

Wilderness-Based Ecosystem Protection in the Northern Rocky Mountains of the United States

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Abstract—Wilderness is a source habitat for grizzly bear (*Ursus arctos*) and bull trout (*Salvelinus confluentus*) populations in the northern Rocky Mountains of the United States, helping sustain these indicators of ecosystem health. The spatial distribution of grizzly bear mortalities has changed since the end of legal hunting seasons, increasing the source potential of wilderness habitat. Due to its fragmented spatial distribution, wilderness alone cannot sustain viable populations, and wilderness refugia must be linked through strategies that include restoration. A wilderness-based reserve network of 140,000 km² is proposed as a foundation for ecosystem protection. A legislative version is before the U.S. Congress as The Northern Rockies Ecosystem Protection Act, H.R. 488.

The northern Rocky Mountains represent a unique landscape in the contiguous United States. It supports free-ranging populations of native large mammals and migratory fish within predator-prey systems, while such systems have largely disappeared from other landscapes. The presence of large areas of federal public lands, wilderness areas and national parks may be a critical factor in the persistence of these unique landscape features.

Biodiversity conservation at the landscape level encompasses thousands of species, about many of which we know little or nothing. For practical reasons, we focus our conservation plans on a few species that serve as indicators of ecosystem health and integrity.

Wilderness habitat was analyzed for its role as refugia for species that are primary indicators of the health of terrestrial and aquatic ecosystems. Data on grizzly bear (*Ursus arctos*), and bull trout (*Salvelinus confluentus*), were analyzed for their relationships with wilderness habitat. These species are listed as threatened under the Endangered Species Act and as Management Indicator Species within the national forest planning process. Both are wide-ranging. Grizzly bears have ranges up to 5,374 km² (2,075 mi²) (Blanchard and Knight 1991), while bull trout undergo lengthy spawning migrations up to 250 km (155 mi) (Fraleigh and Shepard 1989). Both have low reproductive rates (U.S. Fish & Wildlife Service 1993, 1994). Both bull trout (Fraleigh and Shepard 1989) and grizzly bears (French and French 1989) are known to engage in predatory behavior. They fit the definition of umbrella species (Miller and others 1999), under whose span numerous other species might be protected.

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In this paper, I address four central questions:

- Is wilderness habitat a source habitat for grizzly bears?
- Is wilderness habitat a source habitat for bull trout?
- Has legal hunting for grizzly bears had an effect on the spatial distribution of grizzly bear mortalities?
- Can wilderness serve as a foundation for effective ecosystem protection?

Methods

Analysis Area

The area analyzed is the northern Rocky Mountains of the United States, generally bounded by the 49° and 42° latitudes and 119° and 105° W. longitudes (fig. 1). Study regions for grizzly bear are the Greater Yellowstone Ecosystem (GYE) in Wyoming, Montana, and Idaho and the Northern Continental Divide Ecosystem (NCDE) in Montana; the study area for bull trout is the Columbia River Basin generally east of the 119° W. longitude. Relevant data layers for contiguous Canadian lands were unavailable, although protection of habitats in Canada is important to conservation of these international populations.

Analysis of Wilderness and Indicator Species

I define wilderness habitat as congressionally designated wilderness, inventoried roadless areas and roadless national park lands.

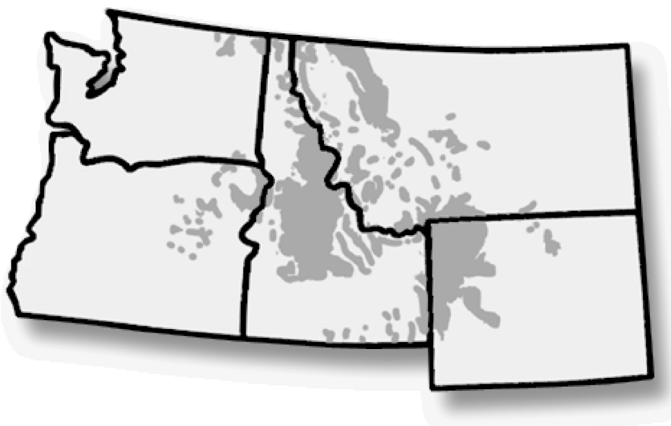


Figure 1—Northern Rockies analysis area.

A Geographic Information System (GIS), Arc/Info 7.11 and ArcView 3.0 with Spatial Analyst (Environmental Systems Research Institute 1997) were used to analyze data layers beginning with a public lands layer (National Center for Geographic Information and Analysis 1996). A digital public lands roadless area database for 27 national forests has been built over several years with information obtained through the Freedom of Information Act process and cooperative arrangements. All recorded GYE grizzly bear mortalities for which Universal Transverse Mercator (UTM) coordinates were available from 1959-1998 (n = 641) (Craighead and others 1988; Montana Department of Fish, Wildlife and Parks 1998), and NCDE mortalities from 1980-1997 (Dood and Pac 1993; Pac and Dood 1999) (n = 229), were digitized for analysis. An undigitized database of NCDE mortalities from 1970-1979 (Dood and others 1986) were duplicated on transparent overlays for analysis. Undigitized mortality data for 1998 were classified based on descriptive information in reports. Mortality locations for the Selkirk Mountains were unavailable, and sample sizes for the Cabinet-Yaak area were considered too low.

Results reported by Mattson and others (1992) suggest that human-habituated grizzly bears which frequented zones within 2 km (1.24 mi) of roads, and within 4 km (2.49 mi) of major developed areas, faced greater mortality risks than those that did not. Thus, for grizzly bears, the analysis of wilderness habitat was modified by buffering roads 2 km on either side, major developments to a 4 km radius, and the perimeters of roadless areas 2 km inward.

Two boundaries were used to evaluate grizzly bear mortalities. The first used grizzly bear recovery area boundaries (U.S. Fish and Wildlife Service 1993). The second used the mortality locations and the computer program HOME RANGER (Hovey 1998) to construct a 97% isopleth distribution boundary. Using the adaptive kernel method, this program can exclude low frequency outlying data points and includes probability density functions. While designed to describe utilization distribution for individual animals, it was found to be useful for describing distribution of grizzly bear populations (Bader, submitted). The minimum convex polygon method (Mohr 1947) was not used since it included vast amounts of agricultural lands and human settlements, and the 98-100% HOME RANGER isopleths were rejected for the same reasons. Grizzly bear mortality rates were calculated for wilderness and nonwilderness habitats and expressed as mortalities per 100 km² (38.61 mi²) for each habitat type. Wilderness (W), nonwilderness (NW) ratios were calculated.

Mortality data were also bifurcated into years when legal hunting seasons were administered, and years in which they were not. Mean annual mortality totals were calculated for hunting and nonhunting years. Mortality reports were analyzed for cause of death and bifurcated into human-caused and all other causes and percentages calculated for each. Mattson (1998) reported that a majority of reported grizzly bear mortalities were human-caused or related.

Since specific demographic and mortality data for bull trout were not available, digital layers for bull trout distribution and strong populations (Lee and others 1997), aquatic strongholds (Quigley and others 1996) and key watersheds from the INFISH aquatic strategy (U.S. Forest Service 1995) were analyzed and the wilderness, nonwilderness percent composition calculated for each.

Digital presence and absence bull trout data were obtained from the Montana Department of Fish, Wildlife and Parks, Idaho Department of Fish and Game, Oregon Department of Fish and Wildlife and Washington Department of Wildlife. These data could not be analyzed for their composition due to data interface problems. These data were overlaid onto the wilderness layer for visual analysis. The bull trout analysis area excluded one small population on the east side of Glacier National Park, and data in Nevada were not available.

Due to major differences in life histories and habitat use, the bull trout analysis made use of a separate definition of wilderness habitat. While most recorded grizzly bear mortality is directly human-caused or related (Bader 1989; Mattson 1998) and available in published reports, bull trout mortality is not so clear, and data were unavailable. Bull trout do not have the behavioral interactions with humans that grizzly bears do where use of habitat directly overlaps. Both grizzly bears and bull trout are impacted by roads, but a major difference for bull trout is that virtually all roadless areas are in higher elevation or headwaters areas, and negative effects from roads, such as sediments, are projected downstream, away from roadless areas rather than towards them. Therefore, roads and roadless area perimeters were not buffered for bull trout analysis.

Results

Wilderness and Grizzly Bears

Analysis of all mortalities from all causes in the GYE from 1959-1998 (n = 641) showed that 36.3% of mortalities occurred in wilderness, compared with 63.7% in nonwilderness, and W/NW = 1: 1.8. The per capita ratio per land area is 1: 3 for the recovery zone and 1: 2.1 for the 97% isopleth.

In the NCDE from 1970-1998 (n = 431), 35.7% of NCDE mortalities occurred in wilderness and 64.3% in nonwilderness, and W/NW = 1: 1.8. The per capita ratio per land area is 1: 1.5 for the recovery zone and 1: 1.3 for the isopleth.

The results of mortalities per 100 km² and their spatial distribution are shown in tables 1 and 2 and figures 2 and 3. The density of mortalities differs, based on sample sizes, years of coverage and whether or not hunting was allowed.

The more important figures are the W/NW ratios. A definite shift from W to NW has occurred as a result of an end

Table 1—Grizzly bear mortalities per 100 km² in wilderness and nonwilderness habitat.

	Wilderness	Nonwilderness	W:NW Ratio
Yellowstone ^a	1.454	4.376	1:3.0
Yellowstone ^b	1.292	2.678	1:2.1
NCDE ¹	0.681	0.998	1:1.5
NCDE ²	0.731	0.867	1:1.2

^aCalculated using U.S. Fish & Wildlife Service grizzly bear recovery area boundary.

^bCalculated using 97% isopleth mortality distribution boundary.

Note: Densities differ based on sample sizes, years of coverage, and analysis area.

Table 2—Grizzly bear mortalities per 100 km² among hunted and unhunted populations.

Area	Wilderness	Nonwilderness	W:NW Ratio
Yellowstone ^a			
Hunted	0.946	2.779	1:2.9
Unhunted	0.508	1.597	1:3.1
Yellowstone ^b			
Hunted	0.858	1.610	1:1.9
Unhunted	0.435	1.069	1:2.5
NCDE ^a			
Hunted	0.624	0.738	1:1.2
Unhunted	0.058	0.259	1:4.5
NCDE ^b			
Hunted	0.670	0.639	1:1
Unhunted	0.062	0.228	1:3.7

^aCalculated using U.S. Fish & Wildlife Service grizzly bear recovery area boundary.

^bCalculated using 97% isopleth mortality distribution boundary.

Note: GYE hunted years are from 1959-1974 and unhunted years are from 1975-1998. NCDE hunted years are from 1980-1991 and unhunted years are from 1992-1997. UTM locations for NCDE mortalities 1970-1979 and 1998 were unavailable and are not included in the figures in this table.

to legal grizzly bear hunting. This has increased the source potential of wilderness habitat for grizzly bears.

As a straight percentage of all mortalities during hunted years in the GYE, 38.7% were in wilderness and 61.3% in nonwilderness, and W/NW = 1: 1.6. During nonhunted years, 32.5% were in wilderness, 67.5% in non-wilderness,

and W/NW = 1: 2.1. In the NCDE during hunted years, 40.2% were in wilderness, 59.8% in nonwilderness, and W/NW = 1: 1.5. During nonhunted years, 10.9% were in wilderness, 89.1% in nonwilderness, and W/NW = 1: 8.2.

This shift is much more pronounced in the NCDE, although sample sizes were smaller. The shift was more pronounced in the GYE within the FWS recovery area; in the NCDE, the shift was more pronounced within the 97% isopleth.

The undigitized NCDE mortalities from 1970-1979 (not included in tables or figures) were 44.1% in wilderness and 55.9% in nonwilderness, and W/NW = 1: 1.3. However, hunting kills were 64.5% in wilderness and 35.5% in nonwilderness, and W/NW = 1.8: 1, while nonhunting kills were just 14.5% in wilderness and 85.5% in nonwilderness, and W/NW = 1: 5.9. UTM coordinates or specific cause other than hunting and nonhunting were not available, and these data underwent no additional analysis.

The undigitized 1998 NCDE mortalities were 5.9% in wilderness and 94.1% in nonwilderness.

The NCDE results were influenced by including some outlying mortalities, which greatly increased the total area of nonwilderness habitat used for calculations. This decreased the density of mortalities for nonwilderness habitat somewhat unrealistically. One can get different results depending on where the analysis boundary is drawn, and the results from the 97% isopleth may mask some of the source/sink effects in localized areas. Despite these effects, the two boundaries both show the same trends, consistent with other reported results (Doak 1995; Dood and others 1986;

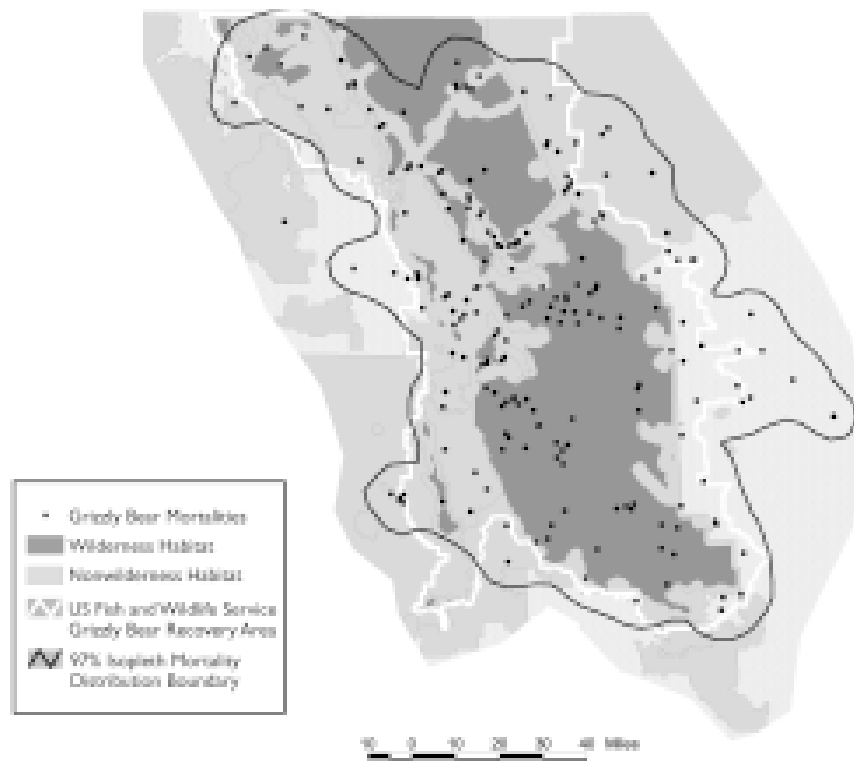


Figure 2—Grizzly bear mortalities in wilderness and nonwilderness habitat in the Northern Continental Divide ecosystem, 1980-1997.



Figure 3—Grizzly bear mortalities in wilderness and nonwilderness habitat in the Greater Yellowstone ecosystem, 1959-1998.

Knight and others 1988; Pease and Mattson 1999). The analysis of mortalities reveals that between 1959-1998, human-caused mortalities accounted for ▲ 90% of mortalities in GYE and ▲ 85% in the NCDE between 1980-1997.

In the GYE, mean annual mortalities were 25.0 in hunted years (1959-1974) and 10.4 in nonhunted years (1975-1998); and in the NCDE, they were 19.1 in hunted years (1967-1991) and 13.0 in nonhunted years (1992-1998).

Wilderness and Bull Trout

Approximately 77.7% of the area inhabited by strong populations of bull trout occurs in wilderness habitat (fig. 4), and W/NW = 3.5: 1. Moreover, virtually all of the remainder is located directly adjacent and downstream from wilderness habitat.

INFISH Priority Watersheds are comprised of ▲ 58.2% wilderness habitat, and W/NW = 1.4: 1. Known aquatic strongholds are comprised of 59.8% wilderness habitat (fig. 5), and W/NW = 1.5: 1. Using known and predicted aquatic strongholds, the amount of wilderness habitat is 49.3% and W/NW ▲ 1: 1.

The different results for bull trout and aquatic strongholds are explained by two factors. The aquatic strongholds data represent strong populations of several native fish species, including watersheds outside the historic range of bull trout. Evidence also suggests that bull trout have more stringent habitat and temperature requirements than other

salmonids (Bitterroot National Forest 1992; Rieman and McIntyre 1993).

The visual comparison of presence and absence data shows a strong link between bull trout presence and wilderness habitat. The demarcation line between presence and absence often coincides with the wilderness and nonwilderness habitat boundary.

Discussion

Source/Sink Relationships

Lambda, the finite rate of population growth, is measured on a scale with 1.0 representing equilibrium. The formal definition of a source habitat is an area with mean lambda > 1, while sink habitats support mean lambda < 1 (Meffe and Carroll 1994). Long-term demographic data on grizzly bear populations are relatively sparse, and few investigators have calculated long-term mean lambda across entire population areas. Therefore, source/sink relationships for grizzly bears are often described in the context of mortality risks or rates. Variations of the source/sink concept have been developed for the GYE grizzly bear population. Doak (1995) describes a source/sink relationship using a definition of “good” and “bad” habitats, where grizzlies in “bad” habitats had per capita mortality rates 4.83 times those in “good” ones.

Pease and Mattson (1999) describe a source/sink relationship where wary, never trapped bears are the source, and

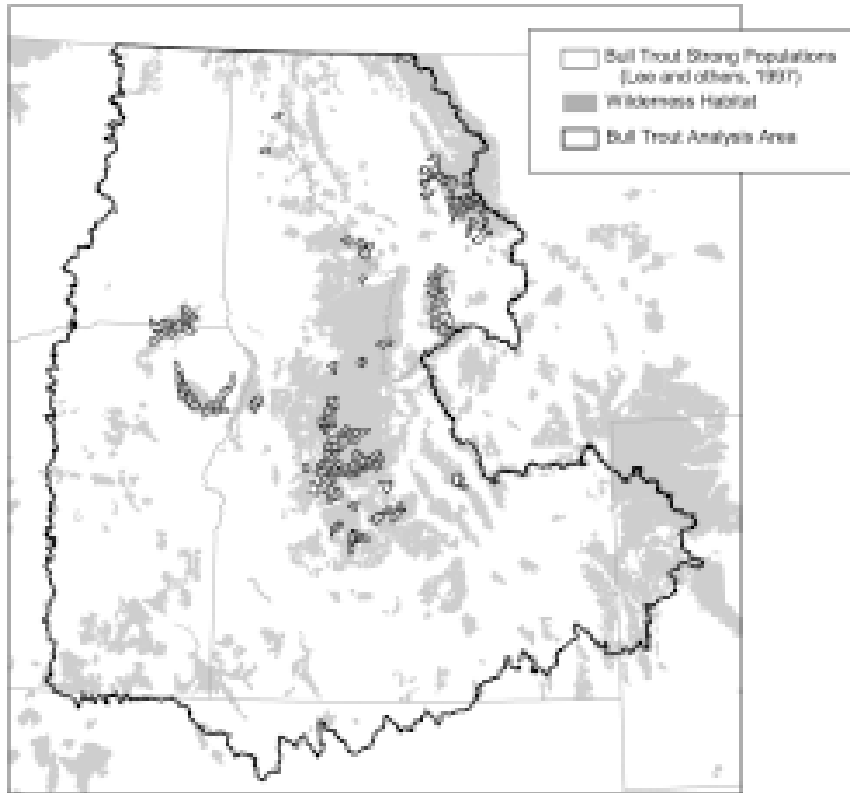


Figure 4—Bull trout strong populations and wilderness habitat in the U.S. Northern Rockies.

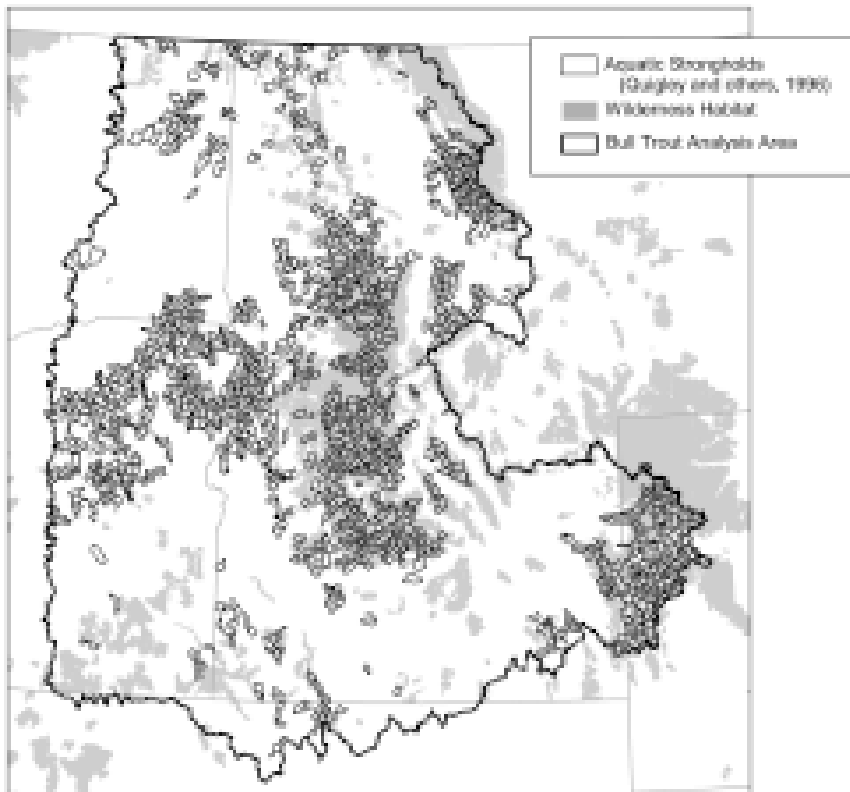


Figure 5—Aquatic strongholds and wilderness habitat in the U.S. Northern Rockies.

habituated, management-trapped bears are the sink. The latter frequent roadside habitats and developed areas and faced mortality rates approximately two times those that did not. Knight and others (1988) identified habitat sinks and found they have large geographic and population-wide effects.

Rieman and McIntyre (1993) used a schematic of historic and current systems to explain bull trout source/sink habitat dynamics. A major problem is that isolated units currently have few opportunities to use alternate habitats, as they have been degraded or blocked off. In many instances, bull trout could not move from one habitat patch to an alternate one, even if alternates were available. In this context, wilderness habitat is not only the best, but often the only refuge for bull trout. In this structure, risks from stochastic events become much greater. Historically, both source and sink areas existed at larger spatial scales, as bull trout are believed to have been extirpated from ³ 50% of their historic range (Thomas 1992), and dams and other barriers have isolated populations.

Wilderness Habitat and Grizzly Bears

Between 1850 and 1920, grizzly bears were extirpated from 95% of their range in the contiguous states (Mattson and others 1995). By 1920, distribution in the U.S. northern Rockies (Merriam 1922) had been reduced to 14 populations in remote areas and national parks. Distribution was further reduced between 1920 and 1970, and grizzlies survived this period of extirpation only in remote wilderness regions > 25,899 km² (10,000 mi²) (Mattson and others 1995). By the 1990s, substantial grizzly bear populations in the contiguous states occurred only in association with large blocks of national park, designated wilderness and other lands with little human intrusion (Metzgar and Bader 1992).

Three populations within extensive wilderness habitat in the Bitterroot Mountains may have been extirpated, but it is believed that they were eliminated by sheepherders and hunters, and not by habitat conditions (U.S. Fish and Wildlife Service 1997), demonstrating that even wilderness is not

entirely effective in providing security for grizzly bears should humans decide to eliminate them.

Human-caused mortality is the force that limits population size of grizzly bears, regardless of where it occurs. The majority of reported grizzly bear mortality in the contiguous states is human caused or related to people and their things (Bader 1989; Mattson 1998). However, the agent most associated with such mortalities are the roads and trails that allow access into grizzly bear habitat. Mattson and Knight (1991) found that secondary roads presented a mortality risk five times that of roadless backcountry areas, ranked second only to primary developments in lethality. They reported that telemetry locations of adult females over a 10-year period coincided with the areas of lowest road and trail densities in the GYE. Several other studies documented that bears avoid roads at all road density levels and that females with cubs select for roadless areas in their use of habitat. Bears avoided areas within 500 m (547 ya) of roads more than expected, and this zone of avoidance ranged up to 3 km (1.86 mi) or more (Archibald and others 1987; Blanchard and Knight 1991; Kasworm and Manley 1990; Mace and Waller 1998; Mattson and others 1987; McLellan and Shackleton 1988; Schallenger and Jonkel 1979; Wakkinen and Kasworm 1997).

The spatial distribution of mortalities (figs. 2 and 3), shows clusters of mortality along major road and railroad access corridors. This pattern is also partially obscured by the fact that some developed areas contained several reported mortalities with the same UTM coordinates, so several dots are piled on top of each other. Several of the clusters within wilderness habitat are near major trailhead access points and areas with high hunter use. The NCDE mortalities from 1970-1979 show the same pattern. The effects of human access on the spatial distribution of grizzly mortalities is clearly visible, which shouldn't be too surprising given that most grizzly mortality is directly human caused or related (fig. 6).

Bears that died in wilderness habitats usually did so because of legal and illegal hunting take, mistaken identity kills, natural causes or because bears came into conflict with

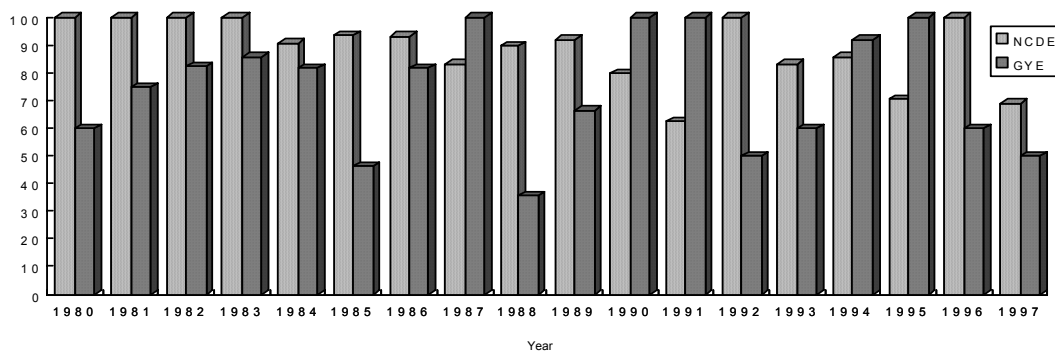


Figure 6—Grizzly bears and human-caused mortality in the Yellowstone and Northern Continental Divide Ecosystem, 1980-1997. Note: GYE data from 1959-1979 were available but not shown in order to provide direct comparison with NCDE data. GYE human-caused or human-related mortalities 1959-1998 were approximately 90% and in the NCDE 1980-1007 were approximately 85%.

“front-country” type situations, such as large camps and cabins in high human-use areas. Poor management, such as unsanitary camp conditions, can elevate mortality risk for grizzly bears and reduce the source potential of wilderness.

There are important management implications in the shift of mortalities from W to NW. If grizzly bear populations are delisted and legal hunting is resumed (the stated intention of state fish and game agencies in Wyoming and Montana), mortality density may shift back to core population areas and increase mortality among wary bears, while nonwary bears in nonwilderness habitats will still face high mortality risks. The source/sink habitat structures in these populations could thus be destabilized, and the source potential of wilderness habitat reduced, as some of the current source area would be converted to sink.

Due to the spatial distribution of wilderness habitat at the core of the GYE and NCDE recovery areas, hunting could have a destabilizing effect on these grizzly bear populations.

Some have theorized that hunting mortality among grizzly bears is compensatory and removes “problem” bears from the ecosystems, reducing the need for control actions (Greer 1976). However, the spatial distribution of the mortalities indicates that hunting mortality may be additive due to the location of numerous hunting mortalities in very remote areas, spatially distant from “problem” bears in front-country areas. Moreover, NCDE hunting mortalities from 1970-1979 were strongly skewed towards wilderness habitat, where more than 85% of the hunting kills occurred. Dood and others (1986) reported that only eight of 81 nonhunting mortalities from 1973-1985 occurred in wilderness. Another corroborating factor indicating that hunting mortality is additive are the mean mortalities per year, which decreased significantly (25.0 to 10.4 in GYE and 19.1 to 13.0 in NCDE) following the end of legal hunting seasons. Dood and others (1986) reported that hunter harvest was the leading cause of NCDE mortalities from 1967-1985. A qualifying factor is that legal hunting mortalities are more likely to be detected or reported.

Another factor is mistaken identity kills. These hunters didn't even know which species they were shooting, let alone if it was a habituated “problem” bear or not, so many of these kills would not be compensatory.

While NCDE sample sizes were smaller, there are indications that the trend from W to NW is strengthening. For example, in 1998 (a poor food source year), mortalities rose to 23 (Pac 1999). The UTMs for 1998 data were unavailable, but those for which enough descriptive information was available to make a determination ($n = 17$), 16, or 94.1% occurred in nonwilderness. Moreover, another 14 mortalities were prevented through direct management intervention (Manley 1998). Previous investigations indicate these bears are living on borrowed time, as once bears are management-trapped, their risk of mortality dramatically increases (Meagher and Fowler 1989; Pease and Mattson 1999). An analysis of more than 100 grizzly bear relocations found that an adult female has only a 60% probability of survival once it comes into contact with humans (Montana Department of Fish, Wildlife and Parks 1992). Most relocations of grizzly bears from one area to another are the result of bear-human interactions in frontcountry locations.

Mattson (1998) and Mattson and others (1992) have demonstrated a strong link between poor food source years and

elevated mortality. They also found that grizzly bears used areas within 5 km (3.1 mi) of roads and 8 km (♣ 5 mi) of developments half as intensively during good food source years. This related effect warrants further analysis, as the nonwilderness sink effect will likely be most pronounced in poor food source years, when bears greatly expand their ranges and make greater use of nonwilderness lands.

Finally, an unknown, but significant percentage of mortalities go unreported, and total actual mortality in both wilderness and nonwilderness habitats is likely higher. However, no reliable estimate of unreported mortality is currently available and we must rely on the data that is available. While natural mortalities within wilderness habitat are unlikely to be detected, the same holds true for natural mortalities within nonwilderness habitats. Moreover, illegal and mistaken identity kills often go unreported since the perpetrators of illegal kills seldom turn themselves in. Since grizzly bear deaths appear to be closely tied to human access, I assume that more such deaths occur on nonwilderness habitats than in wilderness. Several reviewers have demonstrated that a large majority of recorded grizzly bear mortalities were directly human-caused or related (Bader 1989; Craighead and others 1988; Mattson 1998), reducing the likelihood that human-caused mortality is over-reported compared to natural mortality.

Another potential complicating factor is that in the GYE area, grizzly bears within Yellowstone National Park had access to garbage dumps which were often located in frontcountry areas. These dumps were closed by 1971, prior to the cessation of legal hunting seasons.

We can infer from the literature and historical distribution that grizzly bear populations without large cores of wilderness habitat decline, don't grow or remain at low levels that leave them vulnerable to rapid declines and eventual extirpation. Grizzly bears currently inhabit areas in the U.S. northern Rockies that are substantially nonwilderness habitats (Cabinet-Yaak and Selkirk Mountains), but at much lower densities and higher mortality risks than in wilderness habitats (Bader, in prep). Thus, wilderness habitat is the area with the least amount of human access and pressure, providing higher security (lower mortality risk) for grizzly bears.

Wilderness Habitat and Bull Trout

By any measure used, the results show clearly that wilderness habitat is a stronghold for remaining bull trout populations and other salmonids. These findings build upon a growing body of work that reveals a strong link between wilderness habitat, habitat quality and native fish.

Rieman and McIntyre (1993) reported that researchers recognize temperature more consistently than any other factor influencing bull trout distribution. Wilderness habitat, due to its spatial distribution at higher elevations and headwaters areas, contains a disproportionately high percentage of colder waters. These habitats also have generally higher percentages of streamside canopy cover, which helps keep stream temperatures lower.

Bull trout also require very clean water and favor streams with upwelling groundwater for spawning (Fraleley and Shepard 1989). A key determinant is the level of fine sediments - 6.35 mm (♣ .25 in) (Weaver and Fraleley 1991). When

these fine sediments comprise 35% and 40% of the gravel substrate, embryo survival declines by 66% and 77%, respectively. Unmanaged drainages generally have lower fine sediment levels. The occurrence of upwelling groundwater may explain the presence of bull trout within managed watersheds, as the upwelling flushes fine sediments from around embryos.

Sedell and Everest (1990) reported that up to 75% of the pool habitat in the Columbia River Basin has been lost to salmonids, and the only areas where fish habitat quality remained stable or increased were in wilderness.

Numerous studies have reported that with bull trout, strong populations, presence and biomass are inversely related to road densities (Bitterroot National Forest 1992, 1993; Huntington 1995; Quigley and others 1996; Rieman and others 1997; Swanson 1992; U.S. Fish and Wildlife Service 1998). Maxell (1996) found that bull trout spawning activity within the Rock Creek watershed was correlated with roadless sub-drainages.

The leading cause of decline in bull trout populations is logging and roadbuilding (U.S. Fish and Wildlife Service 1994); the agency recommended (1998) that remaining roadless areas within bull trout range be maintained in roadless condition. Region-wide assessments also show that the healthiest watershed conditions substantially correspond with wilderness and roadless drainages (Isaacson 1994; U.S. Forest Service 1994).

Quigley and others (1996) concluded that designated wilderness and potentially unroaded areas are important anchors for remaining strongholds of native fish.

Wilderness habitat plays a disproportionate role in sustaining remaining populations of bull trout and other native fish. Put another way, if all wilderness habitat were somehow lost all at once, extinction risks for bull trout would most certainly rise and distribution decrease. Wilderness habitat is a source habitat for bull trout, with the roaded land-base functioning as a sink habitat, increasing in severity as road densities increase.

Population Viability and Area Requirements

Two figures are important to assessments of population viability. These are the total population size (N) and the effective population size (N_e). The effective population refers to that portion of the population that is of breeding age and that actually breeds (Allendorf and others 1991). Extinction risks become severe whenever $N_e < 50$ (Shaffer and Samson 1985), and $N_e \cdot 500$ has been frequently cited as a minimum goal for species conservation (Franklin 1980; Nunney and Campbell 1993). Just as important is the ratio of N_e : N . If populations become too small, they can enter into an irreversible decline or "extinction vortex" (Gilpin and Soule 1986).

Allendorf and others (1991) calculated that for grizzly bear in the U.S. northern Rockies, N_e : N is $^3 1$: 4. Thus, $N_e = 500$ requires a total $N = 2,000$ for a minimum recovery goal (Metzgar and Bader 1992). Reviewing reported grizzly bear densities, Metzgar and Bader (1992) concluded that a regional $N = 2,000$ requires $\cdot 129,495 \text{ km}^2 (50,000 \text{ mi}^2)$ of secure and connected habitats.

For fish populations, Nelson and Soule (1987) estimated N_e : N $^3 1$: 10 and, applying a 50/500 rule for species viability, recommended total $N = 5000$ to maintain genetic diversity within a closed system. Based on these data, Rieman and McIntyre (1993) concluded that extinction risks for bull trout increase sharply whenever N is $< 1,000$ - $2,000$ or 500 spawning pairs.

Allendorf and others (1997) reported considerable genetic divergence between bull trout populations and that maintenance of genetic diversity requires conservation of essentially all remaining stocks throughout the range of the species. Rieman and McIntyre (1993) recommended that migratory metapopulations be reconnected throughout the current and historic range.

The total area of bull trout watersheds classified as strong, depressed, present (status unknown) and in migration corridors (Lee and others 1997) within the northern Rockies is $^3 103,562 \text{ km}^2 (39,985 \text{ mi}^2)$. Considering that bull trout may have been extirpated from up to 50% of their historic range (U.S. Fish and Wildlife Service 1994), and recovery goals include recolonization of vacant habitats and migratory corridors, recovery habitat for bull trout in the northern Rockies could well be greater than $129,495 \text{ km}^2 (50,000 \text{ mi}^2)$.

Applying the results of Rieman and McIntyre (1993) estimating lower extinction risks for populations $\cdot 50$ - 100 spawning pairs ($N_e = 100$ - 200) and N_e : $N = 1$: 10 (Nelson and Soule 1987), a reasonable rule of thumb for minimum viability within a closed system is N $^3 2,000$ - $5,000$ fish that are sexually mature within one generation ($^3 10$ years), although larger watersheds may require larger goals. With $^3 100$ key bull trout watersheds identified by the states of Montana (1996) and Idaho (1996) and other sources for Oregon (Ratliff and Howell 1992) and Washington (Washington Department of Wildlife 1992), recovery of genetically and demographically viable bull trout populations in the U.S. northern Rockies could well require total $N > 5 \times 10^5$ adult fish, if systems remain closed. Long-term viability requires installation of passage structures or removal of dams and other barriers.

Metapopulation Structures

Source/sink dynamics often occur in association with metapopulation structures, as described by McCullough (1996) and Meffe and Carroll (1994). The metapopulation, (a population or collection of populations, Levins 1969), often occupy patches of source and sink habitats. Populations in sink areas avoid extirpations through "rescue effects" (Brown and Kodric-Brown 1977), whereby immigrants from other patches prevent local extirpations or serve as a source of refounders for vacant patches. Source habitats allow and provide dispersing members of the population to replenish sink patches.

These metapopulation structures provide a mechanism for spreading risk among populations (Rieman and McIntyre 1993) and are believed to significantly increase the likelihood of species persistence as well as genetic viability and variation.

Bull trout likely evolved in a metapopulation structure (Rieman and McIntyre 1993), while grizzly bear populations were once continuous in the contiguous states and were

fragmented by human settlement and activity into a potential metapopulation (Craighead and Vyse 1994).

For more detailed assessments of metapopulation and source/sink concepts, see McCullough (1996); Meffe and Carroll (1994); Rieman and McIntyre (1993).

Limitations to Wilderness

While large blocks of wilderness habitat remain in the northern Rockies, there are limitations to the ability of these habitats to support viable populations of key indicator species and provide comprehensive ecosystem protection. Metzgar and Bader (1992) found none of these semi-isolated blocks are capable of supporting a demographically and genetically viable grizzly bear population. Similarly, bull trout require main river systems for migration in areas where wilderness management is not an option.

Noss (1991a) found that just 19% of Kuchler-Bailey ecosystem types were represented in designated wilderness areas > 100,000 ha (▲ 247,000 ac) in the U.S.

Wilderness habitat also becomes less effective when legal hunting seasons are administered, and it is not always effective at preventing determined efforts at systematic exterminations.

Due to its fragmented spatial distribution, wilderness habitat areas must be linked through other management strategies in order to provide viability. A significant area must also be recovered as wilderness and low road-density habitats. These include special management designations for linkage corridor management, as well as road closure and obliteration efforts that both restore habitat security for grizzly bears and reduce sedimentation, illegal harvest and temperature threats to bull trout.

Due to patterns of wilderness designation, which have often favored higher elevation, more remote areas (Wolke 1991), many designated wilderness areas are comprised of large expanses of alpine terrain of limited value to grizzly bears and other species. In light of its role as critical refugia, more wilderness habitat is needed at lower elevations.

While there are limitations to the ability of wilderness to protect ecosystems and native biodiversity, wilderness is largely accepted as the most secure refugia for a wide array of species, and it is the baseline against which human impacts are measured (Noss 1991b).

A Proposal for Wilderness-Based Ecosystem Protection

Bader (1991, 1999) describes a wilderness-based reserve network for the U.S. northern Rockies. This system has been introduced in legislative form as The Northern Rockies Ecosystem Protection Act, H.R. 488, shown in figure 7.

This network makes use of four essential elements of reserve design identified by Noss (1992). These are cores, buffers, corridors and restoration.

This proposed network, with its 74,415 km² (28,733 mi²) of new wilderness designations, would help provide for connectivity through designated linkage corridors, where road densities would be reduced. Over 2,896 km (1,800 mi) of wild and scenic river designations would maintain connectivity for bull trout and other migratory fish species. One

new national park and preserve area is proposed, and another would be studied for suitability. Another provision is a pilot system of wildland recovery areas totalling 4,030 km² (1,556 mi²), where the process of restoring wilderness habitat, vegetation and low road-density conditions via road closures and obliteration would begin.

The designations are designed to work in concert to achieve ecosystem protection and total 95,705 km² (36,953 mi²); when added to existing wilderness and national park areas these would total ³ 140,000 km² (54,054 mi²), approximately equal to the minimum area requirements for grizzly bears. However, federal legislation cannot provide comprehensive protection for all grizzly bear habitat or for all current and historic bull trout habitat, and many migratory corridors pass through nonfederal lands where wilderness designation is not an option. But this network would protect the core grizzly bear habitat area and virtually all bull trout strongholds, and key habitat areas would be recovered.

The appropriate scale for capturing broader scale environmental phenomena may be 10-15 and as much as 50-100 times the largest disturbance patch (Shugart and West 1981). Wildfires burned ³ 10,460 km² (4,039 mi²) in the northern Rockies in 1988 (National Interagency Fire Center 1999). Therefore, the total minimum dynamic area (Pickett and Thompson 1978) could be ³ 104,606-156,909 km² (40,390-60,585 mi²) and potentially > 5 X 10⁵ km². The proposed network falls within the range of the lower figures. Since the northern Rockies are home to other wide-ranging species, including carnivores such as wolves (*Canis lupus*) that may have minimum area requirements up to 38,849 km² (15,000 mi²) exclusive of corridors (Bader 1991), these figures indicate that large landscapes are required to effect ecosystem protection in the U.S. northern Rockies.

This strategy is designed to work in concert with other efforts, including fish passage proposals, litigation, species listings, conservation easements on private lands and methods to facilitate wildlife movements across major highway and rail corridors, to name a few.

Economic studies by Garrity (1997) and Power (1992) concluded this network can be implemented at a net savings to taxpayers, and with minimal impact on timber industry employment.

Conclusions and Recommendations

Nonwilderness grizzly bear mortalities account for approximately two times those in wilderness, and populations larger than 50 have persisted only in association with large blocks > 10,000 km² (3,861 mi²) of wilderness habitat. The area inhabited by strong populations of bull trout is ³ 78% wilderness habitat. I conclude that wilderness is a source habitat for remnant populations of grizzly bear and bull trout, and the roaded matrix of nonwilderness lands constitute a sink habitat area.

The cessation of legal hunting seasons for grizzly bears has altered the spatial distribution of mortalities and increased the source potential of wilderness habitat. If legal hunting seasons for grizzly bears in the GYE and NCDE areas are reinitiated, the source potential of wilderness habitat would likely be reduced and the source/sink habitat

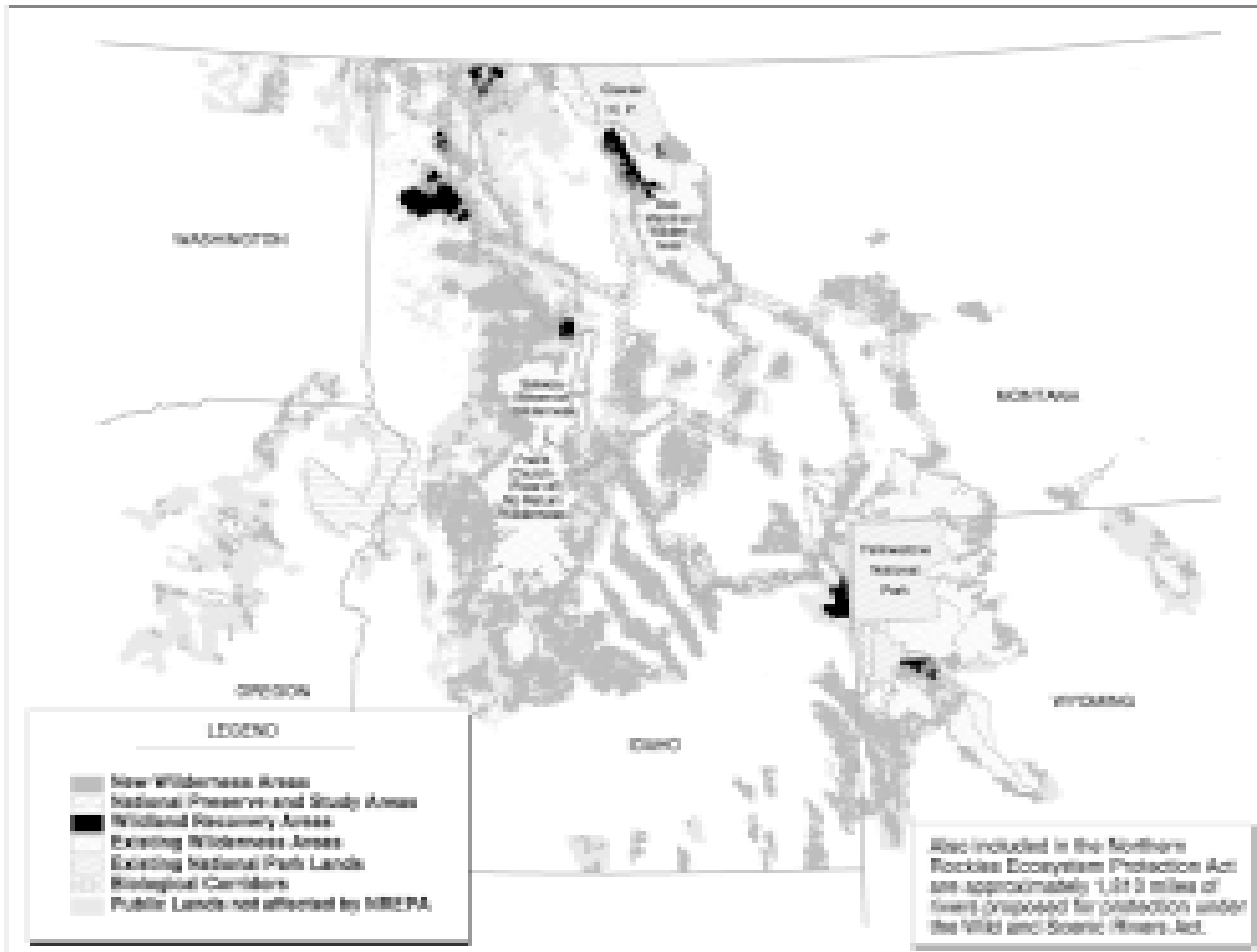


Figure 7—The Northern Rockies Ecosystem Protection Act.

structure disrupted. This could have potentially destabilizing effects, particularly in poor food source years. Bears are still threatened by illegal shootings and mistaken identity kills, which have risen since the cessation of hunting seasons. Moreover, increasing numbers of people are moving into grizzly range, indicating that bear-human conflicts may continue to rise. Thus, the importance of core wilderness source habitats will increase in importance. If increasing the source potential of core areas is a management goal, managers may also wish to consider restricting black bear hunting seasons within occupied grizzly bear range and discourage the use of baits.

Less than 50% of the wilderness habitat area in the U.S. northern Rockies has official legal protection, and it remains vulnerable to degradation. Successful recovery of leading indicator species requires protection of this source habitat area and linkage via other management strategies. Increasing the source area will reduce the overall mortality risk and enhance the possibilities for range expansion of native species, while also preventing local extirpations and range collapse.

Wilderness and unmanaged watersheds may also serve as buffers against climate change effects since forests provide

shade and cooler temperatures (Rieman and others 1997). Sudden climate change could affect bull trout distribution.

Conversely, further losses of wilderness habitat will increase the sink areas and threaten the continued existence of these species. Losses may already have passed critical thresholds. Doak (1995) found that lag effects delay detection of deleterious habitat losses until 10-12 years after they occur. Current wilderness habitat in the northern Rockies may already be too small and fragmented to prevent excessive grizzly bear mortality, particularly in poor food source years. For example, mortality quotas in the NCDE have been exceeded in 1992, 1995, 1997 and 1998 and in four of the last five years in the GYE (Peck 1999).

Due to its fragmented spatial distribution, the current wilderness habitat area, by itself, cannot sustain viable populations of wide-ranging primary indicator species, including grizzly bear and bull trout. Nonwilderness lands also play a vital role in ecosystem functions and recovery habitat, and it may be possible for low road-density lands to serve as source habitats. Wilderness lands do define the core habitat for many species, and effective strategies for landscape scale ecosystem protection in the northern Rockies will have wilderness at their core.

The Wilderness Act remains the only law that specifically protects roadless areas, and its application is a necessary component of effective ecosystem protection. Wilderness habitat is the foundation upon which effective ecosystem protection can be built. Not only is it vital to maintaining remnant populations, it serves as a primary source of individuals to recolonize historic, but currently vacant habitats.

Connectivity of habitats is the key to maintaining the unique landscape features of the U.S. northern Rockies. Functioning metapopulations would greatly enhance the likelihood of recovery and persistence of viable populations of primary indicator species, and urgent action is needed to identify, protect and restore critical linkage areas.

While this analysis is specific to grizzly bear and bull trout, it is reasonable to assume these conclusions generally apply to other species, including elk (*Cervus elaphus*), westslope cutthroat (*Salmo clarki lewisi*), Yellowstone cutthroat (*Oncorhynchus clarki bouvieri*) and other listed salmonids (*Oncorhynchus* spp.).

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