APPENDIX 3-1—OVERVIEW OF THE MAJOR CAUSES LIMITING THE HABITATS AND FISH AND WILDLIFE IN THE UPPER SNAKE PROVINCE

1. Altered Hydrologic Regime

Hydrologic regimes play a major role in determining the biotic composition, structure, and function of aquatic, wetland, and riparian ecosystems. An estimated 25% of the area in the Upper Snake province is highly impacted by altered hydrologic regimes with most impacts occurring in the American Falls and Lake Walcott watersheds (Table 1). The most severely impacted watersheds in terms of altered hydrology include American Falls, Lake Walcott, Upper Snake–Rock, Portneuf, Blackfoot, Willow, Teton, Beaver–Camas, and the Upper and Lower Henrys Fork watersheds (Figure 1). Areas occupied by riparian/herbaceous wetlands and mountain brush appear to be highly impacted by altered hydrologic regimes (Table 2).

Table 1. Comparing the relative percentages of area impacted by altered hydrologic regimes for each watershed in the three subbasins of the Upper Snake province. (Source: ICBEMP 1997.)

Snake Headwaters Subbasin Relative Category	Major Hydrologic Unit (Watershed) [a]				
	GHB	GVT	PAL	SAL	SHW
Very high	<1		8		<1
High	3	7	10	29	2
Moderate	<1		6	12	12
Low	19	<1	30	38	7
Very low	77	93	46	21	79

[a] GHB= Greys–Hoback watershed; GVT= Gros Ventre watershed; PAL=Palisades watershed; SAL=Salt watershed; SHW=Snake Headwaters watershed.

Upper Snake Subbasin Relative Category	Major Hydrologic Unit (Watershed) [a]											
	AMF	BFT	GSE	IFA	LHF	PTF	RFT	TET	UHF	USR	LWT	WIL
Very high	22	31	3	68	24	61	20	62	5	76	21	27
High	46	55	19	31	46	22	31	11	69	15	16	64
Moderate	5	1	7		3	4	8	2		<1	7	
Low	26	11	40	<1	3	7	39	14	22	5	34	9
Very low	<1	2	31		25	6	1	11	4	4	22	<1

[a] AMF=American Falls watershed; BFT=Blackfoot watershed; GSE=Goose watershed; IFA= Idaho Falls watershed; LHF=Lower Henrys Fork watershed; Portneuf watershed; RFT=Raft watershed; TET=Teton watershed; UHF=Upper Henrys Fork waterhshed; LWT=Lake Walcott watershed; WIL=Willow watershed.

Closed Basin Subbasin Relative Category	Major Hydrologic Unit (Watershed) [a]				
	BCM	BCK	BLR	LLR	MDL
Very high	56	<1	7	<1	35
High	30	23	3	14	20

Closed Basin Subbasin Relative Category	**Major Hydrologic Unit (Watershed)** [a]				
	BCM	**BCK**	**BLR**	**LLR**	**MDL**
Moderate			11	10	3
Low	14	46	49	28	21
Very low		30	31	47	20

[a] BCM=Beaver-Camas watershed; BCK=Birch watershed; Big Lost River watershed; Little Lost River watershed; Medicine Lodge watershed.

Table 2. **Relative percentages of impacts to focal habitat types by altered hydrologic regimes in the Upper Snake province. (Source GAP II, Scott *et al*. 2002)**

Focal Habitat Type	**High**	**Low**	**Medium**	**Very High**	**Very Low**
Riparian/herbaceous wetlands	42	18	5	16	19
Open water	12	23	15	39	11
Shrub-steppe	23	38	6	17	17
Pine/fir forest	18	18	5	4	55
Juniper/mountain mahogany	27	38	16	10	9
Whitebark pine	1	24	7	<1	67
Aspen	36	27	3	13	22
Mountain brush	57	10	2	21	9

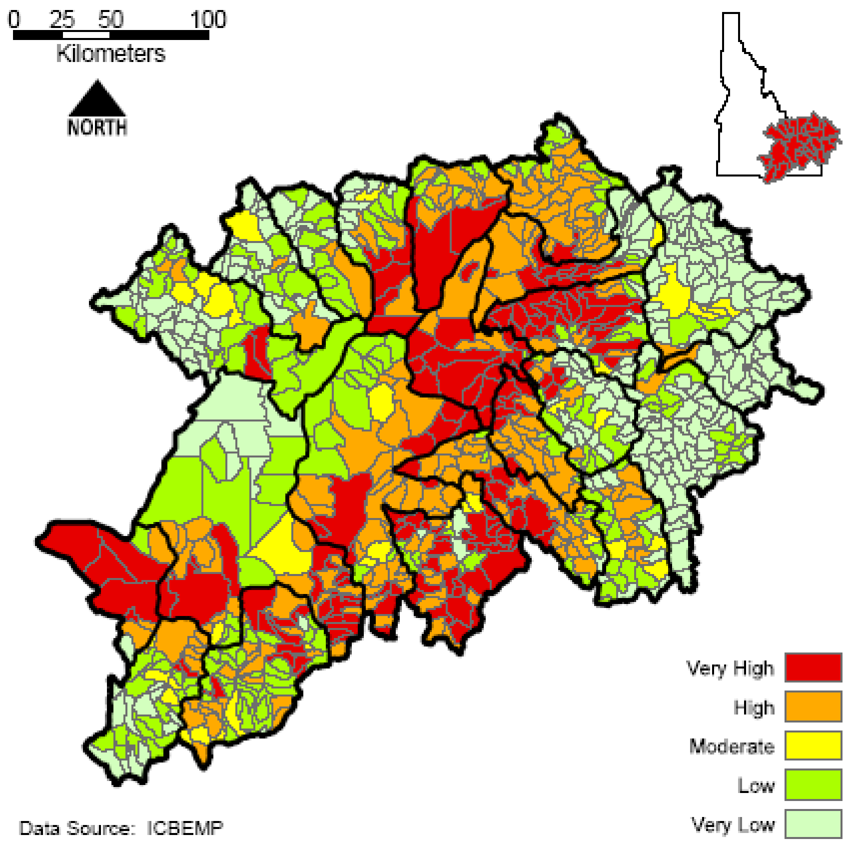

Figure 1. Relative impacts of altered hydrologic regimes in the Upper Snake province (ICBEMP 1997).

1.1 Habitat Loss and Modification

Human activities such as residential and commercial development, recreation, and resource extraction have changed, fragmented, and destroyed natural habitats. Forest and wetland losses increase overland flow and reduce filtration of sediments and pollutants, increasing the likelihood that pollutants will reach streams, rivers, and estuaries (USEPA 2001).

Habitat modification is less obvious, but detrimental nonetheless. For example, when communities build roads over streams, they modify the stream habitat. Road culverts can prevent fish passage and seriously impact fish populations. Road culverts that block fish passage in the Upper Snake province are shown in Figure 2. We estimated 227 road culverts in the Upper Snake province, and of the culverts surveyed for fish passage, only 75 allow passage for juvenile fish, and 1culvert allows passage for adult fish (Table 3).

Table 3. Information on fish passage at road crossings in the Upper Snake province. This inventory represents only the state of available information at the time this report was prepared, and may or may not reflect the appropriate proportions of culverts in a given watershed. Watersheds not included in this table have no inventoried culverts.

		Watershed														
Life Stage	**Status**	**SHW**	**GVT**	**GHB**	**PAL**	**SAL**	**UHF**	**TET**	**BFT**	**PTF**	**LWT**	**RFT**	**GSE**	**BCM**	**LLR**	**Total**
Juv	Allows fish passage	3	5	61		5							1			75
	Passage unknown											1	1		1	3
	No fish passage	7	11	80	8	2	2	1	1	2	1	7	10	2	15	149
	Total	10	16	141	8	7	2	1	1	2	1	8	12	2	16	227
Adult	Allows fish passage														1	1
	Passage unknown											2	1			3
	No fish passage				8	1	2	1	1	2	1	6	9	2	15	48
	Total				8	1	2	1	1	2	1	8	10	2	16	52

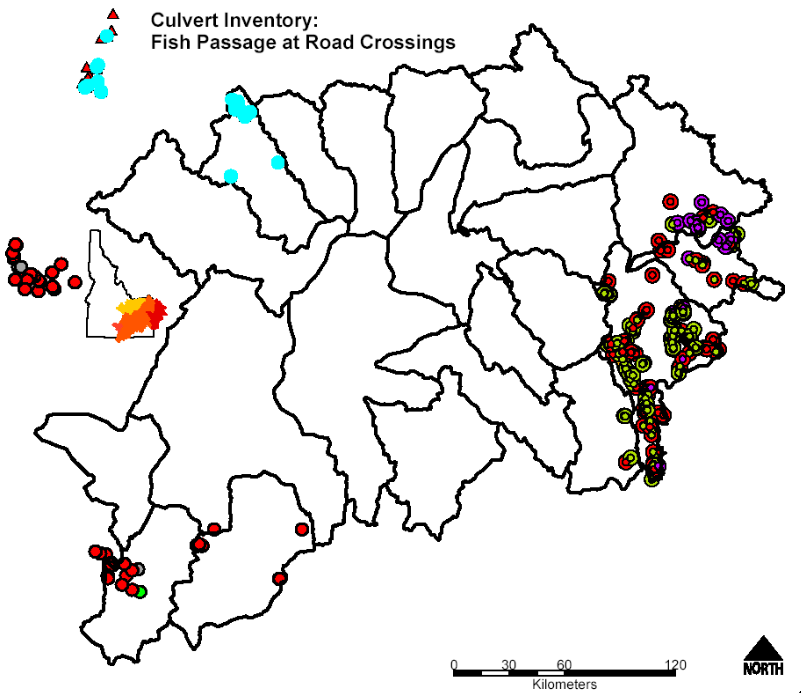

Figure 2. Inventory for road culverts surveyed in the Upper Snake province (National Forest Assessments 2003). Many road culverts remain to be surveyed in the province.

Farm, forestry, and other rural road construction; streamside vehicle operation; and stream crossings can result in significant soil disturbance and create a high potential for increased erosion processes and sediment transport to adjacent streams and surface waters. Road construction involves activities such as clearing existing native vegetation along the road right-of-way; excavating and filling the roadbed to the desired grade; installing culverts and other drainage systems; and installing, compacting, and surfacing the roadbed.

Although most erosion from roadways occurs during the first few years after construction, significant impacts may result from maintenance operations using heavy equipment, especially when the road is located adjacent to a water body. In addition, improper construction and lack of maintenance may increase erosion processes and the risk for road failure (USEPA 2001).

1.2 Hydromodification

If stream flows are lowered, fluctuate, or blocked by physical barriers, these changes can affect many plant and animal species (USFS 1994). These changes can also affect recreational opportunities. Hydromodification is widespread due to efforts to capture, control, store, and divert water. These alterations support drinking water supplies, hydropower, irrigation, flood control, manufacturing uses, and recreation. Few human actions have more significant impacts on a river system than dam construction. Dams change upstream and downstream habitats, water temperatures, water quality, and sediment movement. They also block or slow the movement of materials and organisms throughout a watershed (USEPA 2001) and increase flooding and subsequent loss of property.

More than 19,900 points of water diversion are present in the Upper Snake Province (Figure 3). The majority of these diversions occur in the Big Lost (3,800), Portneuf (3,100), Teton (2,150) and Raft (2,100) watersheds. The Birch watershed has the fewest water diversions in the province, with 100.

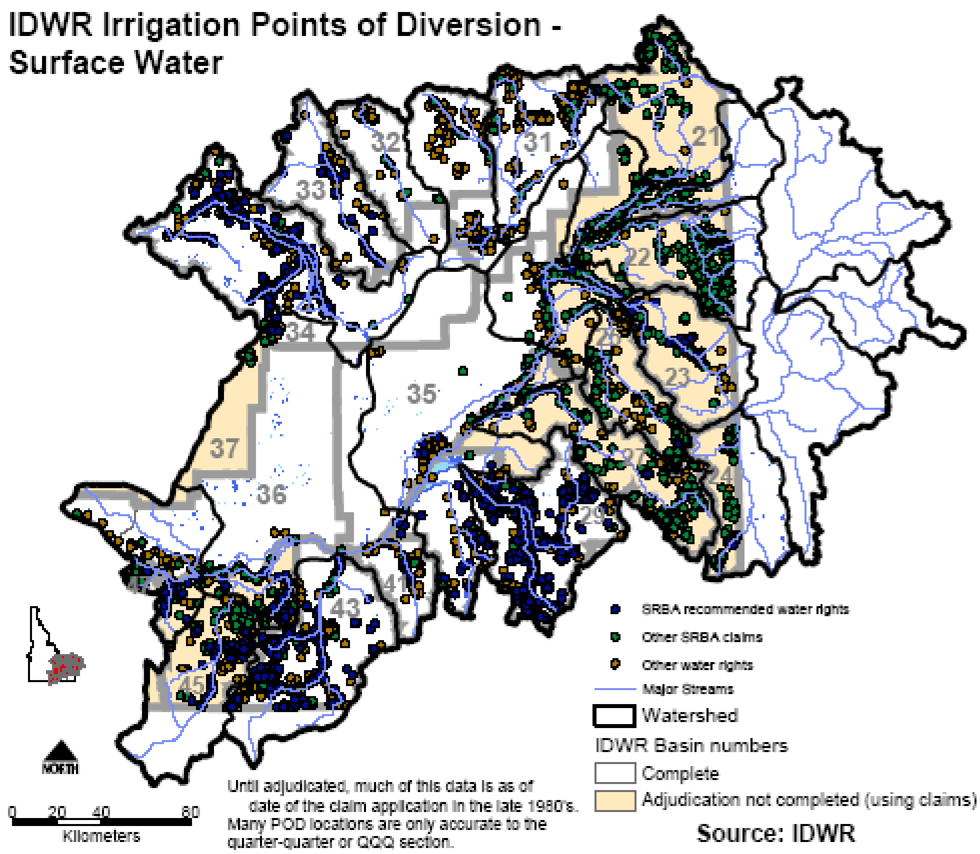

Figure 3. Locations of water diversions in the Upper Snake province (IDWR 2003).

Channelization, which is river and stream channel engineering undertaken for the purpose of flood control, navigation, drainage improvement, and reduction of channel migration potential includes activities such as straightening, widening, deepening, or rating existing stream channels, as well as clearing or snagging operations (Brookes 1990). These forms of hydromodification typically result in more uniform channel cross-sections, steeper stream gradients, a reduction in average pool depths, and altered stream/river flow (USEPA 1993).

Channel-modification activities deprive wetlands of enriching sediments, change the ability of natural systems to both absorb hydraulic energy and filter pollutants from surface waters, and cause interruptions in the different life stages of aquatic organisms (Sherwood *et al*. 1990). A frequent result of channelization and channel-modification activities is a diminished suitability of instream and riparian focal habitat for fish and wildlife. Hardening of banks along waterways eliminates instream and riparian habitat, decreases the quantity of organic matter entering aquatic systems, and increases the movement of nonpoint source pollutants (USEPA 1993).

Increased or fluctuating temperatures can harm fish and other aquatic organisms whose life cycles and breeding success are inextricably linked to water temperature. Thermal modification can eliminate fish species and other aquatic organisms from streams (USEPA 2001).

Completed channel-modification projects usually require regularly scheduled maintenance to preserve them. These maintenance activities may result in continual disturbance of instream and riparian habitat. In some cases, substantial displacement of instream habitat due to the magnitude of the changes in surface water quality; morphology; and composition of the channel, stream hydraulics, and hydrology can occur (USEPA 1993).

The magnitude of stream alteration activities within the Upper Snake province may be examined in terms of the number of alteration permits issued by the U.S. Army Corps of Engineers and the Idaho Department of Water Resources (Figure 4). A total of 2,048 stream alteration permits are issued to participants in the Upper Snake subbasin; 520 are issued to participants in the Closed Basin subbasin; and 291 to participants in the Snake Headwaters subbasin.

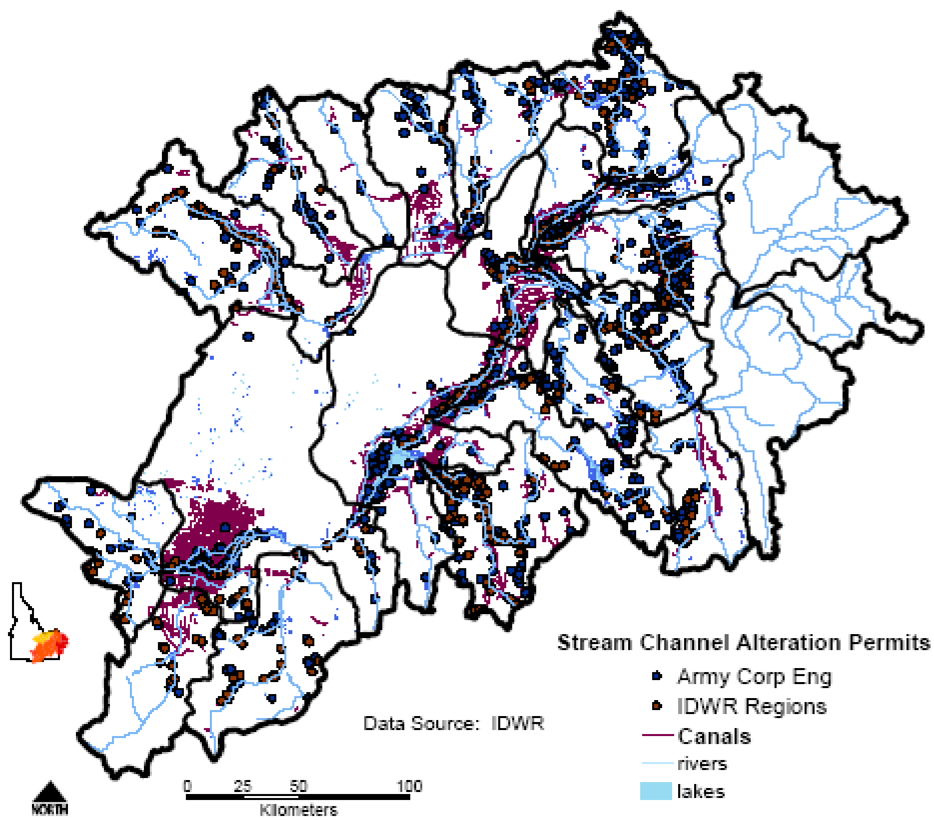

Figure 4. Channelization in the Upper Snake province (IDWR 2003).

Instream hydraulic changes can decrease or interfere with surface water contact to stream bank areas during floods or other high-water events. Channelization and channel modification activities that lead to a loss of surface water contact in stream bank areas also may result in reduced filtering of pollutants by streamside area vegetation and soils. Areas of the stream bank that are dependent on surface water contact (i.e., riparian areas and wetlands) may change in character and function as the frequency and duration of flooding change. Drainage rates from streamside areas were 2.6 times higher in the channelized area than in undisturbed areas during preliminary project activities, and 5.3 times higher following construction (Erickson *et al*. 1979). Schoof (1980) reported several other impacts of channelization, including drainage of wetlands; reduction of oxbows and stream meander; clearing of floodplain hardwood; lowering of groundwater levels; and increased erosion (USEPA 1993).

Channelization and channel-modification projects can also lead to an increased quantity of pollutants and accelerate the rate of delivery of these pollutants to downstream sites. Alterations that increase the velocity of surface water or flushing of the streambed leads to pollutant transport downstream at possibly faster rates. Urbanization has been linked to downstream channelization problems (Anderson 1992).

1.3 Sediment Impaired Waterways

One of the more significant changes in instream habitat associated with channelization and channel modification is in sediment supply and delivery. These changes in sediment supply can include shifts in erosion and deportation areas and increased sedimentation in some areas (Hynson *et al*. 1985, Merigliano 1996). Excessive volumes of sediments entering water bodies can diminish water clarity, alter habitats, impair fish spawning success, and increase treatment costs for drinking water. Timber harvest, mining, agriculture, and construction may cause excessive sedimentation. The removal of vegetation and manipulation of soils by these activities allows wind or water to carry loosened sediments to nearby water bodies. Increases in impervious surfaces decrease infiltration of rainwater into soils and increase surface runoff. These increases in surface runoff increase soil erosion and sediment transport to streams, rivers, and lakes (USEPA 2001).

Approximately 20% of the streams, a total of 71 waterways (2,104 km), in the Upper Snake subbasin are sediment impaired (Table 4).

Table 4. Total lengths (km) of streams impacted by sediments in the Upper Snake province (ICBEMP 1997, USEPA 1998).

Watershed	Total Stream Length (km)	Stream Length (km) Impacted by Sediments	% of Streams Affected by Sediments (by length)
Snake Headwaters Subbasin			
Greys–Hoback	Little Granite Creek	12	1
Gros Ventre	Gros Ventre River	27	5
Salt	Sage Creek	14	1
Snake Headwaters	Pacific Creek	21	2
Upper Snake Subbasin			
American Falls	American Falls Reservoir Bannock Creek Bannock Creek, West Fork McTucker Creek Moonshine Creek Rattlesnake Creek	37 79 6 4 2 3	15

Watershed	Total Stream Length (km)	Stream Length (km) Impacted by Sediments	% of Streams Affected by Sediments (by length)
Blackfoot	Angus Creek	12	37
	Bacon Creek	10	
	Blackfoot River	140	
	Corral Creek	25	
	Diamond Creek	29	
	Dry Valley Creek	16	
	Kendall Creek	5	
	Lanes Creek	27	
	Rawlins Creek	14	
	Sheep Creek	13	
	Slug Creek	33	
	Timothy Creek	10	
	Trail Creek	11	
	Wolverine Creek	17	
Goose	Goose Creek	53	7
	Trapper Creek	27	
Idaho Falls	Snake River	174	61
	Willow Creek	123	
Lake Walcott	Rock Creek, East Fork	18	22
	Raft River	106	
	Rock Creek	20	
	Rock Creek, South Fork	47	
Portneuf	Bell Marsh Creek	10	30
	Dempsey Creek	20	
	Garden Creek	27	
	Gibson Jack Creek	11	
	Goodenough Creek	11	
	Hawkins Creek	22	
	Marsh Creek	85	
	Mink Creek	17	
	Pebble Creek	15	
	Pocatello Creek	8	
	Portneuf River	153	
	Rapid Creek	18	
	Toponce Creek	13	
	Twentyfourmile Creek	21	
	Walker Creek	10	
Raft	Cassia Creek	37	4
	Sublett Creek	11	
Teton	Darby Creek	5	16
	Fox Creek	7	
	Packsaddle Creek	16	
	South Leigh Creek	17	
	Spring Creek	44	
	Teton River	63	
	Teton River, South Fork	32	

Watershed	Total Stream Length (km)	Stream Length (km) Impacted by Sediments	% of Streams Affected by Sediments (by length)
Upper Henrys	Henrys Fork	24	3
Upper Snake–Rock	Dry Creek, West Fork	10	3
Willow	Brockman Creek Corral Creek Cranes Creek Grays Lake Outlet Hancock Creek Hell Creek Homer Creek Lava Creek Long Valley Creek Meadow Creek Mill Creek Sellars Creek Seventy Creek Tex Creek	20 7 22 63 8 22 28 11 10 69 10 13 5 18	50
Closed Basin Subbasin			
Birch	Birch Creek	90	12
Beaver–Camas	Beaver Creek Camas Creek	50 79	14
Big Lost	Antelope Creek Big Lost River Big Lost River, East Fork Cherry Creek Cherry Creek, Middle Fork Muldoon Canyon Star Hope Creek Twin Bridges Creek Wild Horse Creek	71 53 21 41 6 15 25 14 21	12
Little Lost	Badger Creek Deer Creek Dry Creek Little Lost River Sawmill Creek, Main Fork Sawmill Creek Wet Creek	33 8 59 4 8 27 35	19
Medicine Lodge	Edie Creek Irving Creek Medicine Lodge Creek Warm Springs Creek	12 10 42 31	16

2. Land-Use Conversion/ Development/ Fragmentation

The Columbia River basin ecosystem escaped significant human land-use impact until the nineteenth century when settlers and their livestock began to move into the region during the late 1800s.

A major population boom occurred after World War II and has continued since, particularly in metropolitan areas. These urban populations have tapped the water and energy resources of the region and contributed to heavy recreational use, particularly at popular destinations. With more and more people claiming their share of the region's water, energy, and recreational resources, conflicts between mutually exclusive uses such as eco-tourism, recreational off-road vehicles, and ranching are becoming widespread and chronic (Reisner 1993, Ringholz 1996, Talbot and Wilde 1989).

The population of the Columbia River basin has increased sixfold since the beginning of the twentieth century and has more than doubled since the mid-1960s. This growth rate is two-and-a-half times greater than the nation's rate of 39% for that same period.

Population growth is some areas of the Columbia River basin in now outpacing growth in the western United States as a whole, as people flee the urbanization of the Pacific Coast to the intermountain west (USFS 1996).

Idaho is the fastest growing area in the Columbia River basin, with a population growth rate of 28.5%, followed by Washington and Oregon with population growth rates of 21.1 % and 20.4%, respectively (CensusScope 2003). Teton County in eastern Idaho saw its population rise from 3,439 people in 1990 to 6,000 people in 2000, an increase of 74% in just ten years (CensusScope 2003).

Recreation, tourism and quality of life issues play a significant role in population increases across the region. The population growth trend and its related development directly challenge community and environmental quality in many ways. Communities throughout the basin are struggling to deal with the impacts of this population growth to agricultural lands, water quality, forests, wildlife and habitat (Worster 1985).

In the Upper Snake province, the majority of the population resides in the Salt, Idaho Falls, American Falls, Teton, and Portneuf watersheds (Table 5 and Figure 5).

Table 5. Percentage population density classifications by watershed for the three subbasins in the Upper Snake province (ICBEMP 1997).

Snake Headwaters Subbasin Population Density Classification (population per square mile)	Major Hydrologic Unit (Watershed) [a]				
	GHB	GVT	PAL	SAL	SHW
Extremely High ($x > 300$)			<1	11	<1
Very High ($100 < x > 300$)	10	3	1	21	4
High ($60 < x > 100$)	15	23	8	55	76
Medium ($10 < x < 60$)	58	66	74	13	20
Low (1 < x > 10)	17	9	17		
Very Low (x < 1)				11	<1

[a] GHB= Greys–Hoback watershed; GVT= Gros Ventre watershed; PAL=Palisades watershed; SAL=Salt watershed; SHW=Snake Headwaters watershed.

Upper Snake Subbasin Population Density Classification (population per square mile)	Major Hydrologic Unit (Watershed) [a]											
	AMF	BFT	GSE	IFA	LHF	PTF	RFT	TET	UHF	USR	LWT	WIL
Extremely High (x > 300)	1	<1		7	<1	7		5				
Very High (100 < x > 300)	17	9	14	36	16	21	15	15	<1	13	9	1
High (60 < x > 100)	8	19	12	19	18	19	14	19	<1	24	9	17
Medium (10 < x < 60)	73	70	74	37	29	52	65	57	22	63	40	53
Low (1 < x > 10)	<1	3	<1	2	37	<1	6	4	78	<1	42	28
Very Low (x < 1)	<1		<1			<1						

[a] AMF=American Falls watershed; BFT=Blackfoot watershed; GSE=Goose watershed; IFA= Idaho Falls watershed; LHF=Lower Henrys Fork watershed; Portneuf watershed; RFT=Raft watershed; TET=Teton watershed; UHF=Upper Henrys Fork waterhshed; LWT=Lake Walcott watershed; WIL=Willow watershed.

Closed Basin Subbasin Population Density Classification (population per square mile)	Major Hydrologic Unit (watershed) [a]				
	BCM	BCK	BLR	LLR	MDL
Extremely High (x > 300)			<1		
Very High (100 < x > 300)			1		
High (60 < x > 100)	1	2	10	8	6
Medium (10 < x < 60)	19	83	67	77	84
Low (1 < x > 10)	79	15	21	15	10
Very Low (x < 1)					

[a] BCM=Beaver-Camas watershed; BCK=Birch watershed; Big Lost River watershed; Little Lost River watershed; Medicine Lodge watershed.

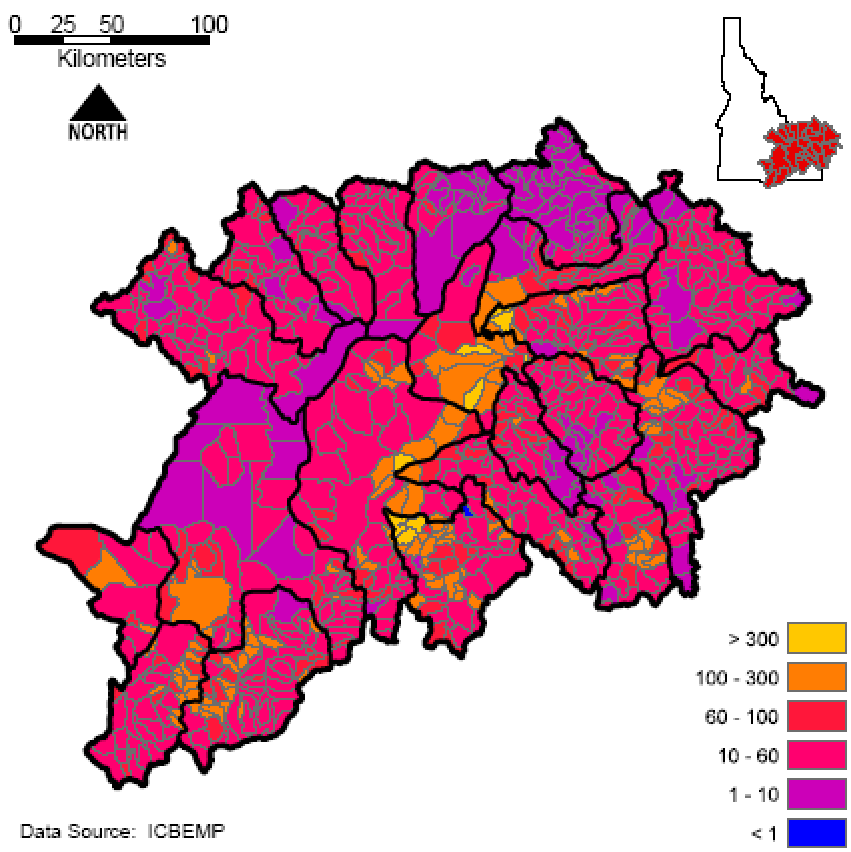

Figure 5. Relative population densities in the Upper Snake province (ICBEMP 1997).

2.1 Development

Land conversion on the urban fringe, also called “sprawl”, is an important issue to address because it has a number of impacts on the natural environment and human activity (Figure 6). Farm and ranch lands, forests, and other open space are transformed into subdivisions, ranchettes, shopping areas with expansive parking lots, and roads. This carves away at wildlife habitat and frequently diminishes wetland/riparian areas. The Natural Resources Conservation Service estimates that 6,461,210 hectares (15,965,998 acres) were converted in the western states between 1992 and 1997. They further estimate that 2,234,658 hectares (5,521,960 acres) of conversion, or about one-third, occurred in non-metropolitan areas (NRCS 2001).

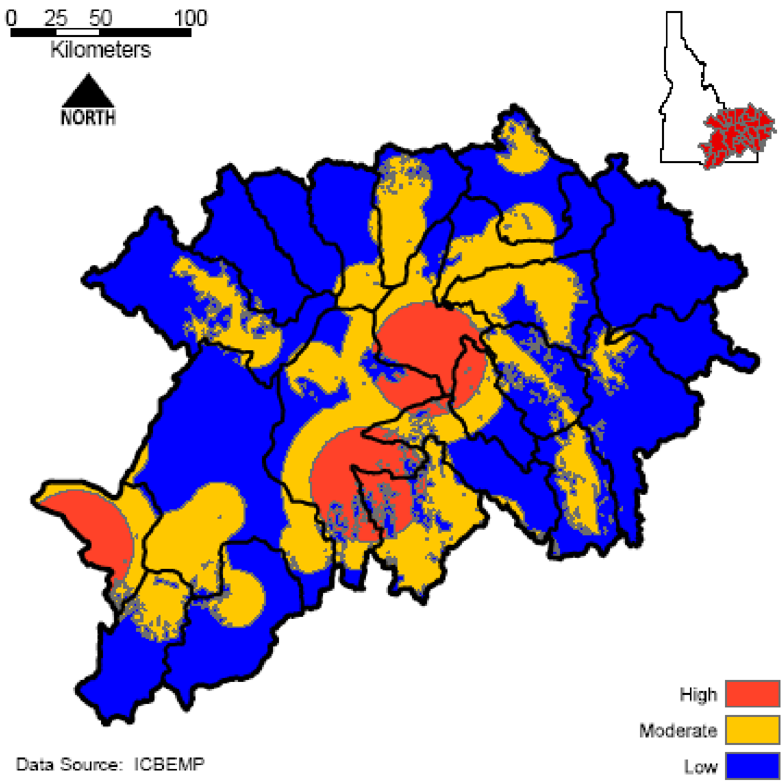

Figure 6. Areas of urban sprawl in the Upper Snake province (based on data collected in 1994 and adjusted by road density [ICBEMP 1997]).

Urban lands grew in Idaho from an estimated 222,658 hectares (550,200 acres) in 1982 to 305,497 hectares (754,900 acres) in 1997. This growth affected primarily natural resource lands (cropland, pastureland, rangeland and forestland) and is a 37% increase in urban lands. From 1982 to 1997, conversions of resource lands to urban lands were estimated at 38,161 hectares (94,298 acres) of cropland; 16,551 hectares (40,898 acres) of pastureland; 9,388 hectares (23,198 acres) of rangeland; and 15,620 hectares (38,598 acres) of forestland. This is an estimated total of 79,720 hectares (196,992 acres) removed from the rural land base for urban uses. The rate of conversion increased from an estimated 4,552 hectares (11,248 acres) per year between 1982 and 1992 to 6,701 hectares per year from 1992 to 1997. This is an increase of 47.2%. The rate of increase was highest on rangeland, followed by pastureland, cropland, and then forestland (Table 6).

Utility Corridors—Human desire to develop relatively secluded areas is generally immediately followed by the introduction of utility corridors for energy supply. These corridors physically fragment ecosystems and habitats by directly removing native vegetation. Additionally, corridors serve as a vector for invasive species, and enhance the potential for human activities. Figure 7 illustrates present and proposed utility corridors in the Upper Snake Province.

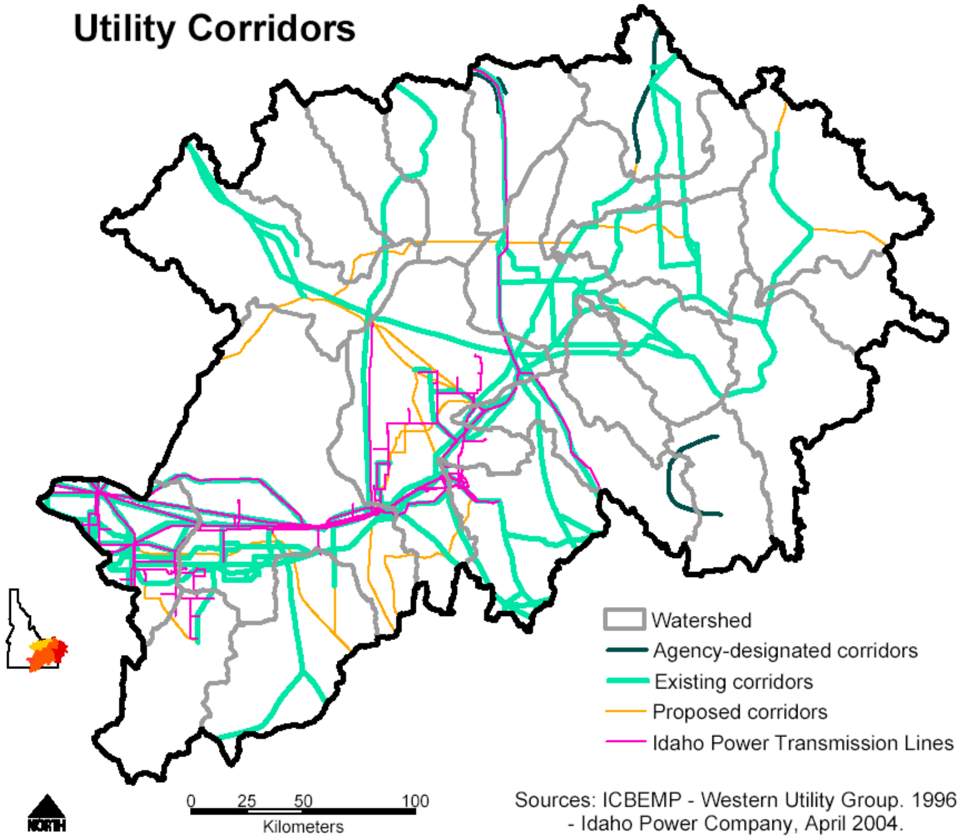

Figure 7. Established and proposed utility corridors in the Upper Snake Province (ICBEMP: Western Utility Group, 1995; Idaho Power Company, 2004).

Table 6. Estimated conversion rates of natural resource lands to urban lands in Idaho, 1982 to 1992 vs. 1992 to 1997, in hectares per year (NRCS 2001).

Natural Resource Land Type	1982-1992	1992-1997	% Change
Cropland	2,278	2,930	+28.6
Pastureland	1,019	1,513	+48.4
Rangeland	360	1,109	+207.9
Forestland	894	1,149	+28.5
Total	4,552	6,701	+47.2

Habitat fragments when new developments (sprawl) divide undisturbed habitats. The resulting fragmentation is particularly harmful to wide ranging species that rely on large territories to draw food and cover. Without adequate continuous habitat, a population of large, wide-ranging animals will eventually disappear from an area, with harmful ripple effects felt throughout the ecosystem (NRCS 2001). Sprawl inevitably translates into more roads, which in turn open up previously undisturbed habitat and open space to additional development.

2.2 Fragmentation

Habitat fragmentation involves the division of large, contiguous areas of habitat into smaller patches have isolated from one another (Figure 8, Table 7). Some habitats (lakes, riparian zones, archipelagos) are naturally fragmented. Some habitat fragmentation results from natural processes such as fires, floods, and insect outbreaks. Habitat fragmentation is an increasingly important issue in conservation biology during as human activities shape the environment and landscape (Weclaw 1998). A key hypothesis is that a reduction in the area of a habitat patch can decrease its suitability for animals to a disproportionately greater degree than the actual reduction in area (Johnson 2001). It is obvious that the numbers of a species are likely to decline if its habitat is reduced; fragmentation effects imply that the value of the remaining habitat also is diminished (Johnson 2001).

Three types of fragmentation effects have been distinguished: patch-size, edge, and isolation (Faaborg *et al*. 1993, Johnson and Winter 1999). Patch-size effects are those that result from differential use or reproductive success associated with habitat patches of different sizes (Johnson 2001). Some patch-size effects may be induced by edge effects, including avoidance, reduced pairing success, predation, interspecific competition, prey availability, and parasitism that may differ near the edge of a habitat from in the interior of a patch (Faaborg *et al*. 1993). Finally, isolation from similar habitat can influence use of a particular habitat patch because of reduced dispersal opportunities. Each of these factors—patch size, edge effects, and isolation—affects the occurrence, density, or reproductive success of animals in a habitat patch.

Habitat fragmentation results in both biotic and abiotic changes to the landscape. Fragmentation affects predator-prey relationships, species composition, dispersal, density, distribution, and population genetics, as well as microclimate variables such as sunlight penetration and temperature (Whitcomb *et al*. 1981, Johnson and Temple 1990, Knopf 1994, Paton 1994, Donovan *et al*. 1995, Greenwood *et al*. 1995, Robinson *et al*. 1995, Weclaw 1998, Winter *et al*. 2000). Although there is insufficient evidence to suggest that habitat fragmentation is entirely undesirable (Schmiegelow *et al*. 1997) it often results in habitat loss that in turn has contributed to extinction of species (Turner 1996).

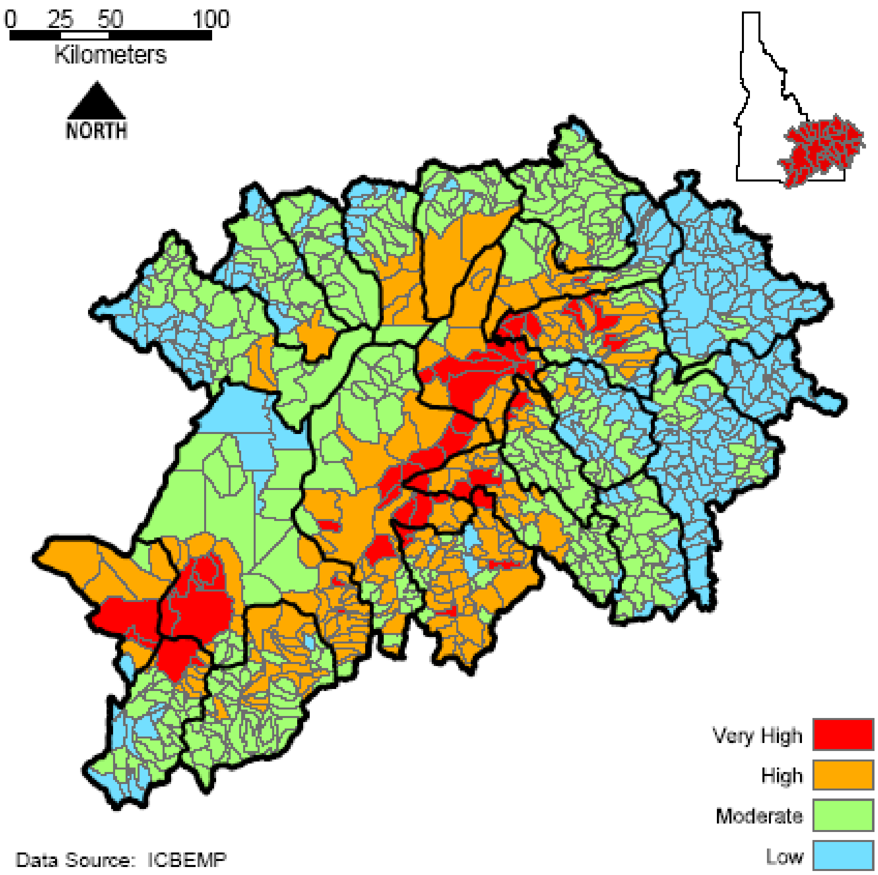

Figure 8. Estimated habitat fragmentation in the Upper Snake province (ICBEMP 1997).

Table 7. Relative percentages of habitat fragmentation by watershed in the Upper Snake province (Source: ICBEMP 1997).

Snake Headwaters Subbasin Relative Category	Major Hydrologic Unit (Watershed) [a]				
	GHB	GVT	PAL	SAL	SHW
Very high			<1		
High	<1		6		<1
Medium	22	7	41	72	5
Low	78	93	53	28	95

[a] GHB= Greys–Hoback watershed; GVT= Gros Ventre watershed; PAL=Palisades watershed; SAL=Salt watershed; SHW=Snake Headwaters watershed.

Upper Snake Subbasin Relative Category	Major Hydrologic Unit (watershed) [a]											
	AMF	BFT	GSE	IFA	LHF	PTF	RFT	TET	UHF	USR	LWT	WIL
Very high	16	12	13	36	<1%	9		21		26	14	6
High	45	39	10	60	35	72	36	45	5	65	22	17
Medium	39	48	49	4	37	13	63	23	91	5	48	77

Low	<1%	2	28		28	6	1	11	4	4	16	<1%

[a] AMF=American Falls watershed; BFT=Blackfoot watershed; GSE=Goose watershed; IFA= Idaho Falls watershed; LHF=Lower Henrys Fork watershed; Portneuf watershed; RFT=Raft watershed; TET=Teton watershed; UHF=Upper Henrys Fork waterhshed; LWT=Lake Walcott watershed; WIL=Willow watershed.

Closed Basin Subbasin Relative Category	Major Hydrologic Unit (watershed) [a]				
	BCM	**BCK**	**BLR**	**LLR**	**MDL**
Very high					
High	50	<1	7	13	34
Medium	46	88	63	56	46
Low	4	12	30	31	20

[a] BCM=Beaver-Camas watershed; BCK=Birch watershed; Big Lost River watershed; Little Lost River watershed; Medicine Lodge watershed.

2.3 Impacts to Winter Range

Land development in big game winter range (i.e., shrub-steppe, native grasslands, and juniper/mountain mahogany habitat types) is a significant wildlife habitat issue, particularly for focal species such as mule deer and Rocky Mountain elk. Subdivision development in winter ranges constitutes a permanent loss of habitat and a permanent reduction in the carrying capacity of the land for big game. The loss of a habitat component already in short supply results in fewer deer and elk for hunters (Trent 2000).

Winter range provides two needs: shelter and food. Although food resources are important, they are not the single reason for winter range selection. Of equal, or more importance is the microclimate of the winter range and how it enhances the ability of animals to minimize their energy loss during a time of food shortage (Trent 2000).

Slope, elevation, aspect, and vegetative cover combine to make some places warmer, more secure, and less snowy. Animals wintering in these areas do not deplete their fat reserves as quickly and are therefore more likely to survive the winter. When winter ranges are lost to subdivisions, this important “place” is lost and cannot be replaced or mitigated by enhancing vegetation in an adjacent area (Trent 2000).

2.4 Roads and Trails

Roads and trails have profound impacts on forest ecosystems. These include direct and indirect effects on individual plant and animal species, as well as broadscale changes in ecosystem structure and function. Askins (1994), Benninger-Truax *et al*. (1992), Ercelawn (1999), Lonsdale (1999), Neumann and Merriam (1972), and Saunders *et al.* (1991) summarize the following impacts of roads and trails:

.

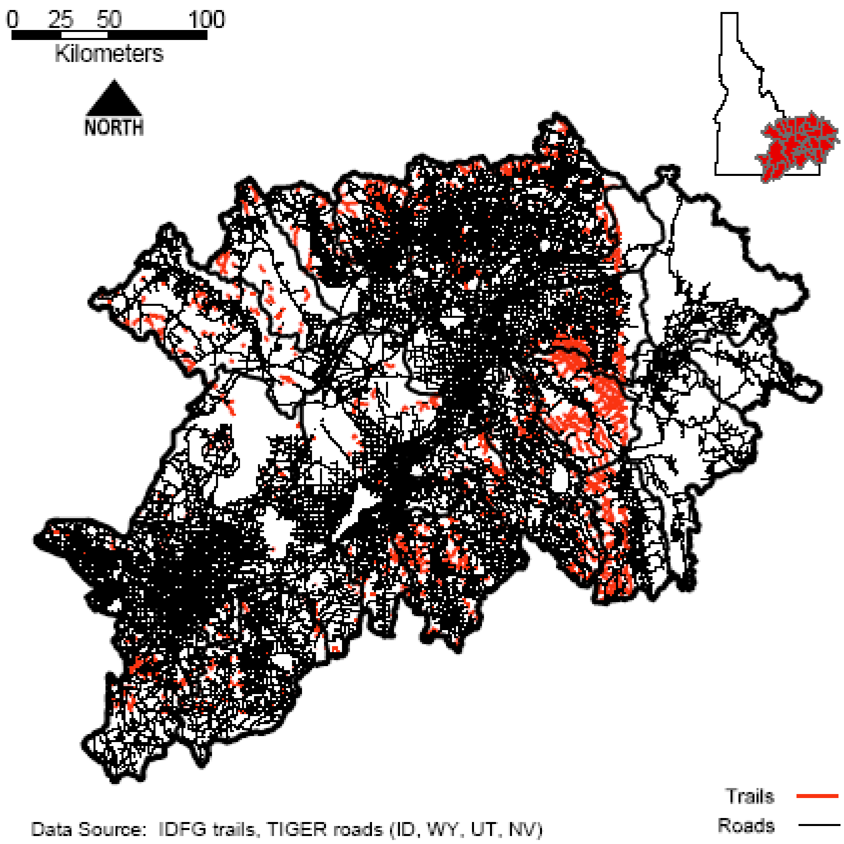

Figure 9. Distribution of roads and trails within the Upper Snake province (IDFG Trails, TIGER Roads [ID, WY, UT, NV] 2000).

- Create barriers to dispersal
- Create a significant source of direct mortality due to collisions
- Cause displacement of sensitive wildlife species
- Cause habitat loss
- Cause loss of ecological complexity
- Reduce reproductive success
- Act as a vector of disease, pest infestations, and/or invasive exotic plants and animals
- Cause degradation of ecosystem function
- Cause degradation of soil resources and water hydrology due to road-building, use, and maintenance activities
- Increase sediment and altered streamflows
- Increased disturbance and harvest of big game animals (both legally and illegally)

Recreational road and trail use is typically defined in terms of hiking, biking, horseback riding, ATVs, snowmobiles, hunting/fishing, and skiing. Impacts typically associated with these activities include trampling; habitat disturbance or modification due to noise, erosion, and soil compaction; introduction of invasive exotics; nutrient loading from animal and human waste; pollution from food waste, litter, and air quality; and increased access to the resource, and subsequent human conflict between competing resource user groups.

Trampling—The effects of trampling are usually limited to one meter from the trail's edge (Dale and Weaver 1974). Some plant species decrease near trails, especially woody plants since they are brittle (Tonnesen and

Ebersole 1997). Grasses and sedges are most tolerant of trampling (Dale and Weaver 1974). Trampling causes compaction of leaf litter and soil, and compaction by horses is greater than by hikers (Whittaker 1978). Trail width increases linearly with logarithmic increase in number of users (width doubles with 10-fold increase in use). Meadow trails are a little wider than forest trails, and trails with both horse and foot traffic are similar in width or slightly narrower than those receiving foot traffic alone. Additionally, trails used by horses and people are deeper than those used by people alone (Dale and Weaver 1974).

Disturbance—Based on an extensive review of the effects of noise and motion from recreationists on birds, Bennett and Zuelke (1999) concluded that disturbance from recreation clearly has at least temporary effects on behavior and movement of birds. Boyle and Samson (1985) documented that direct approaches caused greater disturbance than tangential approaches; rapid movement by joggers was more disturbing than slower hikers; children and photographers were especially disturbing to birds; horses did not seem to disturb birds; and passing or stopping vehicles were less disturbing than people on foot. Wildlife disturbance caused by off-road vehicle use is well documented (Olliff *et al*. 1999). With increasing performance capabilities of snowmobiles, year-round impacts become more pronounced because there are very few areas these newer machines cannot access (Olliff *et al*. 1999).

Nonnative Vegetation—Benninger-Truax *et al.* (1992) documented the introduction of exotics along trails by horses and people—notably where horse manure contained viable seeds of at least eight exotic species. Trail edges have been found to have significantly less native plant cover, and more exotic plant species (Benninger-Truax *et al*. 1992). ATVs have been documented to be a significant factor in the spread of exotic weeds across the landscape (Griggs and Walsh 1981, Trunkle and Fay 1991, Ahlstrand and Racine 1993, Sheley *et al*. 2002).

Nutrient Enrichment—Nutrient enrichment from horse manure and urine likely favors invasion of weedy species along horse trails. Research has shown that experimentally fertilized grasslands undergo a dramatic species change resulting in increased abundance of nonnative grasses, decline of native grasses, and decreased diversity (Wedin and Tilman 1996).

Pollution—Air and water pollution from off-road vehicles can be severe. By design, off-road vehicles expel 20% to 30% of their oil and gasoline unburned into the air and water (Harrison 1976). ATV and snowmobile motors produce 118 times as many pollutants as automobiles on a per-mile basis (California Air Resources Board 1998). Pollution in the form of litter and waste becomes more marked as participation in off-road vehicles activities increases.

Roads and trails are found throughout the Upper Snake province (Figure 9). Every major watershed within the province has been accessed and impacted by roads. Although very few roads occur in the wilderness and protected areas (see Figure 1-16 in the assessment), access can still be gained through extensive trail systems (Figure 9).

3. Altered Fire Regime

Wildfires were once common occurrences throughout the grasslands and forests of the Columbia River basin. Frequent fires maintained an open forest structure in the region’s middle-elevation forests, prevented tree encroachment into mountain meadows and grasslands, and in some areas replaced forested land with grassland (CPLUHNA 2003).

Prior to white settlement, fires likely burned through the region's extensive juniper woodlands every 10 to 30 years, the region's ponderosa pine communities every 1 to 47 years, Rocky Mountain lodgepole pine (*Pinus contorta*) every 25 to 300+ years, Rocky Mountain Douglas-fir (*Pseudotsuga menziesii*) every 25 to 100 years, quaking aspen every 7 to 100 years, and mixed conifer forests every 5 to 25 years. The much wetter and cooler spruce-fir forests atop the highest mountains and plateaus of the region probably went 150 years or more between fires (Fire Sciences Laboratory 2003), but these fires were generally stand-replacing events. Figure 10 illustrates the historical fire regime in the Upper Snake province.

The historical fire regimes changed dramatically with the arrival and settlement of Euro-Americans. Livestock grazing removed much of the grassy fuels that carried frequent surface fires or encouraged annual grasses, roads and trails broke up the continuity of forest fuels and further contributed to reductions in fire frequency and size. Also, the introduced exotic, cheatgrass (*Bromus tectorum*), resulted in unnatural shortened fire-return intervals. Because settlers saw fire as a threat, they actively suppressed it whenever they could. Fire suppression has been one of the great success stories of land management organizations. Over the last 100 years or so, public firefighting agencies such as the U.S. Forest Service, Bureau of Land Management, Bureau of Indian Affairs, and National Park Service have developed an impressive array of firefighting technologies that have remarkably reduced acreage burned by wildfires (Pyne 1982).

Initially, fire suppression was very successful because of low fuel loadings, but without fires to consume them, large fuel loads have accumulated over time (CPLUHNA 2003). Because of heavy fuel accumulations, fires occurring now are more intense and difficult to contain. In recent years (see Figure 11), fires that burned tens and hundreds of thousands of acres have occurred in California, Idaho, Montana, Oregon, Washington, and Wyoming (Martin and Sapsis 1992, Agee 1993, Covington *et al*. 1994, Johnson *et al*. 1994). While most ecosystems occasionally experience very large fires (Romme and Despain 1989), the present-day frequency of large fires is increasing. Figure 12 shows current fire severity in the Upper Snake province, while Figure 13 depicts areas in the province that are most likely to experience severe burns. Table 8 compares the relative percentages of risk by altered fire regimes by watershed in the Upper Snake province. In addition, Figure 14 illustrates fire regime condition class, which is an approximation of ecosystem departure resulting from a change in fire regimes.

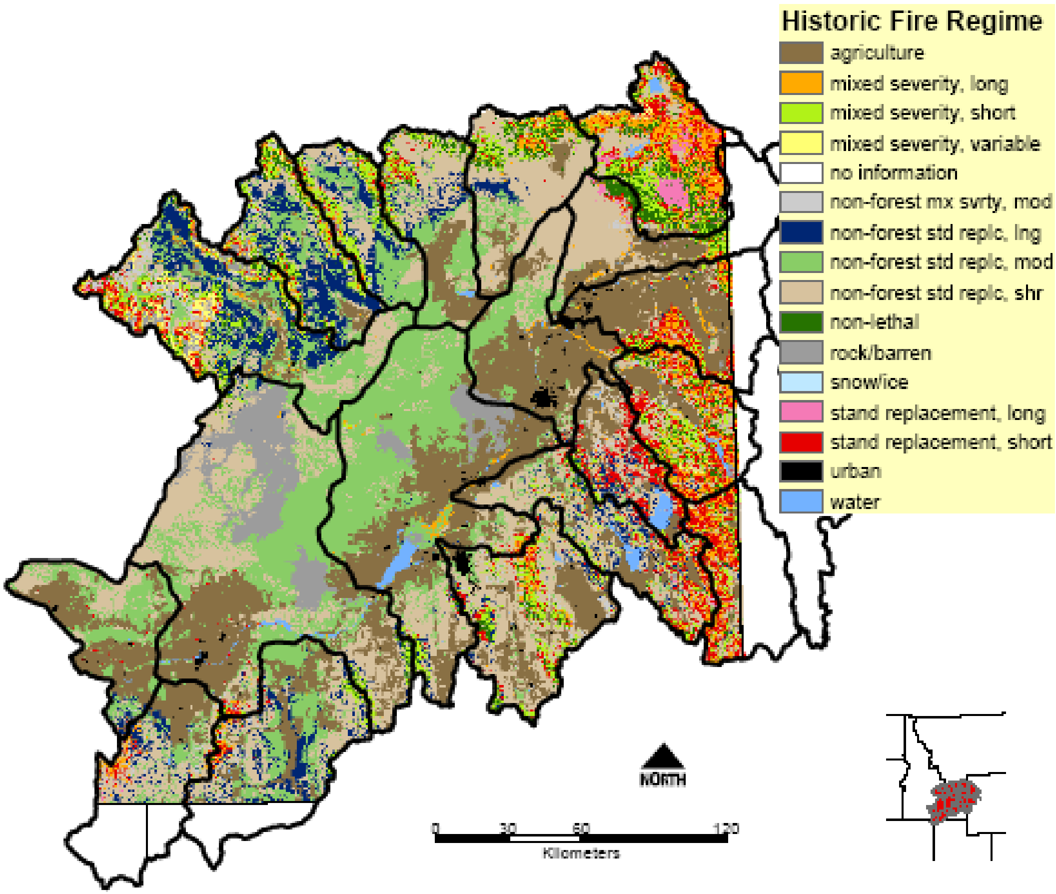

Figure 10. **Historic fire regime in the Upper Snake province (Northern Regional National Fire Plan Cohesive Strategy Assessment Team, Flathead National Forest 2003).**

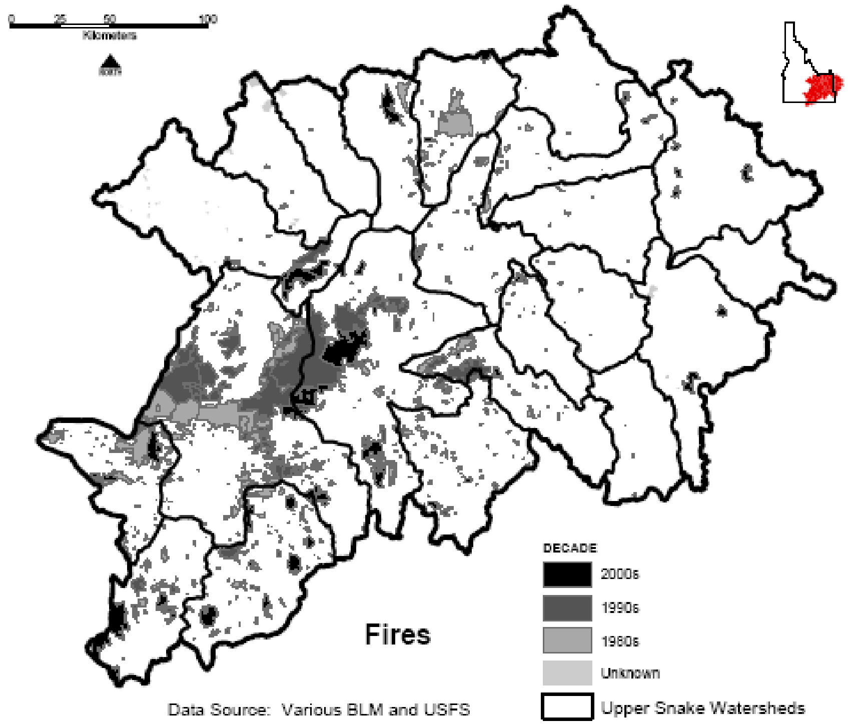

Figure 11. Locations of large (greater than 5-hectare) fires in the Upper Snake province between 1980 and 2000.

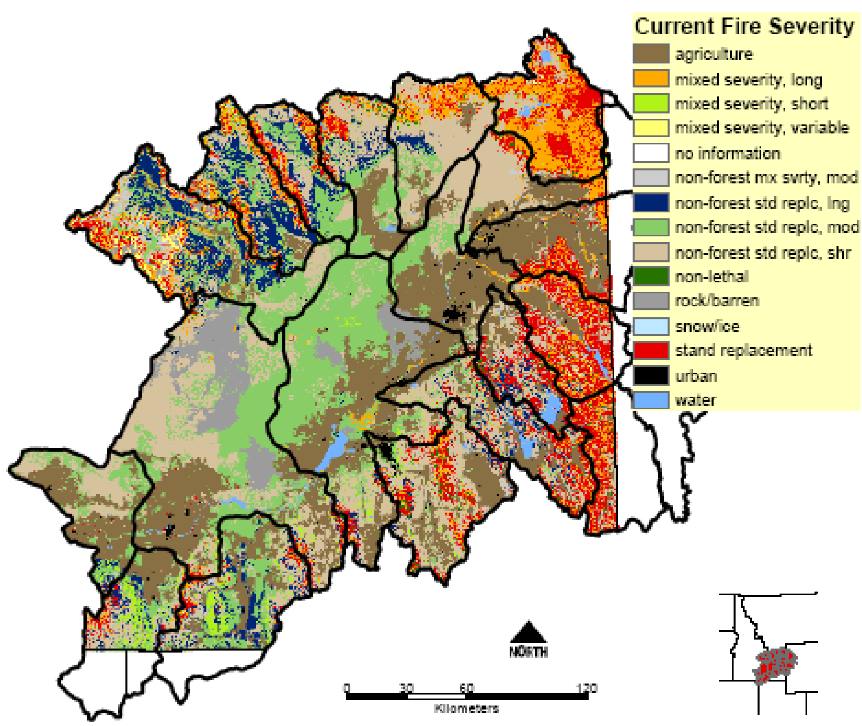

Figure 12. Current fire severity in the Upper Snake province by watershed (Northern Regional National Fire Plan Cohesive Strategy Assessment Team, Flathead National Forest[2003]).

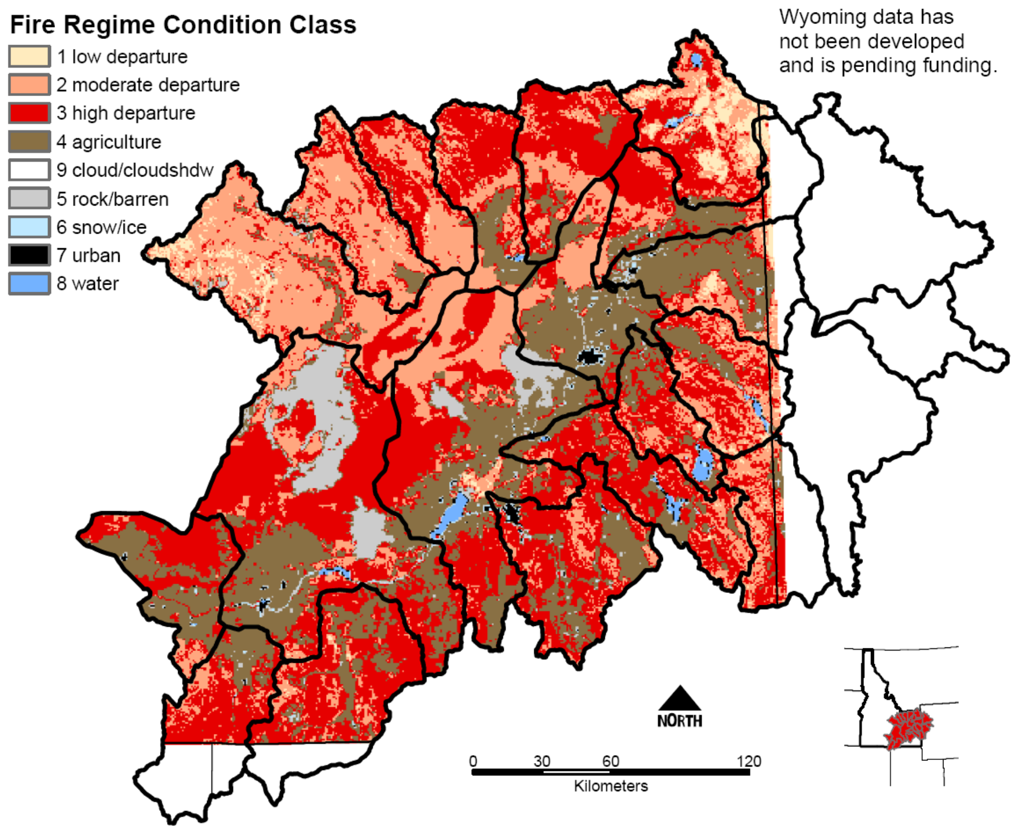

Figure 13. Probability of severe ecological fire effects in the BPW subbasins, Idaho. Fire regime condition class (FRCC) is an approximation of ecosystem departure resulting from a change in fire regimes. (Northern Regional National Fire Plan Cohesive Strategy Assessment Team, Flathead National Forest [2003]).

Table 8. Relative percentages of risk by altered fire regimes by watershed in the Upper Snake province (Northern Regional National Fire Plan Cohesive Strategy Assessment Team, Flathead National Forest). Note that data was only available for the state of Idaho, so areas are standardized to the applicable area. Other areas (agriculture, rock/barren, snow/ice, urban, water, clouds, and cloud shadow) are not included in the analyses.

Snake Headwaters Subbasin Relative Category	Major Hydrologic Unit (Watershed) [a]				
	GHB	GVT	PAL	SAL	SHW
% of watershed analyzed	?	?	94	53	?
Low risk	?	?	22	22	?
Moderate risk	?	?	31	34	?
High risk	?	?	33	36	?
No risk	?	?	22	8	?
Other	?	?	15	8	?

[a] GHB= Greys–Hoback watershed; GVT= Gros Ventre watershed; PAL=Palisades watershed; SAL=Salt watershed; SHW=Snake Headwaters watershed.

Upper Snake Subbasin Relative Category	Major Hydrologic Unit (watershed) [a]											
	AMF	BFT	GSE	IFA	LHF	PTF	RFT	TET	UHF	USR	LWT	WIL
% of watershed analyzed	100	100	61	100	71	100	81	78	99	100	100	100
Low risk	2	8	7		9	5	6	15	45	1	1	5
Moderate risk	25	31	25	29	25	20	25	13	17	14	19	33
High risk	31	39	44	10	36	47	47	11	33	40	40	34
No risk	42	22	44	61	40	28	22	61	5	45	40	28
Other	41	22	23	61	30	28	23	61	5	45	40	28

[a] AMF=American Falls watershed; BFT=Blackfoot watershed; GSE=Goose watershed; IFA= Idaho Falls watershed; LHF=Lower Henrys Fork watershed; Portneuf watershed; RFT=Raft watershed; TET=Teton watershed; UHF=Upper Henrys Fork waterhshed; LWT=Lake Walcott watershed; WIL=Willow watershed.

Closed Basin Subbasin Relative Category	Major Hydrologic Unit (watershed) [a]				
	BCM	BCK	BLR	LLR	MDL
% of watershed analyzed	100	100	100	100	100
Low risk	6	11	19	15	5
Moderate risk	32	57	61	60	37
High risk	52	28	11	16	36
No risk	10	4	9	9	22
Other	11	3	9	9	22

[a] BCM=Beaver-Camas watershed; BCK=Birch watershed; Big Lost River watershed; Little Lost River watershed; Medicine Lodge watershed.

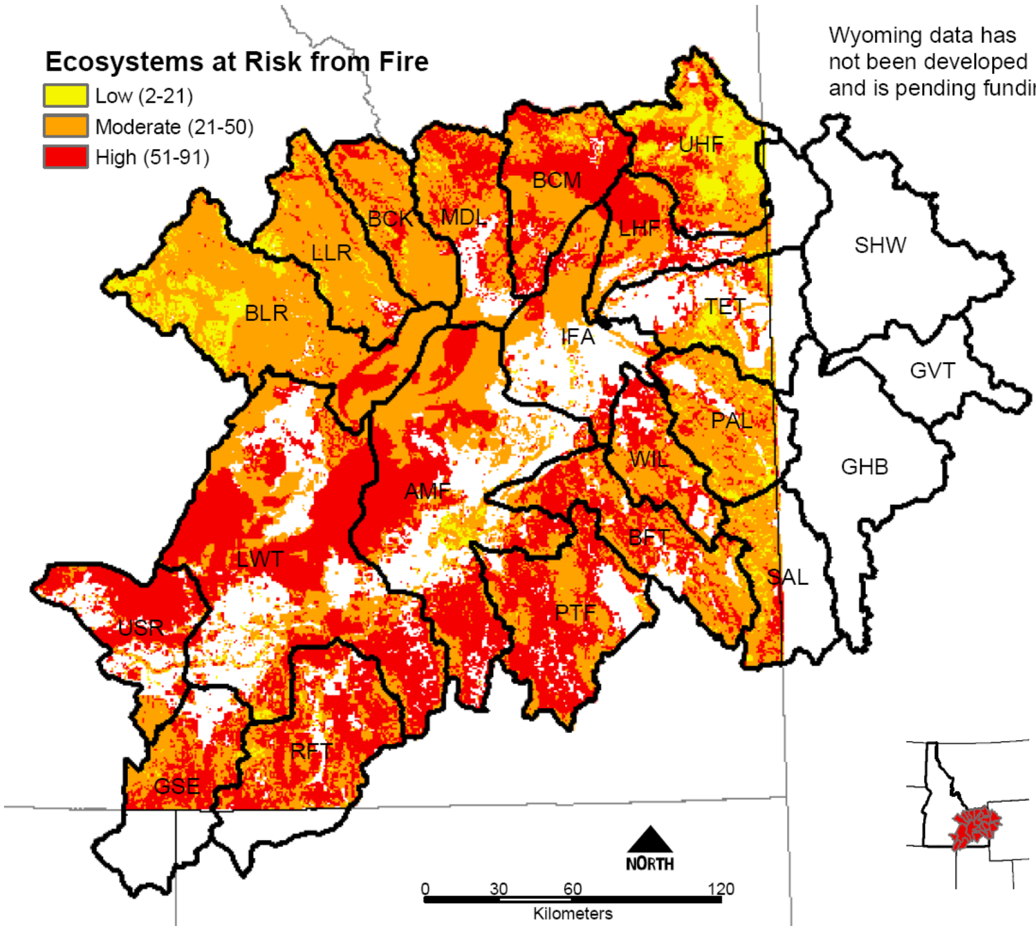

Figure 14. Predicted areas within the Upper Snake province most likely to have severe burns, taking into account FRCC, ignition probability and fire weather hazard (Northern Regional National Fire Plan Cohesive Strategy Assessment Team, Flathead National Forest).

Before the era of fire suppression, fires burned across the landscape at a variety of fire intensities, fire sizes, and fire return intervals based on localized climate, with fire return intervals on a cold/wet to warm/dry gradient. This created a mosaic of stand ages and a variety of vegetation conditions, from meadow and savannah to dense, old forest. Of the various frequencies and intensities of fire, it seems there are few that are entirely detrimental to all organisms. Natural landscapes are often created or maintained by burning, and the plants on these landscapes have ways of dealing with natural fire (INFMS 2003).

Each species has a unique set of characteristics that determines how it is affected by fire. Many plants have adapted to fire by evolving protective mechanisms such as thick bark. Fire may stimulate a positive response in other species, which may get bigger and produce more seeds. Even plants that are killed by fire may have coping mechanisms allowing the species to survive fire, even when individuals are burned. They may have hard seeds that survive until fire readies them to grow, or light, easily dispersed seeds that can quickly reinvade a burned area. Most employ some combination of these strategies (INFMS 2003).

The greatest effect of fire suppression on biological diversity is not on the diversity within a particular habitat (Whittaker 1977), but on the diversity of habitats across a landscape. Landscapes with high diversity resulting from fire perpetuate high species diversity by providing opportunities for the establishment and maintenance of early successional species and communities (Connell 1978, Reice 1994). Fire suppression, on the other hand, increases uniformity in habitats as competition eliminates early successional species, leaving only shade-tolerant understory plants to reproduce. Burned landscapes include habitat types dominated by early successional pines, shrubs, or herbaceous species, whereas unburned landscapes were more uniform in their cover of later successional fir-dominated communities (Stuart 2003).

Fire suppression has helped change the ecosystem dynamics of communities adapted to frequent, low-intensity wildfire. Complex landscapes are made simpler; some early and mid-successional plants and animals are extirpated; shade-tolerant tree populations rapidly expand; and the relative importance of fire as a disturbance agent is reduced, while the importance of insects and pathogens is elevated (Covington *et al*. 1994).

Sagebrush-steppe ecosystems of the Great Basin in the western United States are examples of fireprone ecosystems. Many wildlife species depend on sagebrush steppe ecosystems for survival (Knick and Van Riper III 2002). Unfortunately, a change in the natural fire regime is decreasing the extent of sagebrush ecosystems, and the populations of wildlife species that depend on sagebrush are undergoing steep declines because of habitat loss (Connelly *et al*. 2000, Pyke 2002).

Two major problems resulting from past fire suppression activities are common to the sagebrush ecosystem (Perryman 2003):

1. Longer time periods between fires (lengthened fire intervals) at higher elevations (higher precipitation zones) have allowed various junipers and/or pinyon pines and Douglas fir/lodgepole pine to encroach into mountain sagebrush-grassland communities. In the Great Basin, juniper and pinyon are relatively long-lived species (approximately 1,000 and 600 years, respectively). Depending on specific location, however, 66% to more than 90% of individual trees are less than 130 years old. Fire return intervals have increased from 12 to 25 years to over 100 years. These communities lose the perennial herbaceous understory as the canopy closes in large part because of competition from the encroaching conifers. This encroachment further leads to unmanageable fuel loads and very intense fires resulting in final loss or elimination of perennial herbaceous understory species, and loss of the original sagebrush habitat. Without a healthy herbaceous understory, these disturbed communities become susceptible to cheatgrass or other invasive species establishment, further reducing habitat quality for sagebrush obligates and other species—both wild and domestic—that utilize sagebrush habitats.

2. At mid- and lower elevations, longer fire intervals have created decadent, climax sagebrush systems that dominate very large areas on the landscape. These communities have lost the perennial herbaceous understory in large part because of competition from dense, competitive sagebrush plants. The shrub overstory in these systems is continuous and contiguous leading to fuel continuities that burn hotter and more extensively than normal. These areas have also been invaded by cheatgrass. This species is very successful because there are no perennial, herbaceous species with which

to compete. After extensive fires in these systems, cheatgrass proliferates even more because fire removes sagebrush (and other shrubs)—the only competitor in the system. As fire intervals become shorter because of the fuel loading of the annual brome, areas that a single generation ago were sagebrush grasslands could be converted to annual grasslands dominated by cheatgrass.

4. Grazing/Browsing

One of the most significant human-induced effects on the western landscape has been the widespread introduction of domestic livestock. Brought to the Southwest by the Spanish in the late 1500s, cattle and sheep only began to have a significant impact on the region's biota with their large-scale transportation into the region via the railroads in the late 1800s. By 1890, hundreds of thousands of cattle and/or sheep were grazing on the rangelands of the west (CPLUHNA 2003).

By the time federal forest reserves were proclaimed in the 1890s, ranchers had become accustomed to unregulated use of public lands as range for livestock. As a result of these excessive stocking numbers, once rich grasslands were seriously degraded even before the end of the 1800s, after less than a human generation of use. By the early 1900s, overstocking of sheep in the region's highlands had brought forest regeneration to a halt. The fire ecology of the region's forests, particularly the once grass-rich ponderosa pine forests, was drastically altered, causing significant long-term changes to their structure and composition. By 1912, livestock pressures had penetrated the most remote, timbered, and mountainous areas. Over one hundred years later, the effects of intense grazing in the latter part of the 19th century can still be readily seen in many parts of the West (CPLUHNA 2003).

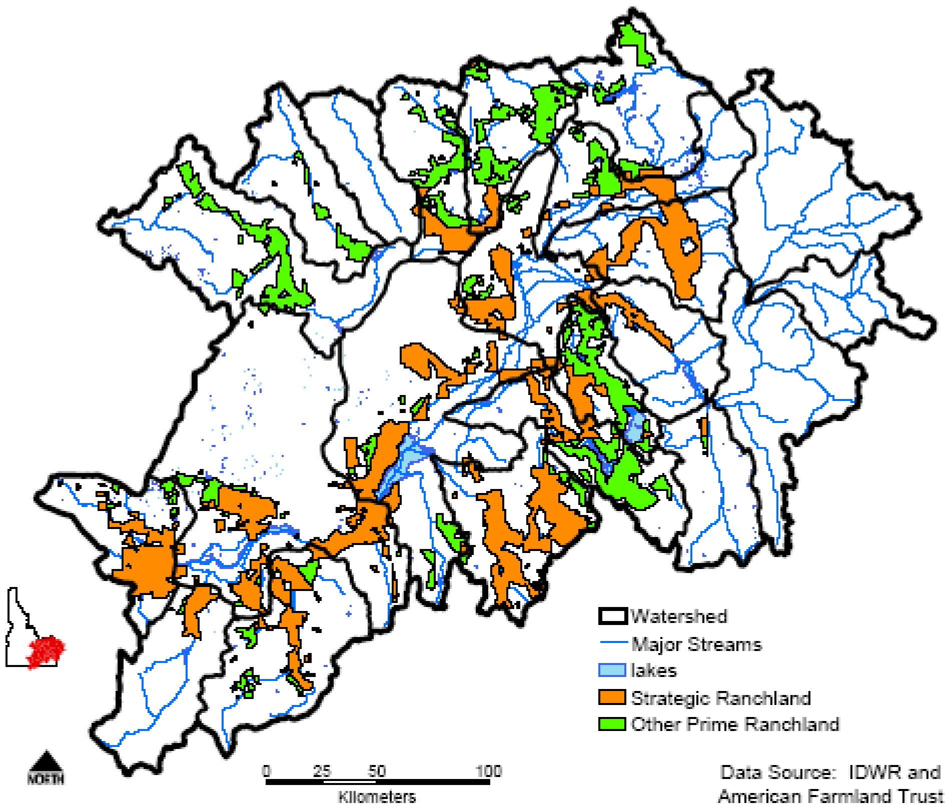

Figure 15. Land ownership and ranchland use in the Upper Snake province, Idaho (IDWR and American Farmland Trust 2003).

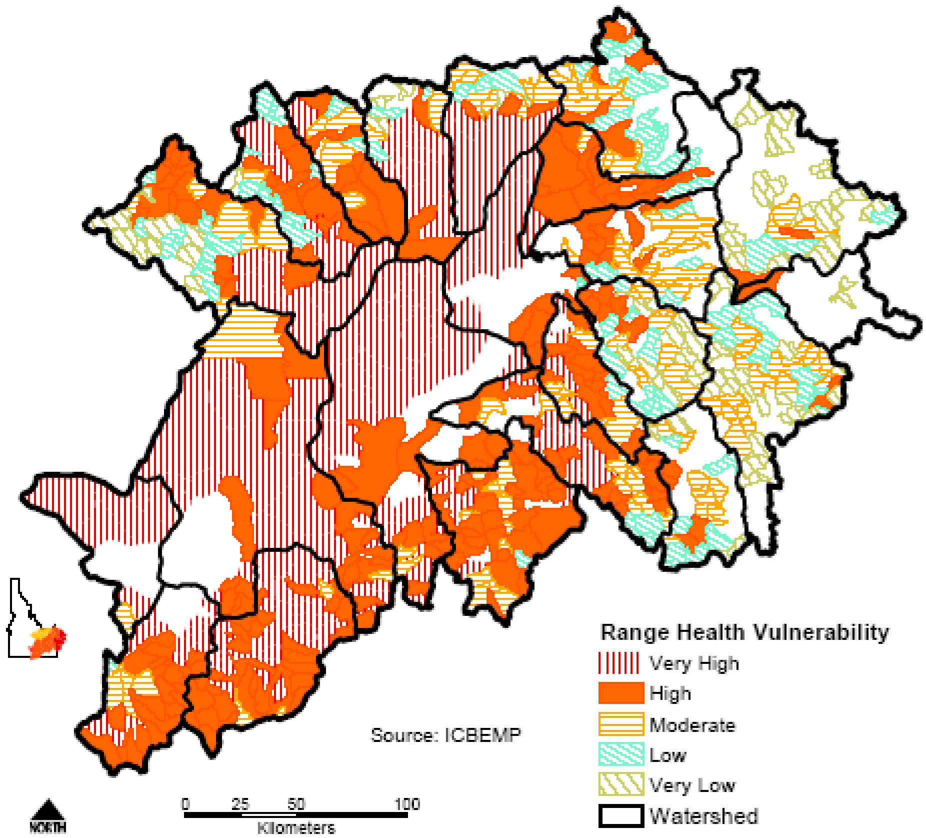

Figure 16. Rangeland condition in the Upper Snake province (ICBEMP 1997).

Livestock have played and continue to play an even more important role in changes to ecosystems in the West. Ninety-one percent of the public land in the western United States is grazed (Belsky and Blumenthal 1997) and 61% of the total area in the Upper Snake province is impacted by grazing and browsing by domestic animals (Figure 17 and Table 9). Undisturbed herbaceous ecosystems across the western United States are rare. Still, a precise determination of the ecological effects of grazing often is difficult to obtain because ungrazed land is extremely rare; exclosures are small; exact figures on grazing intensities are scarce; and approaches to evaluate the effects of grazing are not standardized (Belsky and Blumenthal 1997, Flather *et al.*1994, Fleischner 1994). For example, the status of grazing and browsing by domestic animals in the Upper Snake province is unknown for approximately 51% of the total area (Table 9 and Figure 17).

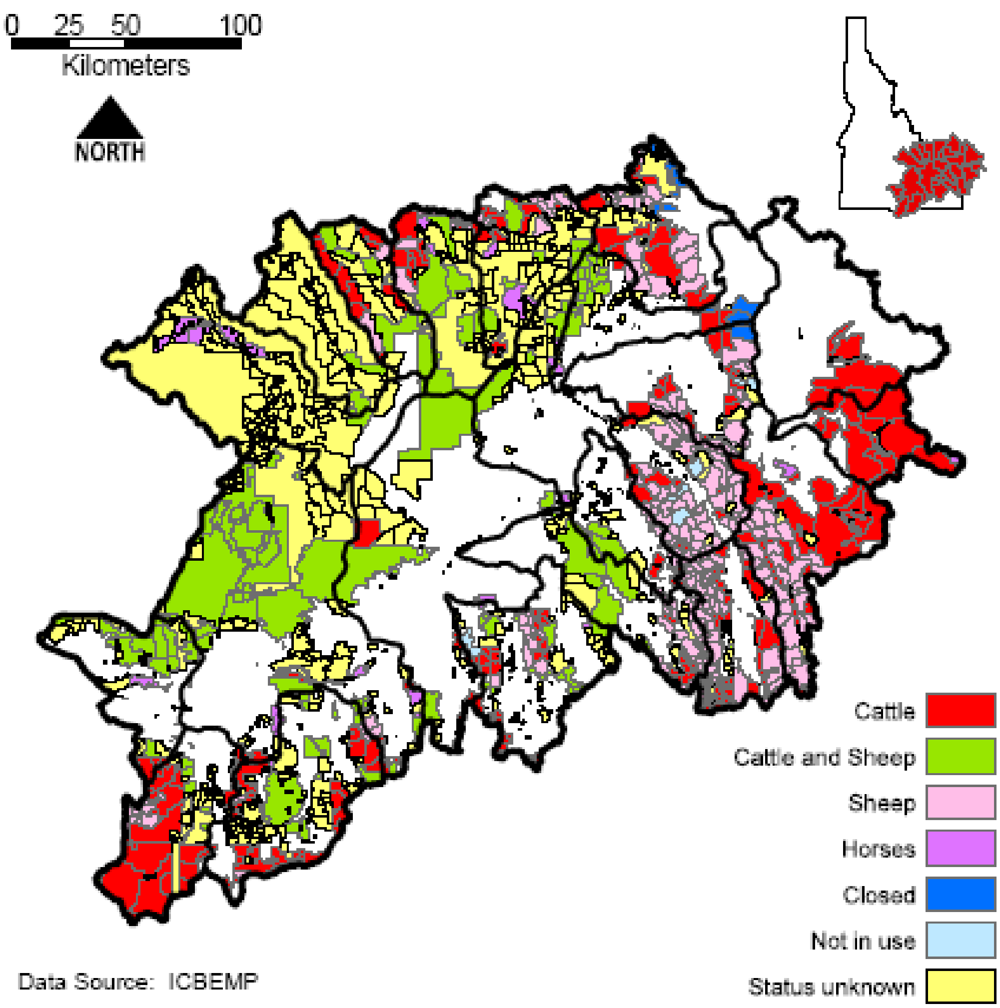

Figure 17. Occurrences of grazing and browsing activities by domestic animals in the Upper Snake province, Idaho (ICBEMP 1997).

4.1 Grazing/Browsing Activity in the Upper Snake Province

Grazing and browsing activities by domestic animals occur throughout the Upper Snake province (Table 9, Figure 17). The grazing and browsing status for many watersheds within the Upper Snake province are unknown, with exceptions being the Snake Headwaters and Gros Ventre watersheds. Comparatively, very little grazing activity (16%) occurs in the Snake Headwaters watershed, whereas, 82% of the Gros Ventre watershed experiences grazing and browsing activities predominating by cattle.

Table 10 presents the percentage of area impacted by grazing for each of the eight focal habitats in the Upper Snake province. Grazing by cattle appears to have the greatest impacts in all the focal habitats, except for aspen habitats where sheep have a greater impact.

Table 9. Percentage of area impacted by grazing/browsing livestock by watershed in the Upper Snake province (Source: ICBEMP 1997, GAP II Scott *et al*. 2002).

Snake Headwaters Subbasin	Major Hydrologic Unit (watershed) [a]				
	GHB	GVT	PAL	SAL	SHW
Cattle	41	78	13	27	16
Cattle and sheep				<1	
Closed					<1
Horses	<1	2		<1	<1
Not in use			5		<1
Sheep	30	<1	54	40	<1
Status unknown	2	1	6	5	<1
Area ungrazed	26	18	22	28	84

[a] GHB= Greys–Hoback watershed; GVT= Gros Ventre watershed; PAL=Palisades watershed; SAL=Salt watershed; SHW=Snake Headwaters watershed.

Upper Snake Subbasin	Major Hydrologic Unit (watershed) [a]											
	AMF	BFT	GSE	IFA	LHF	PTF	RFT	TET	UHF	USR	LWT	WIL
Cattle	3	5	54	<1	5	12	21	14	25	3	2	5
Cattle and sheep	16	24	4	10	14	8	15	<1	<1	26	37	12
Closed					5			3	5			
Horses	<1	<1		2	1	<1	<1	<1		2	1	
Not in use	<1					2		1				
Sheep	<1	13	5		1	6	2	17	24	<1	<1	6
Status unknown	11	15	17	23	7	13	23	5	19	17	25	9
Area ungrazed	70	43	20	65	66	59	39	59	25	52	34	67

[a] AMF=American Falls watershed; BFT=Blackfoot watershed; GSE=Goose watershed; IFA= Idaho Falls watershed; LHF=Lower Henrys Fork watershed; Portneuf watershed; RFT=Raft watershed; TET=Teton watershed; UHF=Upper Henrys Fork waterhshed; LWT=Lake Walcott watershed; WIL=Willow watershed.

Closed Basin Subbasin	Major Hydrologic Unit (watershed) [a]				
	BCM	BCK	BLR	LLR	MDL
Cattle	11	25	<1	<1	14
Cattle and sheep	12	26	2	7	32
Closed					
Horses	5	<1	5		2
Not in use					
Sheep	8	7	<1	<1	9
Status unknown	60	32	85	92	43
Area ungrazed	4	9	8	0	0

[a] BCM=Beaver-Camas watershed; BCK=Birch watershed; Big Lost River watershed; Little Lost River watershed; Medicine Lodge watershed.

Table 10. Percentage of area impacted by grazing domestic animals for each of the focal habitats in the Upper Snake province (Source: ICBEMP 1997, GAP II Scott *et al.* 2002).

Focal Habitat Type	Cattle	Cattle and Sheep	Sheep	Horses	Closed	Not in use	Status Unknown	Not Grazed
Riparian/herbaceous wetlands	19	4	8	<1	<1	<1	18	49
Open water	2	<1	<1	<1		<1	12	86
Shrub-steppe	13	24	3	2	<1	<1	36	22
Pine/fir forest	30	1	22	<1	2	<1	12	30
Juniper/mountain mahogany	26	18	<1	2		<1	30	24
Whitebark pine	14	<1	7	<1	<1	<1	65	13
Aspen	28	5	36	<1	1	1	9	20
Mountain Brush	5	25	2	<1		<1	42	25

4.2 Impacts to Riparian/Wetland Habitats

Riparian areas are critical ecosystems in the semi-arid landscape of the West, yet many have been seriously degraded and others entirely lost due to human activities and land use. The abundance of food, water, and shade, which attracts wildlife to these areas, also attracts livestock. Despite widespread recognition of the problem and attempts to remove or restrict livestock from riparian areas, riparian degradation due to overgrazing is a serious problem (Belsky *et al.* 1999).

The direct effects of livestock grazing on the wetland riparian habitats have been summarized as follows (Harper *et al*. 2003):

- Higher stream temperatures from lack of sufficient woody streamside cover
- Excessive sediment in the channel from bank and upland erosion
- High coliform bacterium counts
- Channel widening from hoof-caused bank sloughing and later erosion by water
- Change in the form of the water column and the channel in which it flows
- Change, reduction, or elimination of vegetation
- Elimination of riparian areas by channel degradation and lowering of the water table
- Gradual stream-channel trenching or braiding depending on soils and substrate composition, with concurrent replacement of riparian vegetation with more xeric plant species

Riparian systems at lower elevations are now increasingly characterized by a reduction of plant species diversity and density. Overgrazing of palatable native species such as willows and cottonwood saplings, combined with the introduction of less palatable nonindigenous species such as Russian olive (*Elaeagnus angustifolia*), has also contributed to changes in overall plant community structure. Road construction associated with grazing operations has caused additional degradation of riparian areas, especially through bank erosion. The carrying capacity of the habitat and fish survival have been reduced by land and water management activities within the province that have affected hydrology, sedimentation, habitat distribution and complexity, and water quality (CBFWA 1999).

Livestock may directly affect fish through trampling or ingestion of adults, larvae, or eggs (Roberts and White 1992). Livestock waste is potentially poisonous to some fish (Cross 1971, Taylor *et al*. 1991), and may increase nitrogen levels, thereby affecting nutrient cycling and encouraging algae growth. High-quality freshwater habitats are critical to the long-term strength and persistence of native resident and anadromous salmonid populations in the Columbia River basin. These fish have generally fared best in areas least disturbed by humans. Grazing and browsing by domestic livestock have the potential to impact salmonid spawning and rearing success.

4.3 Impacts to Shrub-Steppe

Livestock may graze plants that are listed, forage for listed species, or provide cover or protection for listed species. Grazing can also affect the vegetative community and ecosystem functioning (Shreve 1931, Niering *et al*. 1963, Abouholder 1992, USFWS 1999).

Livestock grazing alters the species composition of communities, disrupts ecosystem functioning, and alters ecosystem structure (Fleischner 1994). The main direct impacts from cattle are the grazing of plants and trampling of vegetation and soil (Marlow and Pogacnik 1985). Grazing can alter the prey availability of certain predators by removing herbaceous vegetation, which serves as food, and cover for small mammals (Ward and Block 1995). Grazing can also alter fire regimes, a circumstance that is generally deleterious to ecosystem functioning (USFWS 1999).

A reduction in vegetation cover increases raindrop impact, decreases soil organic matter and soil aggregates, and decreases infiltration rates (Blackburn 1984, Orodho *et al*. 1990). Other detrimental impacts include increased overland flow, reduced soil water content, and increased erosion (DeBano and Schmidt 1989, Guthery *et al*. 1990, Orodho *et al*. 1990). Continuous yearlong grazing can result in large bare areas around water sources and established trails to and from points of livestock concentrations (Platts 1990).

Watershed condition and function can be affected by impacts to vegetation and litter from livestock grazing (Gifford and Hawkins 1978, Busby and Gifford 1981, Blackburn 1984, DeBano and Schmidt 1989, Belnap 1992, Belsky and Blumenthal 1997). Heavy grazing effects are well known and can be severe (Guthery *et al*. 1990, Platts 1990).

4.4 Impacts to Forests

Over the last 100 years, the structure, composition, and dynamics of western, semiarid, interior forests have changed dramatically. These forests, dominated at low elevations by ponderosa pine (*Pinus ponderosa*) and at middle elevations by Douglas-fir (*Pseudotsuga menziesii*), grand fir (*Abies grandis*), and western larch (*Larix occidentalis*), were once commonly described as open woodlands of widely spaced, majestic trees, underlain by dense grass swards (Cooper 1960, Peet 1988, Habeck 1990, Covington and Moore 1994). During the 1900s, most of these forests were clearcut, roaded, and fragmented so that only a small fraction of the original forests remains (Belsky and Blumenthal 1997).

Livestock grazing is occasionally mentioned as contributing to “forest health” problems, but it is simply noted as one of many factors reducing the frequency of surface fire (Belsky and Blumenthal 1997). Nevertheless, a large number of authors have suggested that fire began to decline in frequency and forests began to increase in density soon after livestock were first introduced into the Interior West (Leopold 1924, Weaver 1950, Cooper 1960, Madany and West 1983, Peet 1988).

By the early 1800s in the Southwest, and the late 1800s in the Northwest, virtually all plant communities that supported grass and sedge production, including ponderosa pine and mixed-conifer forests, were heavily stocked with cattle and sheep (Savage and Swetnam 1990, Oliver *et al*. 1994). After they were clearcut and seeded with grasses, even previously dense forests provided “transitory” range for livestock. As shade, drought, water stress, and pests kill small and large trees alike, fuel loads increase. These woody fuels cause what otherwise might be low intensity surface fires to develop into intense conflagrations, resulting in high tree mortality (Belsky and Blumenthal 1997).

Herbaceous Understory

By grazing and trampling herbaceous species, livestock affect understory species composition directly; this differs from the more indirect effects they have on overstory trees (Belsky and Blumenthal 1997). Impacts vary with animal density and distribution: the more evenly grazers are distributed, the lower their impact on any given area (Gillen *et al*. 1984). Unfortunately, cattle show strong preferences for certain environments, leading to high use in some areas and little or no use in others (Belsky and Blumenthal 1997). This is particularly true in western, interior forests, where steep slopes and increasingly dense forests make much of the landscape unattractive (Clary 1975, Roath and Krueger 1982).

Understory Cover and Composition

Livestock also alter understory plant composition as animals select more palatable species, leaving the less palatable ones to increase in dominance (Smith 1967, Hall 1976, Skovlin *et al*. 1976). The effects of livestock grazing on understory composition and biomass are sometimes difficult to distinguish from the effects of tree canopy closure (Smith 1967), which creates shadier, cooler, and moister conditions. However, when Arnold (1950) separated the effects of livestock grazing from those of tree canopy closure, he found that grazing alone was sufficient to reduce the cover of most native bunchgrass species.

Domestic livestock, as well as agriculture, logging, road construction, and other practices that disturb soils, have been instrumental in the establishment of alien weedy species in western forests (Franklin and Dyrness 1973, Johnson *et al*. 1994). Livestock act as vectors for seeds, disturb the soil, and reduce the competitive and reproductive capacities of native species. Exotic weeds have been able to displace native species, in part, because native grasses of the Intermountain West and Great Basin are not adapted to frequent and close grazing (Stebbins 1981, Mack and Thompson 1982). Consequently, populations of native species have been severely depleted by livestock, allowing more grazing-tolerant weedy species to invade. It is possible that in some areas aggressive alien weeds such as cheatgrass (*Bromus tectorum*) and Kentucky bluegrass (*Poa pratensis*) have permanently replaced native herbaceous species (Smith 1967, Laudenslayer *et al*.1989).

Forest Soils and Plant Litter

By consuming aboveground plant biomass, domestic livestock also reduce the amount of biomass available to be converted into litter and, therefore, increase the proportion of bare ground (Belsky and Blumenthal 1997). Schultz and Leininger (1990) found, for example, that grazed areas of a riparian meadow had 50% lower litter cover and 400% more bare ground than ungrazed areas.

Johnson (1956) reported that litter biomass in a ponderosa pine/bunch grass ecosystem was reduced 40% and 60% by moderate and heavy livestock grazing, respectively. Such reductions in litter may have severe consequences on forested ecosystems because litter is critical for slowing overland flow; promoting water infiltration; serving as a source of soil nutrients and organic matter; and protecting the soil from freezing and the erosive force of raindrops (Thurow 1991, Facelli and Pickett 1991).

4.5 Compaction and Infiltration

The rate at which water penetrates the soil surface governs the amount of water entering the ground and the amount running off. Livestock alter these rates by reducing vegetative and litter cover and by compacting the soil (Lull 1959). As a result, livestock grazing is usually associated with decreased water storage and increased runoff (Belsky and Blumenthal 1997). Lower soil moisture contents in turn reduce plant productivity and vegetative cover, creating negative feedback loops that further degrade both the plant community and sod structure (Belsky and Blumenthal 1997). These changes in soil structure may also lead to increased water stress and tree mortality during dry periods, exacerbating the water stress resulting from the higher tree densities. Therefore, disturbance and compaction of forest soils by cattle and sheep may contribute to the increased incidence of water-stress, tree mortality, and fire in western forests (Belsky and Blumenthal 1997).

4.6 Runoff and Erosion

As livestock reduce plant cover and compact the soil, the volume of overland water flow increases (Belsky and Blumenthal 1997). With increasing runoff, soil erosion also increases (Dunford 1954). Smith (1967), for example, found that grazed pastures in a ponderosa pine/bunchgrass range lost 3 to 10 times more sediment than ungrazed pastures. The strong relationship between runoff and erosion was also demonstrated by Forsling (1931), who found that summer rainstorms on grazed subalpine hillsides accounted for 53 to 85% of annual sediment loss. Following elimination of livestock from the watershed, vegetative cover increased 150%, whereas the proportion of annual runoff from summer rainstorms dropped 72%, causing a corresponding 50% drop in sediment loss (Forsling 1931).

4.7 Big Game Impacts and Dietary Overlap with Livestock

Numerous studies have documented the impact of grazing and browsing by big game animals on habitats (Clark 2003). Heavy browsing by big game animals may inhibit shrub and grass cover, alter the plant composition, alter vegetative structure, prevent adequate plant reproduction, or cause direct mortality (Gaffney1941, Korfhage *et al*. 1980, Edgerton 1987, Irwin *et al*.1994, Nolte and Dykzeul 2000). Generally, big game impacts to habitat become significant when animals become so numerous as to exceed the carrying capacity of the habitat. This may occur at spatial and temporal scales depending on the season and the condition of the habitat (e.g., winter range or naturally or artificially altered habitat) (Begon and Mortimer 1986).

Dietary overlap between big game animals and livestock is subject to the specific forage components required by the animals and the timing of ungulate use. Dietary overlap between elk and cattle is most likely to occur on fall cattle range that is used by elk later in the year as winter range (Clark 2003). Dietary overlap between elk and domestic sheep occurs during the summer when both species rely heavily on forbs; however, elk tend to be more selective between forb species than do sheep (Clark 2003). Elk tend to remain on a forb-dominated diet throughout the summer while sheep diets transition from forbs to grasses and browse as the season progresses (Clark 2003).

The diets of cattle and mule deer are most prone to overlap during the spring when mule deer diets contain a substantial amount of graminoids. However, spring mule deer diets are primarily dominated by forbs and browse, while spring cattle diets contain mostly graminoids. Consequently, the degree of diet overlap between cattle and mule deer is relatively small (Clark 2003). The diets of domestic sheep and mule deer overlap during the spring and fall when both ungulates are using browse and forbs. When browse is limited, both domestic sheep and mule deer rely heavily on graminoids (Clark 2003).

Winter bighorn sheep diets and summer-fall cattle diets have the greatest potential for overlap of any seasonal diet combination between these two ungulates. Under this combination, the diets of both cattle and bighorn sheep are dominated by graminoids. However, as with elk and cattle, the differences in seasonal habitat use displayed by cattle and bighorn sheep minimize the potential for dietary competition between these species (Clark 2003). Dietary overlap between domestic sheep and bighorn sheep is not as well understood (Clark 2003).

Dietary overlap between cattle and pronghorn is generally considered minimal, as the two ungulates do not share significant food sources or ranges (Clark 2003). Dietary overlap between domestic sheep and pronghorn is typically the highest during the spring and fall when both species are consuming sizable quantities of browse. However, as with cattle and pronghorn, the degree of similarity between the diets of pronghorn and sheep is generally quite low (Clark 2003).

5. Timber Harvest

Logging began in the vast forests of the west in the 1870s and 1880s when materials and supplies were needed for construction of the transcontinental railroad. Subsequent settlement of the frontier by pioneers and immigrants increased the demand for timber products. In the early 1900s, new technologies allowed greater harvest on terrain previously unavailable for logging. In mid-century, dramatic increases in timber harvest and road building occurred in the National Forests and private lands throughout the West. An agricultural model of sustainable forestry favoring even-aged stands became the standard of timber-harvest operations. During this time, typical harvests removed one-third to two-thirds of the available volume. At these residual stocking rates, stem density increased while tree size and age decreased (CPLUHNA 2003).

Historically, the most important timber species in Idaho were ponderosa pine and western white pine (*Pinus monticola*). Both have declined since 1952, ponderosa pine by 40% and western white pine by 60%. Byler *et al*. (1994) estimated that the extent of western white pine might now be only 10% of what it was in 1900.

Timber harvest has occurred throughout the Upper Snake province (Figure 18, Table 11). Very low to medium harvest activities have occurred in the central Upper Snake subbasin and portions of the Snake Headwaters and Closed Basin subbasins, in the American Falls, Lake Walcott, and Upper Snake–Rock watersheds. The most intense timber harvest activities appear to have occurred in the Salt, Upper Henrys Fork, Willow, Beaver–Camas and Big Lost watersheds (Figure 18, Table 11).

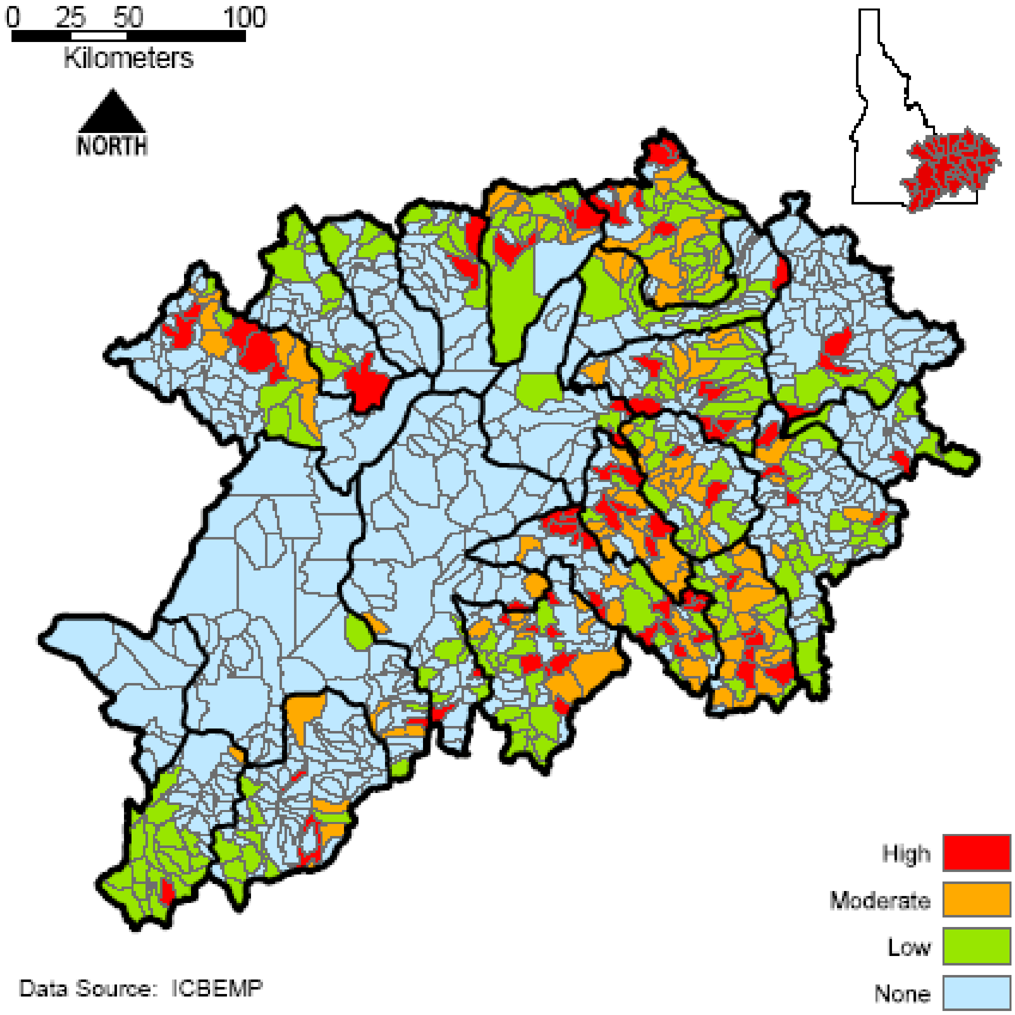

Figure 18. Relative timber harvest impacts in the Upper Snake province, Idaho (ICBEMP 1997).

Table 11. Comparing the relative percentages of timber harvest by watershed in the Upper Snake province (ICBEMP 1997).

Snake Headwaters Subbasin Relative Category	Major Hydrologic Unit (watershed)[a]				
	GHB	GVT	PAL	SAL	SHW
High	5	4	4	14	7
Low	7		24	49	
Medium	33	31	36	32	12
No harvest	56	65	36	5	81

[a] GHB= Greys–Hoback watershed; GVT= Gros Ventre watershed; PAL=Palisades watershed; SAL=Salt watershed; SHW=Snake Headwaters watershed.

Upper Snake Subbasin Relative Category	Major Hydrologic Unit (watershed)[a]											
	AMF	BFT	GSE	IFA	LHF	PTF	RFT	TET	UHF	USR	LWT	WIL
High	2	24	2	3	3	10	5	14	15	<1	<1	25
Low	2	19	2		7	21	11	16	32		2	48
Medium	5	22	56	12	39	32	19	40	42	<1	3	5
No harvest	92	35	40	85	52	37	66	30	11	100	95	22

[a] AMF=American Falls watershed; BFT=Blackfoot watershed; GSE=Goose watershed; IFA= Idaho Falls watershed; LHF=Lower Henrys Fork watershed; Portneuf watershed; RFT=Raft watershed; TET=Teton watershed; UHF=Upper Henrys Fork waterhshed; LWT=Lake Walcott watershed; WIL=Willow watershed.

Closed Basin Subbasin Relative Category	Major Hydrologic Unit (watershed)[a]				
	BCM	BCK	BLR	LLR	MDL
High	12		12	13	10
Low	14		14		
Medium	55	21	11	33	19
No harvest	19	79	62	54	71

[a] BCM=Beaver-Camas watershed; BCK=Birch watershed; Big Lost River watershed; Little Lost River watershed; Medicine Lodge watershed.

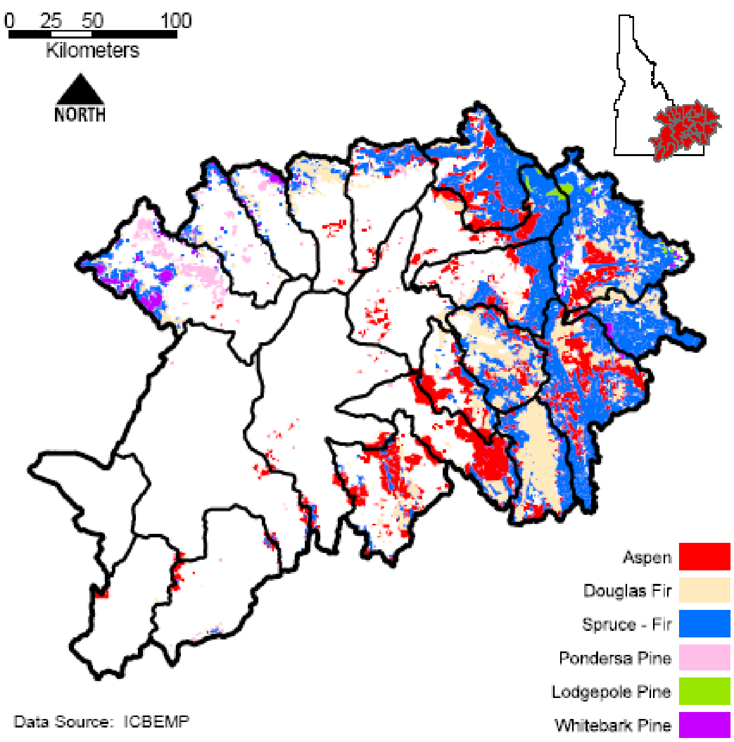

Figure 19. Historical forest species compositions in the Upper Snake province (ICBEMP 1997).

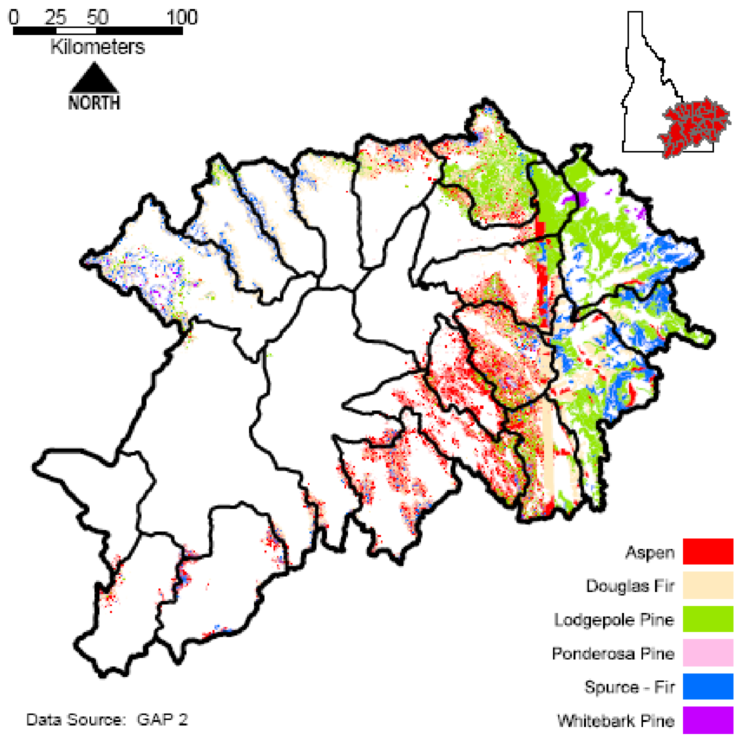

Figure 20. Current forest species compositions in the Upper Snake province (GAP II, Scott *et al.* 2002).

5.1 Effects of Timber Harvesting on Soil

Soil is a primary determinant of long-term site productivity, and timber harvest can produce a variety of changes in soil properties that affect long-term site productivity.

Timber harvest and subsequent site preparation usually result in microclimate changes that influence subsequent biological processes. The most important of these include changes in light, temperature, and moisture. Soil chemistry and microbial processes can be affected in either a beneficial or detrimental manner (Harvey *et al*. 1989).

Timber harvest can cause extensive losses and disturbances of surface organic matter. This potential has important implications for soil chemical, biological, and physical properties (Harvey *et al*. 1987, Jurgensen *et al*. 1990). Timber harvest reduces soil organic matter both by physical loss at time of harvest and by increased microbial activity caused by soil disturbance (Jurgensen *et al*. 1990). Site-preparation techniques, particularly slash piling and windrowing, can cause productivity problems related to organic matter because of the disturbance of large areas of the forest floor (Harvey *et al*. 1987, 1989). Substantive losses of surface organic matter lead to declines in productivity (Powers 1991).

Forest management activities, especially timber harvest and road construction, have been shown to increase erosion rates on forest lands (Megahan 1991). Skid trails and other high-traffic areas are particularly susceptible to erosion (Cullen *et al*. 1991). Debris landslides and gullying cause serious and long-term reductions in site productivity, but the areas affected are small. Surface erosion occurs over much larger areas and reduces site productivity, but the magnitude of the reduction is poorly defined because of the compounding effects of compaction on logged areas and the water repellency of burned areas (Megahan 1991).

Timber harvest can affect both the processes and structures that result in fish habitat. Habitat alterations can adversely affect all life stages of fishes, including migration, spawning, incubation, emergence, and rearing (Lee *et al*.1997). The effects of timber harvest on fish habitat are likely to be varied and dynamic.

Structure

Four major effects of timber harvest on stream structures can be summarized as follows (Chamberlin *et al*. 1991):

1. Increases in peak flows or the frequency of channel modifying flows from increased snowmelt or rain-on-snow events can increase bed scour or accelerate bank erosion.

2. Increases in sediment supply from mass movements or surface erosion, bank destabilization, or instream storage losses can cause aggradation, pool filling, and reduction in gravel quality.

3. Streambank destabilization from vegetation removal, physical breakdown, or channel aggradation adds to sediment supply and generally results in a loss of the channel structures that confine flow and promote the habitat diversity required by fish populations.

4. Loss of stable instream woody debris by direct removal, debris torrents, or gradual attrition as streamside forests are converted to managed stands of smaller trees will contribute to loss of sediment storage sites, fewer and shallower scour pools, and less effective cover for rearing fish.

Streamflow

The hydrologic effects of timber management activities vary with many environmental factors, but Chamberlin *et al*. (1991) suggest that the following broad generalizations apply:

1. Harvest activities such as road building, falling, yarding, and burning can affect watershed hydrology and streamflow much more than can other management activities such as planting and thinning.

2. Clearcutting causes increased snow deposition in forest openings and advances the timing and rate of snowmelt. The effect lasts several decades until stand aerodynamics approach those of the surrounding forest. Snowmelt can be accelerated by large wind-borne energy inputs of warm rain falling on snow.

3. Harvested areas contain wetter soils than unlogged areas do during periods of evapotranspiration and therefore have higher groundwater levels and more potential late-summer runoff. The effects last 3 to 5 years until new root systems occupy the soil.

4. Road systems, skid trails, and landings accelerate slope runoff, concentrate drainage below them, and can increase soil water content.

Water Quality

Stream temperature is affected by eliminated streamside shading, disrupted subsurface flows, reduced stream flows elevated sediments, and morphological shifts toward wider and shallower channels with fewer deep pools (Lee *et al*. 1997). Harvest activities that impose large oxygen demands on streams exacerbate the normal stresses that low flows place on fish (Chamberlin *et al*.1991).

Sediment

Timber harvest can influence both upland erosional processes and the way that forest streams process sediment in their channels. Forest management activities that substantially change the magnitude, timing, or duration of sediment transport and overwhelm the ability of fish to cope with or avoid resulting stress are of most concern (Chamberlin *et al*. 1991). Roads and mass movements associated with roads are the largest sources of sediment production stemming from timber-harvest activities (Cook and O'Laughlin 2000).

Large Woody Debris

Because the supply of large woody debris to stream channels is typically a function of the size and number of trees in riparian areas, it can be profoundly affected by timber-harvest shifts in the composition and size of trees within the riparian area. Large woody debris influences channel morphology, especially in forming pools and instream cover, retaining nutrients, and storing and buffering sediment. Reduction in the amount of large woody debris within streams, or within the distance equal to one site-potential tree height from the stream, can reduce instream complexity. Large woody debris increases the quality of pools by providing hiding cover, slow water refuges, shade, and deep-water areas (Maser *et al*. 1988).

Roads

By far the greatest concerns about timber harvest and water quality result from the issue of roads. Serious degradation of fish habitat can result from poorly planned, designed, located, constructed, or maintained roads. Roads directly affect natural sediment and hydrologic regimes by altering streamflow, sediment loading, sediment transport and deposition, channel morphology, channel stability, substrate composition, stream temperatures, water quality, and riparian conditions within a watershed (Chamberlin *et al*. 1991, Furniss *et al*. 1991, Lee *et al*. 1997).

5.3 Impacts to Wildlife

Timber harvest can have positive, negative, and neutral effects on wildlife habitat, depending on the life requirements of the species inhabiting the area (Cook and O'Laughlin 2000).

One important aspect of the relationship between wildlife and timber harvest is not how many trees are removed but how much vegetation remains as food and cover for the species inhabiting the area. Populations of animals of low mobility and specific habitat requirements (e.g., amphibians, reptiles, small birds, and small mammals) can be adversely affected at the time of a timber harvest, even if the cut is limited to a small area or a single tree. Highly mobile animals (e.g., large birds and mammals) are less affected. The age and size classes of trees that remain after harvest and their spatial relationship is important (Patton 1992)

6. Invasive/Exotics

Invasive plant and animal species—also referred to as exotics, nonnatives, introduced, or nonindigenous species—are organisms that have expanded beyond their native range or have been introduced from other parts of the world. Species are considered invasive if their presence in an ecosystem will cause environmental harm, economic harm, or harm to human health. Invasive species can displace native species, alter predator-prey relationships, destroy crops, and decrease ecosystem resiliency (USEPA 2001). Some species were introduced into the wild intentionally, while others have been introduced unintentionally and expanded on their own. Invasive species are usually nonnative species, and they are often exotic species from another part of the world. Native species can also be characterized as invasive if they dominate their ecosystem because of human-induced changes to that ecosystem (USEPA 2001).

Twenty-three noxious weed species are known to occur within the Upper Snake province in Idaho (Appendix 1-6). In Wyoming, there are an estimated 46 species of noxious and exotic invasive weeds within the Snake Headwaters subbasin. The Gros Ventre, Greys–Hoback, and Snake Headwaters watersheds have the greatest number of noxious and exotic invasive weed species.

6.1 Impacts to Shrub-Steppe

A change in the natural fire regime is decreasing the extent of sagebrush ecosystems, and the populations of wildlife species that depend on sagebrush are undergoing steep declines because of habitat loss (Connelly *et al*. 2000). The invasion of cheatgrass is fueling larger and more frequent fires that are out-competing sagebrush as well as the associated forb and grass species that are native components of that ecosystem (Pyke 2002). It has been estimated that 25% of the original sagebrush ecosystem is now annual cheatgrass/medusahead (*Taeniatherum caput-medusae*)/rye grassland, and an additional 25% of the sagebrush ecosystem has only cheatgrass as an understory constituent (Perryman 2003).

6.2 Impacts to Riparian/Herbaceous Wetlands

A pest weed of Idaho’s aquatic environment is the European purple loosestrife (*Lythrum salicaria*), which was introduced in the early 19th century as an ornamental plant (Malecki *et al*. 1993). Purple loosestrife is capable of invading many wetland types, including freshwater wet meadows, tidal and non-tidal marshes, river and stream banks, pond edges, reservoirs, and ditches. It has been spreading at a rate of 115,000 hectare per year and is changing the basic structure of most of the wetlands it has invaded (Thompson *et al*. 1987). Competitive stands of purple loosestrife have reduced the biomass of 44 native plants and endangered wildlife (Gaudet and Keddy 1988). Loosestrife now occurs in 48 states and costs $45 million per year in control costs and forage losses (ATTRA 1997, Pimentel *et al*. 1999).

A second aquatic weed of concern in the Upper Snake Province is Eurasian watermilfoil (*Myriophyllum spicatum* L.). Eurasian watermilfoil forms large, floating mats of vegetation on the surface of lakes, rivers, and other water bodies, preventing light penetration for native aquatic plants and impeding water traffic. The plant thrives in areas that have been subjected to various kinds of natural and man-made disturbance.

6.3 Impacts to Pine/Fir Forests

An ecologically significant weed to forested habitats in the Upper Snake province is the spotted knapweed (*Centaureai maculosa*). This species infests a variety of natural and semi-natural habitats including barrens, fields, forests, prairies, meadows, pastures, and rangelands. It out-competes native plant species, reduces native plant and animal biodiversity, and decreases forage production for livestock and wildlife. Spotted knapweed may degrade soil and water resources by increasing erosion, surface runoff, and stream sedimentation. It has increased at an estimated rate of 27% per year since 1920 and has the potential to invade about half of all rangeland (35 million acres) in Montana alone (Carpinelli 2003). Spotted knapweed is capable of establishing itself into undisturbed sites; however, disturbance allows for rapid establishment and spread.

6.5 Impacts to Whitebark Pine

The two most serious factors limiting white-bark populations in the Pacific Northwest are altered fire regimes (discussed elsewhere) and the exotic, invasive fungus whitebark pine blister rust (*Cronartium ribicola*).

Although currents and gooseberries (also known as Ribes) are not impacted by white-bark pine blister rust, these plants serve as alternate hosts for the fungus (Ellis and Horst 1998). On the pine in spring, pycnial spores give rise to aecial spores, which may fly 640 to 1,280 km to infect the leaves of Ribes plants. On the alternate host plant, two spore stages follow, which give rise to another airborne spore stage, which infects the pine again (Ellis and Horst 1998).

Although whitebark pine blister rust can damage all North American white pine species, whitebark pine is the most vulnerable, with fewer than one in 10,000 trees resistant to blister rust. Because white-bark pine cones form in the top third of the tree and blister rust tends to kill trees from the top down, a tree's ability to produce seed is eliminated by the rust long before the tree dies (Kendall 1995).

8. References

Abouhalder, F. 1992. Influence of livestock grazing on saguaro seedling establishment. In: C.P. Stone and E.S. Bellantoni, editors. Proceedings of the symposium on research in Saguaro National Monument; Tucson, AZ. p. 57–61.

Agee, J.K. 1993. Fire ecology of Pacific Northwest forests. Island Press, Washington, DC. 493 pp.

Agee, J.K. 1994. Fire and weather disturbances in terrestrial ecosystems of the eastern Cascades. General Technical Report PNW-320. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, OR.

Ahlstrand, G.M., and C.H. Racine. 1993. Response of an Alaska, USA, shrub-tussock community to selected all-terrain vehicle use. Arctic and Alpine Research 25(2):142–149.

American Farmland Trust. 2003. Strategic Ranchland in the Rocky Mountain West Mapping the Threats to Prime Ranchland in Seven Western States http://www.farmland.org/rocky_mountain/strategic_ranchlands1.

Anderson, S. 1992. Studies begin on Kaneohe Bay's toxin problem. University of Hawaii Sea Grant College Program. Makai 14(2):1, 3.

Appropriate Technology Transfer for Rural Areas (ATTRA). 1997. Purple loosestrife: public enemy #1 on federal lands. ATTRA Interior Helper, Washington, DC. Available at http://refuges.fws.gov/NWRSFiles/HabitatMgmt/PestMgmt/LoosestrifeProblem.html.

Arnold, J.F. 1950. Changes in ponderosa pine bunchgrass ranges in northern Arizona resulting from pine regeneration and grazing. Journal of Forestry 48:118–126.

Askins, R.A. 1994. Open corridors in a heavily forested landscape: impact on shrubland and forest interior birds. Wildlife Society Bulletin 22:339–347.

Begon, M., and M. Mortimer. 1986. Population ecology. Sinauer Associates, Sunderland, MA.

Belnap, J. 1992. Potential role of cryptobiotic soil crusts in semiarid rangelands. Paper presented at the Symposium on Ecology, Management, and Restoration of Intermountain Annual Rangelands; May 18–22, 1992; Boise, ID.

Belsky, A.J., and D.M. Blumenthal. 1997. Effects of livestock grazing on stand dynamics and soils in upland forests of the Interior West. Conservation Biology 11:315–427.

Belsky, A.J., A. Matzke, and S. Uselman. 1999. Survey of livestock influences on stream and riparian ecosystems in the western United States. Journal of Soil and Water Conservation 54:419–431.

Bennett, K.A., and E. Zuelke. 1999. The effects of recreation on birds: a literature review. Delaware Natural Heritage Program, Smyrna, DE.

Benninger-Truax, M., J.L. Vankat, and R.L. Schaefer. 1992. Trail corridors as habitat and conduits for movement of plant species in Rocky Mountain National Park, Colorado. Landscape Ecology 6(4):269–278.

Blackburn, W.H. 1984. Impacts of grazing intensity and specialized grazing systems on watershed characteristics and responses. In: Developing strategies for rangeland management. National Research Council/National Academy of Sciences. Westview Press, Boulder, CO. p. 927–983.

Boyle, S.A., and F.B. Samson. 1985. Effects of non-consumptive recreation on wildlife: a review. Wildlife Society Bulletin 13:110–116.

Brookes, A. 1990. Restoration and enhancement of engineered river channels: some European experiences. Regulated Rivers: Research and Management 5:45–56. John Wiley and Sons, Ltd.

Busby, F.E., and G.F. Gifford. 1981. Effects of livestock grazing on infiltration and erosion rates measured on chained and unchained pinyon–juniper sites in southeastern Utah. Journal of Range Management 34:400–405.

Byler, J.W., R.G. Krebill, S.K. Hagle, and S.J. Kegley. 1994. Health of the cedar–hemlock–western white pine forests of Idaho. In: Proceedings; interior cedar–hemlock–white pine forests: ecology and management. Washington State University, Cooperative Extension Service, Pullman, WA.

California Air Resources Board. 2002. A program update for off-road motorcycles and ATVs. Available at http://www.arb.ca.gov/msprog/offroad/mcfactst.htm.

Carpinelli, M. 2003. Spotted knapweed (*Centaurea biebersteinii* DC). Plant Conservation Alliance, Alien Plant Working Group. Available at http://www.nps.gov/plants/alien/fact/cebi1.htm.

CensusScope. 2003. CensusScope: your portal to 2000 census data. Available at http://www.censusscope.org/index.html.

Chamberlin, T.W., R.D. Harr, F.H. Everest. 1991. *Timber harvesting, silviculture, and watershed processes.* In: W.R. Meehan (ed.), “Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats.” American Fisheries Society, Special Publication Number 19. Bethesda, Maryland.

Clark, P.E. 2003. Livestock–big game interactions: a selected review with emphasis on literature from the interior Pacific Northwest. Eastside Ecosystem Management Project. Contract No. 43-OEOO-4-9156. Available at http://www.icbemp.gov/science/clarkpatrick.pdf.

Clary, W.P. 1975. Range management and its ecological basis in the ponderosa pine type of Arizona: the status of our knowledge. Research Paper RM-158. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO.

Colorado Plateau Land Use History of North America (CPLUHNA). 2003. Land use history of North America, Colorado Plateau. Available at http://www.cpluhna.nau.edu/index.htm. Accessed in 2003.

Columbia Basin Fish and Wildlife Authority (CBFWA). 1999. FY 2000 draft annual implementation work plan. Prepared for the Northwest Power Planning Council. Available at http://www.cbfwf.org/products.htm.

Connell, J.H. 1978. Diversity in tropical rain forests and coral reefs. Science 99:1302–1310.

Connelly, J.W., M.A. Schroeder, A.R. Sands, and C.E. Braun. 2000. Guidelines to manage sage grouse populations and their habitats. Wildlife Society Bulletin 28(4):967–985.

Cook, P.S., and J. O’Laughlin. 2000. Toward sustainable forest management. Part II. The role and effects of timber harvesting in Idaho. Policy Analysis Group (PAG) Report Series No. 19. University of Idaho, College of Natural Resources Policy Analysis Group, Moscow, ID. 188 pp.

Cooper, C.F. 1960. Changes in vegetation, structure and growth of southwestern pine forests since white settlement. Ecological Monographs 30:129–164.

Covington, W.W., R.L. Everett, R. Steele, L.L. Irwin, T.A. Daer, and A.N.D. Auclair. 1994. Historical and anticipated changes in forest ecosystems of the Inland West of the United States. In: R.N. Sampson and D.L. Adams, editors. Assessing forest ecosystem health in the Inland West. Food Products Press. p. 13–63.

Covington, W.W., and M.M. Moore. 1994. Southwestern ponderosa forest structure. Journal of Forestry 92:39–47.

Cross, F.B. 1971. Effects of pollution, especially from feed lots, on fishes of the Neosho River basin. Project completion report. Contribution No. 79 A-026-KAN. Kansas Water Resources Institute, Manhattan, KS.

Cullen, S.J., C. Montagne, and H. Ferguson. 1991. Timber harvest trafficking and soil compaction in western Montana. *Soil Science Society of America Journal* 55:1416-1421.

Dale, D., and T. Weaver. 1974. Trampling effects on vegetation of the trail corridors of north Rocky Mountain forests. Journal of Applied Ecology 11:767–772.

DeBano, L.F., and L.J. Schmidt. 1989. Interrelationship between watershed condition and riparian health. In: Practical approaches to riparian resource management. U.S. Department of the Interior, Bureau of Land Management, Billings, MT. p. 45–52.

Donovan, T.M., F.R. Thompson, J. Faaborg, and J.R. Probst. 1995. Reproductive success of migratory birds in habitat sources and sinks. Conservation Biology 9:1380–1395.

Dunford, E.G. 1954. Surface runoff and erosion from pine grasslands of the Colorado Front Range. Journal of Forestry 52:923–927.

Edgerton, P.J., and J.G. Smith. 1987. Influence of ungulates on the development of the shrub understory of an upper slope mixed conifer forest. In: F.D. Provenza, J.T. Flinders, and E.D. McArthur, compilers. Proceedings of a symposium on plant–herbivore interactions. General Technical Report INT-222. U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station, Ogden, UT. p. 162–167.

Ellis, M.A., and L. Horst. 1998. White pine blister rust on currants and gooseberries. Plant pathology fact sheet. HYG-3205-98. Ohio State University Extension, Columbus, OH. Available at http://ohioline.osu.edu/hyg-fact/3000/3205.html.

Ercelawn, A. 1999. End of the road, the adverse ecological impacts of roads and logging: a compilation of independently reviewed research. Natural Resources Defense Council. Available at http://www.nrdc.org/land/forests/roads/intro.asp.

Erickson, R.E., R.L. Linder, and K.W. Harmon. 1979. Stream channelization (PL 83-566): increased wetland losses in the Dakotas. Wildlife Society Bulletin 7(2):71–78.

Faaborg, J., M. Brittingham, T. Donovan, and J. Blake. 1993. Habitat fragmentation in the temperate zone: a perspective for managers. In: D.M. Finch and P.W. Stangel, editors. Status and management of Neotropical migratory birds. General Technical Report RM-229. U.S. Department of Agriculture, Forest Service. p. 331–338.

Facelli, J.M., and S.T.A. Pickett. 1991. Plant litter: its dynamics and effects on plant community structure. Botanical Review 57:1–32.

Fire Sciences Laboratory. 2003. Fire Effects Information System. U.S. Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. Available at http://www.fs.fed.us/database/feis/. Accessed October 2003.

Flather, C.H., L.A. Joyce, and C.A. Bloomgarden. 1994. Species endangerment patterns in the United States. General Technical Report RM 241. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO.

Fleischner, T.L. 1994. Ecological costs of livestock grazing in western North America. Conservation Biology 8(3):629–644.

Forsling, C.L. 1931. A study of the influence of herbaceous plant cover on surface run-off and soil erosion in relation to grazing on the Wasatch Plateau in Utah. Technical Bulletin 220. U.S. Department of Agriculture, Washington, DC.

Franklin, J.F., and C.T. Dyrness. 1973. Natural vegetation of Oregon and Washington. General Technical Report PNW-8. U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station, Portland, OR. 417 pp.

Furniss, M.J., R.D. Roelofs, and C.S. Yee. 1991. Road construction and maintenance. In, *Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats*, American Fisheries Society Special Publication 19:297- 323.

Gaffney, W.S. 1941. The effects of winter elk browsing, South Fork of the Flathead River, Montana. Journal of Wildlife Management 5:427–453.

Gaudet, C.L., and P.A. Keddy. 1988. Predicting competitive ability from plant traits: a comparative approach. Nature 334:242–243.

Gifford, G.F., and R.H. Hawkins. 1978. Hydrologic impact of grazing on infiltration: a critical review. Water Resources Research 14:305–313.

Gillen, R.L., W.C. Krueger, and R.F. Miller. 1984. Cattle distribution on mountain rangeland in northeastern Oregon. Journal of Range Management 37:549–553.

Greenwood, R.J., A.B. Sargeant, D.H. Johnson, L.M. Cowardin, and T.L. Shaffer. 1995. Factors associated with duck nesting success in the prairie pothole region of Canada. Wildlife Monograph 128.

Griggs, G.B., and B.L. Walsh. 1981. The impact, control and mitigation of off-road vehicle activity in Hungry Valley, California. Environmental Geology 3:229–243.

Guthery, F.S., C.A. DeYoung, F.C. Bryant, and D.L. Drawe. 1990. Using short duration grazing to accomplish wildlife habitat objectives. In: K.E. Severson, editor. Can livestock be used as a tool to enhance wildlife habitat? General Technical Report RM-194. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO. p 41–55.

Habeck, J.R. 1990. OId-growth ponderosa pine-western larch forests in western Montana ecology and management. Northwest Environmental Journal 6(2):271–292.

Hall, F.C. 1976. Fire and vegetation in the Blue Mountains—implications for land managers. Proceedings of Tall Timber Fire Ecology Conference 15:155–170.

Harper, J., K. Tate, and M. George. 2003. Grazing effects on riparian areas. Fact Sheet No. 14. University of California, Cooperative Extension. Available at http://danr.ucop.edu/uccelr/h14.htm.

Harrison, R. 1976. Environmental effects of off-road vehicles. Engineering Technology Information System. U.S. Department of Agriculture. San Dimas Equipment Development Center, San Dimas, CA. p. 4–8.

Harvey, A.E., M.F. Jurgensen, M.J. Larsen, and R.T. Graham. 1987. *Decaying Organic Materials and Soil Quality in the Inland Northwest: A Management Opportunity.* USDA Forest Service GTR-INT-225, Ogden, UT. 15pp.

Harvey, A.E., R.T. Meurisse, J.M. Geist, M.F. Jurgensen, G.I. McDonald, R.T. Graham, and N. Stark. 1989. Managing productivity processes in the inland northwest—mixed conifer and pines. In, *Maintaining the Long-Term Productivity of Pacific Northwest Forest Ecosystems,* D.A. Perry, R. Meurisse, B. Thomas, R. Miller, J. Boyle, C.R. Perry, and R.F. Powers, eds. Timber Press, Portland, OR. Pp. 164-184.

Hynson, J.R., P.R. Adamus, J.O. Elmer, T. DeWan, and F.D. Shields. 1985. Environmental features for streamside levee projects. Technical Report E-85-7. U.S. Army Corps of Engineers Waterways Experiment Station, Vicksburg, MS.

Integrated Natural Fuels Management Strategy (INFMS). 2003. The role of fire. U.S. Department of Agriculture, Forest Service, Willamette National Forest, and U.S. Department of the Interior, Bureau of Land Management, Eugene District. Available at http://www.edo.or.blm.gov/infms/HTML/FIRE/BIO.HTM.

Interior Columbia Basin Ecosystem Management Project (ICBEMP). 1997. An assessment of ecosystem components in the

Interior Columbia Basin and portions of the Klamath and Great basins. Volumes 1–4. In: T.M. Quigley and S.J. Arbelbide, editors. Scientific reports and associated spatially explicit datasets. U.S. Department of Agriculture, Forest Service, and U.S. Department of the Interior, Bureau of Land Management.

Irwin, L.L., J.M. Peek, J.G. Cook, R.A. Riggs, and J.M. Skovlin. 1994. Effects of long-term grazing by big game and livestock in the Blue Mountains forest ecosystems. General Technical Report PNW-GTR-325. U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station, Portland, OR. 49 pp.

Johnson, C.G., Jr., R.R. Clausnitzer, P.J. Mehringer, and C.D. Oliver. 1994. Biotic and abiotic processes of Eastside ecosystems: the effects of management on plant and community ecology and on stand and landscape vegetation dynamics. General Technical Report PNW-GTR-322. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, OR. 66 pp.

Johnson, D.H. 2001. Habitat fragmentation effects on birds in grasslands and wetlands: a critique of our knowledge. Great Plains Research 11(2):211–213.

Johnson, D.H., and M. Winter. 1999. Reserve design for grasslands: considerations for bird populations. Proceedings of the George Wright Society Biennial Conference 10:391–96.

Johnson, R.G., and S.A. Temple. 1990. Nest predation and brood parasitism of tallgrass prairie birds. Journal of Wildlife Management 54:106–111.

Johnson, W.M. 1956. The effect of grazing intensity on plant composition, vigor, and growth of pine–bunchgrass ranges in central Colorado. Ecology 37:790–798.

Jurgensen, M.F., A.E. Harvey, R.T. Graham, M.J. Larsen, J.R. Tonn, D.S. Page-Dumroese. 1990. Soil organic matter, timber harvesting, and forest productivity in the Inland Northwest. In: S.P. Gessel, D.S. Lacate, G.F. Weetman, and R.F. Powers, editors. Sustained productivity of forest soils; proceedings of the 7th North American Forest Soils Conference. Faculty of Forestry Publication. University of British Columbia, Vancouver, BC, Canada. p. 392–415.

Kendall, K.C. 1995. Whitebark pine: ecosystem in peril. In: E.T. LaRoe, G.S. Farris, C.E. Puckett, P.D. Doran, and M.J. Mac, editors. Our living resources: a report to the nation on the distribution, abundance, and health of U.S. plants, animals, and ecosystems. U.S. Department of the Interior, National Biological Service, Washington, DC. p. 228–230.

Knick, S.T., and C. Van Riper, III. 2002. Loss of sagebrush ecosystems and declining bird populations in the Intermountain West: priority research issues and information needs. USGS FS-122-02. U.S. Department of the Interior, U.S. Geological Survey. 2 pp.

Knopf, F.L. 1994. Avian assemblages on altered grasslands. Studies in Avian Biology 15:247–257.

Korfhage, R.C., J.R. Nelson, and J.M. Skovlin. 1980. Summer diets of Rocky Mountain elk in northeastern Oregon. Journal of Wildlife Management 44:747–750.

Laudenslayer, W.F., H.H. Darr, and S. Smith. 1989. Historical effects of forest management practices in Eastside pine communities in northeastern California. In: A. Tecle, W.W. Covington, and R.H. Hamre, technical coordinators. General Technical Report RM-

85. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO. p. 26–34.

Lee, D., J. Sedell, B. Rieman, R. Thurow, and J. Williams. 1997. Broad scale assessment of aquatic species and habitats. Volume 3, chapter 4. In: An assessment of ecosystem components in the Interior Columbia Basin and portions of the Klamath and Great basins. General Technical Report PNW-GTR-405. U.S. Department of Agriculture, Forest Service.

Leopold, A. 1924. Grass, brush, timber and fire in southern Arizona. Journal of Forestry 22:1–10.

Lonsdale, W.M. 1999. Global patterns of plant invasions and the concept of invasibility. Ecology 80:1522–1536.

Lull, H.W. 1959. Soil compaction on forest and range lands. Miscellaneous Publication 769. U.S. Department of Agriculture, Washington, DC.

Mack, R.N., and J.N. Thompson. 1982. Evolution in steppe with few large, hooved mammals. American Naturalist 119:757–772.

Madany, M.H., and N.E. West. 1983. Livestock grazing–fire regime interactions within montane forests of Zion National Park, Utah. Ecology 64:661–667.

Malecki, R.A., B. Blossey, S.D. Hight, D. Schroeder, D.T. Kok, and J.R. Coulson. 1993. Biological control of purple loosestrife. BioScience 43(10):680–686.

Marlow, C.B., and T.M. Pogacnik. 1985. Time of grazing and cattle-induced damage to streambanks. In: R.R. Johnson, C.D. Zeibell, D.R. Patton, P.F. Folliott, and R.H. Hamre, technical coordinators. Riparian ecosystems and their management: reconciling conflicting uses. GTR RM-120. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO. p. 279–284.

Martin, R.E., and D.B. Sapsis. 1992. Fires as agents of biodiversity: pyrodiversity promotes biodiversity. In: R.R. Harris and D.C. Erman, technical coordinators. Proceedings of the symposium on biodiversity of northwestern California. Report 29. University of California, Wildland Resources Center, Berkeley, CA. p. 150–157.

Maser, C., R.F. Tarrant, J.M. Trappe, and J.F. Franklin, tech. eds. 1988. *From the Forest to the Sea: A Story of Fallen Trees.* USDA Forest Servuce PNW-GTR-229, Portland, OR.

Megahan, W.F. 1991. Erosion and site productivity in western-montane forest ecosystems. In, *Proceedings—Management and Productivity of Western-Montane Forest Soils,* USDA Forest Service GTR-INT-280, Ogden, UT. Pp. 146- 150.

Merigliano, M.F. 1996. Ecology and management of the South Fork Snake River cottonwood forest. BLM Technical Bulletin BLM-ID-PT96/016+1150. U.S. Department of the Interior, Bureau of Land Management. 100 pp.

Natural Resource Sciences Extention (NRSCE). 2003. White pine blister rust (*Cronartium ribicola*). Forest health notes: a series for the non-industrial private forest landowner. Washington State University, Natural Resource Sciences, Cooperative Extension, Pullman, WA. Available at http://ext.nrs.wsu.edu/forestryext/foresthealth/notes/whitepinerust.htm.

Natural Resources Conservation Service (NRCS). 2001. Conversion of natural resource lands to urban lands, farmsteads, and rural transportation lands, 1997 national resources inventory—Idaho results. U.S. Department of Agriculture, NRCS. Available at http://www.id.nrcs.usda.gov/technical/nri/conversion.html.

Neumann, P.W., and H.G. Merriam. 1972. Ecological effects of snowmobiles. Canadian Field Naturalist 86:207–212.

Niering, W.A., R.H. Whittaker, and C.H. Lowe. 1963. The saguaro: a population in relation to environment. Science 142:15–23.

Nolte, D.L., and M. Dykzeul. 2000. Wildlife impacts on forest resources. In: Human conflicts with wildlife: economic considerations. Proceedings of the third National Wildlife Research Center; August 1–3, 2000; Fort Collins, CO.

Oliver, C.D., L.L. Irwin, and W.H. Knapp. 1994. Eastside forest management practices: historical overview, extent of their applications, and their effects on sustainability of ecosystems. General Technical Report PNW-324. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, OR.

Olliff, T., K. Legg, and B. Kaeding, editors. 1999. Effects of winter recreation on wildlife of the Greater Yellowstone Area: a literature review and assessment. Report to the Greater Yellowstone Coordinating Committee, Yellowstone National Park, WY. 315 pp.

Orodho, A.B., M.J. Trlica, and C.D. Bonham. 1990. Long-term heavy grazing effects on soil and vegetation in the Four Corners region. Southwestern Naturalist 35(1):9–15.

Paton, P.W. 1994. The effect of edge on avian nest success: how strong is the evidence? Conservation Biology 8:17–26.

Patton, David R. 1992. Bridging the gap, achieving effective resource integration. Symposium Proceedings. 10th National Conference, Native Amer. and Wildlife Society. Intertribal Timber Council. Portland, OR.

Peet, R.K. 1988. Forests of the Rocky Mountains. In: M.G. Barbour and W.D. Billings, editors. North American terrestrial vegetation. Cambridge University Press, New York, NY. p. 63–101.

Perryman, B. 2003. What are the consequences of doing nothing? A white paper—eastern Nevada landscape coalition position paper. Available at http://www.envlc.org/white_paper.htm.

Pimentel, D.L., R. Zuniga Lach, and D. Morrison. 1999. Environmental and economic costs associated with nonindigenous species in the United States. Cornell University, College of Agriculture and Life Sciences, Ithaca, NY. Available at http://www.news.cornell.edu/releases/Jan99/species_costs.html.

Platts, W.S. 1990. Managing fisheries and wildlife on rangelands grazed by livestock. Nevada Department of Wildlife, Reno, NV. 462 pp.

Powers, R.F. 1991. Are we maintaining the productivity of forest lands? Establishing guidelines through a network of long-term studies. In: Proceedings—management and productivity of western-montane forest soils. GTR-INT-280. U.S. Department of Agriculture, Forest Service, Ogden, UT. p. 70–81.

Pyke, D.A. 2002. Born of fire-restoring sagebrush steppe. USGS FS-126-02. U.S. Department of the Interior, U.S. Geological Survey. 2 pp.

Pyne, S.J. 1982. Fire in America: a cultural history of wildland and rural fire. Princeton University Press, Princeton, NJ. 654 pp.

Reice, S.R. 1994. Nonequilibrium determinants of biological community structure. American Scientist 82:424–435.

Reisner, M. 1993. Cadillac desert: the American West and its disappearing water. 2nd edition. Penguin Books, New York, NY.

Ringholz, R.C. 1996. Paradise paved: the challenge of growth in the new West. University of Utah Press, Salt Lake City, UT.

Roath, L., and W.C. Krueger. 1982. Cattle grazing and behavior on a forested range. Journal of Range Management 35:332–338.

Roberts, B.C., and R.G. White. 1992. Effects of angler wading on survival of trout eggs and pre-emergent fry. North American Journal of Fisheries Management 12:450–459.

Robinson, S.K., F.R. Thompson, T.M. Donovan, D.R. Whitehead, and J. Faaborg. 1995. Regional forest fragmentation and the nesting success of migratory birds. Science 267:1987–1990.

Romme, W.H., and D.G. Despain. 1989. Historical perspective on the Yellowstone fires of 1988. BioScience 39:695–699.

Saunders, D.A., R.J. Hobbs, and C.R. Margules. 1991. Biological consequences of ecosystem fragmentation: a review. Conservation Biology 5:18–32.

Savage, M., and T.W. Swetnam. 1990. Early 19th century fire decline following sheep pasturing in a Navajo ponderosa pine forest. Ecology 71:2374–2378.

Schmiegelow, F.K.A., C.S. Machtans, and S.J. Hannon. 1997. Are boreal birds resilient to forest fragmentation? An experimental study of short-term community responses. Ecology 78(6):1914–32.

Schoof, R. 1980. Environmental impacts of channel modification. Water Resources Bulletin 16:697–701. In: R.L. Mattingly and E.E. Herricks, editors. 1990. Channelization of streams and rivers in Illinois: procedural review and selected case studies. INENR/re-WR-91/01. Illinois Department of Energy and Natural Resources, Springfield, IL.

Schultz, T.T., and W.C. Leininger. 1990. Differences in riparian vegetation structure between grazed areas and exclosures. Journal of Range Management 43:295–299.

Scott, J.M., C.R. Peterson, J.W. Karl, E. Strand, L.K. Svancara, and N.M. Wright. 2002. A gap analysis of Idaho. Final report. Idaho Cooperative Fish and Wildlife Research Unit, Moscow, ID.

Sherwood, C.R., D.A. Jay, R. Harvey, P. Hamilton, and C. Simenstad. 1990. Historical changes in the Columbia River estuary. Progress in Oceanography 25:299–352.

Sheley, R., M. Manoukian, and G. Marks. 2002. Preventing noxious weeds invasion. MT199517 AG 8/2002. MontGuide fact sheet, Montana State University Extension Service. 4 pp.

Shreve, F. 1931. Physical conditions in sun and shade. Ecology 12:96–104.

Skovlin, J.M., R.W. Harris. G.S. Strickler, and G.A. Garrison. 1976. Effects of cattle grazing methods on ponderosa pine–bunchgrass range in Pacific Northwest. Technical Bulletin 1531. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, OR.

Smith, D.W. 1967. Effects of cattle grazing on a ponderosa pine–bunchgrass range in Colorado. Technical Bulletin 1371.

U.S. Department of Agriculture, Washington, DC.

Stebbins. G.L. 1981. Co-evolution of grasses and herbivores. Annals of the Missouri Botanical Garden 68:75–86.

Stuart, J.D. 2003. Effects of fire suppression on ecosystems and diversity. In: Status and trends of the nation's biological resources. U.S. Geological Survey. Available at http://biology.usgs.gov/s+t/SNT/index.htm.

Talbot, R.K., and J.D. Wilde. 1989. Giving form to the formative: shifting settlement patterns in the eastern Great Basin and northern Colorado Plateau. Utah Archaeology 2:3–18.

Taylor, F.R., L. Gillman, J.W. Pedretti, and J.E. Deacon. 1991. Impact of cattle on two endemic fish populations in the Pahranagat Valley, Nevada. Proceedings of the Desert Fishes Council 21:81.

The Nature Conservancy (TNC). 2003. The Nature Conservancy's invasive species initiative. Informational Pamphlet 02113 01/2003. 4 pp.

Thompson, D.G., R.L. Stuckey, and E.B. Thompson. 1987. Spread, impact, and control of purple loosestrife (*Lythrum salicaria*) in North American wetland. Fish and Wildlife Research 2. U.S. Fish and Wildlife Service, Washington, DC.

Thurow, T.L. 1991. Hydrology and erosion. In: R.K. Heitschmidt and J.W. Stuth, editors. Grazing management—an ecological perspective. Timber Press, Portland, OR. p. 141–159.

Tiger Roads. 2000. Shapefile: geographic coordinates NAD83 for the 48 contiguous states, NAD27 for Alaska, and Old Hawaiian Datum for Hawaii. Provided by U.S. Bureau of the Census.

Tonnesen, A.S., and J.J. Ebersole. 1997. Human trampling effects on regeneration and age structures of Pinus edulis and Juniperus monosperma. Great Basin Naturalist 57:50–56.

Trent, T. 2000. Subdivision development in winter range. Operations Meeting Report Compilation. Idaho Department of Fish and Game. Available at http://www2.state.id.us/lands/Forest%20Legacy/Assessment%20of%20Need%20Breakout%20Files/Appendix%203/winter%20range.pdf. 8 pp.

Trunkle, T., and P. Fay. 1991. Transportation of spotted knapweed seeds by vehicles. Proceedings, Montana Weed Control Association annual conference; January 14–16, 1991; Butte, MT. 33 pp.

Turner, I.M. 1996. Species loss in fragments of tropical rain forest: a review of the evidence. Journal of Applied Ecology 33(2):200–209.

U.S. Environmental Protection Agency (USEPA). 1993. Guidance specifying management measures for sources of nonpoint pollution in coastal waters. EPA 840-B-92-002. USEPA. Available at http://www.epa.gov/owow/nps/MMGI/.

U.S. Environmental Protection Agency and Idaho Department of Environmental Quality (USEPA and IDEQ). 1998. 1998 impaired water. In: Watershed Information Network: surf your watershed. Available at http://www.epa.gov/surf3/. U.S Environmental Protection Agency, Washington, DC.

U.S. Environmental Protection Agency (USEPA). 2001. Protecting and restoring America's watersheds: status, trends and initiatives in watershed management. EPA-

840-R-00-001. USEPA, Office of Water. 56 pp.

U.S. Fish and Wildlife Service (USFWS). 1999. Biological opinion—on-going and long-term grazing on the Coronado National Forest. AESO/SE 2-21-98-F-399. U.S. Fish and Wildlife Service, Arizona Ecological Services Field Office.

U.S. Forest Service (USFS). 1994. Protecting and restoring aquatic ecosystems: new directions for watershed and fisheries research in the USDA Forest Service. U.S. Department of Agriculture, Forest Service, Forest Environment Research Staff, Washington, DC. 13 pp.

U.S. Forest Service (USFS). 1996. U.S. Department of Agriculture, Forest Service, Status of the Interior Columbia Basin: summary of scientific findings. General Technical Report PNW-GTR-385. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, and U.S. Department of the Interior, Bureau of Land Management, Portland, OR. 144 pp.

Ward, J.P., and W.M. Block. 1995. Mexican spotted owl prey ecology. In: Mexican spotted owl recovery plan. U.S. Fish and Wildlife Service, Albuquerque, NM.

Weaver, H. 1950. Shoals and reefs in ponderosa pine silviculture. Journal of Forestry 48:21–22.

Weclaw, P. 1998. Habitat fragmentation: natural vs. human-induced disturbances. University of Alberta, Department of Renewable Resources, AB, Canada. Available at http://www.rr.ualberta.ca/courses/renr575/fragmentationmain.htm.

Wedin, D.A., and D. Tilman. 1996. Influence of nitrogen loading and species composition on the carbon balance of grasslands. Science 274:1720–1723.

Whitcomb R.F., C.S. Robbins, J.F. Lynch, B.L. Whitcomb, M.K. Klimkiewicz, and D. Bystrak. 1981. Effects of forest fragmentation on avifauna of the eastern deciduous forest. In: R.L. Burgess and B.M. Sharpe, editors. Forest island dynamics in man-dominated landscapes. Springer-Verlag, New York, NY. p. 125–206.

Whittaker, P.L. 1978. Comparison of surface impact by hiking and horseback riding in the Great Smoky Mountains National Park. Management Report 24. U.S. Department of the Interior, National Park Service, Southeast Region. 32 pp.

Whittaker, R.H. 1977. Evolution of species diversity in land communities. In: M.K. Hecht, W.C. Steele, and B. Wallace, editors. Evolutionary biology. Volume 10. Plenum Press, New York, NY. p. 1–67.

Winter, M., D.H. Johnson, and J. Faaborg. 2000. Evidence for edge effects on multiple levels in tallgrass prairie. Condor 102(2):256–266.

Worster, D. 1985. Rivers of empire: water, aridity and growth of the American West. 2nd edition. Pantheon Books, New York, NY. 416 pp.