APPENDIX 3-1—OVERVIEW OF THE MAJOR CAUSES LIMITING THE HABITATS AND FISH AND WILDLIFE IN THE SALMON SUBBASIN, IDAHO

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 12% of area in the Salmon subbasin is highly impacted by altered hydrologic regimes, with most impacts occurring in the Little Salmon, Lower Salmon, Middle Salmon–Chamberlain, and Lemhi watersheds (Table 1). According to ICBEMP information, the most severely impacted watersheds in terms of altered hydrology are the Lower Salmon and the Little Salmon (Figure 1). The Pahsimeroi, Lemhi, and South Fork Salmon watersheds have localized subwatersheds with moderate to high hydrological impairment, but the majority of those watersheds are classified as low impairment (Figure 1). Areas occupied by riparian/herbaceous wetlands are highly impacted by altered hydrologic regimes (Table 2).

Table 1. Comparison of the relative percentages of area impacted by altered hydrologic regimes the Salmon subbasin, Idaho. Source: ICBEMP (1997).

Relative Category	Major Hydrologic Unit (Watershed)[a,b]										% Entire Subbasin
	UPS	PAH	LEM	MFU	MFL	MSC	MSP	SFS	LOS	LSA	
Very high	0	4	2	0	0	0	1	0	26	5	3
High	4	5	13	0	0	11	7	2	56	67	12
Medium	6	0	6	0	0	2	5	0	2	5	3
Low	19	45	43	7	6	13	53	34	8	7	24
Very low	72	46	36	93	94	74	33	64	8	17	58

[a] UPS = Upper Salmon, PAH = Pahsimeroi, LEM = Lemhi, MFU = Upper Middle Fork Salmon, MFL = Lower Middle Fork Salmon, MSC = Middle Salmon–Chamberlain, MSP = Middle Salmon–Panther, SFS = South Fork Salmon, LOS = Lower Salmon, and LSA = Little Salmon
[b] Percentages may not sum to 100 due to rounding.

Table 2. Relative percentages of impacts to focal habitats by altered hydrologic regimes in the Salmon subbasin, Idaho. Source: GAP II (Scott *et al*. 2002).

Focal Habitat	Ratings[a]				
	Very Low	Low	Medium	High	Very High
Riparian/herbaceous wetlands	46	31	2	16	5
Shrub-steppe	52	34	4	9	2
Forest	62	22	2	11	2
Native grasslands	47	26	2	17	7
Aspen	70	24	<1	6	?

Focal Habitat	Ratings[a]				
	Very Low	Low	Medium	High	Very High
Juniper/mountain mahogany	60	19	4	14	3
Whitebark pine	86	11	2	<1	<1
Other	48	22	2	21	7
Total area (km^2)	21,138	8,663	994	4,338	1,084

[a] Percentages may not sum to 100 due to rounding.

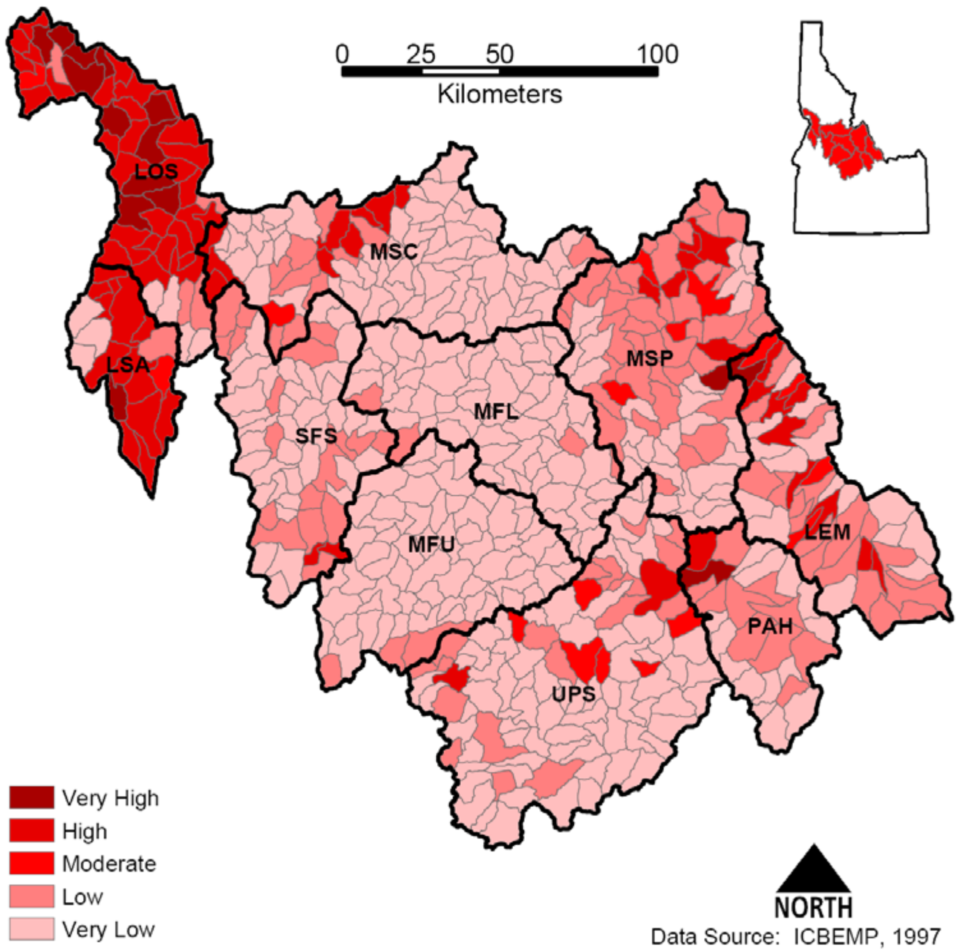

Figure 1. Relative impacts of altered hydrologic regimes in the Salmon subbasin, Idaho.

1.1 Habitat Loss and Modification

Human activities such as residential and commercial development, recreation, and resource extraction have changed, fragmented, and destroyed riparian and aquatic 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 can seriously impact fish populations. Road culverts that block fish passage in the Salmon subbasin are shown in Figure 2. We estimated 552 road culverts in the Salmon subbasin, and of culverts surveyed for fish passage only 17 allow passage for juvenile fish and 44 for adult fish (Table 3). Anadromous fish, species that migrate from freshwater to saltwater and back to freshwater, cannot breed successfully if culverts block their migration routes.

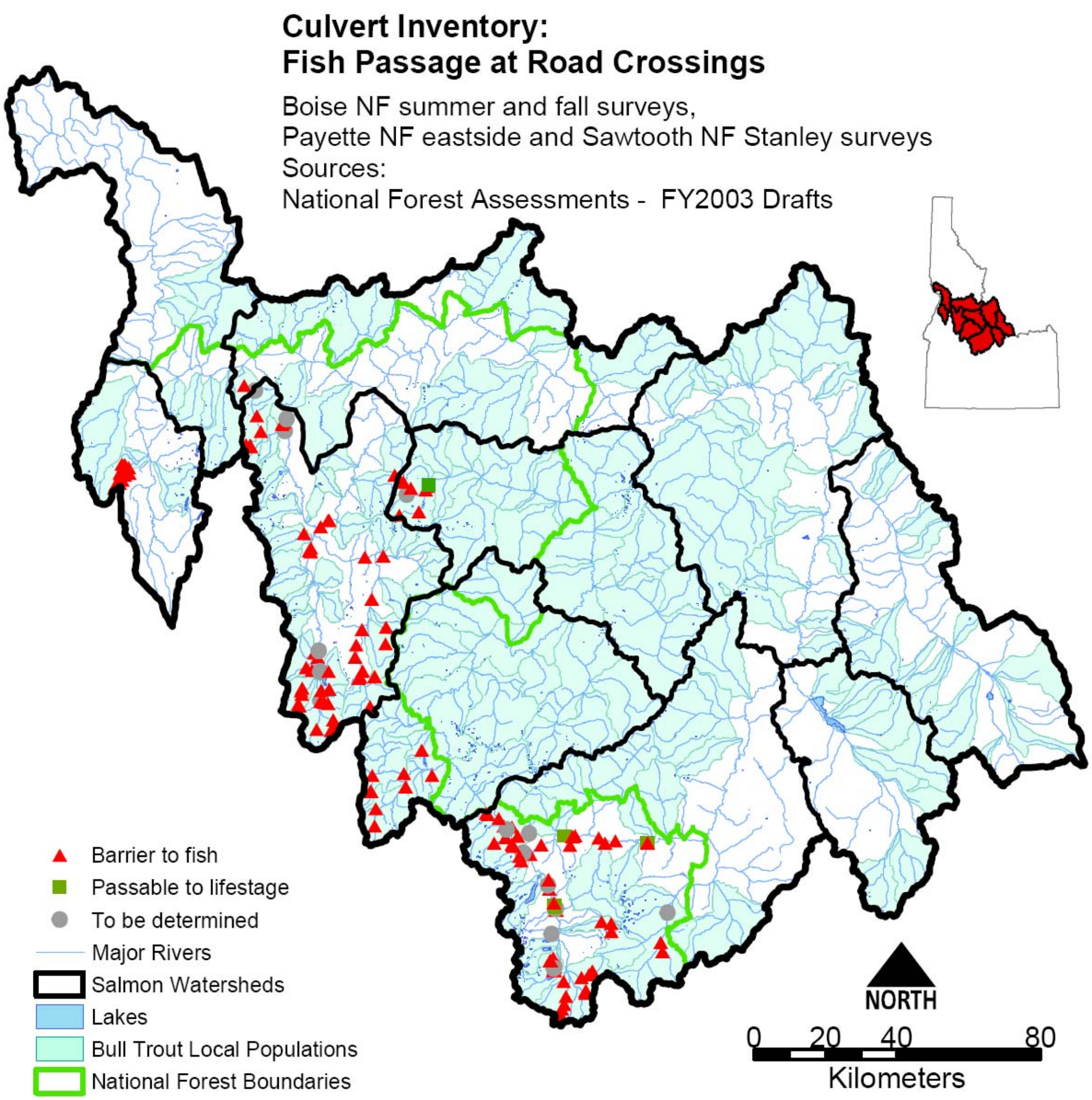

Figure 2. Inventory for road culverts surveyed in the Salmon subbasin, Idaho. Note that many road culverts remain to be surveyed in the subbasin.

Table 3. Information on fish passage at road crossings in the Salmon subbasin, Idaho (Sources: National Forest Assessments, 2003).

Life Stage	Culvert Fish Passage Information	Watershed										Totals
		UPS	PAH	MSP	LEM	MFU	MFL	MSC	SFS	LOS	LSA	
Juvenile	No Fish Passage	82	1	51	13	9	12	1	50	0	23	242
	Passage not determined	14	0	23	0	0	1	0	8	0	0	46
	Allows Fish Passage	5	0	10	1	0	1	0	0	0	0	17
	Unknown Status	115	0	11	8	8	3	0	92	0	10	247
	Totals	216	1	95	22	17	17	1	150	0	33	552
Adult	No Fish Passage	42	1	44	12	8	11	1	48	0	23	190
	Passage not determined	49	0	13	1	1	1	0	10	0	0	75
	Allows Fish Passage	10	0	30	1	0	3	0	0	0	0	44
	Unknown Status	115	0	8	8	8	2	0	92	0	10	243
	Totals	216	1	95	22	17	17	1	150	0	33	2,208

When communities straighten and channelize urban streams and line them with concrete, they modify the vegetative and physical structure of the riverine habitat, increase river velocities during rainstorms, and decrease river volumes during dry periods. Straightened and channelized streams also carry more sediments and chemical pollutants to their receiving waters (USEPA 2001).

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 that use 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 generation, 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.

Over 10,000 points of water diversion are present in the Salmon subbasin (Figure 3). The majority of these diversions occur in the Lemhi (2,950), Middle Salmon–Panther (2,250), Little Salmon (1,500), and Pahsimeroi (850) watersheds. There are 2,585 water diversions in the Upper Salmon watershed. The majority of the diversions in the mainstem accessible to salmon and steelhead are screened according to criteria established by the National Oceanic and Atmospheric Administration's National Marine Fisheries Service (NOAA Fisheries). Most of the pump intakes in the Lower Salmon watershed are also screened. As connectivity is restored to blocked tributaries, these diversions will also require fish screens.

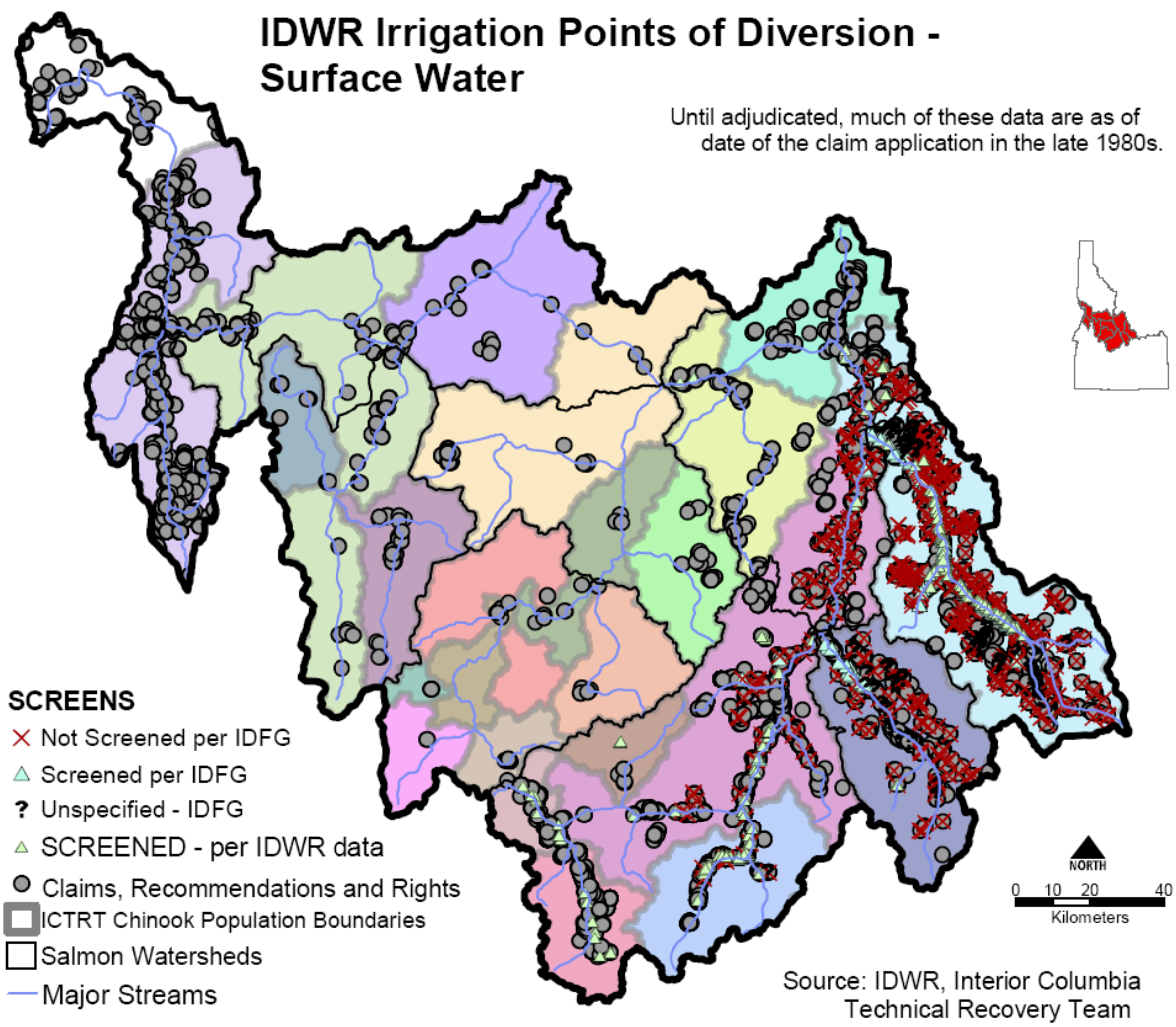

Figure 3. **Locations of water diversions in the Salmon subbasin, Idaho.**

Channelization (or engineering of river and stream channels for flood control, navigation, improvement of drainage, and reduction of channel migration potential) includes activities such as straightening, widening, deepening, or relocating 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, reduced 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 Salmon subbasin may be examined in terms of the number of alteration permits issued by the U.S. Army Corps of Engineers (Figure 4) and the Idaho Department of Water. The most severely altered watersheds in the subbasin are the Lower Salmon, Little Salmon, Upper Salmon, Lemhi, and Middle Salmon–Panther watersheds (Figure 4).

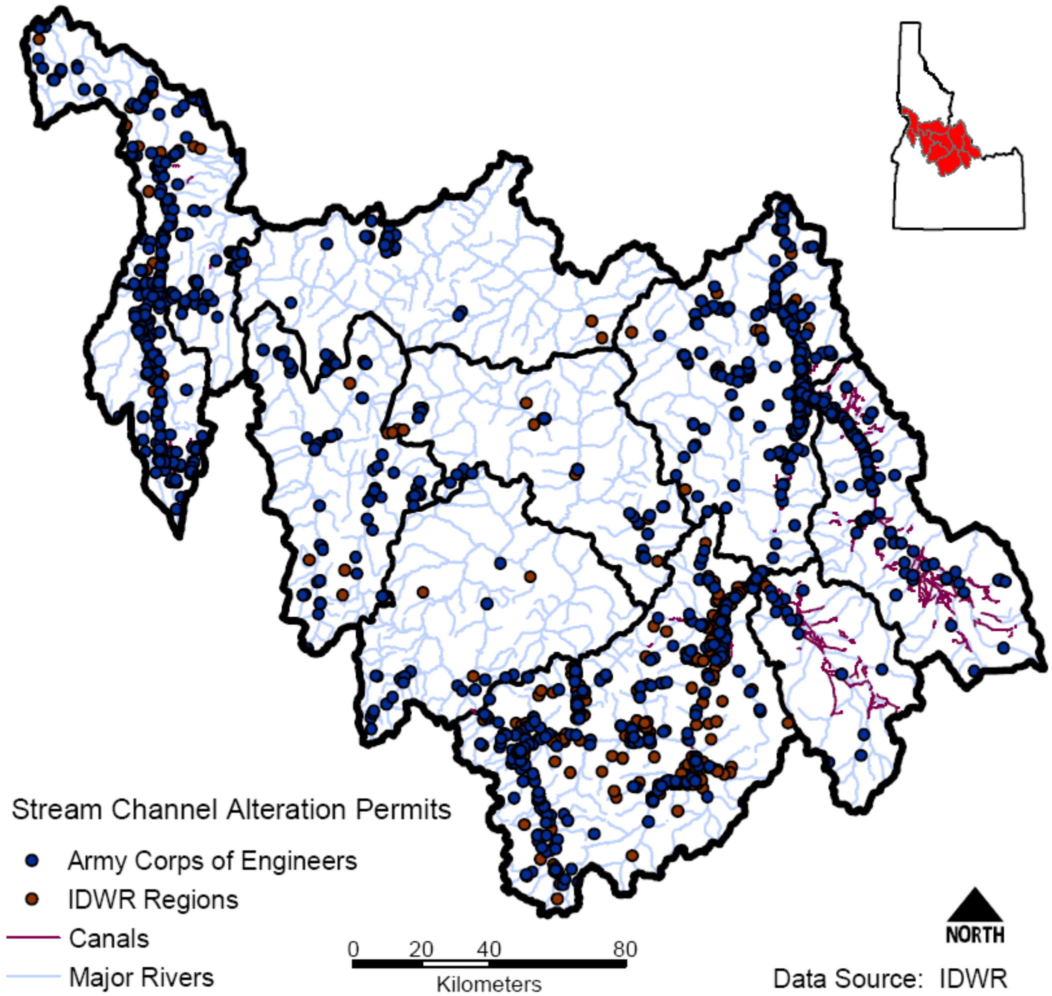

Figure 4. Locations of canals and alteration activities permitted by the Idaho Department of Water Resources and the U.S. Army Corps of Engineers in the Salmon subbasin, Idaho.

Instream hydraulic changes as affected by stream alterations can decrease or interfere with surface water contact in streambank areas during floods or other high-water events. Channelization and channel-modification activities may also result in reduced filtering of pollutants by streamside vegetation and soils. Areas of streambank 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 are 2.6 times higher in channelized area than in undisturbed areas and 5.3 times higher following stream alteration construction (Erickson *et al*. 1979). Schoof (1980) reported impacts of channelization, including drainage of wetlands, reduction of oxbows and stream meander, clearing of floodplain hardwood, lowering of ground-water levels, and increased erosion (USEPA 1993).

Channelization and channel-modification activities can lead to loss of instream and riparian habitat and such ecosystem benefits wildlife migration pathways and suitable conditions for reproduction and growth. Problematic flow modifications have resulted in reversal of flow regimes of some California rivers or streams, and led to the disorientation of anadromous fish that rely on flow to direct them to spawning areas (USEPA 1993). Eroded sediment may cover benthic communities or alter instream habitat (Sherwood *et al*. 1990).

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 10% of the streams, or 85 waterways, in the Salmon subbasin are sediment impaired (Table 4 and

Table 5).

Table 4. Total lengths (km) of streams impacted by sediments in the Salmon subbasin, Idaho. Sources: ICBEMP (1997), USEPA (1998).

Watershed	Total Stream Length (km)	Stream Length (km) Impacted by Sediments	% Streams Affected by Sediments
Upper Salmon	2,439	293	12.0
Pahsimeroi	738	142	19.2
Middle Salmon–Panther	1,939	29	1.5
Lemhi	1,297	139	10.7
Upper Middle Fork Salmon	1,885	115	6.1
Lower Middle Fork Salmon	1,536	12	0.8
Middle Salmon–Chamberlain	2,114	108	5.1
South Fork Salmon	1,617	347	21.4
Lower Salmon	1,446	296	7.0
Little Salmon	684	48	20.5
Totals	15,695	1,529	9.7%

Table 5. List of sediment impaired streams by watershed in the Salmon subbasin, Idaho. Sources: ICBEMP (1997), USEPA (1998).

Watershed	Sediment Impaired Stream	
Upper Salmon	Challis Creek Garden Creek Salmon River Squaw Creek Stanley Lake Creek	Thompson Creek Valley Creek Warm Springs Creek Yankee Fork
Pahsimeroi	Big Creek East Fork Pahsimeroi River Morse Creek	Pahsimeroi River Patterson Creek West Fork Pahsimeroi River
Middle Salmon–Panther	Big Deer Creek Blackbird Creek	Dump Creek
Lemhi	Big Eightmile Creek Big Timber Creek Bohannon Creek Eighteenmile Creek Geertson Creek Hawley Creek	Kenney Creek Kirtley Creek Little Eightmile Creek McDevitt Creek Sandy Creek Wimpey Creek
Upper Middle Fork Salmon	Bear Valley Creek Cache Creek Cook Creek Cub Creek Dagger Creek	Elkhorn Creek Fir Creek Porter Creek Sheep Trail Creek
Lower Middle Fork Salmon	Monumental Creek	
Middle Salmon–Chamberlain	Big Creek Big Mallard Creek Crooked Creek Jersey Creek	Little Mallard Creek Rabbit Creek Rhett Creek Sabe Creek
South Fork Salmon	Bear Creek Curtis Creek Dollar Creek Johnson Creek Meadow Creek Rice Creek South Fork Salmon River	East Fork South Fork Salmon River Secesh River Sugar Creek Trail Creek Trout Creek Tyndall Creek
Lower Salmon	Allison Creek China Creek Cottonwood Creek Cow Creek Deep Creek Deer Creek	Little Slate Creek Little White Bird Creek Maloney Creek Pinnacle Creek Race Creek Rock Creek

Watershed	Sediment Impaired Stream	
	Grave Creek Jungle Creek Kessler Creek Little Boulder Creek	Skookumchuck Creek Slate Creek Tumbull Creek Van Buren Creek
Little Salmon	Big Creek Elk Creek	Indian Creek Shingle Creek

2 Land-Use Conversion/Development/ Fragmentation

The Columbia River basin ecosystem escaped significant human land-use impacts until the nineteenth century when settlers and their livestock began to move into the region in 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 people claiming their share of the region's water, energy, and recreational resources, conflicts between mutually exclusive uses such as ecotourism, recreational off-road vehicles, and ranching are becoming widespread and chronic (Talbot and Wilde 1989, Reisner 1993, Ringholz 1996).

The population of the Columbia River basin has increased six-fold since the beginning of the twentieth century and has more than doubled since the mid-1960s. This growth rate is 2.5 times greater than the nation's rate of 39% for that same period. Population growth in some areas of the Columbia River basin is 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). Ada County in southwestern Idaho saw its population increase from 205,000 people in 1990 to 300,000 people in 2000, an increase of 46% in just ten years (CensusScope 2003).

Recreation, tourism, and quality of life play significant roles 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 Salmon subbasin, the majority of the population resides in seven of the ten watersheds: Lower Salmon, Little Salmon, Lemhi, Pahsimeroi, Middle Salmon–Panther, South Fork Salmon, and Upper Salmon watersheds (Table 6 and Figure 5). Few people reside in the Upper Middle Fork Salmon, Lower Middle Fork Salmon, and Middle Salmon–Chamberlain watersheds, which are large wilderness and protected areas.

Table 6. Percentage of population density classifications by watershed in the Salmon subbasin, Idaho. Source: ICBEMP (1997).

Population Density Classification (population per square mile)	Major Hydrologic Unit (Watershed)									
	UPS	PAH	LEM	MFU	MFL	MSC	MSP	SFS	LOS	LSA
Very Low ($x < 1$)	0	0	0	0	0	0	2	5	0	0
Low ($1 < x > 10$)	22	6	12	73	96	79	45	52	3	7
Medium ($10 < x < 60$)	66	87	80	27	4	21	42	40	89	93
High ($60 < x < 100$)	10	7	5	0	0	0	8	3	4	0
Very High ($100 < x > 300$)	2	0	3	0	0	0	3	0	3	0

[a] Percentages may not sum to 100 due to rounding.

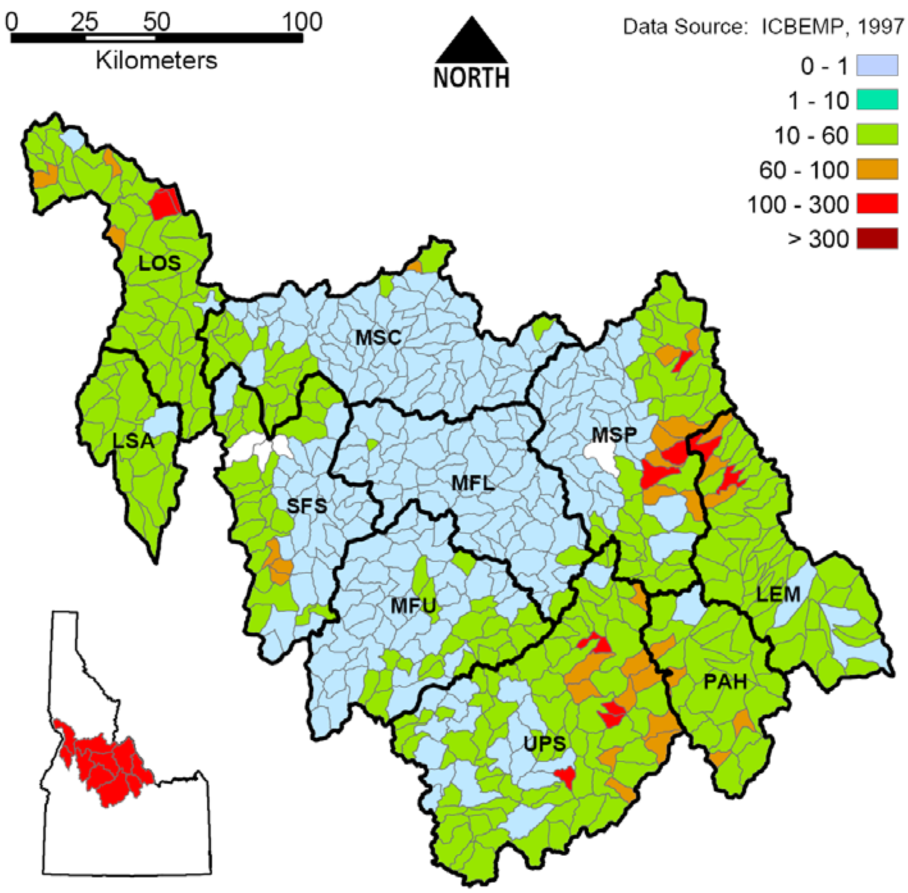

Figure 5. Population densities (people/square mile) in the Salmon subbasin, Idaho.

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. Farm- and ranchland, forests, and other open space are transformed into subdivisions, ranchettes, shopping areas with expansive parking lots, and roads. This conversion carves away at wildlife habitat, and wetland and riparian areas are frequently diminished. 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. It further estimates that 2,234,658 hectares (5,521,960 acres), or about one-third, of the conversion occurred in rural areas (NRCS 2001). Although much of the Salmon subbasin is exempt from urban sprawl, watersheds impacted by development include the Little Salmon, Lower Salmon, Upper Salmon, Lemhi and Middle Salmon–Panther (Figure 6). Based on data collected in 1994, the greatest impacts are in the Lower Salmon and Little Salmon, with 54% and 47% of the watershed area, respectively, impacted by sprawl. The southernmost tip of the Upper Middle Fork Salmon watershed has very high impacts from sprawl. This area is affected by urbanization in the adjacent to the watershed.

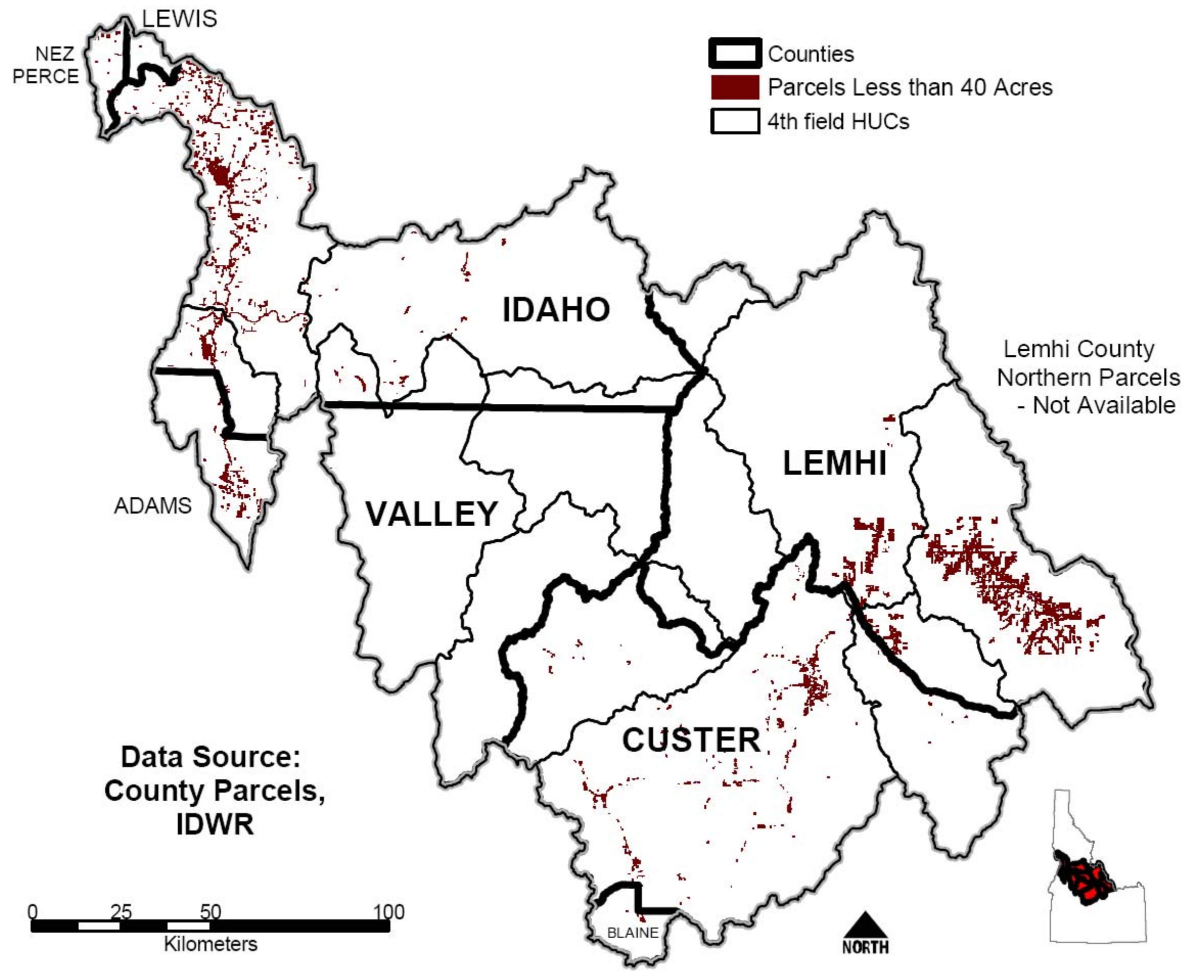

Figure 6. Areas of development in the Salmon subbasin, Idaho. This figure is based on county lot size data collected through 2003.

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 (16,559 acres) per year from 1992 to 1997, resulting in an increase of 47.2%. The rate of increase was highest on rangeland, followed by pastureland, cropland, and then forestland (Table 7).

Table 7. Estimated conversion rates of natural resource lands to urban lands in Idaho, 1982 to 1992 vs. 1992 to 1997, in hectares per year. Source: 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[a]	6,701	+47.2

[a] The discrepancy in addition is due to rounding.

Sprawl fragments habitat when new developments divide undisturbed habitats. The resulting fragmentation is particularly harmful to wide-ranging species that rely on large territories for food and cover. Without adequate continuous habitat, a population of large, wide-ranging animals will eventually disappear from an area, with harmful ripple effects 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. 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 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 habitat for that species is reduced; fragmentation effects imply that the value of the remaining habitat is also 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 do 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 contributes to the extinction of species (Turner 1996).

Because of the large wilderness and protected areas in the Salmon subbasin, the total amount of habitat fragmentation is low or practically absent in the Upper Salmon, Lower Middle Fork Salmon, Upper Middle Fork Salmon, South Fork Salmon, and Middle Salmon–Chamberlain watersheds (Figure 7 and Table 8).

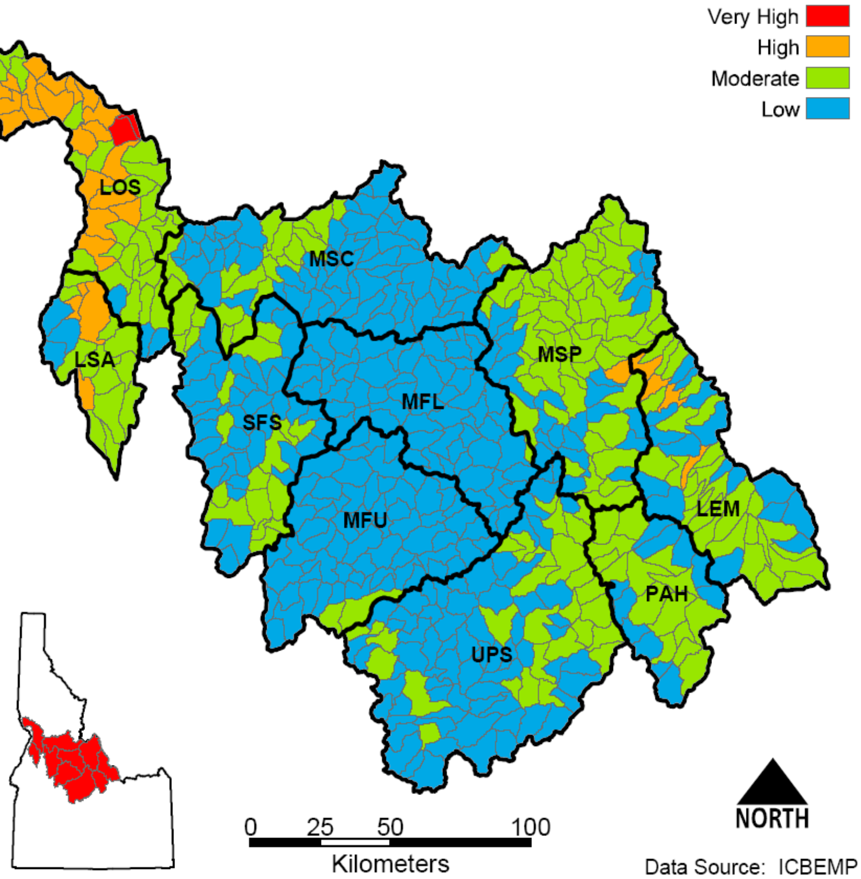

Figure 7. Estimated habitat fragmentation in the Salmon subbasin, Idaho.

Based on tax assessments on land parcel splits (an indicator of development and/or habitat fragmentation) in the Salmon subbasin, the Lemhi, Middle Salmon–Panther, and Upper Salmon watersheds have seen recent increases in habitat fragmentation. These three watersheds are highly valued for their scenic and recreational opportunities, resulting in increased land conversion from rangeland and cropland to residential uses.

Table 8. Comparison of the relative percentages of habitat fragmentation by watershed in the Salmon subbasin, Idaho. Source: ICBEMP (1997).

Relative Category	Major Hydrologic Unit (Watershed)[a]									
	UPS	PAH	LEM	MFU	MFL	MSC	MSP	SFS	LOS	LSA
Very high	0	0	0	0	0	0	0	0	3	0
High	0	0	6	0	0	0	1	0	44	20
Moderate	32	64	57	5	1	24	71	35	47	63
Low	68	36	37	95	99	76	28	65	5	17

[a] Percentages may not sum to 100 due to rounding.

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).

The most severely altered native grassland habitats occur in the Lower Salmon and Little Salmon watersheds, while the Upper Salmon, Pahsimeroi, Lemhi, and Middle Salmon–Panther are moderately altered watersheds.

2.4 Roads and Trails

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

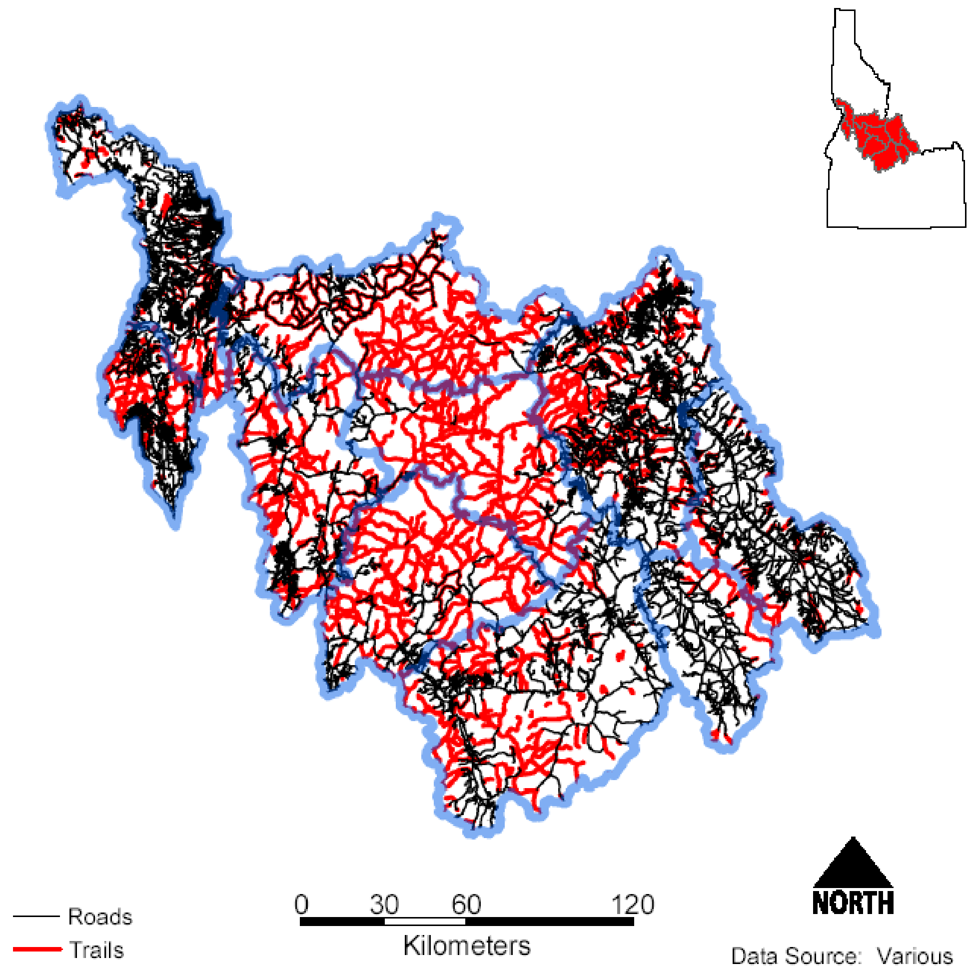

Figure 8. Distribution of roads and trails within the Salmon subbasin, Idaho. Sources: Tiger Roads, Boise National Forest Roads, Trails Payette National Forest Roads, Trails Salmon-Challis National Forest Roads, Trails Sawtooth National Forest Roads.

- Creation of barriers to dispersal
- Significant source of direct mortality due to collisions
- Displacement of sensitive wildlife species
- Habitat loss
- Loss of ecological complexity
- Reduction in species reproductive success
- Vectors of disease, pest infestations, and/or invasive exotic plants and animals
- Degradation of ecosystem function
- Degradation of soil resources and water hydrology due to road-building, use, and maintenance activities
- Increased sediment and altered streamflows
- Increased disturbance and harvest of big game animals (both legal and illegal)

Recreational road and trail use is typically defined in terms of the following recreational activities:

- Hiking
- Biking
- Horseback riding
- Off-roading (all-terrain vehicle use)
- Snowmobiling

- Hunting/fishing
- Skiing

The following impacts are typically associated with these activities, most of which are discussed below:

- Trampling by hikers and horses
- 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 reduced air quality
- Increased access to the resource and subsequent conflict between competing resource user groups.

Trampling—The effects of trampling are usually limited to within a meter of the trail's edge (Dale and Weaver 1974). Trampling causes compaction of leaf litter and soil. The occurrence of some plant species decreases 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). Compaction by horses is greater than by hikers (Whittaker 1978), with horses destroying eight times as much cover and creating an order of magnitude more bare ground than hikers do (Nagy and Scotter 1974). Trail width increases linearly with logarithmic increase in number of users (width doubles with a tenfold increase in use). Trails in meadows are a little wider than trails in forests. Trails with both horse and foot traffic are similar in width or slightly narrower than those receiving foot traffic alone. 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 bird behavior and movement. 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 (Boyle and Samson 1985). 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 since there are very few areas that these newer machines cannot access (Olliff *et al*. 1999).

Nonnative Vegetation—Disturbance in the form of introduced exotics along trails by horses and people has been documented by Benninger-Truax *et al*. (1992) in which horse manure was found to contain 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). All-terrain vehicles 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 is a likely factor that could favor 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). In addition, all-terrain vehicles 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 vehicle activities increases.

Roads and trails are found throughout the Salmon subbasin (Figure 8). Roads and their associated impacts are a significant factor in the Pahsimeroi, Lemhi, Middle Salmon-Panther, Lower Salmon and Little Salmon watersheds (Figure 8). Although very few roads occur in the wilderness and protected areas in the central portion of the subbasin, access can still be gained through extensive trail systems (Figure 8).

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 9 illustrates the historical fire regime in the Salmon subbasin.

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) and firefighting remains a major economic factor in small towns such as Salmon and Challis.

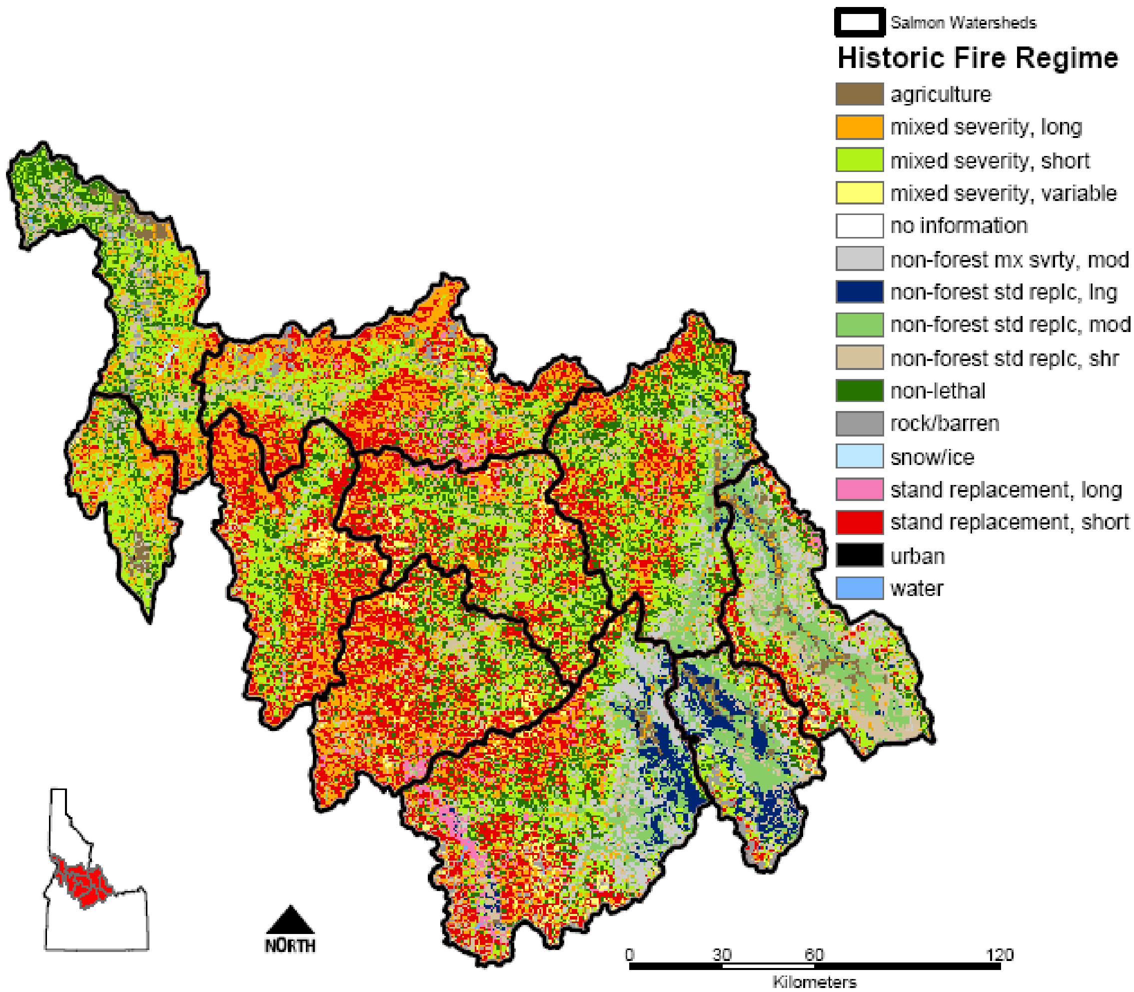

Figure 9. Historic fire regime in the Salmon subbasin, Idaho. Source: Northern Regional National Fire Plan Cohesive Strategy Assessment Team, Flathead National Forest.

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 10), 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 experience very large fires occasionally (Romme and Despain 1989), the present-day frequency of large fires is increasing. Figure 11 shows current fire severity in the Salmon subbasin, while Figure 12 depicts areas in the subbasin that are most likely to experience severe burns. Table 9 compares the relative percentages of risk by altered fire regimes by watershed in the Salmon subbasin. In addition, Figure 13 illustrates fire regime condition class, which is an approximation of ecosystem departure resulting from a change in fire regimes.

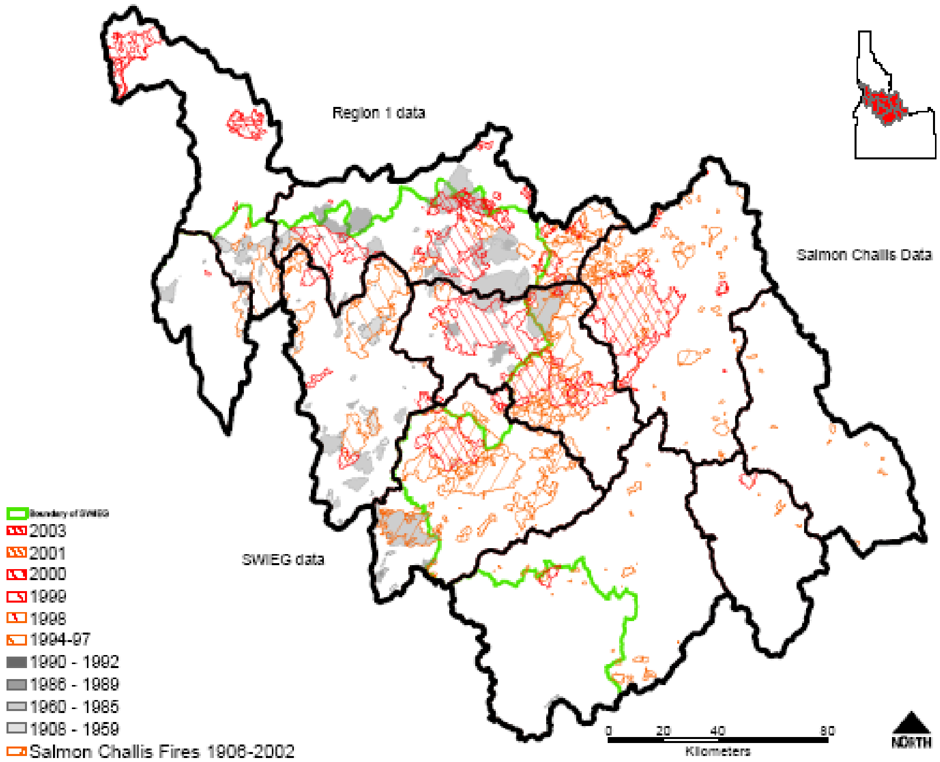

Figure 10. Locations of large (greater than 5-hectare) fires in the Salmon subbasin, Idaho, between 1908 and 2003.

Before the era of fire suppression, fires burned across the landscape at a variety of intensities, sizes, and fire-return intervals based on localized climate and fire-return intervals on a cold/wet to warm/dry gradient. This process 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 that 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 grow bigger and produce more seeds. Even plants that are killed by fire may have coping mechanisms that allow the species to survive, even when individuals are burned. They may have hard seeds that survive until fire readies them to grow, or they may have light, easily dispersed seeds that can quickly reinvade a burned area. Most employ some combination of these strategies (INFMS 2003).

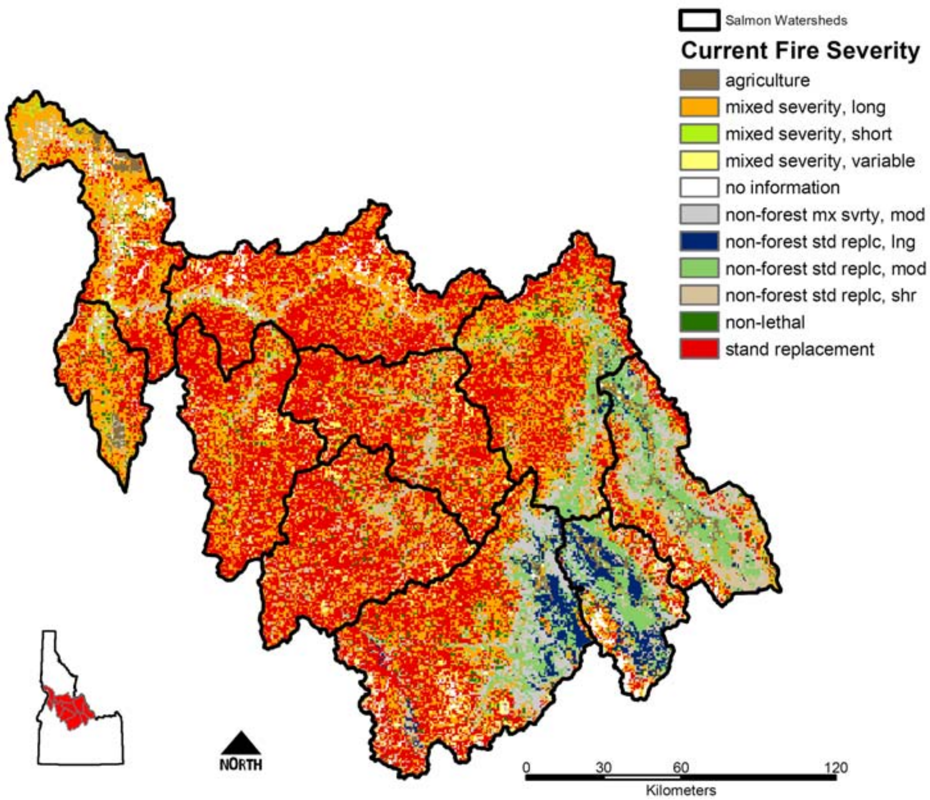

Figure 11. Current fire severity in the Salmon subbasin, by subwatershed, including all fires between 1986 and 1992. Source: Northern Regional National Fire Plan Cohesive Strategy Assessment Team, Flathead National Forest.

The greatest effect of fire suppression on biological diversity is not on the diversity within a particular habitat (Whittaker 1978), 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 included 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 wildfires. 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 and the importance of insects and pathogens elevated (Covington *et al*. 1994).

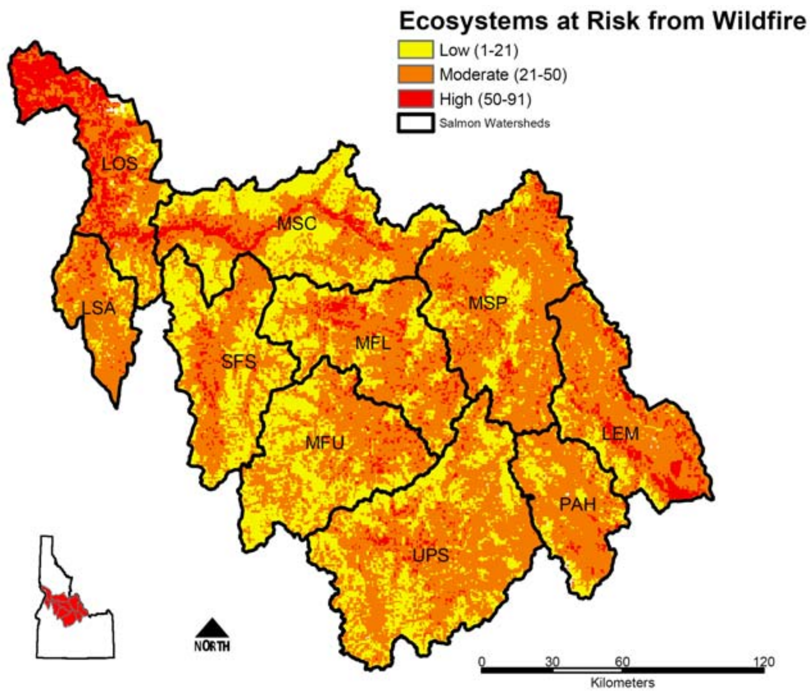

Figure 12. Predicted areas within the Salmon subbasin most likely to have severe burns. Source: Northern Regional National Fire Plan Cohesive Strategy Assessment Team, Flathead National Forest.

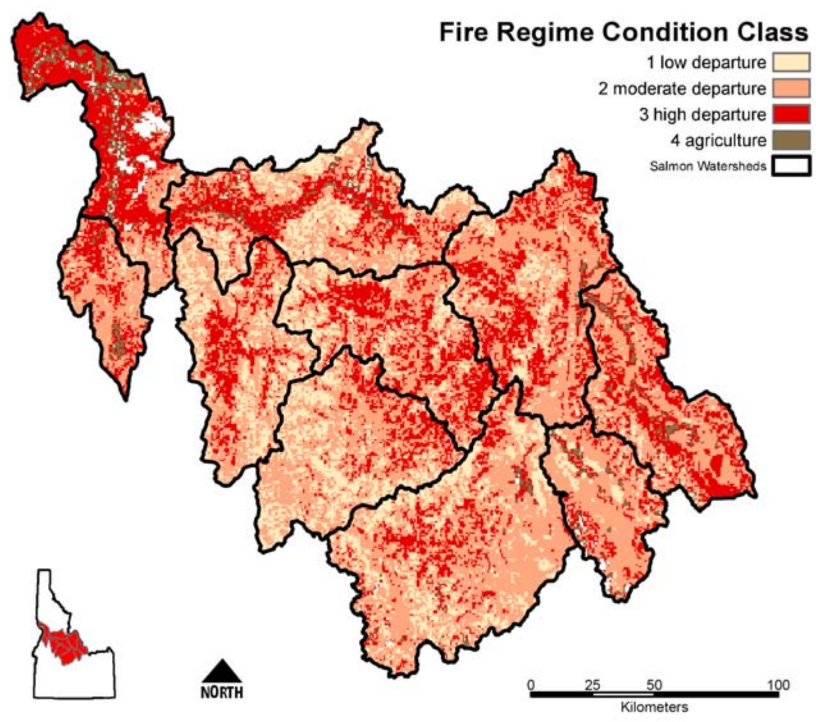

Figure 13. Fire regime condition class in the Salmon subbasin, Idaho. Source: Northern Regional National Fire Plan Cohesive Strategy Assessment Team, Flathead National Forest.

Table 9. Comparison of the relative percentages of risk by altered fire regimes by watershed in the Salmon subbasin, Idaho. (Source: Northern Regional National Fire Plan Cohesive Strategy Assessment Team, Flathead National Forest.)

Relative Category	Major Hydrologic Unit (Watershed)									
	UPS	PAH	LEM	MFU	MFL	MSC	MSP	SFS	LOS	LSA
Low risk	53	34	31	58	43	49	38	53	19	32
Moderate risk	31	45	35	26	29	25	36	29	21	40
High risk	13	11	27	16	27	21	25	18	47	24
No risk	<1	3	5				1		3	3

Sagebrush steppe ecosystems of the Great Basin in the western United States are examples of fire-prone ecosystems. Many wildlife species depend on sagebrush steppe ecosystems for survival (Knick and Van Ripper 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 (*Artesmisia* spp.) 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 (*Juniperus* spp.) and/or pinyon pines (*Pinus monophylla*) 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 over 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 that lead to 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 establishment of cheatgrass or other invasive species, further reducing habitat quality for sagebrush obligates and other species, both wild and domestic, that use 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 due to 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 since 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 competitors in the system. As fire intervals become shorter due to 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 began to have a significant impact on the region's biota with their large-scale transportation into the region via the railroad 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 that federal forest reserves were proclaimed in the 1890s, ranchers had become accustomed to unregulated use of public lands as range for their livestock. As a result of these excessive stocking numbers, once rich grasslands were seriously degraded 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 forest floor in some places was "as bare and compact as a roadbed." 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 pressure had penetrated the most remote, timbered, and mountainous areas. Over 100 years later, the effects of intense grazing in the latter part of the nineteenth century can still be readily seen in many parts of the West (CPLUHNA 2003).

Today in the Salmon subbasin, strategic and prime ranchlands occur primarily in the Upper Salmon, Lower Salmon, Little Salmon, Middle Salmon–Panther, Lemhi and Pahsimeroi watersheds (Figure 14).

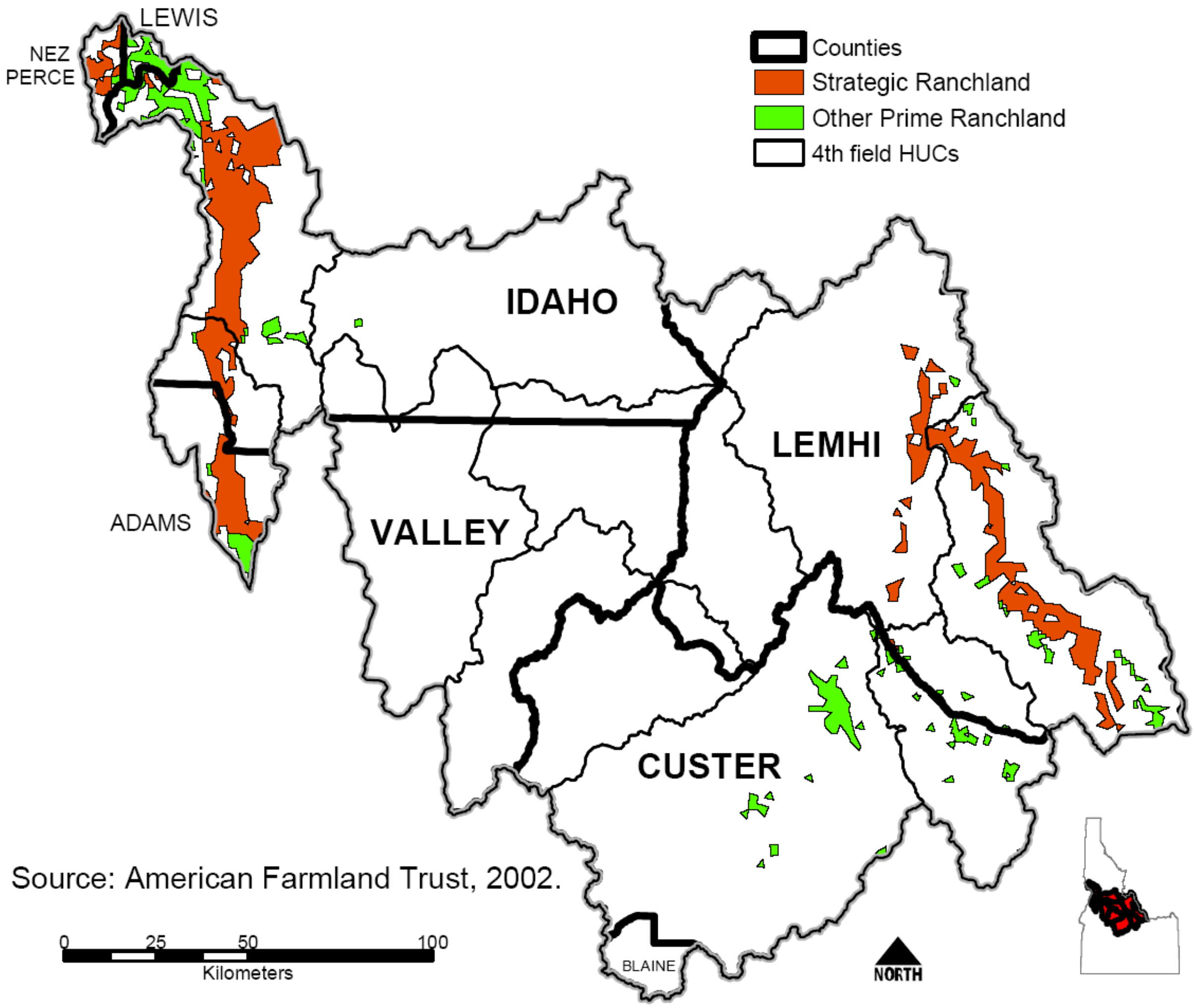

Figure 14. **Strategic and prime ranchland critical to local agricultural economies, plant and wildlife habitat and open space in the Salmon subbasin, Idaho (American Farmland Trust, 2002).**

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 52% of the total area in the Salmon subbasin is impacted by grazing and browsing by domestic animals (Table 10). Undisturbed herbaceous ecosystems across the western United States are rare. Still, a precise determination of the ecological effects of grazing is often difficult to obtain because ungrazed land is extremely rare, exclosures are small, exact figures on grazing intensities are scarce, and approaches for evaluating the effects of grazing are not standardized (Flather *et al.*1994, Fleischner 1994, Belsky and Blumenthal 1997). For example, the status of grazing and browsing by domestic animals in the Salmon subbasin is unknown for approximately 19% of the total area (Table 10).

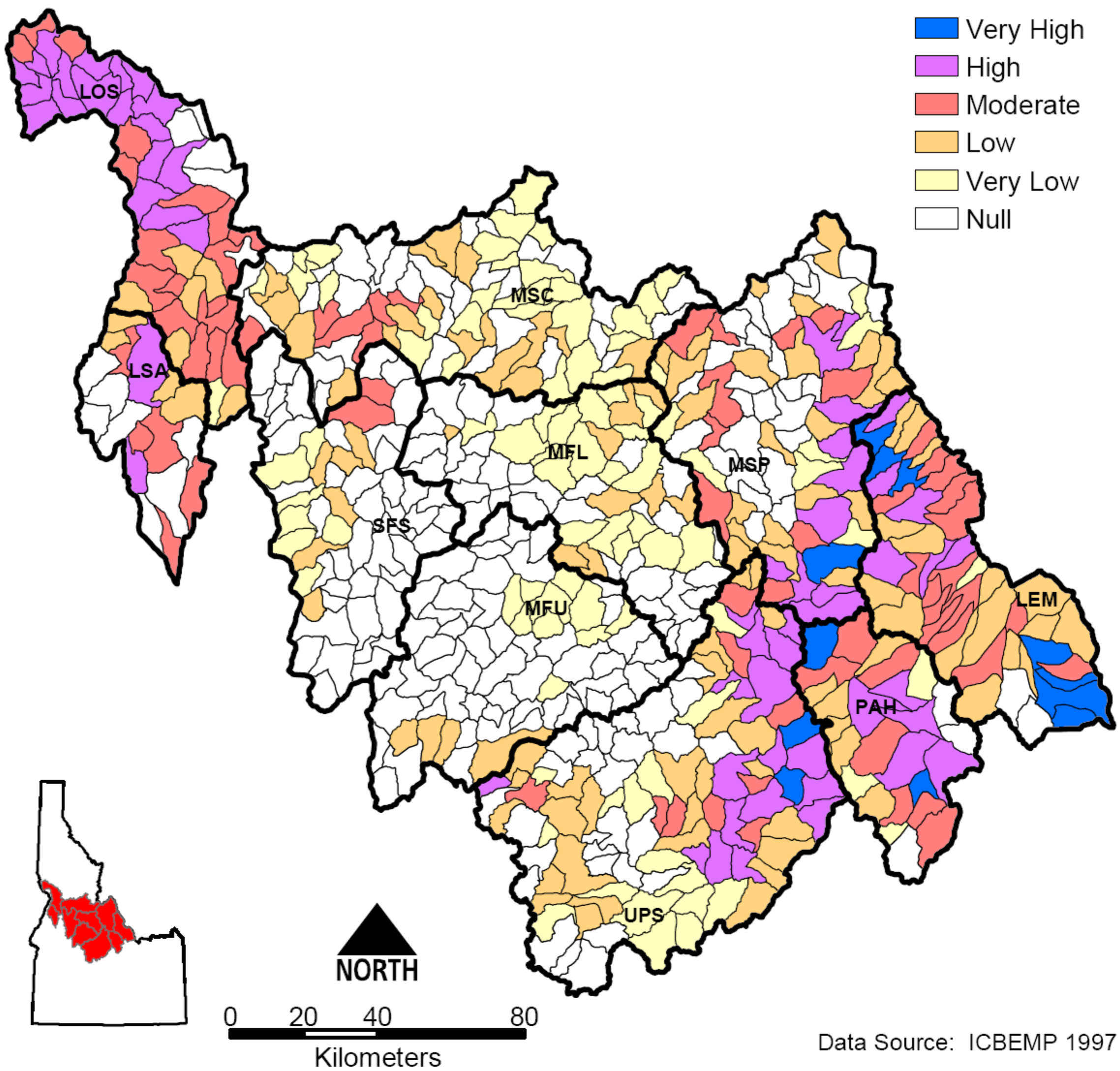

Figure 15. Rangeland condition in the Salmon subbasin, Idaho.

4.1 Grazing/Browsing Activity in the Salmon Subbasin

The majority of grazing and browsing activities occur in the Pahsimeroi, Little Salmon, Lemhi, Upper Salmon, and Middle Salmon–Panther watersheds (Table 10 and Figure 16). Comparatively, very little grazing activity occurs in the Upper Middle Fork Salmon, Lower Middle Fork Salmon, South Fork Salmon, and Middle Salmon–Chamberlain watersheds, some of which have large portions of designated wilderness (Figure 16).

Watersheds that have been identified as having excessive sedimentation and warmer stream temperature due to grazing impacts include the South Fork Salmon, Little Salmon, Pahsimeroi, and Upper Salmon watersheds. Water temperature and sedimentation are important habitat characteristics for resident and anadromous fish populations.

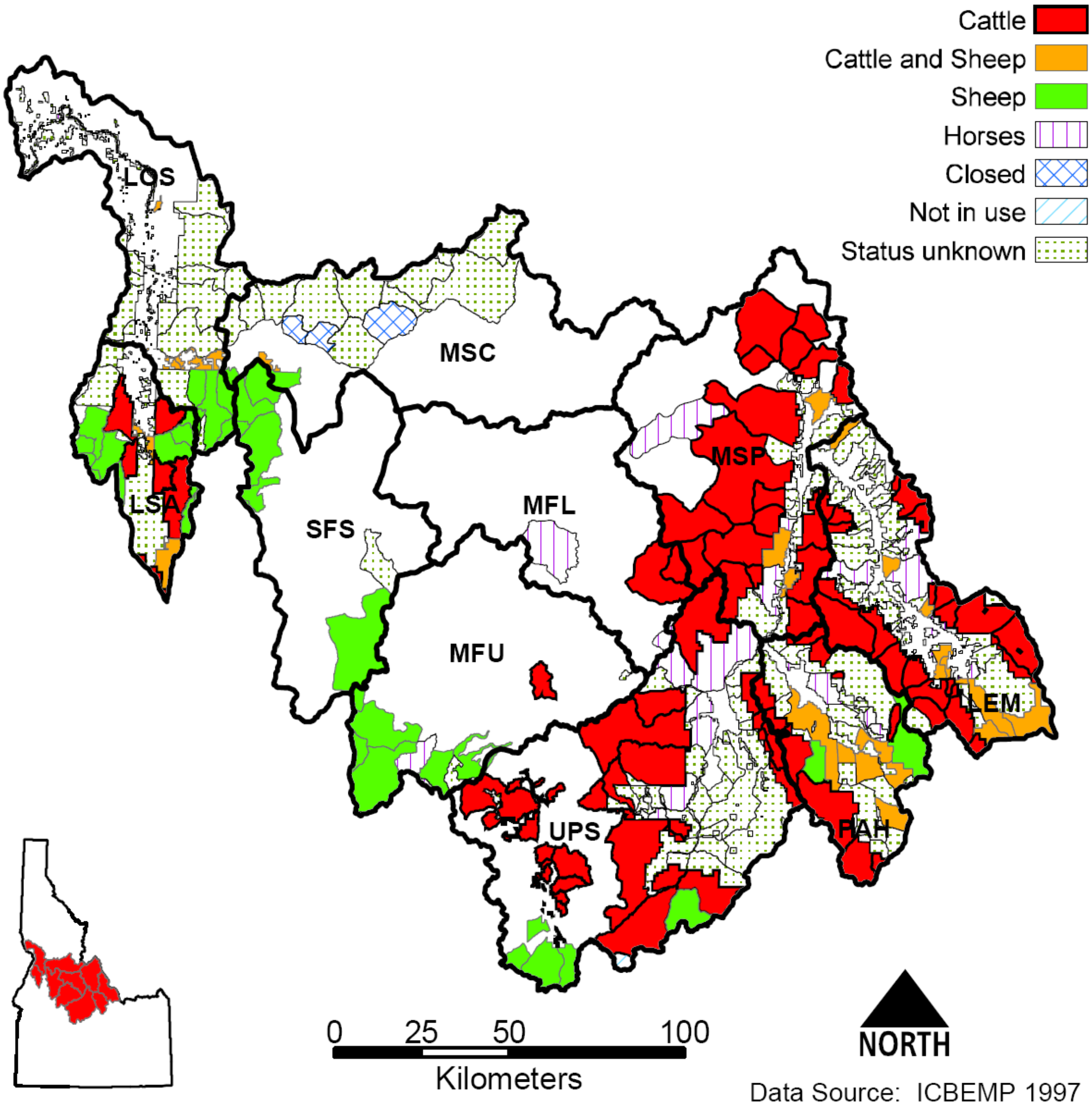

Figure 16. **Occurrences of grazing and browsing activities by domestic animals in the Salmon subbasin, Idaho.**

Table 10. **Percentage of area impacted by grazing/browsing livestock by watershed in the Salmon subbasin, Idaho. Sources: ICBEMP (1997) and GAP II (Scott *et al*. 2002).**

Allocation Description	**Major Hydrologic Unit (Watershed)**										**Total Area (km^2)**
	UPS	**PAH**	**LEM**	**MFU**	**MFL**	**MSC**	**MSP**	**SFS**	**LOS**	**LSA**	
Cattle	37	26	37	1	7		49		<1%	25	7,053
Cattle and Sheep		17	10			<1%	3	<1%	2	5	943

Allocation Description	Major Hydrologic Unit (Watershed)										Total Area (km^2)
	UPS	**PAH**	**LEM**	**MFU**	**MFL**	**MSC**	**MSP**	**SFS**	**LOS**	**LSA**	
Closed						5					221
Horses	8	4	9	2	6		6	<1%	<1%		1,454
Not in use	<1%										25
Sheep	5	9	<1%	15		1		21	8	25	2477
Status unknown	21	39	30	1	<1%	32	9	3	39	31	6,753
Not Allotted for grazing	28	5	15	81	87	62	34	77	52	13	17,314

The percentage of area impacted by grazing for each of the seven terrestrial focal habitats in the Salmon subbasin is presented in Table 11.

Table 11. Percentage of area impacted by grazing domestic animals for each of the seven terrestrial focal habitats in the Salmon subbasin. Sources: ICBEMP (1997) and GAP II (Scott *et al*. 2002).

Focal Habitat	Cattle	Cattle and Sheep	Sheep	Horses	Closed	Not in Use	Status Unknown	Total Area (km^2)
Riparian/herbaceous wetlands	29	4	23	5	<1	<1	39	387
Shrub-steppe	25	11	3	14	<1	<1	47	5,349
Pine/fir forest	46	2	16	6	2	<1	29	10,351
Native grasslands	12	17	3	11	6		50	476
Aspen	26	<1	48	3	4		19	24
Juniper/mountain mahogany	54	4	4	9	<1		29	218
Whitebark pine	64		5	14	<1	<1	17	156
Other[a]	28	2	25	2	1	<1	41	1,961

[a] Includes habitats not grazed.

4.2 Impacts to Riparian/Wetland Habitats

Riparian areas are critical ecosystems in the semiarid 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 from overgrazing is a serious problem (Belsky *et al.* 1999).

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 nonnative 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. Fish survival and the carrying capacity of the habitat have been reduced by land and water management activities within the subbasin 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 that are 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 for 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). Over the last century, most of these forests have been 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 direct effect differs from the more indirect effects livestock 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). Such use 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 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, thereby increasing the proportion of bare ground (Belsky and Blumenthal 1997). For example, Schulz and Leininger (1990) found 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/bunchgrass 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 (Facelli and Pickett 1991, Thurow 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 content, in turn, reduces 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 the habitat become significant when the animals become so numerous as to exceed the carrying capacity of the habitat. Such exceedances 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 among forb species than sheep are (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 because 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).

Idaho forests have undergone significant changes in tree species composition since 1952 (O'Laughlin *et al*. 1993) (Figure 18 and Figure 19). 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.

Douglas-fir increased by roughly 1.2 billion cubic feet, or 15%, holding its position as the largest component of Idaho forests. The second largest component is the aggregation for Engelmann spruce (*Picea engelmannii*), western larch, and other softwoods, primarily western red cedar (*Thuja plicata*) and western hemlock (*Tsuga heterophylla*). Taken together, spruce, larch, cedar, and hemlock increased by more than 30% from 1952 to 1987 (O'Laughlin *et al*. 1993). Lodgepole pine, an early seral species, has declined dramatically (Figure 18 and Figure 19).

Timber harvest has occurred throughout the Salmon subbasin (Figure 17 and Table 12). The most intense timber-harvest activities have occurred in the Little Salmon and Lower Salmon watersheds (Figure 17 and Table 12). Very low to medium harvest activities have occurred in the protected areas of the central Salmon subbasin. The effects of timber harvest to wetland habitats have been described in detail elsewhere.

Intense timber-harvest activities occurring primarily in the Little Salmon and Lower Salmon watersheds have impacted approximately 1,190 and 2,600 km^2, respectively. Other watersheds impacted by timber harvest and the approximate area impacted include the Middle Salmon–Panther (4,180 km^2), Upper Salmon (2,880 km^2), South Fork Salmon (2,120 km^2), Middle Salmon–Chamberlain (1,410 km^2), Pahsimeroi (977 km^2), Upper Middle Fork Salmon (700 km^2), and Lower Middle Fork Salmon (460 km^2).

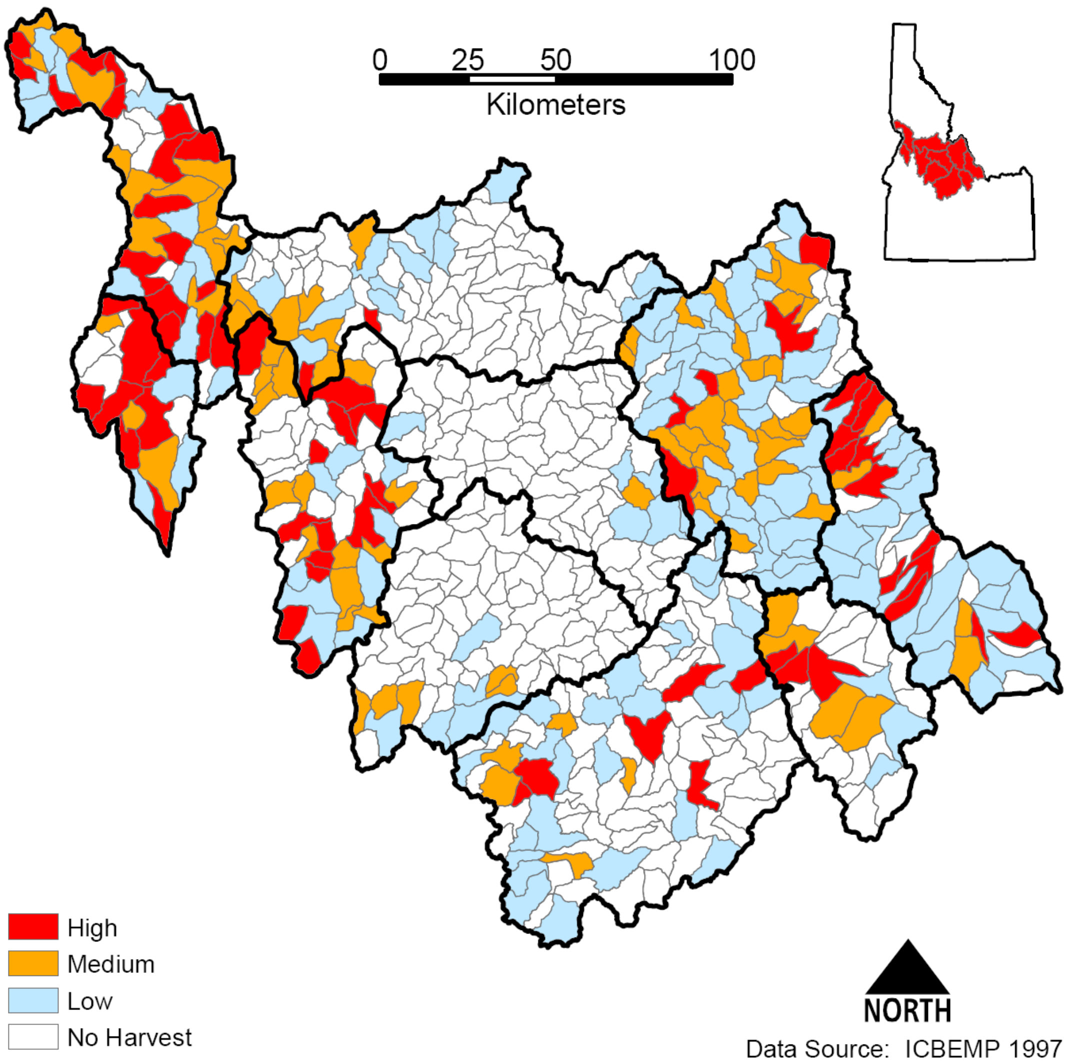

Figure 17. Relative timber-harvest impacts in the Salmon subbasin, Idaho.

Table 12. Comparison of the relative percentages of timber harvest by watershed in the Salmon subbasin, Idaho. Source: ICBEMP (1997).

Relative Category	Major Hydrologic Unit (Watershed)[a]									
	UPS	PAH	LEM	MFU	MFL	MSC	MSP	SFS	LOS	LSA
High	7	11	20	0	0	1	8	24	35	47
Medium	4	22	7	6	1	10	26	20	29	14
Low	34	12	60	12	11	20	55	19	23	18
No harvest	54	55	13	82	87	68	11	37	13	20

[a] Percentages may not sum to 100 due to rounding.

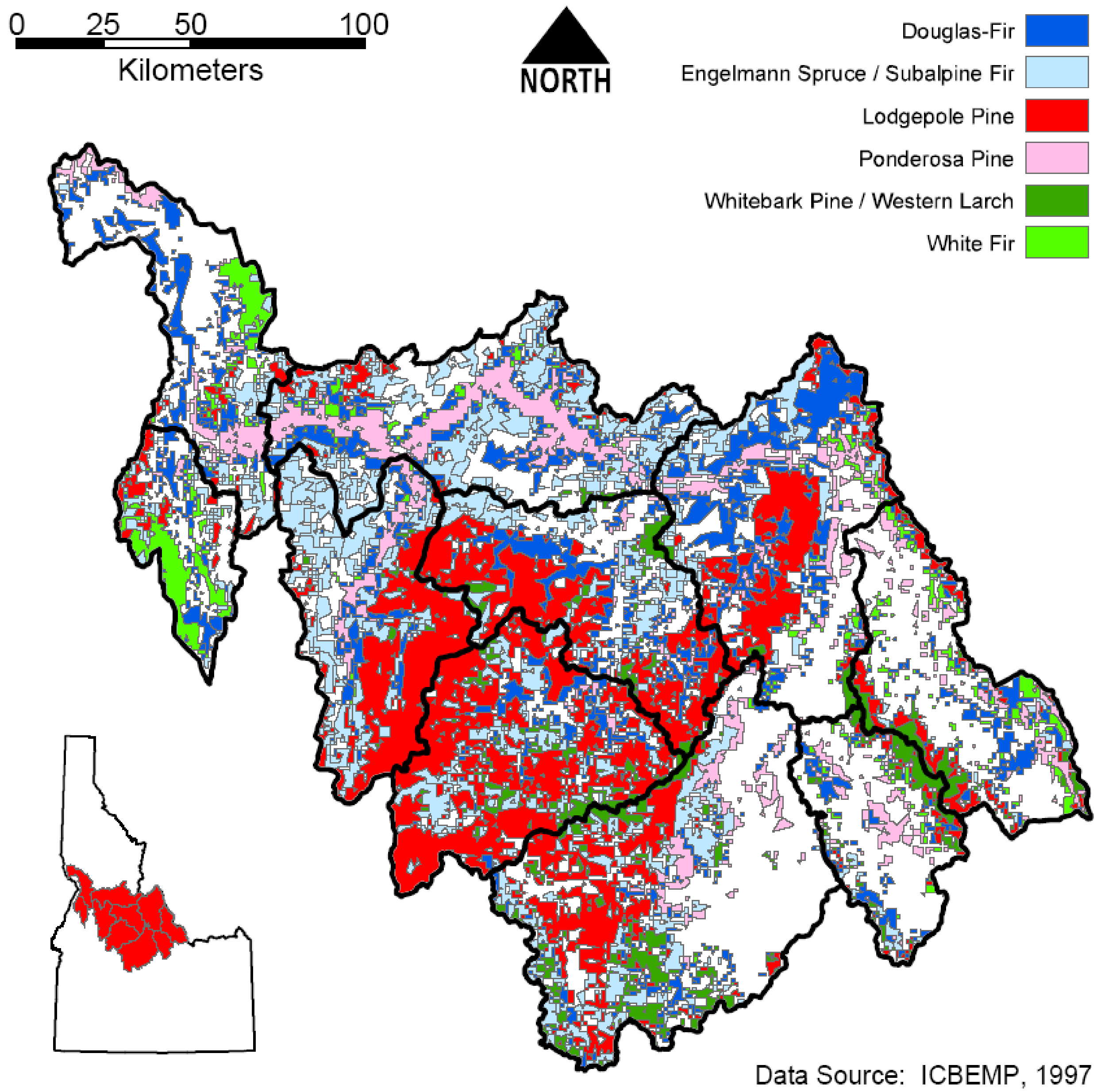

Figure 18. Historical forest species compositions in the Salmon subbasin, Idaho.

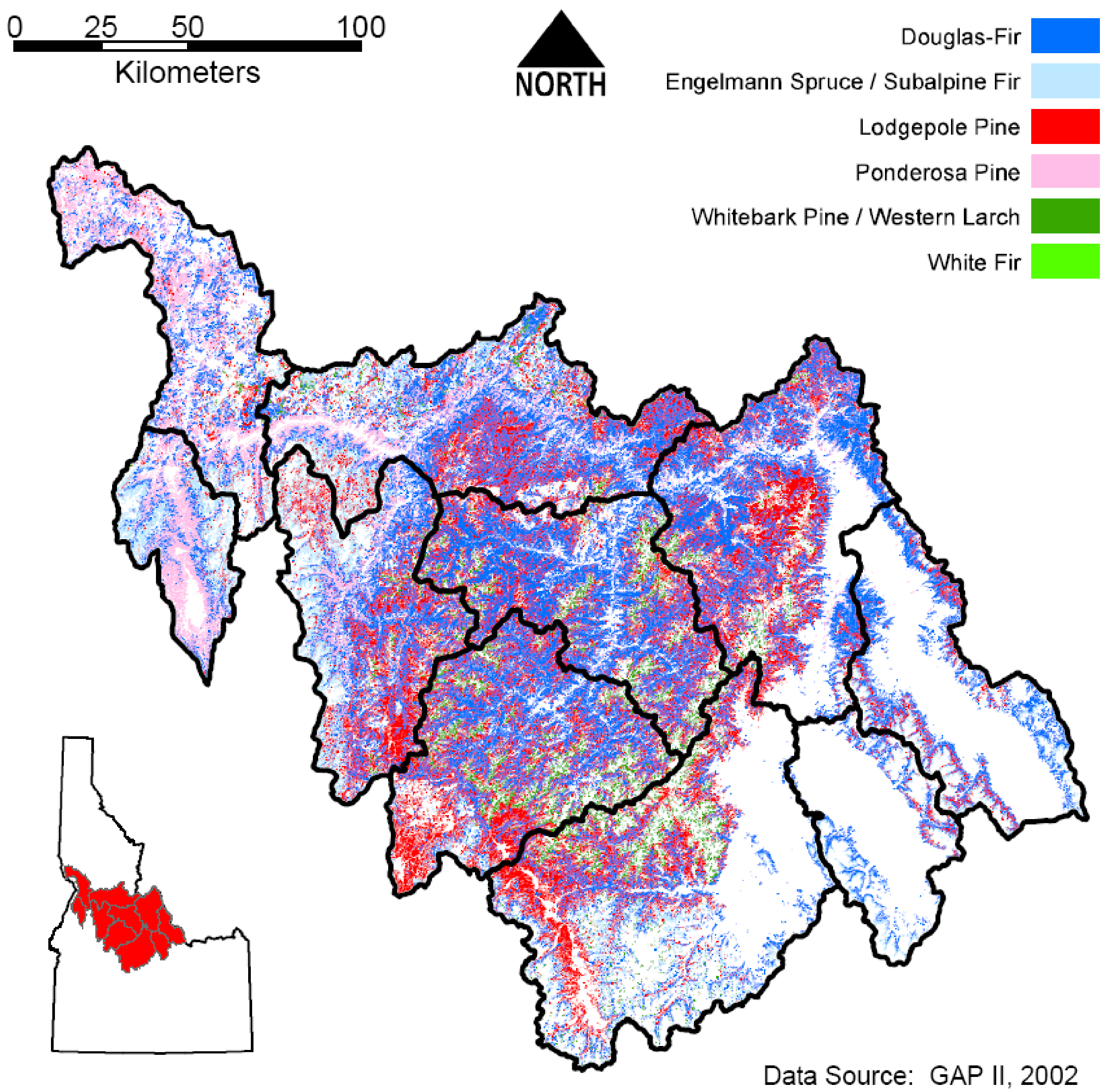

Figure 19. Current forest species compositions in the Salmon subbasin.

5.1 Impacts to 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 ranges or have been introduced from other parts of the world. Species are considered invasive if their presence in an ecosystem causes harm to the environment, economy, or 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 due to human induced changes to that ecosystem (USEPA 2001).

Noxious and invasive exotic weeds significantly impact grassland habitats in the Salmon subbasin and focal terrestrial habitats in the Upper Salmon, Pahsimeroi, Lemhi, Middle Salmon–Panther, Lower Salmon, and Little Salmon watersheds.

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 outcompeting 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 nineteenth century as an ornamental plant (Malecki *et al*. 1993). Purple loosestrife is capable of invading many wetland types, including freshwater wet meadows, tidal and nontidal marshes, river- and streambanks, pond edges, reservoirs, and ditches. It has been spreading at a rate of 115,000 hectares a 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 Salmon subbasin 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 in forested habitats in the Salmon subbasin is the spotted knapweed (*Centaurea maculosa*). This species infests a variety of natural and semi-natural habitats including barrens, fields, forests, prairies, meadows, pastures, and rangelands. It outcompetes 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 the rangeland (35 million acres) in Montana alone (Carpinelli 2003). Spotted knapweed is capable of establishing itself in undisturbed sites; however, disturbance allows for rapid establishment and spread.

6.4 Impacts to Native Grasslands

The most significant invasive weed for native grasslands is cheatgrass, which is discussed within the shrub-steppe section above. Other invasive exotic weeds include yellow starthistle (*Centaurea solstitialis*), spotted knapweed, rush skeletonweed (*Chondrilla juncea*), and whitetop (*Cardaria draba*, also known as hoary cress). Yellow starthistle is especially problematic in the lower Salmon River corridor and Snake River canyon.

6.5 Impacts to Whitebark Pine

The two most serious factors limiting whitebark populations in the Pacific Northwest are altered fire regimes (discussed elsewhere) and the exotic invasive fungus white pine blister rust (*Cronartium ribicola*). Although currants and gooseberries (*Ribes* spp.) are not impacted by whitebark 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: fewer than one in 10,000 trees is resistant to blister rust. Because whitebark 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).

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