit HUCs with a lower degree of human modification (i.e. higher levels of landscape integrity), lighter hues signify a higher degree of human modification—areas without any colors (*white*) have a relatively high degree of human modification. (Color figure online)

landscape integrity near a cluster of status 1&2, or perhaps providing corridors of higher protection to move between reserves. For status 4 lands, areas with high integrity ranks might be considered to have higher value in a prioritization for potential conservation purchase or easement programs. Although this approach is not intended to replace prioritization efforts by individual agencies and organizations, it does give an important complementary perspective by providing an integrated, synthetic, landscape view that crosses land ownership boundaries. Note that locations that are less than the mean standardized value are not portrayed in this map, but should not be interpreted as having no conservation value. Instead, these locations could be viewed through a restoration lens, by identifying those areas that contribute to overall improvements if local stressors to landscape integrity could be ameliorated (Baldwin et al. 2012).

I recognize that there was a practical and opportunistic aspect to the selection of stressors that were included in the final model, as not all stressors have reliable, publicly-available datasets available. A critical advantage of examining potential stressors within the broad framework is that insight can be gained into which threats were most important (impactful) and relevant, and the gaps are made explicit to identify future opportunities for data that would improve the overall human modification model. To that end, the most critical datasets for future improvement to this landscape integrity dataset include stressors that effect disproportionately freshwater resources such as dams, irrigation, and pumping, the proportion of invasive species, likely shifts in biomes due to climate change, the intensity of domestic grazing, and hunting and fishing pressure. Although not emphasized in this paper, this approach supports the monitoring of status and trends in landscape integrity, as the main inputs are time dependent (cover, housing, roads, etc.) so that a landscape integrity dataset could be generated at a 5–10 year interval (e.g., Theobald 2010).

In this paper I developed and provided preliminary applications of an empirically-based, robust measure of ecological integrity at the landscape level. I found that the degree of human modification averaged to be about 0.38 across the US, with reasonable regional variation. Estimates of impact for roughly half of the stressors included here relied on values established by expert judgment, but more than 97 % of the US was dominated by a stressor whose impact was estimated using empirical data. Although improvements could be made to this approach, especially in terms of filling data gaps on invasive species and grazing/hunting intensity, the framework and methodology described here provides important improvements over existing, ad hoc approaches, to provide a foundation on which sound monitoring and evaluation of ecological integrity can be based. Most importantly, landscape-level assessments of ecological integrity should be based on an internally consistent model, comply with decision theory principles, incorporate empirically-derived data to the maximum extent possible, explicitly state the incorporation of the assumed dominant ecological process, and provide validation of their results to the degree possible.

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Appendix N

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CONTRIBUTED PAPER

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Conservation value of national forest roadless areas

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Abstract

Conservation scientists call for establishing additional protected areas amidst ongoing threats of expanding human development. Nevertheless, some existing protected areas are being downsized and demoted of their existing conservation protections. In 2001, the Roadless Area Conservation Rule prohibited road construction and timber harvest in 240,000 km^2 of inventoried roadless areas (IRAs) located on United States Department of Agriculture Forest Service lands. IRAs represent a non-legislative protected status that is in jeopardy of conservation demotion or “degazettement,” and few national protected area assessments recognize the IRA designation. Since the rule's conception two decades ago, little research has been conducted to assess the conservation values of IRAs and the values they could add to the existing system of highly protected areas in the continental United States. To increase understanding of these conservation values, we assessed three aspects of roadless areas: (a) how wild and intact are IRA lands compared to state, national, and protected lands, (b) how do IRAs complement the size, connectedness, and representation of protected lands, and (c) how do IRAs contribute to protection of important ecosystem services (drinking water and annual carbon capture)? Through this analysis we found that many IRAs are among the most wild, undeveloped areas both in the nation and within their respective states. IRAs increase the size of—and reduce isolation between—protected areas, likely buffering them from external stressors. In some places, IRAs protect watersheds that deliver drinking water to hundreds of thousands of people. IRAs also add significantly to the total carbon captured by existing protected areas. The results of our evaluation demonstrate the potential of IRAs to contribute to the conservation value of the U.S. protected area system and to deliver important ecosystem services.

KEYWORDS

buffers, conservation reserves, drinking water, ecosystem services, inventoried roadless areas, protected areas, Roadless Rule

1 | INTRODUCTION

The ecological effects of roads and their related impacts include fragmenting habitat, providing vectors for invasive species, and degrading water quality, among others (Trombulak & Frissell, 2000). While much of the terrestrial Earth is roadless (Ibisch et al., 2016), roadless areas are disproportionately rare inside of the contiguous U.S. (Watts et al., 2007). Associated with the global expansion of road networks, terrestrial ecosystems that are relatively free of human modification (e.g., "wildlands") are being lost to development at nearly twice the rate of their protection (Watson et al., 2016). Additionally, the western United States lost nearly 12,000 km^2 (3 million acres) of open and natural land to development in the decade following 2001 (Theobald, Leinwand, Anderson, Landau, & Dickson, 2019). In response, ecologists have stressed the importance of protecting what is left of the remaining wild places (Belote et al., 2017; Martin, Maris, & Simberloff, 2016; Watson et al., 2018).

In January 2001, recognizing the value of roadless areas, the US Department of Agriculture (USDA) Forest Service established the Roadless Area Conservation Rule, which recognized Inventoried Roadless Areas (IRAs), areas that the agency had identified, inventoried, and found to be free of roads in the 1970s, and prohibited road construction and timber harvest within them in 37 states within the contiguous U.S. (the Rule also covered Alaska, which is not considered here). The primary goal of this policy is to sustain the health and productivity of America's forests so present and future generations can benefit from the resources they offer, but the Rule is also intended to address Forest Service concerns with its ability, due to budget constraints, to maintain the vast amount of existing roads to the environmental standards required by law. Lacking roads and human-infrastructure, IRAs represent relatively wild and undeveloped land, but their potential contribution to the nation's protected area system has not been assessed (but see Aplet, Wilbert, & Morton, 2005).

Because IRAs are an administrative designation of the U.S. Forest Service and not legislatively established by the U.S. Congress, IRAs are not considered part of the U.S. system of protected areas (i.e., units classified with a Gap Analysis Program [GAP] status of 1 or 2) in protected areas databases of the U.S. (USGS GAP, 2012). Protected areas classified as GAP 1 or 2 are governed by mandates to maintain biodiversity, prohibit land cover conversion, and minimize other human stressors. Road building and commercial timber harvesting are prohibited in IRAs, but livestock grazing, motorized recreation, and some mining is permitted. While IRAs provide similar levels of protection to many GAP 1 and 2 areas, their administrative status leaves them vulnerable to efforts to repeal the Roadless Rule completely, as occurred recently in Alaska and Utah (see Weber, 2019). These calls to downgrade the levels of protection from IRAs mirror similar threats to eliminate or reduce the size of other protected areas (Golden Kroner et al., 2019).

Elimination of conservation protections has occurred despite ongoing calls to protect more land to address the growing biodiversity crisis (Aycrigg et al., 2016; Convention on Biological Diversity, 2014; Dinerstein et al., 2019; Jenkins, Van Houtan, Pimm, & Sexton, 2015; Watson et al., 2016). In addition, to facilitate adaptation to a changing climate, conservation scientists have recommended protecting the last remaining wild lands (Watson et al., 2018), connecting existing reserves (Belote et al., 2016), and more fully representing ecological diversity within the protected area system (Aycrigg et al., 2013). Growing the protected area system is also seen as necessary to sustain important ecosystem services upon which humans depend (Dinerstein et al., 2019; Jarvis, 2020).

IRAs are not recognized as formal protected areas but likely represent an example of "other effective area-based conservation measures" (OECMs; IUCN-WCPA, 2019). OECMs are lands not classified as protected but which are "governed and managed in ways that achieve positive and sustained long-term outcomes for the in situ conservation of biodiversity with associated ecosystem functions…" (IUCN-WCPA, 2019). OECMs may also contribute to the ecological representation and connectivity of protected areas. Documenting the role of OECMs is critical to fully account for the contribution to conservation.

Considering the lack of formal recognition that IRAs receive as protected areas, recent calls to repeal the Roadless Rule (Weber, 2019), and ongoing threats to reduce or eliminate protected areas, we undertook an assessment of the potential contributions of IRAs to the protected area system of the contiguous U.S. We focus on three guiding questions throughout the study: (a) How might IRAs add to the protection of the wildest lands in the contiguous U.S. and respective states? (b) How do IRAs contribute to the size, connectedness, and representation of protected areas? (c) How do IRAs add to the ability of protected areas to deliver ecosystem services of drinking water protection and carbon capture?

2 | METHODS

To compare IRAs with currently protected areas in the contiguous United States, we define protected areas as lands designated within the Gap Analysis Program's Protected Areas Database version 10.3 (USGS GAP, 2012) with GAP 1 and 2 status, excluding national fisheries and marine sanctuaries. These protected areas are mostly composed of lands managed by federal and state agencies to maintain biodiversity and limit resource extraction, though private easements are also included as GAP 1 and 2 when their management prevents land use conversion and aims to protect biodiversity. We considered GAP 1 and 2 areas that shared borders as one protected core area. To create cores, we dissolved boundaries of all protected areas directly adjacent to one another. For instance, Yellowstone National Park and the adjacent wilderness areas were treated as one protected core area in our data set. For our analysis, we included 92,556 protected core areas representing 602,901 km^2 (ranging in size from <1 to 24,040 km^2).

Within the IRA database, some IRAs overlap with currently protected GAP 1 and 2 lands. For this analysis, we removed any sections of IRA land that overlap GAP status 1 and 2 lands from the IRA database and assigned those areas to GAP 1 and 2. These areas of overlap between GAP 1 and 2 lands and IRAs occurred mostly in Wilderness Study Areas and represented 4.7% of the total IRA area. IRAs with the same name but delineated by either North, South, East or West, or by sections (A, B, C, etc.) were combined to make up a single IRA spatial unit.

2.1 | IRAs and wildness

To measure wildness, we used existing spatial data on the degree of human modification (Theobald, 2013) but reverse-ordered the data so that higher values are associated with "wilder" (i.e., less human-modified) lands. Human modification was quantified using data on roads, railways, transmission lines, land cover, and human population density (see Theobald, 2013 for more details). The human modification map is closely correlated with earlier maps of wildness (Aplet, Thomson, & Wilbert, 2000) and has been used in recent research to identify the wildest lands in the lower 48 US states (Belote et al., 2017). In order to compare wildness between IRAs and protected areas, we extracted all wildness values across each designation. We then classified wildness values into deciles based on the distribution across the contiguous US and assigned each decile an integer 1 (lowest decile) through 10 (highest decile). We classified the data in this way to more clearly evaluate how much of the wildest lands were protected in GAP 1 and 2 lands and how IRAs might contribute to protecting the wildest lands.

We also extracted mean wildness for all IRAs and compared them with wildness values within their respective states as well as wildness values across the contiguous US. This allowed us to compare the mean wildness of IRAs to the state-wide and full national distribution of gradients in wildness.

2.2 | IRAs and the size of protected areas

To determine how IRAs contribute to the size of protected areas, we calculated total area of protected lands (GAP 1 and 2 lands excluding national fisheries and marine sanctuaries), as well as the total area contained within IRAs. We also analyzed how IRAs effectively increase the size of protected areas by locating all IRAs directly adjacent to existing protected areas. We then compared the area of the protected system as it stands today to what it would be with the inclusion of these IRAs. We also analyzed how IRAs adjacent to individual protected core areas would increase the size of those cores. We spatially joined the protected core areas to adjacent IRAs. We then summed the area of IRAs adjacent to each core and compared the area of cores before and after the spatial join by calculating the percent and absolute increase in area of each protected core after adding adjacent IRAs.

Following this analysis and after looking at the maps of IRAs and their locations with respect to protected lands, we noticed that some IRAs appear to form larger land unit complexes with protected areas at their core with adjacent IRAs (identified above) and additional IRAs very close to those. These complexes may form large intact and relatively connected core areas for maintaining wildlife movement (Gaston, Jackson, Cantú-Salazar, & Cruz-Piñón, 2008; Tucker et al., 2018). To analyze the extent to which IRAs may create complexes with core protected areas in this way, we selected all IRAs within 100 m of a protected area and joined them to protected areas (Step 1). Then, we selected all IRAs within 100 m of the layer created in Step 1 and continued this stepped process until no IRAs were within 100 m of the previous step. In addition to the 100-m distance, we repeated the analysis using 5 and 10 km stepped build out distances to account for wide ranges in animal dispersal distances (Bowman, Jaeger, & Fahrig, 2002).

2.3 | IRAs and ecological representation

We were interested in whether IRAs contained ecosystems not otherwise well-protected in existing protected areas. We relied on the GAP national land cover data set

(USGS, 30-m resolution) to evaluate ecological representation of protected areas and IRAs (Aycrigg et al., 2013, 2016; Dietz, Belote, Aplet, & Aycrigg, 2015). We extracted landcover data within protected areas, as well as protected areas with the inclusion of IRAs to calculate increase in representation. We focused on the "formation" (n = 31) level of ecosystem classification to summarize results, as it provided a sufficiently detailed yet tractable number of classes in the hierarchy of national vegetation classification. We removed human-altered land cover formations from our assessment, including: Current and Historic Mining Activity, Developed and Urban, Introduced and Semi Natural Vegetation, Open Water, Pasture and Hay Field Crop, Recently Disturbed or Modified, Row and Close Grain Crop Cultural Formation, and Woody Horticultural Crop. We assessed which formations were located in protected areas and IRAs, and we used the percentage of each as a measure of ecological representation within the lower 48 United States.

2.4 | IRAs and ecosystem services

Potential importance of IRAs for drinking water supply was assessed by examining the percentage of IRAs that were also surface water and groundwater protection areas (henceforth referred to as drinking water protection areas or DWPAs) as delineated by the US Environmental Protection Agency (U.S. Environmental Protection Agency, 2017). A water protection area is defined as all US Geological Survey National Hydrography Dataset plus catchments located 1 day's water time of travel (24 hr) upstream from a surface water intake or groundwater facility point. Drinking water protection areas were calculated for all active systems and surface water facilities (i.e., intakes, reservoirs, infiltration galleries, and springs) and groundwater facilities (i.e., wells) with valid spatial locations as of July 2017. Percent DWPA area was calculated by dividing the 12-digit Hydrologic Unit Code (HUC12) area located within a DWPA by the total area of the HUC12. Exact locations of drinking water facilities or DWPAs are not included in the data set. To determine how the addition of IRAs to the protected area network would increase protection of important watersheds, we first performed an intersection of GAP 1 and 2 lands and all HUC12 watersheds that touch NFS lands. From these data, we calculated the percentage of each watershed that was covered by GAP 1 and 2 lands. The overall percentage of DWPA was then compared against the percentage of the watershed covered by GAP 1 and 2 lands to examine potential importance of these protected lands to drinking water supplies. In order to assess how adding IRAs to the protected area network could aid in preserving water quality in the lower 48, we then merged the IRA shapefile with the GAP 1 and 2 shapefile and repeated the steps above. Related tables for each HUC12 watershed detail the water providers served by the watershed. These water-provider data include the total population served as reported by the water-providers to the USEPA.

We quantified carbon capture of existing protected areas and IRAs using an estimate of net primary productivity (NPP, g C m^{-2} yr^{-1}) provided by the Numerical Terradynamic Simulation Group at the University of Montana. NPP data represent the average annual estimates from the years 2000 to 2012 at 1-km resolution (method describe in Zhao, Heinsch, Nemani, & Running, 2005). We converted the original raster units (g C m^{-2} yr^{-1}) to kg C km^{-2} yr^{-1} to stay consistent with the 1-km resolution. We then extracted and summed NPP data for protected areas and IRAs to quantify how much more annual carbon fixation IRAs might contribute to protected areas (reporting values as total Mg of carbon per year for all protected areas and IRAs). Protected areas and IRAs <1 km^2 were removed from this analysis to ensure NPP estimates were based on at least one full pixel value. Excluding these small units removed only 1.1% of the total area from protected cores and 0.01% area from IRAs. We then stratified the contiguous U.S. into three regions (west, interior west, and east) to evaluate how the contribution of IRAs to protected area carbon capture varies across these broad regions. The West included California, Oregon, and Washington. The Interior West included Idaho, Nevada, Utah, Arizona, Montana, Wyoming, Colorado, New Mexico, North and South Dakota, Nebraska, Kansas, Oklahoma, and Texas. The East included all other contiguous states. As an ancillary analysis, we calculated the average elevation of IRAs and protected areas.

3 | RESULTS

3.1 | IRAs and wildness

Currently, protected areas (GAP 1 and 2 lands excluding national fisheries and marine sanctuaries) encompass 602,901 km^2 of land within the contiguous U.S., representing 8% of land area, and IRAs account for another 2% (161,708 km^2) (Table 1). Despite their relatively small area, GAP 1 and 2 lands protect some of the wildest places in the contiguous U.S., including 24% (195,025 km^2) of the top 10% wildest lands in the contiguous U.S. IRAs, if added to the conservation network, would increase that amount to 31%. Furthermore, if all IRAs were added to the protected network, nearly

TABLE 1 The table below displays protected area (GAP 1 and 2) area and the amount and percent of IRA land directly adjacent to a protected area, as well as IRA area within 100 m, 5 km, and 10 km of a protected area

Description	Area (km^2)	Percentage of all IRA area	Percentage increase in PA area
Protected areas	602,901		
Adjacent IRAs	93,818	58%	15.6%
IRAs within 100m of a protected area	99,448	61%	16.5%
IRAs within 5km of a protected area	135,906	84%	22.5%
IRAs within 10km of a protected area	149,899	93%	24.9%
All IRAs	161,708	100%	26.8%

115,000 km^2 of the top 20% wildest places in the lower 48 would be added to conservation reserves (Figure 1).

Ninety-six percent of IRAs are wilder than the median value for the contiguous U.S. (i.e., the majority of IRAs are among the wildest half) (Supplemental S1 and S2). Additionally, 21% of IRAs are in the top 10% of wildest places in the lower 48, while 8% of IRAs are in the top 5%. In some states, IRAs considered relatively wild at the national level are overshadowed by the abundance of wild lands at the state level. For example, Idaho

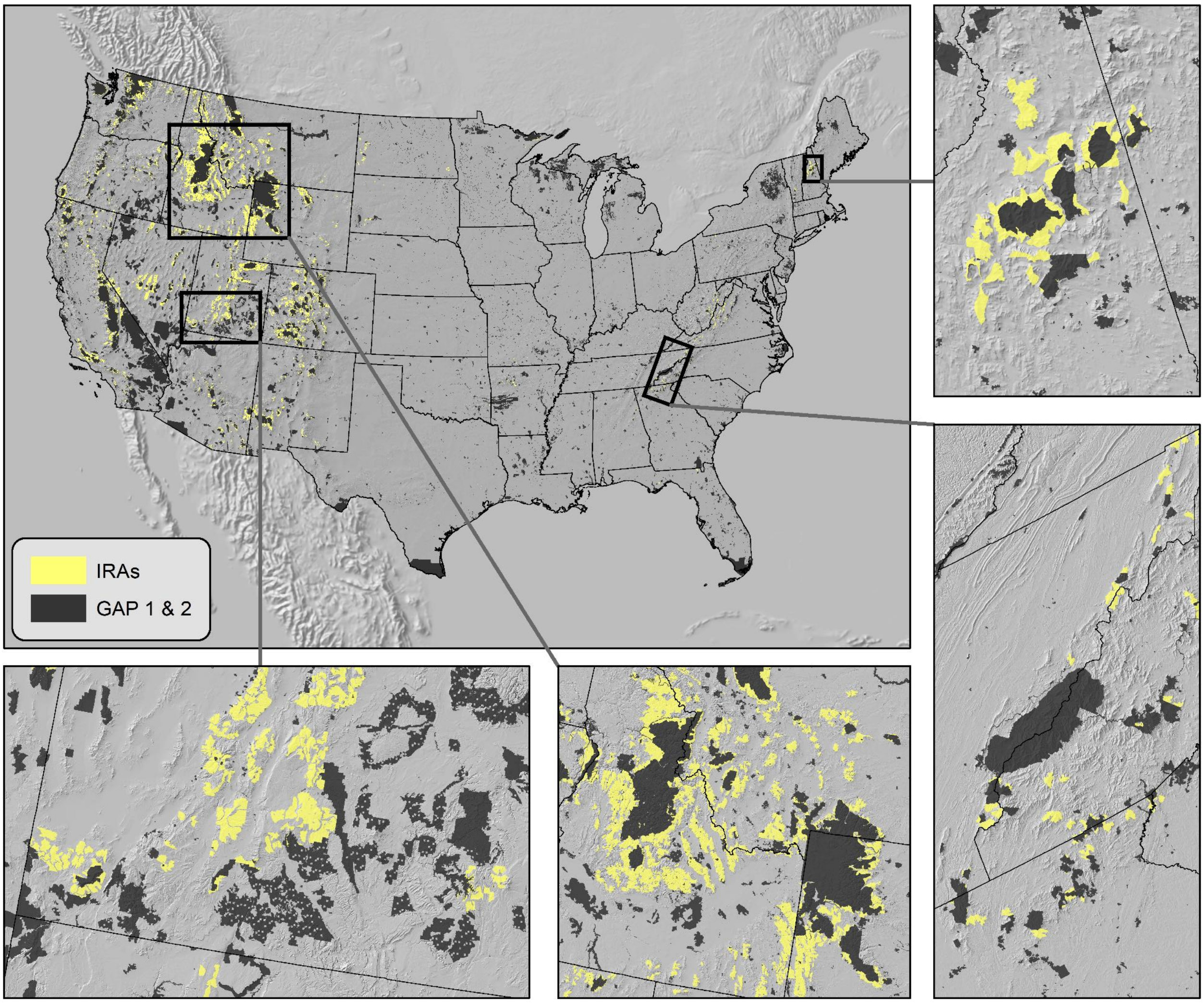

FIGURE 1 Map of protected areas (GAP 1 and 2 lands, black) and USDA Forest Service Inventoried Roadless Areas (yellow)

has 52 IRAs in the top 5% wildest places in the contiguous U.S. but not a single IRA in the top 10% wildest places in the state. Alternatively, Virginia has six IRAs in the top 5% wildest places in the state, but not a single IRA in the top 25% wildest places in the contiguous U.S. These patterns provide statewide and national context for the relative wildness of IRAs.

3.2 | IRAs and the size of protected areas

As described above, the protected area system of the contiguous U.S. currently includes 602,901 km^2. Incorporating IRAs in that system increases its total area by 27% to 764,609 km^2. Across the contiguous U.S., 1,127 IRAs are directly adjacent to a protected core, increasing the size of protected cores by 93,818 km^2 or 15.6% (Table 1). In fact, 6 of the top 10 largest protected cores have adjacent IRAs, which on average add 25% to their total area (Table 2). In some cases, very small conservation easements adjacent to IRAs could increase their size by three orders of magnitude relative to their size without including IRAs. In many cases, IRAs are not immediately adjacent to protected cores but add substantially to the distributed protected area system within a region (e.g., southern Utah and the Southern Appalachians in Figure 2). In all, 149,899 km^2 of IRAs (93% of IRA area) lie within 10 km of a protected core (Table 1).

While 93,818 km^2 of IRAs are directly adjacent to protected areas, an additional 5,630 km^2 are within 100-m of a protected area, and 14,091 km^2 more are within 100 m of those IRAs (Figure 3). If one were to continue building out from IRAs within 100 m of each other, 120,159 km^2 of IRA land would be connected to a protected area via a network of proximal units. With the inclusion of IRAs within this 100-m build-out system, the amount of land within the protected network would increase from 602,901 to 723,059 km^2. Based on the build-out steps using a 5-km threshold between edges, the amount of land in the protected system would increase to 758,039 km^2. Finally, at the 10-km threshold, the amount of protected land would increase to 761,815 km^2 (Figure 3).

TABLE 2 The 10 largest protected core areas determined after combining adjacent Gap 1 and 2 units with shared borders

Core area	Largest protected area	Size of the largest protected area (km^2)	Size of core area (km^2)	Size of adjacent IRAs (km^2)	Percent increase with IRAs included
Greater Yellowstone	Yellowstone National Park	8,900	24,040	6,878	29
Central Idaho	Frank Church-River of No Return Wilderness	9,579	17,633	6,767	38
Death Valley	Death Valley National Park	13,649	16,421	231	1
South Sierra	Yosemite National Park	3,028	13,605	2,337	17
Mojave Desert	Mojave Wilderness	2,813	12,839	0	NA
Lower Rio Grande[a]	Laguna Atascosa National Wildlife Refuge	183	12,725	0	NA
Crown of the Continent	Bob Marshall Wilderness	4,301	10,490	3,373	32
North Cascades	North Cascades National Park	2028	9,548	2,921	31
Desert National Wildlife Refuge	Desert National Wildlife Refuge	6,540	8,990	0	NA
Everglades	Everglades National Park	6,227	7,013	0	NA

Note: Here we name the core area, the largest protected area within these cores, the size of the largest protected area, the size of the entire core area, size of adjacent IRAs, and percent increase in area when IRAs are added to the cores. Six of these largest core areas included adjacent IRAs expanding the total area by an average of 25%. In these cases, IRAs likely serve as key buffers around core protected areas.

[a]The Protected Areas Database (PAD) includes a large GAP 2 "protected area" in the southern tip of Texas, which we include here for the sake of consistency with the PAD, but the area consists mostly of as-yet unprotected private land within the purchase area boundary of the Lower Rio Grande National Wildlife Refuge, which, at this time, is smaller than the nearby Laguna Atacosa NWR adjacent to the purchase area.

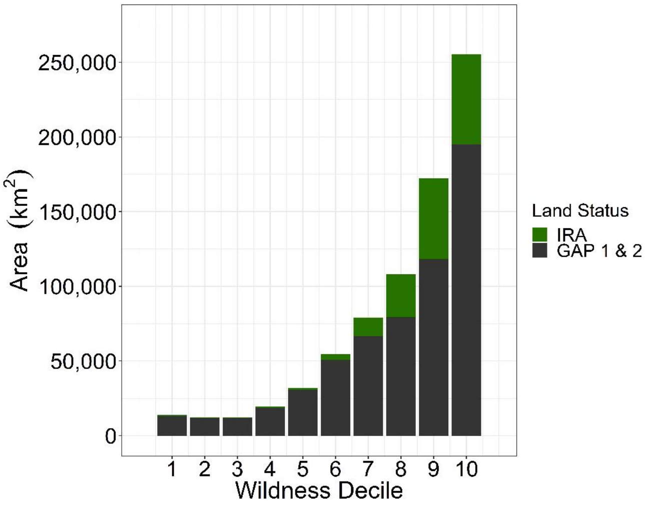

FIGURE 2 Comparison of wildness values within IRAs (green) and protected lands (grey). Values are binned by deciles of all wildness values in the lower 48, for example, the last bar represents GAP 1 and 2 and IRA lands in the top 10% wildest places in the lower 48

FIGURE 3 Cumulative area within the protected network with the inclusion of IRA stepping stones at 100-m, 5-, and 10-km increments. If IRAs within 10-km of a protected network were included in the protected network, as well as IRAs within 10-km of those IRAs and so on, the protected network would increase in size from 600,000 to over 750,000 km^2. See Table 2 for step descriptions

3.3 | IRAs and ecological representation

By far, the most common "formation" in the protected area system is Cool Temperate Forest and Woodland, representing 40.6% of GAP 1and 2 vegetation, followed by Cool Semi-Desert Scrub and Grassland (17.8%) and Warm Desert and Semi-Desert Scrub and Grassland (13.5%), which together account for over 70% of protected area vegetation (Table 3). The composition of IRAs is even more tilted toward Cool Temperate Forest and Woodland (67.0%) and Cool Semi-Desert Scrub and Grassland (13.0%).

The best represented vegetation formations in the protected area system are Mangrove (89.6% of all Mangrove is in GAP 1 and 2), Temperate and Boreal Alpine Tundra (70%), Tropical Dry Forest and Woodland (43.2%), Benthic Vascular Saltwater Vegetation (42.1%), and Temperate to Polar Scrub and Herb Coastal Vegetation (41.2%), but together these vegetation types make up less than 5% of the protected area system. On average, 25% (median = 17.9%) of the current extent of each formation is represented in the protected area system (Table 3). If all IRAs were added to the protected area system, the average representation of formations would increase to 27%. Formations with the greatest absolute increase in representation when including IRAs in the protected network are Temperate and Boreal Alpine Tundra (19.2%), Cool Temperate Forest and Woodland (5.8%), and Temperate and Boreal Cliff, Scree and Other Rock Vegetation (5.4%). Formations with the greatest percentage increase in representation when including IRAs in the protected network include Temperate Grassland and Shrubland (57.4%), Cool Temperate Forest and Woodland (52.2%), and Mediterranean Scrub and Grassland (35.5%) (Table 3).

3.4 | IRAs and ecosystem services

There are 17,598 HUC-12 watersheds that intersect National Forest System lands, of which 10,292, or 58%, have some fraction (>0%) of their area within a drinking water protection area, supplying water for over 48 million people. To examine the range of importance of these watersheds to protecting drinking water supply, the percentage of the watershed covered by GAP 1 and 2 lands and/or IRA lands is compared to the percentage of the watershed covered by a DWPA. This provides a rough estimate of overall importance. However, EPA data do not contain specific information on which part of the watershed is a DWPA. It is therefore possible that the area of watersheds protected in GAP 1 and 2 lands and DWPAs do not intersect, especially when the percent area of GAP 1 and 2 and DWPA is low. As the percentage of the watershed covered by both a DWPA and GAP 1 and 2 and/or IRA increases, the likelihood of them intersecting increases. To reduce the probability of non-intersection, we narrowed our analysis to only those protected watersheds with at least 50% of their area covered by GAP 1 and 2 and/or IRAs, and 50% of their area covered by a DWPA.

TABLE 3 Ecosystem representation of protected areas and IRAs, as well as representation levels currently in conservation reserves and their potential levels if all IRAs were added to the protection network

National Vegetation Class—formation and land use class	GAP 1 and 2 area (km^2)	IRA area (km^2)	Combined area (km^2)	Current representation	Potential representation
Barren	1,006	23	1,049	12.2%	12.5%
Benthic vascular saltwater vegetation	83	0	84	42.1%	42.1%
Boreal Flooded and Swamp Forest	6,031	38	6,189	19.5%	19.6%
Boreal Forest and Woodland	5,879	140	6,136	23.2%	23.8%
Cool Semi-Desert Scrub and Grassland	92,245	20,421	114,511	10.5%	12.9%
Cool Temperate Forest and Woodland	209,958	105,443	319,600	11.5%	17.3%
Mangrove	2,319	0	2,366	89.6%	89.6%
Mediterranean scrub and grassland	6,946	2,326	9,410	11.9%	15.8%
Salt marsh	13,500	35	13,805	15.5%	15.5%
Temperate and Boreal Alpine Tundra	19,459	5,324	25,172	70.0%	89.2%
Temperate and Boreal Cliff, Scree and Other Rock Vegetation	3,814	971	4,861	21.1%	26.5%
Temperate and Boreal Freshwater Aquatic Vegetation	0	0	0	4.9%	4.9%
Temperate Flooded and Swamp Forest	30,268	1,005	31,879	10.1%	10.4%
Temperate Grassland and Shrubland	24,447	13,548	38,484	3.0%	4.7%
Temperate to Polar bog and fen	1,507	18	1,555	28.1%	28.4%
Temperate to polar freshwater marsh, wet Meadow and Shrubland	7,969	1,499	9,628	15.5%	18.4%
Temperate to Polar Scrub and Herb Coastal Vegetation	794	33	843	41.2%	42.9%
Tropical dry Forest and Woodland	13	0	14	43.2%	43.2%
Tropical Flooded and Swamp Forest	337	0	344	17.9%	17.9%
Tropical freshwater marsh, wet Meadow and Shrubland	2,404	0	2,453	31.3%	31.3%
Tropical scrub and herb coastal vegetation	12	0	13	35.0%	35.0%
Warm Desert and Semi-Desert Scrub and grassland	69,648	1,382	72,423	15.7%	16.0%
Warm Temperate Forest and Woodland	18,562	5,225	24,158	4.9%	6.3%

Note: Classifications are from the National Vegetation Classification—formation or land use class.

Figure 4 displays a range of values for the intersection of protected lands, protected lands with the addition of IRAs, and DWPAs. At its broadest, 489 watersheds are at least 50% covered by a drinking water protection area and at least 50% covered by GAP 1 and 2 lands. If all IRAs were added to the protected area network, the number of watersheds within this class would increase to 780, extending protection to 291 watersheds. At the highest level of protection, there are 145 watersheds that are at least 90% covered by a DWPA and at least 90% covered by GAP 1 and 2 lands. If all IRAs were added to conservation reserves, the number of

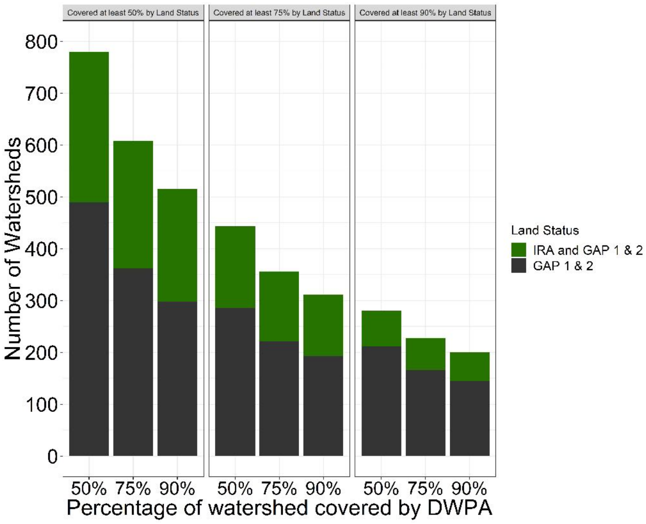

FIGURE 4 HUC-12 watersheds with varying degrees of Inventoried Roadless Area and/or GAP 1 and 2 protection and drinking water protection areas (DWPA).The three panels represent watersheds that are at least 50% (left), 75% (middle), or 90% (right) covered by IRAs and/or GAP 1 and 2 lands, and each bar represents a varying degree of drinking water protection cover (e.g., first bar represents watersheds that are at least 50% in a DWPA)

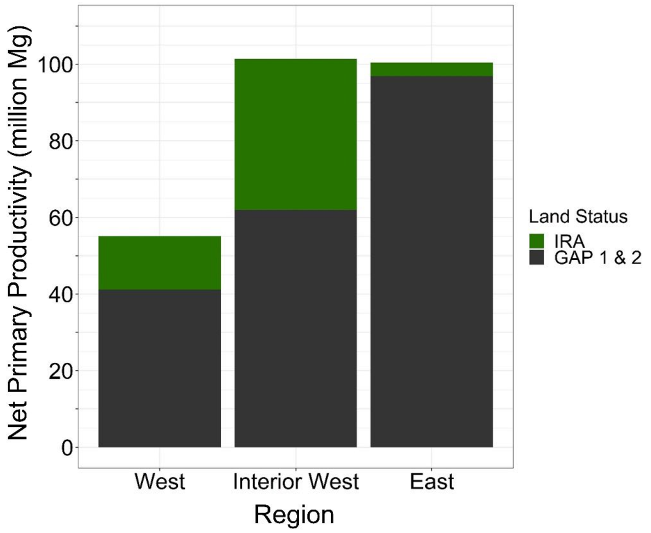

FIGURE 5 Total carbon captured (estimated using net primary productivity) in protected areas (GAP 1 and 2 lands) as well as within Inventoried Roadless Areas in three regions of the contiguous US. Only Gap 1 and 2 lands and IRAs larger than 1 km^2 were included in this analysis

watersheds within this high level of protection would increase to 200, extending protection to 55 additional watersheds.

Across the contiguous US, the existing protected area system captures 199,978,833 megagrams of carbon per year, and IRAs add 29% more total NPP to that captured by the existing system. Within the West, Interior West, and East regions, IRAs contribute 34.0, 63.6, and 3.7% additional NPP to the existing protected areas, despite increasing the area protected by only 19.2, 46.9, and 3.1%, respectively (Figure 5, Supplemental S3). In the West, IRAs capture 13,989,603 megagrams of carbon per year, 39,404,180 megagrams in the Interior West, and 3,607,583 megagrams in the East (Supplemental S3). It is important to note that NPP does not represent carbon sequestration; it represents only the rate at which carbon is fixed by plants (photosynthesis minus live plant respiration) and does not account for losses from decomposition, fire, and so forth. (Lovett, Cole, & Pace, 2006).

4 | DISCUSSION

Our results suggest that IRAs serve important roles in land conservation within the contiguous United States. IRAs maintain some of the wildest places in the country and within states where they occur. IRAs also reduce the isolation of—and provide buffers for—national parks, wilderness areas, and other existing protected lands. Furthermore, some IRAs contain important watersheds for drinking water throughout the US and capture significant amounts of carbon each year relative to existing protected areas. Together, our findings suggest that IRAs are key components of the U.S. system of conservation lands.

Protecting wild lands has been essential to conservation since its inception, and recent research confirms its effectiveness (DiMarco, Ferrier, Harwood, Hoskins, & Watson, 2019). Wild places have more intact ecological processes with fewer local extinctions (DiMarco et al., 2019; Watson et al., 2018). Wild protected areas are therefore more likely to be able to sustain their biodiversity into the future (Aplet & McKinley, 2017; Belote et al., 2017). At the same time, wildlands around the world are disappearing at an alarming rate (Watson, Darling, et al., 2016). Theobald et al. (2019) report that between 2001 and 2017, the U.S. lost almost one hectare of natural area every minute. Our results show that IRAs house some of the wildest places in the U.S. as well as some of the last wild places in states that have been mostly developed (Figure 1; Supplemental S2). These results compare well with Aplet et al. (2005), who used a different metric of wildness to find that the vast majority of the wildest land in the U.S. occurs on federal land in the West and that on the national forests, almost as much of the wildest land occurs in IRAs as in wilderness.

In addition to urging protection of the wildest places, conservation scientists continue to call for increasing the size of protected areas and protected area networks

(Belote et al., 2017; Cantú-Salazar & Gaston, 2010; Gaston et al., 2008). For instance, the Aichi Targets of the Convention on Biological Diversity recommends increasing the proportion of land protected to 17% (CBD, 2014), and recent calls have been made to increase protected areas to cover as much as 50% of land (Pimm, Jenkins, & Li, 2018). It is generally agreed that larger protected areas can contain more species (Cantú-Salazar & Gaston, 2010; Gaston et al., 2008) and more intact ecological processes (Cantú-Salazar & Gaston, 2010), and existing protected areas may be too small to adequately achieve their intended purpose (Hansen et al., 2011). Many conservation scientists have recommended that core protected areas include a "buffer" of adjacent lands that can increase the effective size of reserves (Belote & Wilson, 2020; Hansen et al., 2014; Martino, 2001; Shafer, 1999; Wade & Theobald, 2010). We found that many large protected areas are bordered by IRAs, and these IRAs add to the overall area protected from intensive commercial development and other human land use pressures. With nearly 60% of all IRA area sharing a border with protected areas, and more than 90% of all IRA area within 10-km of one, IRAs increase the effective area of these protected lands. In fact, we have shown that 74% of all IRA land in the lower 48 is connected to a protected area via a close (< 100 m) complex of conservation units composed of IRAs and protected areas (Figure 3). If this distance were extended to 10 km, 98% of all IRA area in the lower 48 would be included in protected area-IRA complexes, increasing the size of the protected network by nearly 159,000 km^2 (a 26% increase in area). The function of stepping-stones will depend on the condition of interstitial lands between IRAs. In some cases, restoration of degraded lands may be required to provide opportunities for establishing large functional protected areas and roadless complexes.

IRAs enhance the size of core protected areas and buffer them from anticipated development (see Hansen et al., 2014; Martinuzzi et al., 2015). For instance, the protected core centered on Yellowstone National Park increases by 29% when considering the surrounding 6,878 km^2 of IRAs. The Greater Yellowstone Ecosystem is expected to experience increased human pressures from residential development, recreation, and land use (Hansen & Phillips, 2018). Other protected areas that are surrounded by IRAs may benefit as well based on the relative increase in effective size. For example, the Great Gulf Wilderness in New Hampshire is only surrounded by 60 km^2 of IRA land, but this represents a nearly 265% increase in the size of this protected area. In Montana, a 5 km^2 segment of the Blackfoot-Clearwater Wildlife Management Area shares a border with a 3,346 km^2 IRA on the southern end of the Bob Marshall Wilderness Complex. The connection among core protected areas and IRAs likely provides for continuous unfragmented ecological processes across these units. The role IRAs play in buffering protected areas from development may be even more critical in the future as developed areas continue to expand (Sohl et al., 2014).

In addition to size, the connectedness of protected areas has been identified as a critical consideration for sustaining biodiversity (Belote et al., 2016; Saura et al., 2018). Protected areas—if not intentionally connected—could become isolated, resulting in loss of genetic diversity of populations and ultimately loss of species (Gaston et al., 2008; Rayfield, Fortin, & Fall, 2011). In some cases, stepping-stones between protected areas can help connect large protected areas (Belote et al., 2016; Hannah et al., 2014). Stepping-stones function by reducing the distance between protected areas and providing temporary refuge for organisms moving across the "matrix" between core protected areas. Belote et al. (2016) showed that IRAs may serve as important stepping stones between Yellowstone National Park and the Bob Marshall Wilderness-Glacier National Park protected areas (see Supplemental Figure S5 in Belote et al., 2016). While land between IRAs and protected areas may include roads and be susceptible to human development, recognizing the value of IRAs as part of complexes of core protected areas should be an important consideration of conservation planners working to maintain large blocks of intact and wild lands and enhance the size of the national system of protected areas.

While IRAs could serve as buffers around protected areas, reduce their isolation, and increase their connectivity, adding them to the system would not dramatically increase the representation of ecosystem types (i.e., vegetation formations). Of the 23 formations currently in the protected area system, 10 are already above the 17% Aichi threshold set by the Convention on Biological Diversity in 2010 (CBD, 2014), and only two (Cool Temperate Forest and Woodland and Temperate to Polar Freshwater Marsh, Wet Meadow and Shrubland) would move above 17% if IRAs were included in the protected area system. IRAs would not contribute any new formations to the system. Using a classification system consisting of 83 ecoregions, DeVelice and Martin (2001) found a similarly modest increase in the number of ecoregions achieving a 12% threshold (the informal scientific standard before 2010) if roadless areas were added to the nationwide protected area system. In studies conducted at a regional scale, Strittholt and DellaSala (2001) found that roadless areas in the Klamath-Siskiyou ecoregion added 96 new types to the 42 (out of a possible 214) types already protected above a 25% threshold, and Crist, Wilmer, and Aplet (2005) found that protecting the roadless areas of the northern Rocky Mountain states would add six vegetation types to the 12 (out of 29) that exceeded a 12% threshold and add a new type, Bur oak woodland, to the protected area system. All three of these studies found IRAs to occupy generally lower elevations than the existing protected area system, accounting for the increase in representation. In contrast, we found IRAs to occur at higher average elevation (Supplemental S3), likely because our analysis included all GAP 1 and 2 lands, not just those on national forests.

In order to better understand the value of national forest IRAs, we also examined the role IRAs play in delivering ecosystem services, a critical rationale for conserving natural areas (Balmford et al., 2002). Watershed delivery of drinking water (Keeler et al., 2012) is among the most critical ecosystem services for mitigating climate change and providing for human well-being (Costanza et al., 1997). Road-building and roads can impair watershed condition (Potyondy & Geier, 2011), and IRAs may play a key role in protecting watersheds important for delivering drinking water (DellaSala, Karr, & Olson, 2011). IRAs add significantly to protection of watersheds important for delivering drinking water to people, increasing the number of watersheds dominated by DWPAs that are also dominated by protected areas by as much as 60%. Our intention in this analysis was to quantify the contribution of IRAs to watershed protection, as these values were highlighted in the 2001 Roadless Rule. The Forest Service's analysis of the Rule acknowledged that "[r]oads have long been recognized as the primary human-caused source of soil and water disturbances in forested environments" (USDA Forest Service, 2000, pp. 3–44). Roads cause multiple negative impacts on water quality and hydrology (Gucinski, Furniss, Ziemer, & Brookes, 2001) and can continue to disrupt hydrology for years, even after closure (Sosa-Pérez & MacDonald, 2017). By restricting road building in IRAs, the Roadless Rule helps protect water quality for watersheds upon which people depend (DellaSala et al., 2011).

In addition to water delivery, the ability of ecosystems to capture atmospheric carbon through photosynthesis and vegetation productivity is increasingly recognized as a critical ecosystem service (Dinerstein et al., 2019). The carbon captured and stored by ecosystems is a key service that mitigates the would-be impacts of human-induced climate change (Naidoo et al., 2008), and protecting carbon-capturing ecosystems from development has emerged as a key conservation strategy to aid climate change mitigation (Dinerstein et al., 2019). Our results show that IRAs add disproportionately, relative to the area added, to the carbon captured by existing protected areas. Of course, the type of management and strategies for best maintaining an ecosystem's ability to capture and store carbon will vary by disturbance history and context (Kashian, Romme, Tinker, Turner, & Ryan, 2006; Law et al., 2018). Further work is needed to understand the co-benefits and tradeoffs between carbon storage and management aimed at addressing fire mitigation or timber harvest (Buotte, Law, Ripple, & Berner, 2020; Johnston & Radeloff, 2019; Onaindia, Fernández de Manuel, Madariaga, & Rodríguez-Loinaz, 2013).

IRAs may be good candidates to consider as "other effective area-based conservation measures" (OECMs) when assessing national and international conservation targets. Our work responds to calls to identify and assess OECM conservation value (IUCN-WCPA, 2019). However, while the Roadless Rule currently affords protections from road building and commercial timber harvests, IRAs remain vulnerable to conservation demotion unless protected by law.

We note that while our assessment did not include Alaska due to lack of some data sets we used in the contiguous US (e.g., human modification, our index of wildness), IRAs located on the Tongass National Forest may be similarly valuable and yet threatened by removal of the protections afforded by the Roadless Rule. Before such changes are made to demote IRAs' status on the Tongass or other national forests, an assessment of their conservation value should be conducted to more fully understand the roles IRAs play in maintaining ecological values.

5 | CONCLUSIONS

Even as conservation scientists call for additional area to be conserved, other researchers have identified the potential threat posed by downgrading, downsizing, and degazettement of protected areas (Golden Kroner et al., 2019). IRAs are currently administered under the Roadless Rule and are not formally protected by legislation. Since 2001, many wilderness areas have been designated from existing IRAs. In fact, 74% of all wilderness areas designated on US Forest Service land since 2000 were first IRAs. However, lacking legislative protection, IRAs may also be candidates for downgrading by the U.S. Congress or administrative rule changes. Such downgrading could open these important conservation lands to commercial resource extraction, road building, or other damaging activities. Given their conservation potential and uncertain status, it is critical that their values be assessed before any decision is made about downgrading. Because of their special qualities, maintaining roadless areas has been a goal of conservation for at least 80 years (Marshall & Dobbins, 1936), and our results reveal their continuing conservation value. Through their effect on protected area quality, size, and connectivity, and their influence on valued ecosystem services, national forest IRAs make a valuable addition to the nation's protected area system.

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CONFLICT OF INTEREST

The authors declare no conflict of interest.

AUTHOR CONTRIBUTIONS

Gregory H. Aplet and R. Travis Belote conceived of the study. McKinley J. Talty and Kelly Mott Lacroix conducted analyses. McKinley J. Talty, Kelly Mott Lacroix, Gregory H. Aplet, and R. Travis Belote wrote the paper.

DATA AVAILABILITY STATEMENT

All data are publicly available.

ETHICS STATEMENT

No ethics approval was required for this research.

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of this article.

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Appendix O

Transportation Infrastructure and Access on National Forests and Grasslands
A Literature Review
May 2014

Introduction

The Forest Service transportation system is very large with 374,883 miles (603,316 km) of system roads and 143,346 miles (230,693 km) of system trails. The system extends broadly across every national forest and grasslands and through a variety of habitats, ecosystems and terrains. An impressive body of scientific literature exists addressing the various effects of roads on the physical, biological and cultural environment – so much so, in the last few decades a new field of "road ecology" has emerged. In recent years, the scientific literature has expanded to address the effects of roads on climate change adaptation and conversely the effects of climate change on roads, as well as the effects of restoring lands occupied by roads on the physical, biological and cultural environments.

The following literature review summarizes the most recent thinking related to the environmental impacts of forest roads and motorized routes and ways to address them. The literature review is divided into three sections that address the environmental effects of transportation infrastructure on forests, climate change and infrastructure, and creating sustainable forest transportation systems.

I. Impacts of Transportation Infrastructure and Access to the Ecological Integrity of Terrestrial and Aquatic Ecosystems and Watersheds
II. Climate Change and Transportation Infrastructure Including the Value of Roadless Areas for Climate Change Adaptation
III. Sustainable Transportation Management in National Forests as Part of Ecological Restoration

I. Impacts of Transportation Infrastructure and Access to the Ecological Integrity of Terrestrial and Aquatic Ecosystems and Watersheds

It is well understood that transportation infrastructure and access management impact aquatic and terrestrial environments at multiple scales, and, in general, the more roads and motorized routes the greater the impact. In fact, in the past 20 years or so, scientists having realized the magnitude and breadth of ecological issues related to roads; entire books have been written on the topic, e.g., Forman et al. (2003), and a new scientific field called "road ecology" has emerged. Road ecology research centers have been created including the Western

Transportation Institute at Montana State University and the Road Ecology Center at the University of California - Davis.[1]

Below, we provide a summary of the current understanding on the impacts of roads and access allowed by road networks to terrestrial and aquatic ecosystems, drawing heavily on Gucinski et al. (2000). Other notable recent peer-reviewed literature reviews on roads include Trombulak and Frissell (2000), Switalski et al. (2004), Coffin (2007), Fahrig and Rytwinski (2009), and Robinson et al. (2010). Recent reviews on the impact of motorized recreation include Joslin and Youmans (1999), Gaines et al. (2003), Davenport and Switalski (2006), Ouren et al. (2007), and Switalski and Jones (2012). These peer-reviewed summaries provide additional information to help managers develop more sustainable transportation systems

Impact on geomorphology and hydrology

The construction or presence of forest roads can dramatically change the hydrology and geomorphology of a forest system leading to reductions in the quantity and quality of aquatic habitat. While there are several mechanisms that cause these impacts (Wemple et al. 2001 , Figure 1), most fundamentally, compacted roadbeds reduce rainfall infiltration, intercepting and concentrating water, and providing a ready source of sediment for transport (Wemple et al. 1996, Wemple et al. 2001). In fact, roads contribute more sediment to streams than any other land management activity (Gucinski et al. 2000). Surface erosion rates from roads are typically at least an order of magnitude greater than rates from harvested areas, and three orders of magnitude greater than erosion rates from undisturbed forest soils (Endicott 2008).

[1] See http://www.westerntransportationinstitute.org/research/roadecology and http://roadecology.ucdavis.edu/

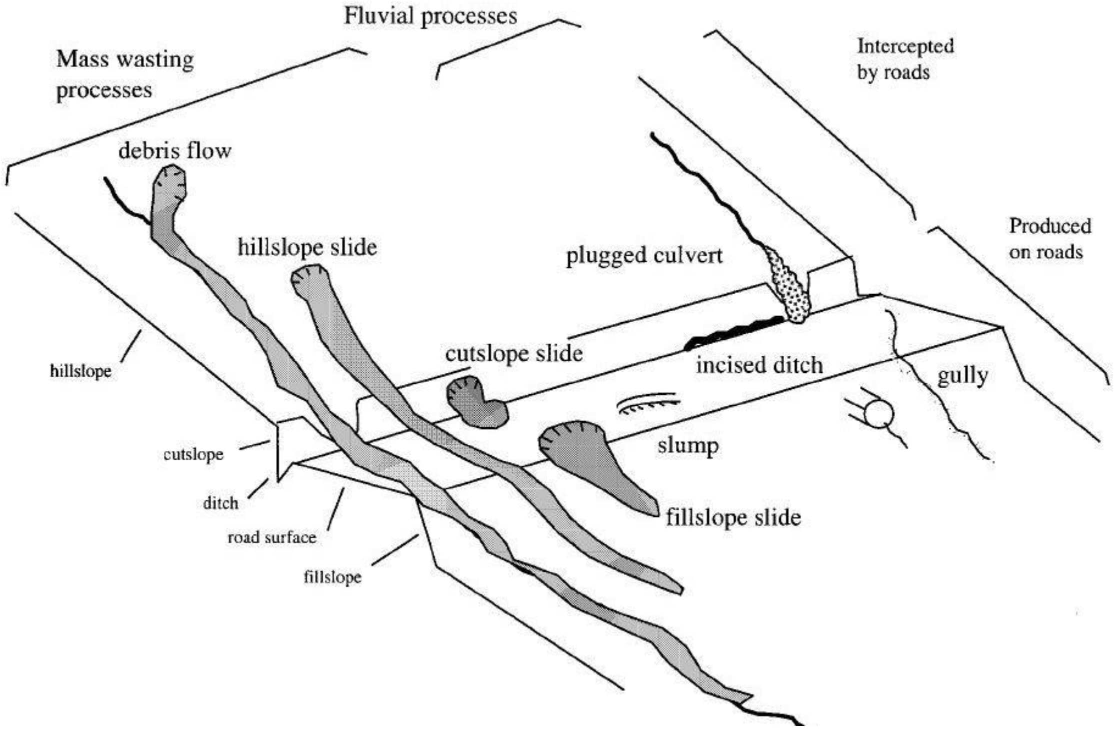

Figure 1: Typology of erosional and depositional features produced by mass-wasting and fluvial processes associate with forest roads (reprinted from Wemple et al. 2001)

Erosion of sediment from roads occurs both chronically and catastrophically. Every time it rains, sediment from the road surface and from cut- and fill-slopes is picked up by rainwater that flows into and on roads (fluvial erosion). The sediment that is entrained in surface flows are often concentrated into road ditches and culverts and directed into streams. The degree of fluvial erosion varies by geology and geography, and increases with increased motorized use (Robichaud et al. 2010). Closed roads produce less sediment, and Foltz et al. (2009) found a significant increase in erosion when closed roads were opened and driven upon.

Roads also precipitate catastrophic failures of road beds and fills (mass wasting) during large storm events leading to massive slugs of sediment moving into waterways (Endicott 2008; Gucinski et al. 2000). This typically occurs when culverts are undersized and cannot handle the volume of water, or they simply become plugged with debris. The saturated roadbed can fail entirely and result in a landslide, or the blocked stream crossing can erode the entire fill down to the original stream channel.

The erosion of road- and trail-related sediment and its subsequent movement into stream systems affects the geomorphology of the drainage system in a number of ways. The magnitude of their effects varies by climate, geology, road age, construction / maintenance practices and storm history. It directly alters channel morphology by embedding larger gravels as well as filling pools. It can also have the opposite effect of increasing peak discharges and scouring channels, which can lead to disconnection of the channel and floodplain, and lowered base flows (Furniss et al. 1991; Joslin and Youmans 1999). The width/depth ratio of the stream changes which then can trigger changes in water temperature, sinuosity and other geomorphic factors important for aquatic species survival (Joslin and Youmans 1999; Trombulak and Frissell 2000).

Roads also can modify flowpaths in the larger drainage network. Roads intercept subsurface flow as well as concentrate surface flow, which results in new flowpaths that otherwise would not exist, and the extension of the drainage network into previously unchannelized portions of the hillslope (Gucinski et al. 2000; Joslin and Youmans 1999). Severe aggradation of sediment at stream structures or confluences can force streams to actually go subsurface or make them too shallow for fish passage (Endicott 2008; Furniss et al. 1991).

Impacts on aquatic habitat and fish

Roads can have dramatic and lasting impacts on fish and aquatic habitat. Increased sedimentation in stream beds has been linked to decreased fry emergence, decreased juvenile densities, loss of winter carrying capacity, and increased predation of fishes, and reductions in macro-invertebrate populations that are a food source to many fish species (Rhodes et al. 1994, Joslin and Youmans 1999, Gucinski et al. 2000, Endicott 2008). On a landscape scale, these effects can add up to: changes in the frequency, timing and magnitude of disturbance to aquatic habitat and changes to aquatic habitat structures (e.g., pools, riffles, spawning gravels and in-channel debris), and conditions (food sources, refugi, and water temperature) (Gucinski et al. 2000).

Roads can also act as barriers to migration (Gucinski et al. 2000). Where roads cross streams, road engineers usually place culverts or bridges. Culverts in particular can and often interfere with sediment transport and channel processes such that the road/stream crossing becomes a barrier for fish and aquatic species movement up and down stream. For instance, a culvert may scour on the downstream side of the crossing, actually forming a waterfall up which fish cannot move. Undersized culverts and bridges can infringe upon the channel or floodplain and trap sediment causing the stream to become too shallow and/or warm such that fish will not migrate past the structure. This is problematic for many aquatic species but especially for anadromous species that must migrate upstream to spawn. Well-known native aquatic species affected by roads include salmon such as coho (*Oncorhynchus kisutch*), chinook (*O. tshawytscha*), and chum (*O. keta*); steelhead (*O. mykiss*); and a variety of trout species including bull trout (*Salvelinus confluentus*) and cutthroat trout (*O. clarki*), as well as other native fishes and amphibians (Endicott 2008).

Impacts on terrestrial habitat and wildlife

Roads and trails impact wildlife through a number of mechanisms including: direct mortality (poaching, hunting/trapping) changes in movement and habitat use patterns (disturbance/avoidance), as well as indirect impacts including alteration of the adjacent habitat and interference with predatory/prey relationships (Wisdom et al. 2000, Trombulak and Frissell 2000). Some of these impacts result from the road itself, and some result from the uses on and around the roads (access). Ultimately, roads have been found to reduce the abundance and distribution of several forest species (Fayrig and Ritwinski 2009, Benítez-López et al. 2010).

Table 1: Road- and recreation trail-associated factors for wide-ranging carnivores (Reprinted from Gaines et al. (2003)[2]

[2] For a list of citations see Gaines et al. (2003)

Focal species	Road-associated factors	Motorized trail-associated factors	Nonmotorized trail-associated factors
Grizzly bear	Poaching Collisions Negative human interactions Displacement or avoidance	Poaching Negative human interactions Displacement or avoidance	Poaching Negative human interactions Displacement or avoidance
Lynx	Down log reduction Trapping Collisions Disturbance at a specific site	Disturbance at a specific site Trapping	Disturbance at a specific site
Gray wolf	Trapping Poaching Collisions Negative human interactions Disturbance at a specific site Displacement or avoidance	Trapping Disturbance at a specific site	Trapping Disturbance at a specific site
Wolverine	Down log reduction Trapping Disturbance at a specific site Collisions	Trapping Disturbance at a specific site	Trapping Disturbance at a specific site

Direct mortality and disturbance from road and trail use impacts many different types of species. For example, wide-ranging carnivores can be significantly impacted by a number of factors including trapping, poaching, collisions, negative human interactions, disturbance and displacement (Gaines et al. 2003, Table 1). Hunted game species such as elk (*Cervus canadensis*), become more vulnerable from access allowed by roads and motorized trails resulting in a reduction in effective habitat among other impacts (Rowland et al. 2005, Switalski and Jones 2012). Slow-moving migratory animals such as amphibians, and reptiles who use roads to regulate temperature are also vulnerable (Gucinski et al. 2000, Brehme et al. 2013).

Habitat alteration is a significant consequence of roads as well. At the landscape scale, roads fragment habitat blocks into smaller patches that may not be able to support successfully interior forest species. Smaller habitat patches also results in diminished genetic variability, increased inbreeding, and at times local extinctions (Gucinski et al. 2000; Trombulak and Frissell 2000). Roads also change the composition and structure of ecosystems along buffer zones, called edge-affected zones. The width of edge-affected zones varies by what metric is being discussed; however, researchers have documented road-avoidance zones a kilometer or more away from a road (Table 2). In heavily roaded landscapes, edge-affected acres can be a significant fraction of total acres. For example, in a landscape area where the road density is 3 mi/mi^2 (not an uncommon road density in national forests) and where the edge-affected zone is estimated to be 500 ft from the center of the road to each side, the edge-affected zone is 56% of the total acreage.

Table 2: A summary of some documented road-avoidance zones for various species (adapted from Robinson et al. 2010).

Species	Avoidance zone m (ft)	Type of disturbance	Reference
Snakes	650 (2133)	Forestry roads	Bowles (1997)
Salamander	35 (115)	Narrow forestry road, light traffic	Semlitsch (2003)
Woodland birds	150 (492)	Unpaved roads	Ortega and Capen (2002)
Spotted owl	400 (1312)	Forestry roads, light traffic	Wasser et al. (1997)
Marten	<100 (<328)	Any forest opening	Hargis et al. (1999)
Elk	500–1000 (1640-3281)	Logging roads, light traffic	Edge and Marcum (1985)
	100–300 (328-984)	Mountain roads depending on traffic volume	Rost and Bailey (1979)
Grizzly bear	3000 (9840)	Fall	Mattson et al. (1996)
	500 (1640)	Spring and summer	
	883 (2897)	Heavily traveled trail	Kasworm and Manley (1990)
	274 (899)	Lightly traveled trail	
	1122 (3681)	Open road	Kasworm and Manley (1990)
	665 (2182)	Closed road	
Black bear	274 (899)	Spring, unpaved roads	Kasworm and Manley (1990)
	914 (2999)	Fall, unpaved roads	

Roads and trails also affect ecosystems and habitats because they are also a major vector of non-native plant and animal species. This can have significant ecological and economic impacts when the invading species are aggressive and can overwhelm or significantly alter native species and systems. In addition, roads can increase harassment, poaching and collisions with vehicles, all of which lead to stress or mortality (Wisdom et al. 2000).

Recent reviews have synthesized the impacts of roads on animal abundance and distribution. Fahrig and Rytwinski (2009) did a complete review of the empirical literature on effects of roads and traffic on animal abundance and distribution looking at 79 studies that addressed 131 species and 30 species groups. They found that the number of documented negative effects of roads on animal abundance outnumbered the number of positive effects by a factor of 5. Amphibians, reptiles, most birds tended to show negative effects. Small mammals generally showed either positive effects or no effect, mid-sized mammals showed either negative effects or no effect, and large mammals showed predominantly negative effects. Benítez-López et al. (2010) conducted a meta-analysis on the effects of roads and infrastructure proximity on mammal and bird populations. They found a significant pattern of avoidance and a reduction in bird and mammal populations in the vicinity of infrastructure.

Road density[3] thresholds for fish and wildlife

[3] We intend the term "road density" to refer to the density all roads within national forests, including system roads, closed roads, non-system roads administered by other jurisdictions (private, county, state), temporary roads and motorized trails. Please see Attachment 2 for the relevant existing scientific information supporting this approach.

It is well documented that beyond specific road density thresholds, certain species will be negatively affected, and some will be extirpated. Most studies that look into the relationship between road density and wildlife focus on the impacts to large endangered carnivores or hunted game species, although high road densities certainly affect other species – for instance, reptiles and amphibians. Gray wolves (*Canis lupus*) in the Great Lakes region and elk in Montana and Idaho have undergone the most long-term and in depth analysis. Forman and Hersperger (1996) found that in order to maintain a naturally functioning landscape with sustained populations of large mammals, road density must be below 0.6 km/km^2 (1.0 mi/mi^2). Several studies have since substantiated their claim (Robinson et al. 2010, Table 3).

A number of studies at broad scales have also shown that higher road densities generally lead to greater impacts to aquatic habitats and fish density (Table 3). Carnefix and Frissell (2009) provide a concise review of studies that correlate cold water fish abundance and road density, and from the cited evidence concluded that "1) no truly "safe" threshold road density exists, but rather negative impacts begin to accrue and be expressed with incursion of the very first road segment; and 2) highly significant impacts (e.g., threat of extirpation of sensitive species) are already apparent at road densities on the order of 0.6 km/km^2 (1.0 mi/mi^2) or less" (p. 1).

Table 3: A summary of some road-density thresholds and correlations for terrestrial and aquatic species and ecosystems (reprinted from Robinson et al. 2010).

Species (Location)	**Road density (mean, guideline, threshold, correlation)**	**Reference**
Wolf (Minnesota)	0.36 km/km2 (mean road density in primary range); 0.54 km/km^2 (mean road density in peripheral range)	Mech et al. (1988)
Wolf	>0.6 km/km^2 (absent at this density)	Jalkotzy et al. (1997)
Wolf (Northern Great Lakes region)	>0.45 km/km^2 (few packs exist above this threshold); >1.0 km/km^2 (no pack exist above this threshold)	Mladenoff et al. (1995)
Wolf (Wisconsin)	0.63 km/km^2 (increasing due to greater human tolerance	Wydeven et al. (2001)
Wolf, mountain lion (Minnesota, Wisconsin, Michigan)	0.6 km/km^2 (apparent threshold value for a naturally functioning landscape containing sustained populations)	Thiel (1985); van Dyke et al. (1986); Jensen et al. (1986); Mech et al. (1988); Mech (1989)
Elk (Idaho)	1.9 km/km^2 (density standard for habitat effectiveness)	Woodley 2000 cited in Beazley et al. 2004
Elk (Northern US)	1.24 km/km^2 (habitat effectiveness decline by at least 50%)	Lyon (1983)
Elk, bear, wolverine, lynx, and others	0.63 km/km^2 (reduced habitat security and increased mortality)	Wisdom et al. (2000)
Moose (Ontario)	0.2-0.4 km/km2 (threshold for pronounced response)	Beyer et al. (2013)
Grizzly bear (Montana)	>0.6 km/km^2	Mace et al. (1996); Mattson et al. (1996)
Black bear (North Carolina)	>1.25 km/km^2 (open roads); >0.5 km/km2 (logging roads); (interference with use of habitat)	Brody and Pelton (1989)
Black bear	0.25 km/km^2 (road density should not exceed)	Jalkotzy et al. (1997)
Bobcat (Wisconsin)	1.5 km/km^2 (density of all road types in home range)	Jalkotzy et al. (1997)

Large mammals	>0.6 km/km^2 (apparent threshold value for a naturally functioning landscape containing sustained populations)	Forman and Hersperger (1996)
Bull trout (Montana)	Inverse relationship of population and road density	Rieman et al. (1997); Baxter et al. (1999)
Fish populations (Medicine Bow National Forest)	(1) Positive correlation of numbers of culverts and stream crossings and amount of fine sediment in stream channels (2) Negative correlation of fish density and numbers of culverts	Eaglin and Hubert (1993) cited in Gucinski et al. (2001)
Macroinvertebrates	Species richness negatively correlated with an index of road density	McGurk and Fong (1995)
Non-anadromous salmonids (Upper Columbia River basin)	(1) Negative correlation likelihood of spawning and rearing and road density (2) Negative correlation of fish density and road density	Lee et al. (1997)

Where both stream and road densities are high, the incidence of connections between roads and streams can also be expected to be high, resulting in more common and pronounced effects of roads on streams (Gucinski et al. 2000). For example, a study on the Medicine Bow National Forest (WY) found as the number of culverts and stream crossings increased, so did the amount of sediment in stream channels (Eaglin and Hubert 1993). They also found a negative correlation with fish density and the number of culverts. Invertebrate communities can also be impacted. McGurk and Fong (1995) report a negative correlation between an index of road density with macroinvertebrate diversity.

The U.S. Fish and Wildlife Service's Final Rule listing bull trout as threatened (USDI Fish and Wildlife Service 1999) addressed road density, stating:

> "… assessment of the interior Columbia Basin ecosystem revealed that increasing road densities were associated with declines in four non-anadromous salmonid species (bull trout, Yellowstone cutthroat trout, westslope cutthroat trout, and redband trout) within the Columbia River Basin, likely through a variety of factors associated with roads (Quigley & Arbelbide 1997). Bull trout were less likely to use highly roaded basins for spawning and rearing, and if present, were likely to be at lower population levels (Quigley and Arbelbide 1997). Quigley et al. (1996) demonstrated that when average road densities were between 0.4 to 1.1 km/km^2 (0.7 and 1.7 mi/mi^2) on USFS lands, the proportion of subwatersheds supporting "strong" populations of key salmonids dropped substantially. Higher road densities were associated with further declines" (USDI Fish and Wildlife Service 1999, p. 58922).

Anderson et al. (2012) also showed that watershed conditions tend to be best in areas protected from road construction and development. Using the US Forest Service's Watershed Condition Framework assessment data, they showed that National Forest lands that are protected under the Wilderness Act, which provides the strongest safeguards, tend to have the healthiest watersheds. Watersheds in Inventoried Roadless Areas – which are protected from road building and logging by the Roadless Area Conservation Rule – tend to be less healthy than watersheds in designated Wilderness, but they are considerably healthier than watersheds in the managed landscape.

Impacts on other resources
Roads and motorized trails also play a role in affecting wildfire occurrence. Research shows that human-ignited wildfires, which account for more than 90% of fires on national lands, is almost five times more likely in areas with roads (USDA Forest Service 1996a; USDA Forest Service 1998). Furthermore, Baxter (2002) found that off-road vehicles (ORVs) can be a significant source of fire ignitions on forestlands. Roads can affect where and how forests burn and, by extension, the vegetative condition of the forest. See Attachment 1 for more information documenting the relationship between roads and wildfire occurrence.

Finally, access allowed by roads and trails can increase of ORV and motorized use in remote areas threatening archaeological and historic sites. Increased visitation has resulted in intentional and unintentional damage to many cultural sites (USDI Bureau of Land Management 2000, Schiffman 2005).

II. Climate Change and Transportation Infrastructure including the value of roadless areas for climate change adaptation

As climate change impacts grow more profound, forest managers must consider the impacts on the transportation system as well as from the transportation system. In terms of the former, changes in precipitation and hydrologic patterns will strain infrastructure at times to the breaking point resulting in damage to streams, fish habitat, and water quality as well as threats to public safety. In terms of the latter, the fragmenting effect of roads on habitat will impede the movement of species which is a fundamental element of adaptation. Through planning, forest managers can proactively address threats to infrastructure, and can actually enhance forest resilience by removing unneeded roads to create larger patches of connected habitat.

Impact of climate change and roads on transportation infrastructure
It is expected that climate change will be responsible for more extreme weather events, leading to increasing flood severity, more frequent landslides, changing hydrographs (peak, annual mean flows, etc.), and changes in erosion and sedimentation rates and delivery processes. Roads and trails in national forests, if designed by an engineering standard at all, were designed for storms and water flows typical of past decades, and hence may not be designed for the storms in future decades. Hence, climate driven changes may cause transportation infrastructure to malfunction or fail (ASHTO 2012, USDA Forest Service 2010). The likelihood is higher for facilities in high-risk settings—such as rain-on-snow zones, coastal areas, and landscapes with unstable geology (USDA Forest Service 2010).

Forests fragmented by roads will likely demonstrate less resistance and resilience to stressors, like those associated with climate change (Noss 2001). First, the more a forest is fragmented (and therefore the higher the edge/interior ratio), the more the forest loses its inertia characteristic, and becoming less resilient and resistant to climate change. Second, the more a forest is fragmented characterized by isolated patches, the more likely the fragmentation will interfere with the ability of species to track shifting climatic conditions over time and space. Noss (2001) predicts that weedy species with effective dispersal mechanisms might benefit from fragmentation at the expense of native species.

Modifying infrastructure to increase resilience
To prevent or reduce road failures, culvert blow-outs, and other associated hazards, forest managers will need to take a series of actions. These include replacing undersized culverts with larger ones, prioritizing maintenance and upgrades (e.g., installing drivable dips and more outflow structures), and obliterating roads that are no longer needed and pose erosion hazards (USDA Forest Service 2010, USDA Forest Service 2012a, USDA Forest Service 2011, Table 4).

Olympic National Forest has developed a number of documents oriented at oriented at protecting watershed health and species in the face of climate change, including a 2003 travel management strategy and a report entitled Adapting to Climate Change in Olympic National Park and National Forest. In the travel management strategy, Olympic National Forest recommended that 1/3rd of its road system be decommissioned and obliterated (USDA Forest Service 2011a). In addition, the plan called for addressing fish migration barriers in a prioritized and strategic way – most of these are associated with roads. The report calls for road decommissioning, relocation of roads away from streams, enlarging culverts as well as replacing culverts with fish-friendly crossings (USDA Forest Service 2011a, Table 4).

Table 4: Current and expected sensitivities of fish to climate change on the Olympic Peninsula, associated adaptation strategies and action for fisheries and fish habitat management and relevant to transportation management at Olympic National Forest and Olympic National Park (excerpt reprinted from USDA Forest Service 2011a).

Current and expected sensitivites	Adaptation strategies and actions
Changes in habitat quantity and quality	• Implement habitat restoration projects that focus on re-creating watershed processes and functions and that create diverse, resilient habitat.
Increase in culvert failures, fill-slope failures, stream adjacent road failures, and encroachment from stream-adjacent road segments	• Decommission unneeded roads. • Remove sidecast, improve drainage, and increase culvert sizing on remaining roads. • Relocate stream-adjacent roads.
Greater difficulty disconnecting roads from stream channels	• Design more resilient stream crossing structures.
Major changes in quantity and timing of streamflow in transitional watersheds	• Make road and culvert designs more conservative in transitional watersheds to accommodate expected changes.
Decrease in area of headwater streams	• Continue to correct culvert fish passage barriers. • Consider re-prioritizing culvert fish barrier correction projects.
Decrease in habitat quantity and connectivity for species that use headwater streams	• Restore habitat in degraded headwater streams that are expected to retain adequate summer streamflow (ONF).

In December 2012, the USDA Forest Service published a report entitled "Assessing the Vulnerability of Watersheds to Climate Change." This document reinforces the concept expressed by Olympic National Forest that forest managers need to be proactive in reducing erosion potential from roads:

> "Road improvements were identified as a key action to improve condition and resilience of watersheds on all the pilot Forests. In addition to treatments that reduce erosion, road improvements can reduce the delivery of runoff from road segments to channels, prevent diversion of flow during large events, and restore aquatic habitat connectivity by providing for passage of aquatic organisms. As stated previously, watershed sensitivity is determined by both inherent and management-related factors. Managers have no control over the inherent factors, so to improve resilience, efforts must be directed at anthropogenic influences such as instream flows, roads, rangeland, and vegetation management….
>
> [Watershed Vulnerability Analysis] results can also help guide implementation of travel management planning by informing priority setting for decommissioning roads and road reconstruction/maintenance. As with the Ouachita NF example, disconnecting roads from the stream network is a key objective of such work. Similarly, WVA analysis could also help prioritize aquatic organism passage projects at road-stream crossings to allow migration by aquatic residents to suitable habitat as streamflow and temperatures change" (USDA Forest Service 2012a, p. 22-23).

Reducing fragmentation to enhance aquatic and terrestrial species adaptation
Decommissioning and upgrading roads and thus reducing the amount of fine sediment deposited on salmonid nests can increase the likelihood of egg survival and spawning success (McCaffery et al. 2007). In addition, this would reconnect stream channels and remove barriers such as culverts. Decommissioning roads in riparian areas may provide further benefits to salmon and other aquatic organisms by permitting reestablishment of streamside vegetation, which provides shade and maintains a cooler, more moderated microclimate over the stream (Battin et al. 2007).

One of the most well documented impacts of climate change on wildlife is a shift in the ranges of species (Parmesan 2006). As animals migrate, landscape connectivity will be increasingly important (Holman et al. 2005). Decommissioning roads in key wildlife corridors will improve connectivity and be an important mitigation measure to increase resiliency of wildlife to climate change. For wildlife, road decommissioning can reduce the many stressors associated with roads. Road decommissioning restores habitat by providing security and food such as grasses and fruiting shrubs for wildlife (Switalski and Nelson 2011).

Forests fragmented by roads and motorized trail networks will likely demonstrate less resistance and resilience to stressors, such as weeds. As a forest is fragmented and there is more edge habitat, Noss (2001) predicts that weedy species with effective dispersal mechanisms will increasingly benefit at the expense of native species. However, decommissioned roads when seeded with native species can reduce the spread of invasive species (Grant et al. 2011), and help restore fragmented forestlands. Off-road vehicles with large knobby tires and large undercarriages are also a key vector for weed spread (e.g., Rooney 2006). Strategically closing and decommissioning motorized routes, especially in roadless areas, will reduce the spread of weeds on forestlands (Gelbard and Harrison 2003).

Transportation infrastructure and carbon sequestration
The topic of the relationship of road restoration and carbon has only recently been explored. There is the potential for large amounts of carbon (C) to be sequestered by reclaiming roads. When roads are decompacted during reclamation, vegetation and soils can develop more

rapidly and sequester large amounts of carbon. A recent study estimated total soil C storage increased 6 fold to 6.5 x 107g C/km (to 25 cm depth) in the northwestern US compared to untreated abandoned roads (Lloyd et al. 2013). Another recent study concluded that reclaiming 425 km of logging roads over the last 30 years in Redwood National Park in Northern California resulted in net carbon savings of 49,000 Mg carbon to date (Madej et al. 2013, Table 5).

Kerekvliet et al. (2008) published a Wilderness Society briefing memo on the impact to carbon sequestration from road decommissioning. Using Forest Service estimates of the fraction of road miles that are unneeded, the authors calculated that restoring 126,000 miles of roads to a natural state would be equivalent to revegetating an area larger than Rhode Island. In addition, they calculate that the net economic benefit of road treatments are always positive and range from US$0.925-1.444 billion.

Table 5. Carbon budget implications in road decommissioning projects (reprinted from Madej et al. 2013).

Road Decommissioning Activities and Processes	Carbon Cost	Carbon Savings
Transportation of staff to restoration sites (fuel emissions)	X	
Use of heavy equipment in excavations (fuel emissions)	X	
Cutting trees along road alignment during hillslope recontouring	X	
Excavation of road fill from stream crossings		X
Removal of road fill from unstable locations		X
Reduces risk of mass movement		X
Post-restoration channel erosion at excavation sites	X	
Natural revegetation following road decompaction		X
Replanting trees		X
Soil development following decompaction		X

Benefits of roadless areas and roadless area networks to climate change adaptation
Undeveloped natural lands provide numerous ecological benefits. They contribute to biodiversity, enhance ecosystem representation, and facilitate connectivity (Loucks et al. 2003; Crist and Wilmer 2002, Wilcove 1990, The Wilderness Society 2004, Strittholt and Dellasala 2001, DeVelice and Martin 2001), and provide high quality or undisturbed water, soil and air (Anderson et al. 2012, Dellasalla et al. 2011). They also can serve as ecological baselines to help us better understand our impacts to other landscapes, and contribute to landscape resilience to climate change.

Forest Service roadless lands, in particular, are heralded for the conservation values they provide. These are described at length in the preamble of the Roadless Area Conservation Rule (RACR)[4] as well as in the Final Environmental Impact Statement (FEIS) for the RACR[5], and

[4] Federal Register .Vol. 66, No. 9. January 12, 2001. Pages 3245-3247.

include: high quality or undisturbed soil, water, and air; sources of public drinking water; diversity of plant and animal communities; habitat for threatened, endangered, proposed, candidate, and sensitive species and for those species dependent on large, undisturbed areas of land; primitive, semi-primitive non- motorized, and semi-primitive motorized classes of dispersed recreation; reference landscapes; natural appearing landscapes with high scenic quality; traditional cultural properties and sacred sites; and other locally identified unique characteristics (e.g., include uncommon geological formations, unique wetland complexes, exceptional hunting and fishing opportunities).

The Forest Service, National Park Service, and US Fish and Wildlife Service recognize that protecting and connecting roadless or lightly roaded areas is an important action agencies can take to enhance climate change adaptation. For example, the Forest Service National Roadmap for Responding to Climate Change (USDA Forest Service 2011b) establishes that increasing connectivity and reducing fragmentation are short and long term actions the Forest Service should take to facilitate adaptation to climate change.[6] The National Park Service also identifies connectivity as a key factor for climate change adaptation along with establishing "blocks of natural landscape large enough to be resilient to large-scale disturbances and long-term changes" and other factors. The agency states that: "The success of adaptation strategies will be enhanced by taking a broad approach that identifies connections and barriers across the landscape. Networks of protected areas within a larger mixed landscape can provide the highest level of resilience to climate change."[7] Similarly, the National Fish, Wildlife and Plants Climate Adaptation Partnership's Adaptation Strategy (2012) calls for creating an ecologically-connected network of conservation areas.[8]

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[5] Final Environmental Impact Statement, Vol. 1, 3–3 to 3–7

[6] Forest Service, 2011. *National Roadmap for Responding to Climate Change*. US Department of Agriculture. FS-957b. Page 26.

[7] National Park Service. *Climate Change Response Program Brief.* http://www.nature.nps.gov/climatechange/adaptationplanning.cfm. Also see: National Park Service, 2010. *Climate Change Response Strategy*. http://www.nature.nps.gov/climatechange/docs/NPS_CCRS.pdf. Objective 6.3 is to "Collaborate to develop cross-jurisdictional conservation plans to protect and restore connectivity and other landscape-scale components of resilience."

[8] See http://www.wildlifeadaptationstrategy.gov/pdf/NFWPCAS-Chapter-3.pdf. Pages 55- 59. The first goal and related strategies are:

> Goal 1: Conserve habitat to support healthy fish, wildlife, and plant populations and ecosystem functions in a changing climate.
>
> > Strategy 1.1: identify areas for an ecologically-connected network of terrestrial, freshwater, coastal, and marine conservation areas that are likely to be resilient to climate change and to support a broad range of fish, wildlife, and plants under changed conditions.
> >
> > Strategy 1.2: Secure appropriate conservation status on areas identified in Strategy 1.1 to complete an ecologically-connected network of public and private conservation areas that will be resilient to climate change and support a broad range of species under changed conditions.
> >
> > Strategy 1.4: Conserve, restore, and as appropriate and practicable, establish new ecological connections among conservation areas to facilitate fish, wildlife, and plant migration, range shifts, and other transitions caused by climate change.

Crist and Wilmer (2002) looked at the ecological value of roadless lands in the Northern Rockies and found that protection of national forest roadless areas, when added to existing federal conservation lands in the study area, would 1) increase the representation of virtually all land cover types on conservation lands at both the regional and ecosystem scales, some by more than 100%; 2) help protect rare, species-rich, and often-declining vegetation communities; and 3) connect conservation units to create bigger and more cohesive habitat "patches."

Roadless lands also are responsible for higher quality water and watersheds. Anderson et al. (2012) assessed the relationship of watershed condition and land management status and found a strong spatial association between watershed health and protective designations. Dellasalla et al. (2011) found that undeveloped and roadless watersheds are important for supplying downstream users with high-quality drinking water, and developing these watersheds comes at significant costs associated with declining water quality and availability. The authors recommend a light-touch ecological footprint to sustain the many values that derive from roadless areas including healthy watersheds.

III. Sustainable Transportation Management in National Forests as Part of Ecological Restoration

At 375,000 miles strong, the Forest Service road system is one of the largest in the world – it is eight times the size of the National Highway System. It is also indisputably unsustainable – that is, roads are not designed, located, or maintained according to best management practices, and environmental impacts are not minimized. It is largely recognized that forest roads, especially unpaved ones, are a primary source of sediment pollution to surface waters (Endicott 2008, Gucinski et al. 2000), and that the system has about 1/3rd more miles than it needs (USDA Forest Service 2001). In addition, the majority of the roads were constructed decades ago when road design and management techniques did not meet current standards (Gucinski et al. 2000, Endicott 2008), making them more vulnerable to erosion and decay than if they had been designed today. Road densities in national forests often exceed accepted thresholds for wildlife.

Only a small portion of the road system is regularly used. All but 18% of the road system is inaccessible to passenger vehicles. Fifty-five percent of the roads are accessible only by high clearance vehicles and 27% are closed. The 18% that is accessible to cars is used for about 80% of the trips made within National Forests.[9] Most of the road maintenance funding is directed to the passenger car roads, while the remaining roads suffer from neglect. As a result, the Forest Service currently has a $3.7 billion road maintenance backlog that grows every year. In other words, only about 1/5th of the roads in the national forest system are used most of the time, and the fraction that is used often is the best designed and maintained because they are higher level access roads. The remaining roads sit generally unneeded and under-maintained – arguably a growing ecological and fiscal liability.

Current Forest Service management direction is to identify and implement a sustainable transportation system.[10] The challenge for forest managers is figuring out what is a sustainable road system and how to achieve it – a challenge that is exacerbated by climate change. It is

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[9] USDA Forest Service. Road Management Website Q&As. Available online at http://www.fs.fed.us/eng/road_mgt/qanda.shtml.

[10] See Forest Service directive memo dated March 29, 2012 entitled "Travel Management, Implementation of 36 CFR, Part 202, Subpart A (36 CFR 212.5(b))"

reasonable to define a sustainable transportation system as one where all the routes are constructed, located, and maintained with best management practices, and social and environmental impacts are minimized. This, of course, is easier said than done, since the reality is that even the best roads and trail networks can be problematic simply because they exist and usher in land uses that without the access would not occur (Trombulak and Frissell 2000, Carnefix and Frissell 2009, USDA Forest Service 1996b), and when they are not maintained to the designed level they result in environmental problems (Endicott 2008; Gucinski et al. 2000). Moreover, what was sustainable may no longer be sustainable under climate change since roads designed to meet older climate criteria may no longer hold up under new climate scenarios (USDA Forest Service 2010, USDA Forest Service 2011b, USDA Forest Service 2012a, AASHTO 2012).

Forest Service efforts to move toward a more sustainable transportation system

The Forest Service has made efforts to make its transportation system more sustainable, but still has considerable work to do. In 2001, the Forest Service tried to address the issue by promulgating the Roads Rule[11] with the purpose of working toward a sustainable road system (USDA 2001). The Rule directed every national forest to identify a minimum necessary road system and identify unneeded roads for decommissioning. To do this, the Forest Service developed the Roads Analysis Process (RAP), and published Gucinski et al. (2000) to provide the scientific foundation to complement the RAP. In describing the RAP, Gucinski et al. (2000) writes:

> "Roads Analysis is intended to be an integrated, ecological, social, and economic approach to transportation planning. It uses a multiscale approach to ensure that the identified issues are examined in context. Roads Analysis is to be based on science. Analysts are expected to locate, correctly interpret, and use relevant existing scientific literature in the analysis, disclose any assumptions made during the analysis, and reveal the limitations of the information on which the analysis is based. The analysis methods and the report are to be subjected to critical technical review" (p. 10).

Most national forests have completed RAPs, although most only looked at passenger vehicle roads which account for less than 20% of the system's miles. The Forest Service Washington Office in 2010 directed that forests complete a Travel Analysis Process (TAP) by the end of fiscal year 2015, which must address all roads and create a map and list of roads identifying which are likely needed and which are not. Completed TAPs will provide a blueprint for future road decommissioning and management, they will not constitute compliance with the Roads Rule, which clearly requires the identification of the minimum roads system and roads for decommissioning. Almost all forests have yet to comply with subpart A.

The Forest Service in 2005 then tried to address the off-road portion of this issue by promulgating subpart B of the Travel Managemenr Rule,[12] with the purpose of curbing the most serious impacts associated with off-road vehicle use. Without a doubt, securing summer-time travel management plans was an important step to curbing the worst damage. However, much work remains to be done to approach sustainability, especially since many national forests used the travel management planning process to simply freeze the footprint of motorized routes, and did not try to re-design the system to make it more ecologically or socially sustainable. Adams

[11] 36 CFR 215 subpart A

[12] 36 CFR 212 subpart B

and McCool (2009) considered this question of how to achieve sustainable motorized recreation and concluded that:

> As the agencies move to revise [off-road vehicle] allocations, they need to clearly define how they intend to locate routes so as to minimize impacts to natural resources and other recreationists in accordance with Executive Order 11644....[13]
>
> …As they proceed with designation, the FS and BLM need to acknowledge that current allocations are the product of agency failure to act, not design. Ideally, ORV routes would be allocated as if the map were currently empty of ORV routes. Reliance on the current baseline will encourage inefficient allocations that likely disproportionately impact natural resources and non-motorized recreationists. While acknowledging existing use, the agencies need to do their best to imagine the best possible arrangement of ORV routes, rather than simply tinkering around the edges of the current allocations.[14]

The Forest Service only now is contemplating addressing the winter portion of the issue, forced by a lawsuit challenging the Forest Service's inadequate management of snowmobiles. The agency is expected to issue a third rule in the fall of 2014 that will trigger winter travel management planning.

Strategies for identifying a minimum road system and prioritizing restoration

Transportation Management plays an integral role in the restoration of Forestlands. Reclaiming and obliterating roads is key to developing a sustainable transportation system. Numerous authors have suggested removing roads 1) to restore water quality and aquatic habitats Gucinski et al. 2000), and 2) to improve habitat security and restore terrestrial habitat (e.g., USDI USFWS 1993, Hebblewhite et al. 2009).

Creating a minimum road system through road removal will increase connectivity and decrease fragmentation across the entire forest system. However, at a landscape scale, certain roads and road segments pose greater risks to terrestrial and aquatic integrity than others. Hence, restoration strategies must focus on identifying and removing/mitigating the higher risk roads. Additionally, areas with the highest ecological values, such as being adjacent to a roadless area, may also be prioritized for restoration efforts. Several methods have been developed to help prioritize road reclamation efforts including GIS-based tools and best management practices (BMPs). It is our hope that even with limited resources, restoration efforts can be prioritized and a more sustainable transportation system created.

GIS-based tools

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[13] Recent court decisions have made it clear that the minimization requirements in the Executive Orders are not discretionary and that the Executive Orders are enforceable. See

- *Idaho Conservation League v. Guzman ,* 766 F. Supp. 2d 1056 (D. Idaho 2011) (Salmon-Challis National Forest TMP) .
- *The Wilderness Society v. U.S. Forest Service*, CV 08-363 (D. Idaho 2012) (Sawtooth-Minidoka district National Forest TMP).
- *Central Sierra Environmental Resource Center v. US Forest Service*, CV 10-2172 (E.D. CA 2012) (Stanislaus National Forest TMP).

[14] Page 105.

Girvetz and Shilling (2003) developed a novel and inexpensive way to analyze environmental impacts from road systems using the Ecosystem Management Decision Support program (EMDS). EMDS was originally developed by the United States Forest Service, as a GIS-based decision support tool to conduct ecological analysis and planning (Reynolds 1999). Working in conjunction with Tahoe National Forest managers, Girvetz and Shilling (2003) used spatial data on a number of aquatic and terrestrial variables and modeled the impact of the forest's road network. The network analysis showed that out of 8233 km of road analyzed, only 3483 km (42%) was needed to ensure current and future access to key points. They found that the modified network had improved patch characteristics, such as significantly fewer "cherry stem" roads intruding into patches, and larger roadlessness.

Shilling et al. (2012) later developed a recreational route optimization model using a similar methodology and with the goal of identifying a sustainable motorized transportation system for the Tahoe National Forest (Figure 2). Again using a variety of environmental factors, the model identified routes with high recreational benefits, lower conflict, lower maintenance and management requirements, and lower potential for environmental impact operating under the presumption that such routes would be more sustainable and preferable in the long term. The authors combined the impact and benefit analyses into a recreation system analysis "that was effectively a cost-benefit accounting, consistent with requirements of both the federal Travel Management Rule (TMR) and the National Environmental Policy Act" (p. 392**).**

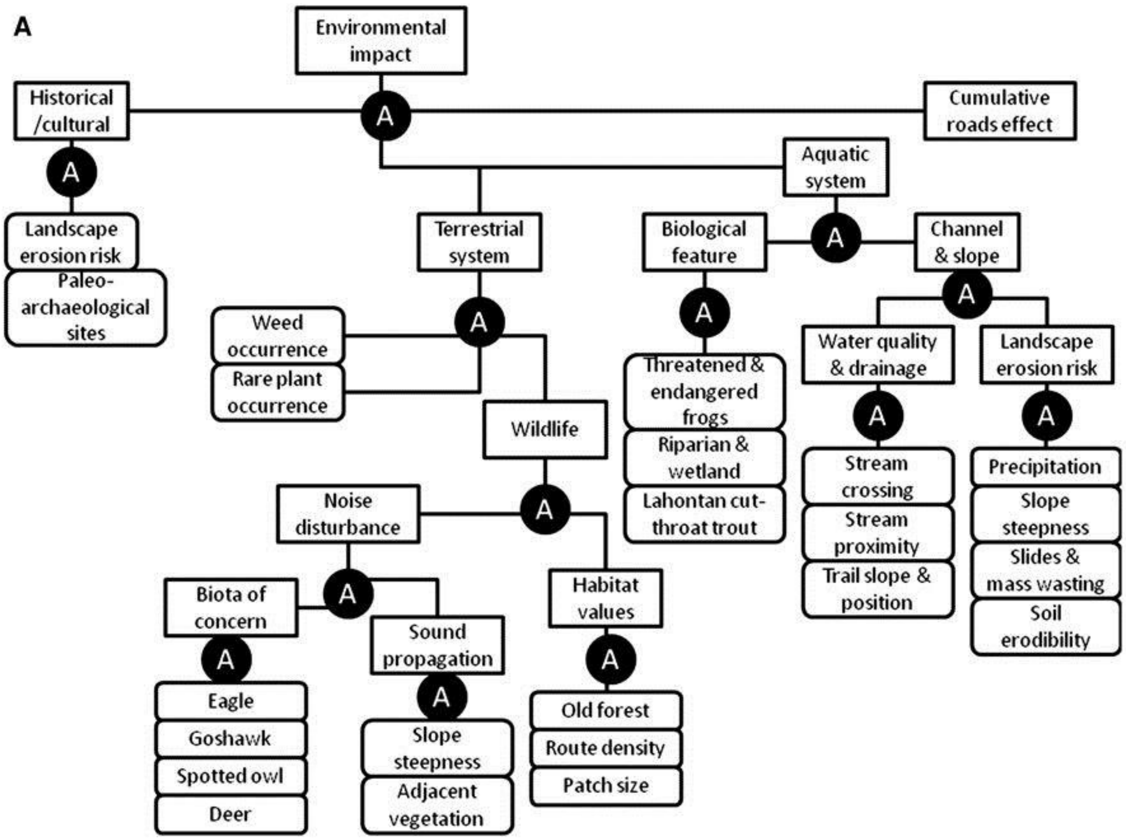

Figure 2: A knowledge base of contributions of various environmental conditions to the concept ‘‘environmental impact’’ [of motorized trails]. Rectangles indicate concepts, circles indicate Boolean logic operators, and rounded rectangles indicate sources of environmental data. (Reprinted from Shilling et al. 2012)

The Wilderness Society in 2012 also developed a GIS decision support tool called “RoadRight” that identifies high risk road segments to a variety of forest resources including water, wildlife, and roadlessness (The Wilderness Society 2012, The Wilderness Society 2013). The GIS system is designed to provide information that will help forest planners identify and minimize road related environmental risks. See the summary of and user guide for RoadRight that provides more information including where to access the open source software.[15]

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[15] The Wilderness Society, 2012. Rightsizing the National Forest Road System: A Decision Support Tool. Available at http://www.landscapecollaborative.org/download/attachments/12747016/Road+decommissioning+model+-overview+2012-02-29.pdf?version=1&modificationDate=1331595972330.

The Wilderness Society, 2013.
RoadRight: A Spatial Decision Support System to Prioritize Decommissioning and Repairing Roads in

Best management practices (BMPs)
BMPs have also been developed to help create more sustainable transportation systems and identify restoration opportunities. BMPs provide science-based criteria and standards that land managers follow in making and implementing decisions about human uses and projects that affect natural resources. Several states have developed BMPs for road construction, maintenance and decommissioning practices (e.g., Logan 2001, Merrill and Cassaday 2003, USDA Forest Service 2012b).

Recently, BMPs have been developed for addressing motorized recreation. Switalski and Jones (2012) published, "*Off-Road Vehicle Best Management Practices for Forestlands: A Review of Scientific Literature and Guidance for Managers*." This document reviews the current literature on the environmental and social impacts of off-road vehicles (ORVs), and establishes a set of Best Management Practices (BMPs) for the planning and management of ORV routes on forestlands. The BMPs were designed to be used by land managers on all forestlands, and is consistent with current forest management policy and regulations. They give guidance to transportation planners on where how to place ORV routes in areas where they will reduce use conflicts and cause as little harm to the environment as possible. These BMPs also help guide managers on how to best remove and restore routes that are redundant or where there is an unacceptable environmental or social cost.

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Attachments

Attachment 1: Wildfire and Roads Fact Sheet

Attachment 2: Using Road Density as a Metric for Ecological Health in National Forests: What Roads and Routes should be Included? Summary of Scientific Information

Roads and Fire: A Proven Relationship

Photo: Lou Anegli Digital

Roaded Forests Are at a Greater Risk of Experiencing Wildfires than Unroaded Forests

- A wildland fire ignition is almost twice as likely to occur in a roaded area than in a roadless area. (USDA 2000, Table 3-18)
- The location of large wildfires is often correlated with proximity to busy roads. (Sierra Nevada Ecosystem Project, 1996)
- High road density increases the probability of fire occurrence due to human-caused ignitions. (Hann, W.J., et al. 1997)
- Unroaded areas have lower potential for high-intensity fires than roaded areas because they are less prone to human-caused ignitions. (DellaSala, et al. 1995)
- The median size of large fires on national forests is greater outside of roadless areas. (USDA 2000, Table 3-22)
- A positive correlation exists between lightning fire frequency and road density due to increased availability of flammable fine fuels near roads. (Arienti, M. Cecilia, et al. 2009)
- Human caused wildfires are strongly associated with access to natural landscapes, with the proximity to urban areas and roads being the most important factor (Romero-Calcerrada, et al. 2008)

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Human Activity and Wildfire

- Sparks from cars, off-road vehicles, and neglected campfires caused nearly 50,000 wildfire ignitions in 2000. (USDA 2000, Fuel Management and Fire Suppression Specialist Report, Table 4.)
- More than 90% of fires on national lands are caused by humans (USDA 1996 and 1998)
- Human-ignited wildfire is almost 5 times more likely to occur in a roaded area than in a roadless area (USDA 2000, Table 3-19).

There are 375,000 miles of roads in our national forests.

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Roads and Fire: A Proven Relationship

Photo: USDA Forest Service, Coconino National Forest

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Attachment 2: Using Road Density as a Metric for Ecological Health in National Forests: What Roads and Routes should be Included?
Summary of Scientific Information
Last Updated, November 22, 2012

I. Density analysis should include closed roads, non-system roads administered by other jurisdictions (private, county, state), temporary roads and motorized trails.

Typically, the Forest Service has calculated road density by looking only at open system road density. From an ecological standpoint, this approach may be flawed since it leaves out of the density calculations a significant percent of the total motorized routes on the landscape. For instance, the motorized route system in the entire National Forest System measures well over 549,000 miles.[1] By our calculation, a density analysis limited to open system roads would consider less than 260,000 miles of road, which accounts for less than half of the entire motorized transportation system estimated to exist on our national forests.[2] These additional roads and motorized trails impact fish, wildlife, and water quality, just as open system roads do. In this section, we provide justification for why a road density analysis used for the purposes of assessing ecological health and the effects of proposed alternatives in a planning document should include closed system roads, non-system roads administered by other jurisdictions, temporary roads, and motorized trails.

Impacts of closed roads

It is crucial to distinguish the density of roads physically present on the landscape, whether closed to vehicle use or not, from "open-road density" (Pacific Rivers Council, 2010). An open-road density of 1.5 mi/mi² has been established as a standard in some national forests as protective of some terrestrial wildlife species. However, many areas with an open road density of 1.5 mi/mi² have a much higher inventoried or extant hydrologically effective road density, which may be several-fold as high with significant aquatic impacts. This higher density occurs because many road "closures" block vehicle access, but do nothing to mitigate the hydrologic alterations that the road causes. The problem is

[1] The National Forest System has about 372,000 miles of system roads. The forest service also has an estimated 47,000 miles of motorized trails. As of 1998, there were approximately 130,000 miles of non-system roads in our forests. Non-system roads include public roads such as state, county, and local jurisdiction and private roads. (USFS, 1998) The Forest Service does not track temporary roads but is reasonable to assume that there are likely several thousand miles located on National Forest System lands.

[2] About 30% of system roads, or 116,108 miles, are in Maintenance Level 1 status, meaning they are closed to all motorized use. (372,000 miles of NFS roads - 116,108 miles of ML 1 roads = 255,892). This number is likely conservative given that thousands of more miles of system roads are closed to public motorized use but categorized in other Maintenance Levels.

further compounded in many places by the existence of "ghost" roads that are not captured in agency inventories, but that are nevertheless physically present and causing hydrologic alteration (Pacific Watershed Associates, 2005).

Closing a road to public motorized use can mitigate the impacts on water, wildlife, and soils only if proper closure and storage technique is followed. Flow diversions, sediment runoff, and illegal incursions will continue unabated if necessary measures are not taken. The Forest Service's National Best Management Practices for non-point source pollution recommends the following management techniques for minimizing the aquatic impacts from closed system roads: eliminate flow diversion onto the road surface, reshape the channel and streambanks at the crossing-site to pass expected flows without scouring or ponding, maintain continuation of channel dimensions and longitudinal profile through the crossing site, and remove culverts, fill material, and other structures that present a risk of failure or diversion. Despite good intentions, it is unlikely given our current fiscal situation and past history that the Forest Service is able to apply best management practices to all stored roads,[3] and that these roads continue to have impacts. This reality argues for assuming that roads closed to the public continue to have some level of impact on water quality, and therefore, should be included in road density calculations.

As noted above, many species benefit when roads are closed to public use. However, the fact remains that closed system roads are often breached resulting in impacts to wildlife. Research shows that a significant portion of off-road vehicle (ORV) users violates rules even when they know what they are (Lewis, M.S., and R. Paige, 2006; Frueh, LM, 2001; Fischer, A.L., et. al, 2002; USFWS, 2007.). For instance, the Rio Grande National Forest's Roads Analysis Report notes that a common travel management violation occurs when people drive around road closures on Level 1 roads (USDA Forest Service, 1994). Similarly, in a recent legal decision from the Utah District Court , *Sierra Club v. USFS*, Case No. 1:09-cv-131 CW (D. Utah March 7, 2012), the court found that, as part of analyzing alternatives in a proposed travel management plan, the Forest Service failed to take a hard look at the impact of continued illegal use. In part, the court based its decision on the Forest Service's acknowledgement that illegal motorized use is a significant problem and that the mere presence of roads is likely to result in illegal use.

In addition to the disturbance to wildlife from ORVs, incursions and the accompanying human access can also result in illegal hunting and trapping of animals. The Tongass National Forest refers to this in its EIS to amend the Land and Resources Management Plan. Specifically, the Forest Service notes in the EIS that Alexander Archipelego wolf mortality due to legal and illegal hunting and trapping is related not only to roads open to motorized access, but to all roads, and that *total road densities* of 0.7-1.0 mi/mi^2 or less may be necessary (USDA Forest Service, 2008).

As described below, a number of scientific studies have found that ORV use on roads and trails can have serious impacts on water, soil and wildlife resources. It should be expected that ORV use will continue to

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[3] The Forest Service generally reports that it can maintain 20-30% of its open road system to standard.

some degree to occur illegally on closed routes and that this use will affect forest resources. Given this, roads closed to the general public should be considered in the density analysis.

Impacts of non-system roads administered by other jurisdictions (private, county, state)

As of 1998, there were approximately 130,000 miles of non-system roads in national forests (USDA Forest Service, 1998). These roads contribute to the environmental impacts of the transportation system on forest resources, just as forest system roads do. Because the purpose of a road density analysis is to measure the impacts of roads at a landscape level, the Forest Service should include all roads, including non-system, when measuring impacts on water and wildlife. An all-inclusive analysis will provide a more accurate representation of the environmental impacts of the road network within the analysis area.

Impacts of temporary roads

Temporary roads are not considered system roads. Most often they are constructed in conjunction with timber sales. Temporary roads have the same types environmental impacts as system roads, although at times the impacts can be worse if the road persists on the landscape because they are not built to last.

It is important to note that although they are termed temporary roads, their impacts are not temporary. According to Forest Service Manual (FSM) 7703.1, the agency is required to "Reestablish vegetative cover on any unnecessary roadway or area disturbed by road construction on National Forest System lands within 10 years after the termination of the activity that required its use and construction." Regardless of the FSM 10-year rule, temporary roads can remain for much longer. For example, timber sales typically last 3-5 years or more. If a temporary road is built in the first year of a six year timber sale, its intended use does not end until the sale is complete. The timber contract often requires the purchaser to close and obliterate the road a few years after the Forest Service completes revegetation work. The temporary road, therefore, could remain open 8-9 years before the ten year clock starts ticking per the FSM. Therefore, temporary roads can legally remain on the ground for up to 20 years or more, yet they are constructed with less environmental safeguards than modern system roads.

Impacts of motorized trails

Scientific research and agency publications generally do not decipher between the impacts from motorized trails and roads, often collapsing the assessment of impacts from unmanaged ORV use with those of the designated system of roads and trails. The following section summarizes potential impacts resulting from roads and motorized trails and the ORV use that occurs on them.

Aquatic Resources
While driving on roads has long been identified as a major contributor to stream sedimentation (for review, see Gucinski, 2001), recent studies have identified ORV routes as a significant cause of stream sedimentation as well (Sack and da Luz, 2004; Chin et al.; 2004, Ayala et al.; 2005, Welsh et al;. 2006). It has been demonstrated that sediment loss increases with increased ORV traffic (Foltz, 2006). A study by

Sack and da Luz (2004) found that ORV use resulted in a loss of more than 200 pounds of soil off of every 100 feet of trail each year. Another study (Welsh et al., 2006) found that ORV trails produced five times more sediment than unpaved roads. Chin et al. (2004) found that watersheds with ORV use as opposed to those without exhibited higher percentages of channel sands and fines, lower depths, and lower volume – all characteristics of degraded stream habitat.

Soil Resources [4]

Ouren, et al. (2007), in an extensive literature review, suggests ORV use causes soil compaction and accelerated erosion rates, and may cause compaction with very few passes. Weighing several hundred pounds, ORVs can compress and compact soil (Nakata et al., 1976; Snyder et al., 1976; Vollmer et al., 1976; Wilshire and Nakata, 1976), reducing its ability to absorb and retain water (Dregne, 1983), and decreasing soil fertility by harming the microscopic organisms that would otherwise break down the soil and produce nutrients important for plant growth (Wilshire et al., 1977). An increase in compaction decreases soil permeability, resulting in increased flow of water across the ground and reduced absorption of water into the soil. This increase in surface flow concentrates water and increases erosion of soils (Wilshire, 1980; Webb, 1983; Misak et al., 2002).

Erosion of soil is accelerated in ORV-use areas directly by the vehicles, and indirectly by increased runoff of precipitation and the creation of conditions favorable to wind erosion (Wilshire, 1980). Knobby and cup-shaped protrusions from ORV tires that aid the vehicles in traversing steep slopes are responsible for major direct erosional losses of soil. As the tire protrusions dig into the soil, forces far exceeding the strength of the soil are exerted to allow the vehicles to climb slopes. The result is that the soil and small plants are thrown downslope in a "rooster tail" behind the vehicle. This is known as mechanical erosion, which on steep slopes (about 15° or more) with soft soils may erode as much as 40 tons/mi (Wilshire, 1992). The rates of erosion measured on ORV trails on moderate slopes exceed natural rates by factors of 10 to 20 (Iverson et al., 1981; Hinckley et al., 1983), whereas use on steep slopes has commonly removed the entire soil mantle exposing bedrock. Measured erosional losses in high use ORV areas range from 1.4-242 lbs/ft^2 (Wilshire et al., 1978) and 102-614 lbs/ft^2 (Webb et al., 1978). A more recent study by Sack and da Luz (2003) found that ORV use resulted in a loss of more than 200 lbs of soil off of every 100 feet of trail each year.

Furthermore, the destruction of cryptobiotic soils by ORVs can reduce nitrogen fixation by cyanobacteria, and set the nitrogen economy of nitrogen-limited arid ecosystems back decades. Even small reductions in crust can lead to diminished productivity and health of the associated plant community, with cascading effects on plant consumers (Davidson et al., 1996). In general, the deleterious effects of ORV use on cryptobiotic crusts is not easily repaired or regenerated. The recovery time for the lichen component of crusts has been estimated at about 45 years (Belnap, 1993). After this time the crusts may appear to have regenerated to the untrained eye. However, careful observation will reveal that the 45 year-old crusts will not have recovered their moss component, which will take an additional 200 years to fully come back (Belnap and Gillette, 1997).

[4] For a full review see Switalski, T. A. and A. Jones (2012).

Wildlife Resources [5]

Studies have shown a variety of possible wildlife disturbance vectors from ORVs. While these impacts are difficult to measure, repeated harassment of wildlife can result in increased energy expenditure and reduced reproduction. Noise and disturbance from ORVs can result in a range of impacts including increased stress (Nash et al., 1970; Millspaugh et al., 2001), loss of hearing (Brattstrom and Bondello, 1979), altered movement patterns (e.g., Wisdom et al. 2004; Preisler et al. 2006), avoidance of high-use areas or routes (Janis and Clark 2002; Wisdom 2007), and disrupted nesting activities (e.g., Strauss 1990).

Wisdom et al. (2004) found that elk moved when ORVs passed within 2,000 yards but tolerated hikers within 500 ft. Wisdom (2007) reported preliminary results suggesting that ORVs are causing a shift in the spatial distribution of elk that could increase energy expenditures and decrease foraging opportunities for the herd. Elk have been found to readily avoid and be displaced from roaded areas (Irwin and Peek, 1979; Hershey and Leege, 1982; Millspaugh, 1995). Additional concomitant effects can occur, such as major declines in survival of elk calves due to repeated displacement of elk during the calving season (Phillips, 1998). Alternatively, closing or decommissioning roads has been found to decrease elk disturbance (Millspaugh et al., 2000; Rowland et al., 2005).

Disruption of breeding and nesting birds is particularly well-documented. Several species are sensitive to human disturbance with the potential disruption of courtship activities, over-exposure of eggs or young birds to weather, and premature fledging of juveniles (Hamann et al., 1999). Repeated disturbance can eventually lead to nest abandonment. These short-term disturbances can lead to long-term bird community changes (Anderson et al., 1990). However when road densities decrease, there is an observable benefit. For example, on the Loa Ranger District of the Fishlake National Forest in southern Utah, successful goshawk nests occur in areas where the localized road density is at or below 2-3 mi/mi^2 (USDA, 2005).

Examples of Forest Service planning documents that use total motorized route density or a variant

Below, we offer examples of where total motorized route density or a variant has been used by the Forest Service in planning documents.

- The Mt. Taylor RD of the Cibola NF analyzed open and closed system roads and motorized trails together in a single motorized *route* density analysis. Cibola NF: Mt. Taylor RD Environmental Assessment for Travel Management Planning, Ch.3, p 55. http://prdp2fs.ess.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5282504.pdf.

- The Grizzly Bear Record of Decision (ROD) for the Forest Plan Amendments for Motorized Access

[5] For a full review see:Switalski, T. A. and A. Jones (2012).

Management within the Selkirk and Cabinet-Yaak Grizzly Bear Recovery Zones (Kootenai, Lolo, and Idaho Panhandle National Forests) assigned route densities for the designated recovery zones. One of the three densities was for Total Motorized Route Density (TMRD) which includes open roads, restricted roads, roads not meeting all reclaimed criteria, and open motorized trails. The agency's decision to use TMRD was based on the Endangered Species Act's requirement to use best available science, and monitoring showed that both open and closed roads and motorized trails were impacting grizzly. Grizzly Bear Plan Amendment ROD. Online at cache.ecosystem-management.org/48536_FSPLT1_009720.pdf.

- The Chequamegon-Nicolet National Forest set forest-wide goals in its forest plan for both open road density and total road density to improve water quality and wildlife habitat.

> *I decided to continue reducing the amount of total roads and the amount of open road to resolve conflict with quieter forms of recreation, impacts on streams, and effects on some wildlife species.* ROD, p 13.

Chequamegon-Nicolet National Forest Land and Resource Management Plan Record of Decision. Online at http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5117609.pdf.

- The Tongass National Forest's EIS to amend the forest plan notes that Alexander Archipelago wolf mortality due to legal and illegal hunting and trapping is related not only to roads open to motorized access, but to all roads, and that *total road densities* of 0.7-1.0 mi/mi^2 or less may be necessary.

> *Another concern in some areas is the potentially unsustainable level of hunting and trapping of wolves, when both legal and illegal harvest is considered. The 1997 Forest Plan EIS acknowledged that open road access contributes to excessive mortality by facilitating access for hunters and trappers. Landscapes with open-road densities of 0.7 to 1.0 mile of road per square mile were identified as places where human-induced mortality may pose risks to wolf conservation. The amended Forest Plan requires participation in cooperative interagency monitoring and analysis to identify areas where wolf mortality is excessive, determine whether the mortality is unsustainable, and identify the probable causes of the excessive mortality.*
>
> *More recent information indicates that wolf mortality is related not only to roads open to motorized access, but to all roads, because hunters and trappers use all roads to access wolf habitat, by vehicle or on foot. Consequently, this decision amends the pertinent standard and guideline contained in Alternative 6 as displayed in the Final EIS in areas where road access and associated human caused mortality has been determined to be the significant contributing factor to unsustainable wolf mortality. The standard and guideline has been modified to ensure that a range of options to reduce mortality risk will be considered in these areas, and to specify that total road densities of 0.7 to 1.0 mile per square mile or less may be necessary.* ROD, p 24.

Tongass National Forest Amendment to the Land and Resource Management Plan Record of Decision and Final EIS. January 2008. http://tongass-fpadjust.net/Documents/Record_of_Decision.pdf

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Appendix P

February 2, 2024

The Honorable Thomas J. Vilsack
Secretary
U.S. Department of Agriculture
1400 Independence Ave., S.W.
Washington, DC 20250

Submitted via webform to: https://cara.fs2c.usda.gov/Public//CommentInput?Project=65356.

Dear Secretary Vilsack:

The Wilderness Society (TWS) appreciates the opportunity to provide scoping comments on the USDA's notice of intent (NOI) and preliminary proposed action to amend all national forest land management plans to provide consistent direction for management of old-growth forests ("proposed amendment"). TWS is keenly interested in the USDA's efforts, as described in the NOI, to "conserve and steward existing and recruit future old-growth forest conditions and to monitor their condition," with the intent to "foster the long-term resilience of old-growth forest conditions and their contributions to ecological integrity across the National Forest System."

TWS commends the USDA Forest Service for taking this important step to conserve old-growth and mature forests in accordance with President Biden's Executive Order 14072 and following up on the USDA's advance notice of proposed rulemaking (ANPR) on climate resilience. Over the past few decades, the Forest Service has significantly shifted its perspective on the primary value and role of older forests from timber supply and limitations on cultural burning resulting in excessive vegetation in many fire-prone forests. In fact, thinning coupled with prescribed burning has often become an essential tool to remove excessive fuels and reduce wildfire risk to older forests in fire-adapted ecosystems.

Yet, many forest plans – even the 1994 Northwest Forest Plan and plans recently revised under the 2012 Planning Rule – still do not provide adequate direction to protect older forests from commercial logging or to improve their resilience to wildfire and other climate change impacts. This proposed amendment – with suggested improvements and if implemented to its fullest extent – has the potential to effectively provide consistent direction across the National Forest System to affirmatively manage for old-growth forest conditions, to enhance old-growth forest characteristics, and to address threats to their long-term persistence in a collaborative manner that involves local communities and addresses Tribal interests.

Several key aspects of the proposed amendment must be strengthened or clarified to ensure that its intent is achieved upon implementation. For example, as discussed in the "Implementation" section and elsewhere in our comments, the long-term success of the proposed amendment largely hinges on the quality of each *Adaptive Strategy for Old-Growth Forest Conservation*, since they are the mechanism by which future old-growth forest conditions are recruited. The

Forest Service should provide each unit with sufficiently detailed guidance on effective Tribal engagement and stakeholder collaboration, as well as a model *Adaptive Strategy* describing the process for prioritization of areas for the retention and recruitment of old-growth forest conditions, monitoring, and adaptive management. We also note the need for information required for completion of an *Adaptive Strategy* and the need for strong agency leadership and accountability. Without clarity on the requirements and expectations of collaborative adaptive management, and sufficient guidance and support from the Washington Office as units develop their *Adaptive Strategy*, the intent of this proposed amendment will not be fully realized.

These comments begin with an analysis of the legal framework for the proposed amendment, including the National Forest Management Act and EIS alternatives. Next, we provide detailed comments on each of the plan components and other plan content proposed in the amendment. Third, as mentioned above, we comment on implementation of the proposed amendment. Finally, we discuss impacts specific to Eastern forests, the Northwest Forest Plan amendment process, and the Tongass National Forest.[1]

I. Legal Framework and Issues

In our comments on the ANPR, TWS recommended that the Forest Service conserve old-growth and mature forests through a federal rule, exercising its broad rulemaking authority under the 1897 Organic Act. While TWS continues to support this regulatory approach, in part because of its legal durability, we also see merit in USDA's proposal to adopt a nationwide forest planning amendment as a complementary approach. Following are some legal issues that the proposed amendment process must address.

<u>NFMA</u>

The USDA clearly has the legal authority to adopt a nationwide forest plan amendment. The National Forest Management Act (NFMA) of 1976 gives the Forest Service broad authority to amend forest plans. Specifically, Section 6(f) of the NFMA states that forest plans "shall … be amended *in any manner whatsoever* after final adoption after public notice…." 16 USC 1604(f)(4) (emphasis added). Furthermore, the 2012 Planning Rule specifically states that the Secretary of Agriculture can act as the responsible official for approval of plan amendments. 36 CFR 219(b)(3). These and an array of other authorities provide ample legal and scientific support for the proposed amendment, as we discussed at length in our letter for the administrative record for the Advance Notice of Proposed Rulemaking (ANPR) on Climate Resilience addressed to Chris French and dated July 20, 2023, which we hereby incorporate by reference.

A notable condition that the NFMA places on plan amendments is that "if such amendment would result in a significant change in such plan," the plan must be amended "in accordance with the provisions of subsections (e) and (f) of this section …." 16 USC 1604(f)(4). We urge the agency to pay special attention to the requirement for significant plan amendments in subsection (e)(2): "determine forest management systems, harvesting levels, and procedures in the light of

[1] TWS endorses the comments submitted by Silvix Resources on behalf of an informal coalition of conservation groups in which TWS is a participant.

all of the [multiple] uses… and the availability of lands and their suitability for resource management."

While the NOI at least implicitly acknowledges that the proposed amendment would significantly change the forest plans, we are concerned that the NOI does not appear to recognize all of the associated requirements in section 6(e)(2) of the NFMA. In particular, the NOI's initial listing of the substantive provisions of the 2012 Planning Rule that will govern the proposed amendment process does not mention any of the rule's forest management provisions in 36 CFR 219.11. That section of the Planning Rule, which is titled "Timber requirements based on the NFMA," includes provisions on timber land suitability, timber harvest for purposes of timber production, and timber harvest for purposes other than timber production, along with various limitations on timber harvest.

The EIS for the proposed amendment should address the NFMA's requirements for significant plan amendments. In particular, for each alternative considered, the EIS should "determine the forest management systems, harvesting levels, … and the availability of lands and their suitability for resource management." 16 USC 1604(e)(2). For example, the EIS should estimate not only the different old-growth timber harvest levels of the alternatives – for the purposes of both "proactive stewardship" and "economic reasons" – but also the amount of old-growth forest currently classified as suitable for timber production.

The EIS for the Roadless Area Conservation Rule, completed in 2000, provides a good model for addressing the timber harvest and suitability issues at a national scale. In that EIS, the Forest Service estimated that approximately 9 million acres of roadless areas were classified as suitable for timber production, out of a national total of 47 million acres of suitable timber land. Roadless Rule FEIS, p. 3-194. The EIS included a table that divided the 9 million acres into each of the nine Forest Service regions. FEIS, p. 3-195. Similarly, the EIS provided information about past and planned future timber sales in roadless areas. From 1993 to 1999, 783 million board feet of timber was sold on approximately 80,000 acres of roadless areas, of which one-third was salvage logged. FEIS, p. 3-199. Projected future timber sales in roadless areas for the next 5 years (under the No Action alternative) were estimated to total 1.1 billion board feet on 94,600 acres. FEIS, p. 3-200. Again, a table displayed the timber sale projections for each region. In the effects analysis, the EIS estimated the different amounts of timber that could be harvested from roadless areas under each of the four alternatives considered. In addition, the EIS discussed the different timber harvest systems – for stewardship and production purposes, salvage, helicopter logging, etc. – that have been and would be employed in the roadless area timber sales.

<u>NEPA</u>

We agree with USDA that the proposed amendment process should involve preparation of an environmental impact statement (EIS). Section 6(g)(1) of NFMA requires the Forest Service to adopt regulations to guide the forest planning process, including "direction on when and for what plans an environmental impact statement is required." The 2012 Planning Rule requires an EIS for plan revisions, but for plan amendments the NEPA documentation can be an EIS, an EA, or a CE, "depending upon the scope and scale of the amendment and its likely effects." 36 CFR 219.13(b)(3). The 2012 Planning Rule (as amended in 2016) clarifies that "a proposed amendment that may create a significant environmental effect and thus requires preparation of an [EIS] is considered a significant change in the plan for the purposes of the NFMA and therefore requires a 90-day comment period for the proposed plan and draft [EIS].” Id. A nationwide plan amendment providing new long-term management direction for millions of acres of old-growth forest is certainly a major federal action significantly affecting the environment and therefore requires preparation of an EIS.

As discussed above, it is important that the EIS contains a reasonably thorough discussion of the effects that the proposed amendment would have on timber suitability, timber harvest levels, and harvest methods. We are concerned that the Forest Service’s continued insistence in the NOI that “tree cutting is now a relatively minor threat” to mature and old-growth forests could result in an EIS that fails to disclose information about timber suitability and timber harvest in older forests that is both of high interest to the public and legally required by NEPA and NFMA.

The EIS also must consider a range of reasonable alternatives. An obvious and reasonable alternative to the proposed plan amendment, which focuses on protection of old-growth forest conditions, would be an amendment that applies the same “Standards for Management Actions Within Old-Growth Forest Conditions” to mature forests. We request that the EIS analyze such an alternative.

The range of alternatives should also evaluate different management standards for the Tongass National Forest, apart from the rest of the national forests. Specifically, the EIS should analyze alternatives that include and eliminate the exception for the Tongass proposed in Standard #4.

Durability

Given the importance of older forests, the significance of the threats to their persistence, and the extraordinary amount of time it takes for forests to reach old-growth forest conditions, making policies to conserve older forests durable is essential. As a general matter, we are concerned that the proposed amendment process that USDA is pursuing may not be sufficiently durable to ensure the sustainability of older forests far into the future. For example, there must be commitment by the agency to the goals of this proposed amendment across multiple planning cycles because the achievement of resilient old-growth forest conditions may take centuries in some forest types, especially those that are poorly represented among existing old growth. Durability is a major reason that TWS has advocated for a federal rule to conserve older forests, and we continue to believe that a rule to complement the proposed amendment is another important step that should be taken to ensure older forests persist throughout the National Forest System.

For the same reasons, we also urge the USDA to find ways to make the proposed amendment as durable as possible. For example, the Forest Service could amend the 2012 Planning Rule to require all forest plans to maintain and restore old-growth forests. In the meantime, the Record of Decision for the proposed amendment should specify that any changes to this proposed amendment suggested by local national forests must be approved by the Forest Service Chief or the Secretary of Agriculture.

II. Comments on Proposed Amendment

Below, we comment on specific sections of the proposed amendment and suggest changes to help achieve its intent.

<u>Goal</u>

We enthusiastically support the Goal of promoting tribal sovereignty and co-stewardship and encourage the recommitment expressed in the Management Approach, for example, to be repeated wherever appropriate throughout the proposed amendment. To that end, we suggest adding a Desired Condition of greater tribal inclusion and use of Indigenous Knowledge to complement the proposed Goal. Government-to-government relations, including tribal consultation and developing co-stewardship agreements, take time and must be rooted in reciprocity and trust. We realize that time is of the essence in initiating these arrangements where they do not exist but caution the agency not to let urgency corrupt these essential processes.

We support tribal co-stewardship of older forests on national forest system lands and appreciate that the Goal promotes tribal co-stewardship. We want to take this opportunity to call attention to the comment letter submitted by the Intertribal Timber Council (ITC) to the Forest Service in response to the agency's ANPR on Climate Resilience. ITC offered insightful answers to the questions posed in the ANPR that have relevance to this proposed amendment process, including opportunities for tribal co-management.

While we are excited about the inclusion of the tribal sovereignty and co-stewardship Goal, we question why there is not a second Goal reflecting the intent to foster the long-term resilience of old-growth forest conditions, their qualities, and their contributions to ecological integrity. Every forest plan should include that as a Goal, given that is the proposed amendment's intent.

Last, we are unclear whether the addition of the proposed Goal applies only to the old-growth conservation provisions of the proposed amendment or whether by including it among the Goals of every amended plan, it would guide every aspect of plan implementation. Please clarify.

<u>Management Approach - *Adaptive Strategy*</u>

In many ways, the Management Approach is the lynchpin of the entire proposed old-growth forest initiative. Without the *Adaptive Strategies*, there is no Guideline and therefore no planning mechanism for the recruitment of future old growth. The Standards will protect old growth only until it is lost to natural or human-caused disturbances, and mature forest will be left vulnerable to logging or uncharacteristic disturbances before it can reach the old-growth stage.

Such a critical part of the overall policy cannot be relegated to the status of a non-compulsory, "unenforceable" element of forest plans. The *Adaptive Strategy* must either be made a compulsory and enforceable plan component (i.e., included in a Standard), or the Guideline must be made a Standard, thus requiring the development of an *Adaptive Strategy* to ensure compliance with the amended plan.

The requirement that the *Adaptive Strategy* be completed within two years of finalization of the proposed amendment seems to present an impossible timeline. The Management Approach commits the agency to consultation with tribes and a collaborative process on each forest or group of forests to produce an *Adaptive Strategy*. The experience of the Collaborative Forest Landscape Restoration Program suggests that it will take at least a year to even stand up a credible collaborative group, let alone for it to gel enough to reach agreement on a process that will produce sufficient results.

In addition, it would seem impossible for these collaboratives to "prioritize areas for the retention and promotion of old growth" without adequate information about the location of old-growth and mature forests, which the agency doesn't appear to have a plan for producing, at least for every unit. Because of the significant process required to convene collaborative groups to develop the *Adaptive Strategies*, the time required to consult with tribes, the as-yet-undeveloped information needed to complete an *Adaptive Strategy*, and the work that must be done to identify priority areas and design a program of work for both current and future old growth, we recommend extending the timeline for completion of *Adaptive Strategies* to four years.

Even with an extended timeline, there is no guarantee that all national forests will complete their *Adaptive Strategies* on time. We recommend that the proposed amendment address this possibility by stipulating that if the *Adaptive Strategy* is not adopted in accordance with the Management Approach within four years, then the provisions of the Standards for Management Actions within Old-Growth Forest Conditions shall apply to both old-growth forest conditions and forests that do not meet old-growth definitional conditions until the *Adaptive Strategy* is adopted. This stipulation would help to ensure that older forests are conserved pending adoption of the *Adaptive Strategy*, while also giving forest supervisors an incentive to complete the *Adaptive Strategy* on time.

We find the bullet “Identify criteria used to indicate conditions where plan components will apply” to be confusing, redundant and unnecessary. Plan components should apply across the plan area, where conditions are appropriate to fulfilling the *Adaptive Strategy*. We are also concerned that this language could be interpreted to mean that collaboratives should develop their own criteria for identifying mature and old-growth forests, which would undermine the important need for consistency and make effective monitoring impossible. We recommend it be deleted.

In addition to this deletion, we recommend the inclusion of another bullet requiring the *Adaptive Strategy* to include an adaptive management plan. It is essential to the proposed amendment's success that the *Adaptive Strategy* be revisited regularly and evaluated against monitoring data. The *Adaptive Strategy* should include a detailed plan for collaborative adaptive management that anticipates and addresses challenges that other collaborative adaptive management processes have faced.[2]

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[2] Cheng, A.S., Aplet, G.H., and A. EM Waltz. 2019. Challenges and opportunities for collaborative adaptive management in forest landscape restoration. Pages 119-136 in Butler, W.H., and Schultz, C.A. (eds.). A New Era for Collaborative Forest Management. Routledge: New York and London.

Part (b) of section 1 of the Management Approach should be tightened up to make clear how many units may join to create a Strategy. We suspect that this is included to make room for the Northwest Forest Plan amendment, but it seems to leave the door wide open to ad hoc groupings that may not serve the purpose of the proposed amendment -- for example, single plans developed at the regional level. This section should provide more guidance about the nature of multi-unit plans that could qualify. Also, we are concerned about how it will be determined if these plans "meet the intent" of the proposed amendment and lack of clarity on who makes this decision. This section, at the very least, should be edited to make clear that any such plan must have been developed through a collaborative process.

In addition, we are concerned that the second sentence of part (b) will create a powerful incentive for units with an existing old-growth management strategy to use that strategy as their *Adaptive Strategy*. Existing old-growth strategies were developed without the context of national level mapping and threat assessment or the benefit of the dialogue surrounding the ANPR. They also may not have been developed through a collaborative process, which is an essential part of the Management Approach and is necessary to determine if the strategy "meets the intent" of the proposed amendment. At the very least, part (b) should be modified to make clear that any "already existing strategy or other document" must have been developed through a collaborative process beyond the traditional public engagement efforts for plan revisions for the same purposes as the proposed amendment and that the decision to substitute an existing strategy is subject to approval by the Chief.

As an example, the recently revised Nantahala-Pisgah forest plan includes an "old-growth network" that meets some of the requirements for an *Adaptive Strategy*. However, the plan's old-growth direction still does not contribute to "a consistent approach to manage for old-growth forest conditions" as described in the proposed amendment. While the purpose of the proposed amendment includes both "maintaining and developing old-growth forest conditions," the direction in the Nantahala-Pisgah plan excludes thousands of acres of field-verified old growth as well as areas identified under the previous plan as desirable for a well-distributed, representative old-growth network. These areas would not be subject to the non-degradation Standard of the proposed amendment. Instead, the plan's old-growth network incorporates areas "unlikely to be prioritized" for timber harvests, which resulted in the inclusion of many younger stands that do little to ensure old growth will "be persistent over the long term."

<u>Desired Conditions</u>

We are concerned that the phrase "maintained and improved relative to the existing condition over time" in Desired Condition #1 leaves unclear the timeframe to which "existing" applies and may be interpreted as setting the area of old-growth forest conditions in 2024 as the desired condition. We support a simplified statement that makes clear that the desired condition is for the amount and distribution of old-growth forest conditions to be "increasing."

In the second Desired Condition statement, we support a change to make clear that it is the objective of stewardship activities to "<u>enhance old-growth forest characteristics</u> and foster an increasing trend…." We strongly recommend this change here (and elsewhere, as appropriate, throughout the entire proposed amendment) to clarify that stewardship activities should not simply increase the amount and distribution of forests that meet the minimum threshold definitions for classification as old growth; rather, they should aim to improve the representation of the *qualities* of old-growth forests, including old and large trees with complex features reflecting their age (e.g., large branches, thick furrowed bark, cavities) and other characteristics, like abundant large snags and down wood, where appropriate. We realize these characteristics will vary with forest type, but they should not be diminished through treatments aimed only at increasing "resilience."

Objective

We appreciate the attention to "measurable improvements" in the Objective; however, we are concerned that the Objective apparently would only apply for the first ten years following adoption of the proposed amendment. We recommend additional language making clear that the Objective applies in perpetuity (e.g., insert "and every ten years thereafter" following "within ten years"). We're also concerned about the phrase "at least one landscape" -- both because the term landscape is undefined and the Objective too modest. Also, the term "old-growth desired conditions" is inconsistent with previous language referring to "old-growth forest conditions" and does not reflect the need to account for old-growth forest *characteristics* in forests that have not yet reached criteria defining old-growth forest conditions. We suggest changes to the language accordingly. Finally, the last sentence can be made more parsimonious through a proper definition of "stewardship" that includes retention, recruitment, and natural succession.

Standards

We applaud the language of Standard #1 that prevents the degradation of old-growth composition, structure, or ecological processes, as well as the intent of Standard #2 to promote old-growth characteristics. However, we are concerned that the language of both Standards leaves the door open to activities that will diminish old-growth character. Specifically, we are concerned that the phrase that follows "ecological processes" in Standard #1 could be interpreted to allow degradation as long as old-growth forest conditions persist somewhere in the "plan area."

Similarly, Standard #2 seems to allow for degradation of old-growth characteristics as long as stewardship activities are aimed at achieving "one" of the listed objectives. For example, it appears that stewardship activities may reduce the "amount, density, and distribution of old trees" all the way down to the minimum threshold definitions for classification as old growth as long as the activities are conducted for the purpose of facilitating the "return of appropriate fire disturbance regimes." This kind of "thinning to the minimum" has for several years represented a misguided agency approach to managing old growth. As a result, many large trees that contribute to old-growth character have been logged and sold in fuel treatment projects (over the objections of citizens concerned about old-growth conservation) without technically contributing to a reduction in old-growth area.

We encourage two changes to address these shortcomings. First, we recommend striking the phrase "in a manner that prevents the long-term persistence" following "ecological processes" and the phrase "in the plan area" at the end of Standard #1 to make clear that degradation of composition, structure, or ecological processes within old-growth stands is a hard line that may not be crossed. Second, we recommend that Standard #2 include language to make clear that the minimum criteria for classification of old growth used in the federal inventory are not to be used to guide stewardship. Instead, all actions should enhance old-growth character and not drive stands toward the minimum threshold. In addition, we suggest striking all the language of Standard #2 following "proactive stewardship," and defining "stewardship" in the glossary as actions that enhance old-growth characteristics, including activities that promote the objectives now included in Standard #2.

While TWS supports science-based active management in appropriate areas, we are concerned that the proposed amendment places no restrictions on road construction associated with active management, particularly commercial thinning. The negative ecological impacts from roads and road construction include habitat fragmentation, water quality degradation, and the spread of invasive species. The impacts can be pervasive and profound and are well-documented.[3] Accordingly, we recommend that the proposed amendment provide a definition of "vegetation management" that includes associated road construction or reconstruction. That would make it clear that the non-degradation requirement in Standard 1 and the proactive stewardship requirement in Standard 2 both apply to road building.

We are also very concerned about the broad implications of Section (b)(v.), the exception for "cases where it is determined that the direction in this amendment is not relevant or beneficial to a particular forest ecosystem type." We acknowledge that there may be situations on the national forests where it simply does not make sense to manage for old-growth forest conditions, such as in plantations of exotic species or where native species have been planted "off site" for the purpose of timber production, shelterbelts, etc.; however, we feel that the current language is far too broad and leaves too much to discretion for arbitrary decision-making. The simplest solution is to delete the exception. At the very least, it should be restricted to list the specific conditions to which it would apply or to apply it only to situations of ecological or ecocultural restoration or scientific research.

We appreciate the intent behind Standard #3; however, we are concerned that it leaves open the possibility that "economic reasons" may be considered a secondary purpose in the "statement of purpose and need" for an ecological restoration project. The latitude this affords is inappropriate, since economic reasons should not guide proactive stewardship in old-growth forests. We recommend striking "primary" from the standard.

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[3] Gucinski, H., M. J. Furniss, R. R. Ziemer, and M. H. Brookes. 2001. Forest roads: a synthesis of scientific information. Gen. Tech. Rep. PNWGTR-509. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, OR. http://www.fs.fed.us/pnw/pubs/gtr509.pdf. Trombulak S., and C. Frissell. 2000. Review of Ecological Effects of Roads on Terrestrial and Aquatic Communities. Conservation Biology 14(1): 18-30. Switalski, T.A., J.A. Bissonette, T.H. DeLuca, C.H. Luce, and M.A. Madej. 2004. Benefits and impacts of road removal. Frontiers in Ecology and the Environment. 2(1): 21-28. Available at: http://www.fs.fed.us/rm/pubs_other/rmrs_2004_switalski_t001.pdf. Coffin, A. 2006. From roadkill to road ecology: A review of the ecological effects of roads. Journal of Transport Geography 15: 396-406. Fahrig, L., and T. Rytwinski. 2009. Effects of roads on animal abundance: an empirical review and synthesis. Ecology and Society 14(1): 21. Available at: http://www.ecologyandsociety.org/vol14/iss1/art21/. Robinson, C., P.N. Duinker, and K.F. Beazley. 2010. A conceptual framework for understanding, assessing, and mitigation effects for forest roads. Environmental Review 18: 61-86.

Regarding Standard #4, which provides a partial exemption of the Tongass National Forest, see our comments in the "Regional Issues" section below.

Guideline

We very much appreciate the language of the Guidance that provides for the recruitment of future old growth. This is the only part of the proposed amendment that addresses this crucial aspect of old forest conservation and is an essential part of the proposed amendment. That said, as a Guideline, it lacks the "teeth" of a Standard, and, absent an *Adaptive Strategy* that identifies sufficient future old growth in priority areas, the Guideline could be rendered meaningless. If a Forest simply refuses to produce an adequate *Adaptive Strategy*, there is, in effect, no Guideline.

Relying for such a critical element of old forest conservation on "optional plan content" and an "unenforceable" plan component undermines the entire intent of the proposed amendment and is inconsistent with Executive Order 14072. The proposed amendment should be modified to reinforce the conservation of future old growth. One possible solution is for either the Management Approach or the Guideline to be elevated to a Standard so that the agency can be held accountable for this aspect of old-growth conservation; alternatively, a new Standard could be added to provide a strong incentive to comply with the intent of the proposed amendment. We reiterate our suggestion that a new Standard be added stating: "If the *Adaptive Strategy* is not adopted in accordance with the Management Approach within four years of the date of this proposed amendment, then the provisions of the Standards for Management Actions within Old-Growth Forest Conditions shall apply to both old-growth forest conditions and forests that do not meet old-growth definitional conditions until the *Adaptive Strategy* is adopted in accordance with the Management Approach."

We also recommend that the proposed amendment define "landscape-level proactive stewardship activities." If "stewardship" is defined sufficiently, it should be unnecessary to modify it with "landscape-level" or "proactive." If the term is retained, it must be defined. Included within a sufficient definition of "stewardship" should be "activities" that extend well beyond "vegetation management" to include protective activities, like travel management decisions that mitigate impacts from off-highway motor vehicles, road decommissioning and restoration, etc. It should also include decisions to allow natural succession to proceed unaided.

Monitoring

We are encouraged by the commitment to monitoring evident in the monitoring section, especially the Chief's commitment to developing the National Old-Growth Monitoring Network. It is not clear that a nationwide plan amendment can actually drive the establishment of the Network, but we are nevertheless pleased to see it referenced. The creation of the Network will require effort supplemental to the proposed amendment (see discussion of monitoring in the Implementation Issues section below). Similarly, it is unclear whether a national amendment would create substantive requirements to provide the "regular updates on measurable changes in unit-level old-growth forest conditions," as required by the monitoring section.

Delivering on these commitments will require strong leadership and follow-through (see discussion of leadership below). Because a nationwide plan amendment appears to have limited authority to establish a monitoring strategy, we recommend that, concurrent with the drafting of the EIS, the Chief develop and publish a document, similar to the Wildfire Crisis Strategy,[4] committing the Forest Service to a nationwide old forest monitoring strategy and dedicate resources to support staff and public participation in the strategy.

We are very concerned that the Management Approach section says the *Adaptive Strategy for Old-Growth Forest Conservation* – which is critical to the success of the proposed amendment – is anticipated to be published as part of either the "broader scale monitoring strategy" or the "biennial monitoring report." This is an unreliable element of the proposed amendment. Despite the requirement in the 2012 Planning Rule that monitoring plans be modified by 2016 to "meet the requirements of [the Rule]," most forests don't have a plan-level monitoring program. In addition, even the forests that have completed planning since the 2012 Rule have a spotty record of completion of plan-level monitoring programs (or the "biennial reports"). Therefore, it is unclear what the inclusion of monitoring questions and indicators will achieve or what the fate of the *Adaptive Strategies* will be without the existence of a broader-scale monitoring strategy or publication of a biennial monitoring report. This is a significant weakness of the proposed amendment.

In addition, some aspects of the monitoring section would benefit from additional attention. For example, the name "*Adaptive Old-Growth Conservation and Management Strategy*" is different from the "*Adaptive Strategy for Old-Growth Conservation and Management*" referred to in the Management Approach section. The first indicator refers to "changes in trends in amounts and distributions;" this is not the same as what is asked in the first monitoring question, which refers to "amount, representativeness, redundancy, and connectivity." At the very least, the indicator should match the question. Even better would be if the biennial monitoring report included information about the status of "old-growth forest characteristics" or the qualities of old-growth and older forests included in priority areas.

In addition, the wording "changes in trends" is awkward. It would seem to require reporting only on rates of change, not on status. We recommend changing it to "changes in status and trends."

Finally, the second monitoring question is restricted only to management activities "within old-growth forest." It leaves out changes in future old growth. The question should be redrafted to read: "Are vegetation management activities promoting desired old-growth forest composition, structure, pattern, and ecological conditions?"

III. Implementation

While amending the forest plans is an important step to conserving older forests and their associated values across the National Forest System, several additional steps must accompany the proposed amendment to ensure it achieves its intent.

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[4] USDA Forest Service. January 2022. Confronting the Wildfire Crisis: A Strategy for Protecting Communities and Improving Resilience in America's Forests. FS-11871. Available at: https://www.fs.usda.gov/sites/default/files/fs_media/fs_document/Confronting-the-Wildfire-Crisis.pdf.

First, the proposed amendment includes Standards intended to prevent degradation of stands where old-growth forest conditions are currently expressed. If these Standards are to be effective, units will bear the responsibility of field verifying whether any given stand meets the minimum criteria of old-growth forest conditions during project development. We are concerned about the lack of a formal process to resolve the inevitable conflicts upon implementation of the proposed amendment as to whether a stand meets the minimum criteria for old-growth forest conditions, and, therefore, whether the Standards apply. While the agency has existing inventory protocols to support project-level planning, such as the Common Stand Exam, we are skeptical that these sampling designs will be sufficient to determine whether a stand meets all the criteria for old-growth forest conditions. We are aware of instances where such ambiguities result in disputes between agency staff and key stakeholders, which further erodes trust between the public and the agency. For example, nearly all the definitions of old-growth forest conditions developed by the Regions include a minimum stand age, yet stand age is notoriously one of the most challenging variables to estimate in the field due to complex stand histories, uneven age distributions, the consequences of which trees to select for ageing, and the difficulties of interpreting and verifying tree ring counts. We expect there to be disagreement among agency staff and key stakeholders, including collaboratives charged with developing the *Adaptive Strategies*, as to whether a stand currently meets the definition of old-growth forest conditions. We encourage USDA to detail a formal dispute resolution process, including a more thorough field reconnaissance, that can be triggered when stands meet some, but not all, of the minimum criteria of old-growth forest conditions. We understand this may be viewed as burdensome, but we believe that correctly identifying stands that meet current old-growth forest conditions is paramount to the successful implementation of the proposed amendment.

Second, a critical element to the development of adequate *Adaptive Strategies* is delivery of the best available spatial information that describes the current distribution of old-growth forest characteristics consistent with “definitions and inventories (that) have been established for forests exhibiting old-growth conditions.” Without an *Adaptive Strategy* that identifies all existing old growth and sufficient “areas that currently do not meet old-growth definitional conditions... as a priority for the future contribution of the development of those conditions over time,” the proposed amendment will be unable to achieve its intent “to manage for old-growth forest conditions with sufficient distribution, abundance, and ecological integrity...to be persistent over the long term, in the context of climate amplified stressors.”

The Forest Service must make available to units and associated collaboratives the best scientific information on the location of forests exhibiting old-growth characteristics. At a minimum, all units should have access to the same set of spatial information describing where on the landscape old-growth forest characteristics may be more likely expressed. This information must be of sufficiently high resolution to serve the intent of the proposed amendment; data that are too coarse (e.g., firesheds) will inevitably blur important fine scale heterogeneity and mask opportunities to retain and proactively steward old-growth forests. TWS published such a map to demonstrate how permanent inventory plots administered by the Forest Inventory and Analysis (FIA) program can first be classified as meeting the criteria for old-growth forest conditions and subsequently paired with imputation techniques to map old-growth forest conditions throughout the country.[5]

We are keenly aware and appreciate that the USDA Forest Service has invested considerable resources into the BIGMAP project, an effort to more accurately describe the spatial distribution of forest attributes by employing sophisticated plot imputation techniques beyond those used by TWS. Unfortunately, the broader scientific community is unable to take part in the coproduction of maps describing the spatial distribution of old-growth forest conditions in a manner consistent with regional definitions of old growth because of privacy laws intended to maintain the integrity of the FIA plot network. These restrictions convey an exceptional responsibility on the agency to deliver the highest quality spatial information describing the distribution of old-growth forest conditions, since no other spatial data products exist. A more aspirational process would be to pair these predictions with the best available data at the unit level, including information collected by citizen groups, to provide a top down/bottom up estimate of the location of forests exhibiting old-growth characteristics (i.e., old-growth and mature forests).

Third, we applaud the language of the Management Approach requiring the *Adaptive Strategy* to "prioritize areas for the retention and promotion of old-growth forest conditions." We believe that such an "area-based approach" is essential to identifying sufficient current and future old growth to achieve the proposed amendment's intent "to manage for old-growth forest conditions with sufficient distribution, abundance, and ecological integrity (composition, structure, function, connectivity) to be persistent over the long term…." The goal of the area-based approach should be to identify areas for conservation of current and future old-growth forest conditions with the following parameters:

- in every forest type;
- with sufficient redundancy to endure expected disturbances;
- well-distributed across each forest (including both reserves and unreserved areas);
- oriented to facilitate habitat connectivity and minimize fragmentation; and
- composed of patches of sufficient size to support old-growth-dependent species.

The system of old-growth conservation areas should be of sufficient size and distribution to achieve long-term persistence in the face of climate-amplified stressors. The system could potentially be created without requiring a total area target to be determined in advance (e.g., through an analysis of historical old forest distribution).[6] Conservation areas will likely need to be on the order of several thousand acres to meet habitat requirements of old-growth-dependent species, as was the case in the various conservation strategies developed in the 1990s for old growth in the Pacific Northwest and Sierra Nevada and likely larger than even the "Large-sized Areas" recommended in the Region 8 Old-Growth Guidelines.

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[5] Barnett K., Aplet G.H. and Belote R.T. 2023. Classifying, inventorying, and mapping mature and old-growth forests in the United States. Front. For. Glob. Change 5:1070372. doi:10.3389/ffgc.2022.1070372.

[6] We believe that upon synthesis of the above information sources, the total area by which the standards apply on each unit must be equal to or greater than the estimated area of current old-growth forest conditions provided by the strategic-level inventory. This will provide necessary credibility that the plan components – specifically the standards on non–degradation of current old–growth forest conditions – are implemented to their fullest potential due to imperfect information.

Priority areas for future old growth should be oriented around existing patches of old growth to the maximum extent feasible but should not be limited to occurrences of existing old growth. Because the prioritization process is essential to the success of the proposed amendment, we recommend the Forest Service convene, at the earliest possible date, a team/Committee of Scientists or even a Federal Advisory Committee with significant representation by scientists and tribal representatives and Indigenous Knowledge holders to refine this process for use by the collaborative groups.

Fourth, while we agree that the collaborative process can be an appropriate mechanism to organize a diverse public when developing *Adaptive Strategies*, we believe that the process and expectations of collaboration should not be left to individual units to invent in isolation. We encourage the USDA Forest Service to share a model collaborative process that details its form and function to increase the likelihood that an adequate *Adaptive Strategy* will be produced. Such a model should include the steps to convene and conduct collaboration and draw upon syntheses of critical factors to success and the many lessons learned from related collaborative programs.[7]

In addition, sufficient technical support must be available to all units that draws upon the breadth of expertise found across USDA Forest Service programmatic areas, Tribal Nations, the academic community, non-governmental organizations, and other relevant entities. Besides identifying priority areas, as described above, the collaborative groups developing the *Adaptive Strategies* should detail the types of stewardship activities that are (in)appropriate across the landscape, as well as the expected consequences of those activities on the retention and recruitment of old-growth forest characteristics, and describe the process for adaptive management – including across planning cycles - that will be used during collaborative implementation, including the process for updating the *Adaptive Strategy* over time. Development of the *Adaptive Strategy* cannot be a "one and done" event.

Fifth, given the importance of adaptive management to the success of the *Adaptive Strategies*, and the foundational role that monitoring plays in adaptive management, we encourage the USDA Forest Service to hasten development of the National Old-Growth Monitoring Network and not wait until the conclusion of the amendment process. The monitoring network's role in informing adaptive management must be articulated in the context of collaboration. Such a monitoring network must be conceived to address both the plan-level monitoring questions included in the proposed amendment and report on status and trends upward to the national level. Careful consideration of the scale at which old-growth forest characteristics are expressed and affected by management activities will be essential so that inferences drawn from the monitoring are statistically robust and relevant to the adaptive management process.

An effective monitoring program is likely to require information that captures changes in old-growth forest conditions at a finer spatial scale than can be achieved through existing strategic-level monitoring programs like FIA. For example, our experience participating in collaborative forest landscape restoration indicates that a relatively high density of monitoring plots will be necessary to detect change in condition, given the significant variability in pre-treatment forest conditions, multiple treatment objectives, and alternative proactive stewardship activities.

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[7] Butler, W.H, and Schultz, C.A. 2019. A New Era for Collaborative Forest Management: Policy and Practice Insights from the Collaborative Forest Landscape Restoration Program. Routledge: New York and London.

Furthermore, reliance on FIA data alone will fail to capture important characteristics of old growth in frequent fire ecosystems where proactive stewardship activities often seek to restore horizontal heterogeneity and the distribution of individuals, clumps, and openings.[8] FIA plots simply have too small footprints to monitor such desired conditions. Wall-to-wall remote sensing products can be useful to describe status and trends at broad spatial extents when validated by FIA data, as is the case of monitoring under the Northwest Forest Plan,[9] but they are likely insufficient to inform adaptive management within planning units.[10]

These issues are exacerbated when proactive stewardship activities intended to develop old-growth forest characteristics are not easily measured through traditional inventory methods or easily quantified through metrics describing forest structure, function, or composition, but nonetheless are permitted under the proposed amendment (e.g., cultural uses). The co-stewardship process must be articulated and built into the development of the *Adaptive Strategy*, including monitoring, from the beginning. For these reasons, we encourage the USDA Forest Service to prioritize the careful development of an adequate monitoring network to address the above challenges.

Sixth, as we've discussed, the Management Approach and development of the *Adaptive Strategy* is the cornerstone of the proposed old-growth forest policy. We suggested revisions to the proposed plan components to help ensure the successful development of the *Adaptative Strategy.* In addition to these suggestions, we also encourage the Forest Service to adopt a performance measure related to the development of the *Adaptive Strategy*. The agency adopted a performance measure to ensure expedited compliance with the 2005 Travel Management Rule and its requirement that units develop Motor Vehicle Use Maps (MVUMs), and nearly all national forest units published MVUMs within a relatively short timeline.

To complement our suggested performance measure to accelerate development of *Adaptive Strategies,* the agency may consider carefully selecting a few "early implementer" units that have already made significant progress towards solving several of the implementation challenges (e.g., tribal co-stewardship, collaboration, mapping of existing old growth and mature forest, monitoring, supportive leadership) to create strong models for other forests to follow. It would be wise to select units from across the National Forest System that have strong familiarity with old-growth forest management. While it is assumed that these units will face fewer barriers in the development of *Adaptive Strategies*, their experiences handling the many unforeseen challenges

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[8] Churchill, D.J., Larson, A.J., Dahlgreen, M.C., Franklin, J.F., Hessburg, P.F., and J.A. Lutz. 2013. Restoring forest resilience: From reference spatial patterns to silvicultural prescriptions and monitoring. Forest Ecology and Management (291): 442-457.
[9] Davis, Raymond J.; Bell, David M.; Gregory, Matthew J.; Yang, Zhiqiang; Gray, Andrew N.; Healey, Sean P.; Stratton, Andrew E. 2022. Northwest Forest Plan—the first 25 years (1994–2018): status and trends of late-successional and old-growth forests. Gen. Tech. Rep. PNW-GTR-1004. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 82 p. https://doi.org/10.2737/ PNW-GTR-1004.
[10] Bell, David M.; Gregory, Matthew J.; Palmer, Marin; Davis, Raymond. 2023. Guidance for forest management and landscape ecology applications of recent gradient nearest neighbor imputation maps in California, Oregon, and Washington. Gen. Tech. Rep. PNW-GTR-1018. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 41 p. https://doi. org/10.2737/pnw-gtr-1018.

could nonetheless help the agency identify key knowledge gaps and programmatic areas for long-term investment.

Finally, we believe that the proposed amendment is unlikely to succeed without strong, supportive leadership from the Chief on down. With so many critical issues left unresolved by the proposed amendment, it will take firm and steady guidance from everyone in a leadership position to ensure the success of this initiative. We are encouraged to see in the proposed amendment that the Chief is committed to establishing a National Old-Growth Monitoring Network; however, success of the Network will require not just a few staff positions allocated at the top but commitment from regional foresters, forest supervisors, and district rangers to ensure that the necessary data are collected and evaluated, especially given the dismal history of forest-level monitoring in the agency.

Similarly, the collaborative process for development and implementation of the *Adaptive Strategies* will need to be supported at every level. Historically, support for collaboration has been spotty, and even functional collaboratives have suffered from turnover due to the agency's system of transfers and details. All staff need to be assured that it is their duty to participate in collaboration in good faith, and these commitments should be repeatedly and publicly demonstrated.[11]

Leadership must also set the tone for cooperation within the Forest Service. Success of this proposed amendment will depend on good working relationships between the National Forest System, Research and Development, and the Forest Inventory and Analysis program. Old rivalries and resentments cannot be allowed to obstruct the teamwork that will be needed. We believe the Forest Service would do well to establish a system of rewards and awards for demonstrating commitment to old-growth conservation, dedication to collaboration, and a cooperative spirit in support of old-growth inventory and monitoring.

The spirit of collaboration and cooperation must be extended to tribes wherever possible. Federal policy requires "consultation," which has historically translated only to "notification." We are encouraged by the language of the proposed amendment committing the Forest Service to "co-stewardship," but we also recognize that a "culture of co-stewardship" must be built within the agency. Formal government-to-government agreements that outline the co-stewardship arrangements between tribal nations and the federal government may need to be prioritized,[12] and tribal nations should be welcomed directly into the collaborative process, which must be grounded in a spirit of reciprocity, not extraction of Indigenous Knowledge. We encourage the Forest Service to consider providing grant funding to Tribes that are interested in participating in the process, similar to how the agency extends financial support to States to engage as cooperating agencies in other NEPA processes.

IV. Regional Issues

[11] Cheng, A.S., Aplet, G.H., and A. EM Waltz. 2019. Challenges and opportunities for collaborative adaptive management in forest landscape restoration. A New Era for Collaborative Forest Management. Routledge.
[12] For examples see Memorandum Of Understanding between the Chippewa National Forest and Leech Lake Band of Ojibwe; Memorandum Of Understanding between the Superior National Forest and the Bois Forte, Grand Portage, and Fond du Lac Bands of Lake Superior Chippewa.

Eastern Old Growth

Region 8 old-growth guidance directed each unit to develop a network of small, medium, and large patches to be managed for old-growth conditions, and many units have designated areas for those networks. These networks may not be sufficient to meet the patch-size needs of all old-growth-associated species or all needs identified through consultation and collaboration, but the portions not already in old-growth condition should be included in the priority areas identified in the Adaptive Strategy. To ignore or arbitrarily dismiss these networks in whole or in part would undermine the credibility of agency old-growth planning, including this proposed amendment. Designating stands for an old-growth network and then removing them before old-growth conditions have been achieved would run counter to the purpose of the amendment. Unfortunately, that is precisely what happened when the revised Nantahala-Pisgah Forest Plan created a new old-growth network that excluded over 2,000 acres of small old-growth patches that were included under the prior plan and in accordance with the Region 8 guidance. Networks should be durable, as long as they continue to meet the purpose of the amendment, and help provide the long-term consistency necessary for successful stewardship and recruitment.

Across the eastern US, many units have scarce remnants of old growth within extensive landscapes of mature forest. This differs significantly from pre-colonial conditions when old growth was typically abundant. Hence, these units have both a substantial old-growth deficit and a major opportunity to restore it. In coming decades, newly recruited old growth from harvested stands will dominate some units' old-growth cover. However, stands that have just reached old-growth status will often differ from those that were never industrially logged in several key aspects: species composition, structural traits such as tree size and coarse woody debris, and services such as carbon storage and habitat quality. To make informed decisions that enhance the value and ecological resilience of old-growth conditions, these differences among different old-growth stands need to be captured. Evaluating old-growth conditions requires assessing not only the extent and distribution, but also stand-level characteristics.

Where old-growth forest conditions remain, they may be difficult to recognize. Industrial logging operations circa 1900 bypassed forest stands most often due to low commercial value, commonly associated with poor growing conditions. Relatively small trees frequently dominate these stands. As a result, the stands do not match stereotypical images of old growth, and their age may go unnoticed. Additionally, natural uneven age structures within most eastern forest types can complicate the identification and aging of the oldest age class. In formerly open stands, fire suppression has also allowed younger cohorts to fill in around older age classes. These challenges highlight the need for careful review of stands in proposed projects, taking these factors into account.

Across much of the Eastern US, the risk of stand-replacing disturbances remains low. Consequently, there are limited opportunities for active management to mitigate threats to those forests because there are few threats. Indeed, fire suppression has been so effective in the East that there is a deficit of stand-replacing fire relative to the natural range of variation. Active management, in particular timber harvests, can also create a threat through its potential to introduce invasive species. In the Guideline and elsewhere, the proposed amendment should make clear that "proactive stewardship" includes natural succession.

Northwest Forest Plan Amendment

At the same time that the USDA is amending all national forest management plans to conserve old-growth forests, it is also amending the Northwest Forest Plan (NWFP) to address five interrelated issues, one of which is "improving conservation and recruitment of mature and old-growth forest conditions…." 88 Fed. Reg. 87393, 87395 (Dec. 18, 2023). The NOI mentions the concurrent NWFP amendment process in its discussion of "areas of agreement" that emerged from the ANPR comments – specifically, the agreement that "differences in threats and conditions in different regions and ecosystems will require additional consultation with Tribes … and place-based collaboration to develop geographically informed adaptive management strategies." NOI at 88044. The NOI states that the USDA's appointment of a NWFP Federal Advisory Committee (FAC) last year to guide the NWFP amendment is an example of tribal consultation and place-based collaboration to develop a geographically informed adapted management strategy.

TWS commends the USDA both for instituting a NWFP amendment process to conserve and recruit mature and old-growth forests in the Pacific Northwest and for establishing the NWFP FAC. Obviously, it is important that the proposed Amendment work together with the NWFP Amendment as much as possible. Key outcomes for Pacific Northwest forests include:

- Greater protection for the 1.5 million acres of old-growth and late successional/mature forests that the NWFP purposely made available for commercial logging when it was adopted in 1994 but largely remain unlogged and intact today.
- Increased ecological restoration activity in relatively dry, frequent-fire forests in the eastern and southern portions of the NWFP area to reduce their vulnerability to extreme wildfire and drought exacerbated by climate change.
- Greater co-stewardship of resources that are important to tribes in the NWFP areas, based on traditional knowledge, treaty rights, and an indigenous ethic of reciprocity.

We hope these outcomes will be achieved under the suggested actions for the proposed amendment and the NWFP amendment. There is much that the two amendment processes can learn from each other, and we strongly encourage the Forest Service to make sure that the agency planners for the two processes are consistently interacting.

Tongass Exemption

The proposed amendment singles out the Tongass National Forest for exclusion from its protections by creating an exception for old-growth logging under the Southeast Alaska Sustainability Strategy (SASS). This exception is deeply concerning. The SASS sets out in many respects a positive direction for the Tongass. Indeed, the Forest Service restored the Roadless Rule on the Tongass, has been redirecting its resources to invest in forest restoration, recreation, and resilience, and is centering collaboration and partnerships in its work. TWS supports these changes, and we believe that protecting the Tongass' old growth is critical. Exempting the Tongass from the protections being considered for every other national forest in the country seems unnecessary and inconsistent with the purpose and need of the nationwide land management plan amendment as well as President Biden's Executive Order 14072.

TWS supports the primary purposes of the SASS and understands the Forest Service's desire to avoid having conflicting policies related to old growth on the Tongass. The SASS has very little durability and could be easily amended and/or revoked by any future administration with no public input or additional process. TWS sees the proposed amendment as an opportunity to enhance the durability of a key aspect of the SASS, namely ending large-scale, old-growth timber harvest and focusing resources to support forest restoration, recreation, climate resilience, and sustainable young-growth management.[13] Indeed, this is precisely what the proposed plan Standards attempt to address.

Further, an important part of the SASS is to ensure Tribal Nations will be provided opportunities to describe, identify, or remove cultural wood to maintain for future generations or for uses such as totem poles, canoes, and tribal artisan use. As put forward, the proposed amendment would provide for an exception to Standards 2 and 3 to allow for culturally significant uses. Thus, it appears that the proposed amendment (without the Tongass exception) and the SASS are, in many ways, consistent with each other.

We offer three recommendations. It is imperative that the Forest Service consult with Southeast Alaska Tribes regarding this plan amendment and its proposed exception for old-growth logging on the Tongass. The Forest Service must analyze an alternative that does not provide an exception for the Tongass. If this exception is analyzed in an alternative in the DEIS, we request that it be modified as follows:

- Clarify that the exception to these standards be provided on a case-by-case basis for individual projects.
- The approval to grant the exception should be elevated from the Alaska Region Forester to the Chief.

In conclusion, TWS greatly appreciates the USDA Forest Service's proposal to conserve old-growth forests through a nationwide forest plan amendment. We look forward to working with you to achieve a strong, durable, and implementable policy to provide the immense benefits of old-growth forests to current and future generations.

Sincerely,

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[13] Available online at : https://www.usda.gov/media/press-releases/2021/07/15/usda-announces-southeast-alaska-sustainability-strategy-initiates.

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