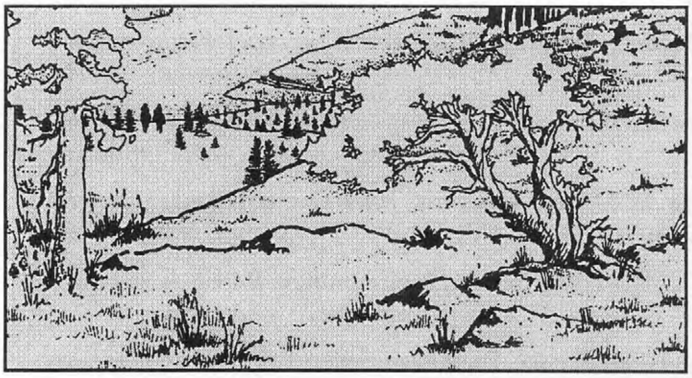


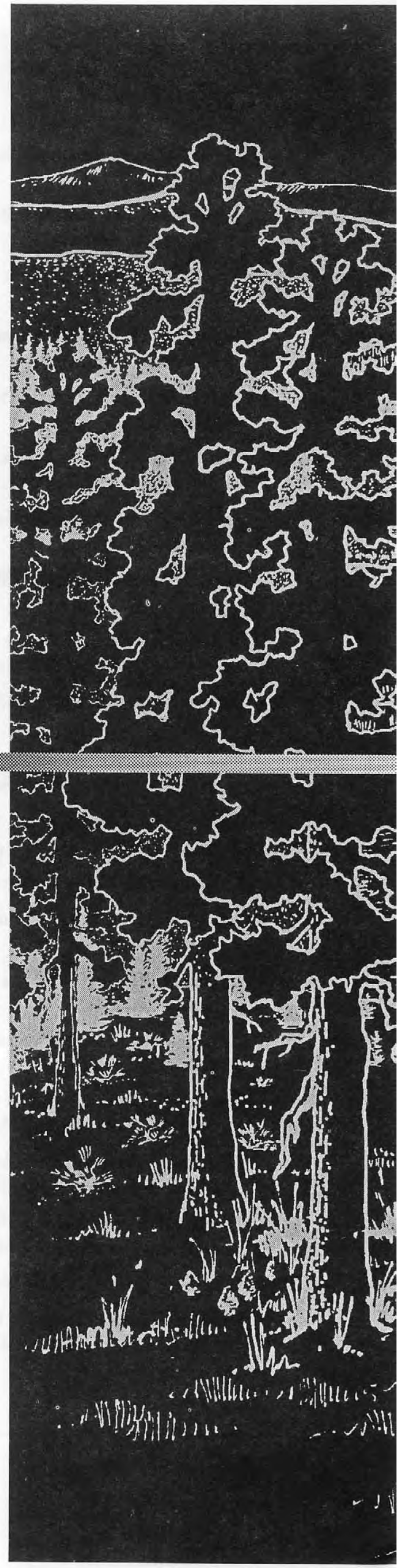
CHAPTER 4

Broadscale Assessment of Aquatic Species and Habitats

Danny C. Lee, James R. Sedell*,*
Bruce E. Rieman, Russell F. Thurow*,*
*Jack E. Williams**

David Burns, James Clayton, Lynn Decker,
Robert Gresswell, Robert House, Phil Howell,
Kristine M. Lee, Ken MacDonald,
John McIntyre, Shaun McKinney, Tracy Noel,
Jim E. O'Connor, C. Kerry Overton,
Doug Perkinson, Ken Tu,
and Pat Van Eimeren

**contact authors*



Danny C. Lee, Research Biologist. USDA Forest Service, Intermountain Research Station, Boise, ID.

James R. Sedell, Research Biologist. USDA Forest Service, Pacific Northwest Research Station, Corvallis, OR.

Bruce E. Rieman, Research Fisheries Biologist Service, Intermountain Research Station, Boise, ID.

Russell F. Thurow, Fisheries Research Scientist. USDA Forest Service, Intermountain Research Station, ID.

Jack E. Williams, Aquatic Scientist. Bureau of Land Management, Idaho State Office, Boise, ID.

David Burns, Fisheries Biologist. USDA Forest Service, Intermountain Region, Payette National Forest, McCall, Idaho.

James Clayton, Soil Scientist. USDA Forest Service, Intermountain Research Station, Boise, ID.

Lynn Decker, Biologist. USDA Forest Service, Pacific Southwest Region, San Francisco, CA.

Robert Gresswell, Fisheries Biologist. USDI Fish and Wildlife Service, detailed to Pacific Northwest Research Station, Corvallis, OR.

Robert House, Fisheries Biologist. USDI Bureau of Land Management, Idaho State Office, Boise, ID.

Phil Howell, Fisheries Biologist. USDA Forest Service, Pacific Northwest Region, Umatilla National Forest, North Fork Ranger District, Uriah, OR.

Kristene M. Lee, Biologist. USDA Forest Service, Intermountain Region, Ogden, UT.

Ken MacDonald, Fisheries biologist. USDA Forest Service, Pacific Northwest Region, Wenatchee National Forest, Wenatchee, WA.

John McIntyre, Scientist Emeritus, USDA Forest Service, Intermountain Research Station, Boise, ID.

Shaun McKinney, Fisheries Biologist. USDA Forest Service, Umatilla National Forest, Pendleton, OR.

Tracy Noel, Biological Technician. USDA Forest Service, Pacific Northwest Research Station, Corvallis, OR.

Jim E. O'Connor, Fisheries Biologist. USDA Forest Service, Pacific Northwest Research Station, Walla Walla, WA.

C. Kerry Overton, Technology Transfer Specialist. USDA Forest Service, Intermountain Research Station, Boise, ID.

Doug Perkinson, Fisheries Biologist. USDA Forest Service, Northern Region, Kootenai National Forest, Libby, MT.

Ken Tu, Forester. USDA Forest Service, Umatilla National Forest, Walla Walla, WA.

Pat Van Eimeren, Fisheries Biologist. USDA Forest Service, Northern Region, Flathead National Forest, Kalispell, ID.

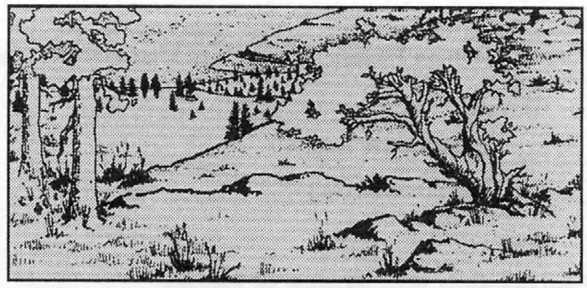
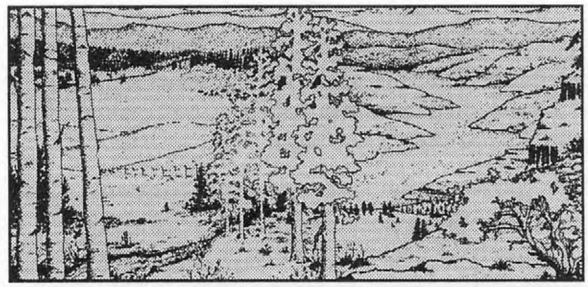


TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF MAPS



INTRODUCTION

The idea of rebuilding the fish and wildlife of an industrialized ecosystem is heroically optimistic—a hope that might not have occurred to anyone except those who had rehabilitated the Willamette basin in Oregon or Lake Washington near Seattle. The extension of those learning experiences to the multi jurisdictional, multi functional situation of the Columbia basin — a large ecosystem — requires coordinated action and learning on a new scale

- Kai N. Lee (1993)

In this chapter, we report on a broad-scale scientific assessment of aquatic resources conducted as part of the Interior Columbia Basin Ecosystem Management Project. Our assessment area, collectively referred to as the Basin, includes the Columbia River Basin east of the crest of the Cascade Mountains (Washington, Oregon, Idaho, western Montana, and small portions of Nevada, Utah, and Wyoming), and those portions of the Klamath Basin and the Great Basin in Oregon.

To many people, salmon characterize the Pacific Northwest. Salmon have been a cornerstone of the culture, economy, recreation, and history of the region. Salmon, trout, and other salmonids require relatively clean, cold-water environments, and because they are among the better known and monitored aquatic resources, salmonids, as a group, provide sensitive barometers to the overall health of aquatic habitats in the region (Marcot and others 1994).

Salmon is not the only fishery resource within the Basin. Fifty-two native fish species occur in the Columbia River system; thirteen of these species are found nowhere else (McPhail and Lindsey 1986). Including subspecies and the Oregon portions of the upper Klamath and Great Basin systems, the entire Basin supports 88 native fish taxa, of which 28 are narrowly distributed endemics. Forty-five of the 88 taxa are considered to be threatened, endangered, sensitive, or otherwise of special concern. In general, the fish faunas in the Great Basin and upper Klamath areas are less diverse than in the Columbia River system, but they are more highly specialized and contain a greater percentage of narrowly distributed endemic fishes.

Native fishes and other aquatic faunas throughout the assessment area are on the decline. Chinook and sockeye salmon in the Snake River are listed as endangered under the Endangered Species Act. Bull trout, once widely distributed in central Oregon, Idaho, and western Montana, warrant protection (U.S. Government 1995). Genetically pure populations of Yellowstone cutthroat trout are limited to a fraction of their historical stream habitat in the upper Snake River drainage (Gresswell 1995; Varley and Gresswell 1988). Only a small portion of the historic range of westslope cutthroat trout in Idaho and Montana still sustains genetically pure populations (McIntyre and Rieman 1995; Rieman and Apperson 1989). Redband trout within the Basin are poorly understood, but many subbasins appear to contain genetically unique strains that have declined concomitant with habitat degradation (Behnke 1992; Williams and others 1989). Such changes in salmonid populations may be indicative of broad declines in other aquatic resources such as stream habitats and riparian areas in the Columbia River basin.

The aquatic resources in the upper Klamath and Great Basin portions of the assessment area also have declined. For example, members of the sucker family (Catostomidae) in the upper Klamath basin, which supported important Native American subsistence fisheries and sport fisheries until the early 1980s, are now listed as endangered pursuant to the Endangered Species Act (U.S. Government 1988). Other native fishes of the Upper Klamath basin have declined as water quality has been reduced and the abundance of non-native species has increased. Most native fishes of the relatively sparse fish fauna in the Great Basin are federally listed as endangered, threatened, or are candidates for Endangered Species Act of 1973 (Public Law 93-205, as amended) protection because of activities that degrade important riparian habitats. Aquatic macroinvertebrates, including a surprisingly diverse springsnail (Hydrobiidae) fauna in the Great Basin, also have declined (Hershler 1995).

Many factors contribute to the decline of the fishery resources in the Basin. Dams and hydroelectric operations, introductions of hatchery and other non-native species, excessive harvest, and other factors have been identified. Degradation and loss of freshwater habitats, however, are a consistent and pervasive problem facing the aquatic faunas in the Basin and throughout much of the western United States (Nehlsen and others 1991; Williams and others 1989; Young 1995). Management of Federal lands can play a critical role in the condition of aquatic systems. Continued declines of fisheries resources suggest that past management practices have been insufficient to reverse the trend. Broad-scale ecosystem approaches are needed to halt habitat degradation, to maintain existing high-quality habitats, and to aid in the recovery of declining fish and aquatic invertebrate resources.

Restoring the general condition of streams and their watersheds in the Basin will yield benefits beyond conservation of fishes. Overall biodiversity, from microbes that perform valuable ecosystem services to top carnivores, may benefit from restorative actions. Other important but often unappreciated natural processes, such as recharging underground aquifers and providing large woody debris to stream systems, would also be maintained (Maser and Sedell 1994; Naiman 1992). Water quality, recreation, drought resistance, and flood protection would be enhanced by improving watershed condition. We are beginning to understand and appreciate the natural geomorphic, hydrologic, and biotic processes that shape stream systems. To effectively apply these processes to stream restoration, however, we also need a clearer understanding of the effects of human activities on natural processes.

In creating this assessment we addressed four primary objectives. First, we broadly characterized the geophysical and biological settings that define the natural potential of the Basin to support aquatic resources. Second, we identified anthropogenic factors that affect aquatic habitats and the species they support, either directly or through indirect disruption of natural processes. Special emphasis was given to effects of Federal land management. Third, we conducted a broad-scale assessment of the current condition of aquatic habitats and species. Though we focused primarily on fishes, ancillary information was also gathered on molluscs, other invertebrates, and aquatic plants. Finally, we worked to synthesize our information such that it might provide a regional context for Federal management agencies to devise proper management strategies for aquatic and riparian habitats.

This broad-scale assessment of aquatic resources is organized in six sections. The first section describes the primary elements of change, both natural and human-caused, that affect the integrity of aquatic ecosystems across the Basin. In the second section, the relationship between small-scale stream features, landscape-scale environmental features, and human activities is examined using stream-inventory data. Next, we describe the distribution and status of fishes in the Basin, identify critical and high integrity areas, and explore linkages between fish communities, the landscape, and human activities. The following section describes rare and sensitive species in the assessment area. Ecological risks and management opportunities at various scales are then discussed, and the last section contains information and research needs for improved management and conservation of aquatic resources.

The breadth and depth of this assessment require close attention to spatial scale. For comparative purposes, 13 Ecological Reporting Units (ERUs) were defined based on bedrock, hydrological characteristics, vegetation patterns, and climate (table 4.1, map 4.1). At finer scales, topography was used to define a hierarchical system of increasingly smaller watersheds (table 4.2, fig. 4.1). Within the Basin, for example, 164 large subbasins are defined. Examples of subbasins include the Bitterroot River in Montana, South Fork Boise River in Idaho, Middle Fork John Day River in Oregon, and Wenatchee River in Washington. The subbasins are further divided into watersheds, which average approximately 22,820 hectares in size. These watersheds are then divided into smaller subwatersheds, each approximately 7,830 hectares in size. These divisions follow the hierarchical framework of aquatic ecological units recently described by Maxwell and others (1995), which provides further descriptions of these elements. Subwatersheds form the basic sample unit for analysis and description of many of the elements discussed in the following sections. ERU boundaries are not congruent with subbasin boundaries but do follow subwatershed boundaries. That is, subwatersheds within subbasins may be divided among more than one ERU.

Table 4.1— Summary characterization of Ecological Reporting Units.

Northern Cascades	
Bedrock:	Cenozoic andesites and basalts with extensive metamorphism
Topography:	steep volcanic foothills and mountains with narrow intermountain valleys and highly dissected stream networks
Vegetation:	gradient of pine/fir with sagebrush through fir/cedar to fir/hemlock as elevation increases
Elevation:	152-3,658 m (500-12,000 ft)
Climate:	25-381 cm (10-150 inches) annual precipitation
Southern Cascades	
Bedrock:	Cenozoic andesites, tephra, and basalts
Topography:	moderately steep volcanic foothills and mountains with narrow intermountain valleys highly dissected with stream networks
Vegetation:	pine/fir with sagebrush, juniper, and meadows at low elevations; proportions of white fir, silver fir, and meadows increasing with elevation
Elevation:	152-3,048 m (500-10,000 ft); extremely variable
Climate:	25-254 cm (10-100 in)annual precipitation; extremely variable
Upper Klamath	
Bedrock:	Cenozoic volcanics and sediments
Topography:	steep volcanic mountains, foothills, and plains and low to moderately dissected stream networks
Vegetation:	sagebrush, juniper, and pine with meadows and fir at low elevations; gradient from white and red fir to silver fir mixed with meadows at the higher elevations
Elevation:	457-2,743 m (1,500-9,000 ft)
Climate:	51-127 cm (20-50 in) annual precipitation

Table 4.1 (continued).

Northern Great Basin

Bedrock:	Cenozoic alluvial and volcanic materials
Topography:	low relief plains with some isolated mountains with low dissected stream networks; area contains the closed basins of Lake Abert, and Summer, Warner, Harney, Malhuer, and Alvord lakes
Vegetation:	vegetation is primarily salt desert shrub, sagebrush, and juniper with some meadows
Elevation:	1,219-2,195 m (4,000-7,200 ft); little variation
Climate:	15-51 cm (6-20 in) annual precipitation

Columbia Plateau

Bedrock:	Cenozoic basalts
Topography:	low relief plains and breaks with few stream networks
Vegetation:	sagebrush, bluebunch wheatgrass, and Idaho fescue
Elevation:	61-1,372 m (200-4,500 ft) elevation
Climate:	23-51 cm (9-20 in) annual precipitation

Blue Mountains

Bedrock:	Paleozoic and Cenozoic sediments and Cenozoic basalts
Topography:	low to moderate relief plains, foothills, and mountains with narrow intermountain valleys and breaks; stream networks uncommon
Vegetation:	sagebrush and grasslands mixed with pine and fir at the low elevations; grand fir more common at the mid-elevations with small amounts of subalpine fir mixed with meadows at the higher elevations
Elevation:	762-3,048 m (2,500-10,000 ft)
Climate:	25-127 cm (10-50 in) annual precipitation

Northern Glaciated Mountains

Bedrock:	Precambrian and Cenozoic sediments and metasedimentary materials
Topography:	low to steep relief foothills and mountains with narrow to wide intermountain valleys and highly dissected stream networks
Vegetation:	fir and pine mixed with grasslands, sagebrush, and meadows at lower elevations; grand fir and cedar (sometimes hemlock) in middle elevations; and subalpine fir mixed with meadows at the higher elevation
Elevation:	305-2,896 m (1,000-9,500 ft)
Climate:	38-254 cm (15-100 in) annual precipitation

Upper Clark Fork

Bedrock:	Precambrian and Mesozoic granite and metasedimentary materials
Topography:	steep glaciated mountain ranges; moderately steep foothills with moderately wide intermountain valleys and highly dissected stream networks
Vegetation:	fir and pine on foothills mixed with cottonwood, aspen, grasslands, sagebrush, and meadows in the valleys at the lower elevations; subalpine fir mixed with meadows more common at the middle to higher elevations
Elevation:	914-2,896 m (3,000-9,500 ft)
Climate:	36-203 cm (14-80 in) annual precipitation

Lower Clark Fork

Bedrock:	Precambrian metasedimentary materials
Topography:	steep mountain ranges with steep foothills and narrow mountain valleys and breaks with highly dissected and incised stream networks
Vegetation:	fir and pine on foothills mixed with grasslands and meadows in the valleys at lower elevations; fir and cedar more common at middle elevations; subalpine fir mixed with meadows at higher elevations
Elevation:	366-2,591 m (1,200-8,500 ft)
Climate:	76-203 cm (30-80 in) annual precipitation

Table 4.1 (continued).

Owyhee Uplands

Bedrock:	Cenozoic basalts
Topography:	low relief plains with isolated breaks, mountains and foothills, with low dissection and few stream networks
Vegetation:	salt desert shrub, sagebrush, and juniper with some meadows
Elevation:	1,219-2,438 m (4,000-8,000 ft)
Climate:	18-38 cm (7-15 in) annual precipitation

Upper Snake

Bedrock:	Cenozoic basalts
Topography:	low relief plains with low dissection and few stream networks
Vegetation:	salt desert shrub, sagebrush, and juniper with some meadows
Elevation:	914-1,829 m (3,000-6,000 ft)
Climate:	13-30 cm (5-12 in) annual precipitation

Snake River Headwaters

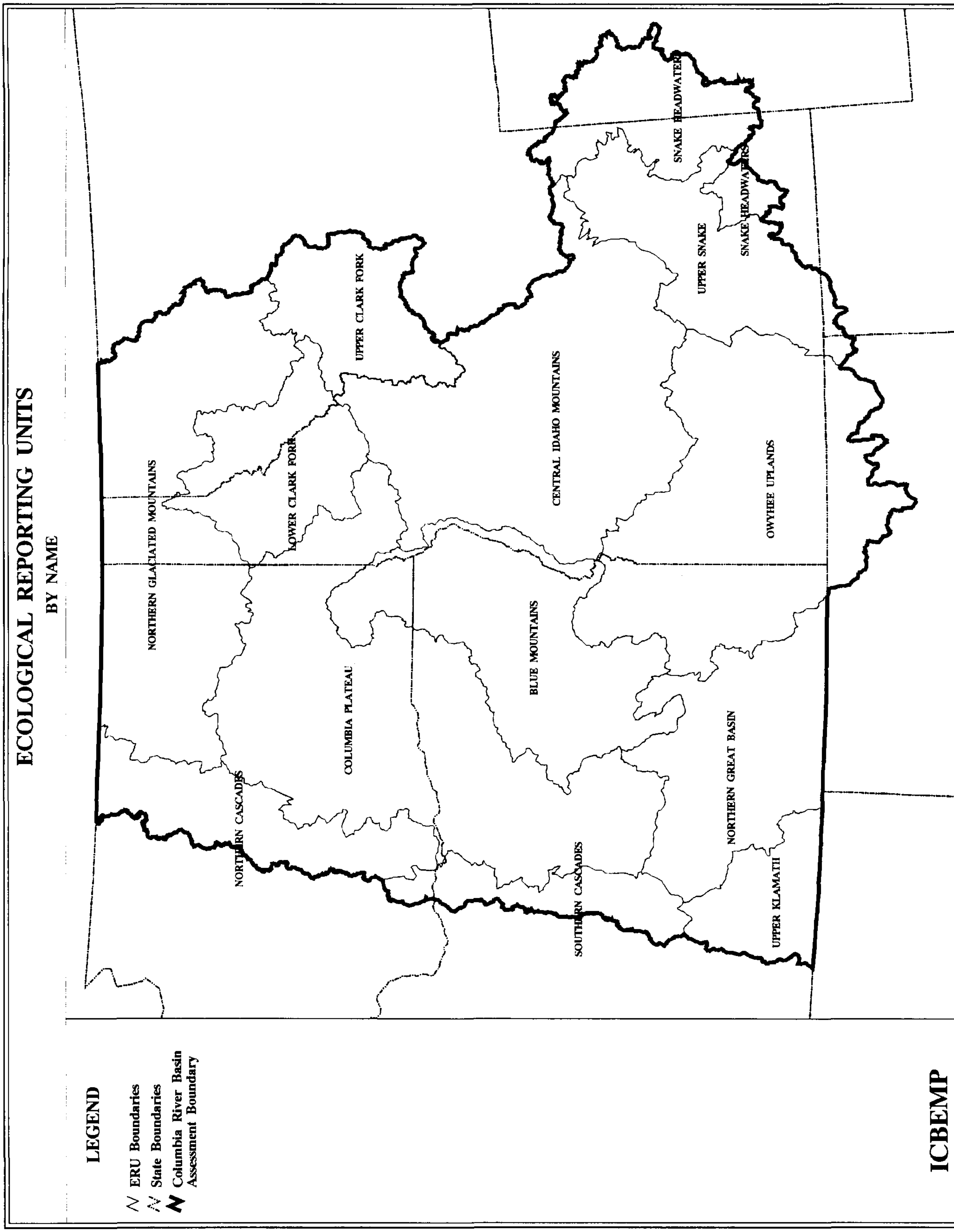
Bedrock:	Precambrian to Cenozoic complex sediments, metasediments, metamorphics, and volcanics
Topography:	steep glaciated and nonglaciated mountain ranges with intermountain valleys, foothills, and high plains; stream dissection low in the valleys and plains; foothills and mountains highly dissected with numerous stream networks
Vegetation:	fir and pine mixed with sagebrush and meadows on high plains, foothills, and lower mountain slopes; subalpine fir mixed with meadows and mountain sagebrush are found at the higher elevations
Elevation:	1,524-3,962 m (5,000-13,000 ft)
Climate:	41-114 cm (16-45 in) annual precipitation

Central Idaho Mountains

Bedrock:	Cenozoic and Mesozoic granite and Cenozoic volcanics and Precambrian quartzite
Topography:	glaciated and nonglaciated mountain ranges and foothills; narrow intermountain valleys and breaks of moderate to steep relief highly dissected by stream networks
Vegetation:	fir mixed with sagebrush, grasslands, and meadows on valleys, breaks, and lower mountain slopes; subalpine fir mixed with meadows and mountain sagebrush and grasslands more common at the higher elevations
Elevation:	914-3,352 m (3,000-11,000 ft)
Climate:	30-203 cm (12-80 in) annual precipitation

Table 4.2— Hierarchical framework of hydrology for the assessment area.

Hydrological Unit	Total Number within Assessment Area	Average Size
Subbasins	164	356,496 hectares (880,890 acres)
Watersheds	2,562	22,820 hectares (56,387 acres)
Subwatersheds	7,467	7,830 hectares (19,347 acres)



Map 4.1— Ecological Reporting Units in the Interior Columbia Basin Ecosystem Management Project area.

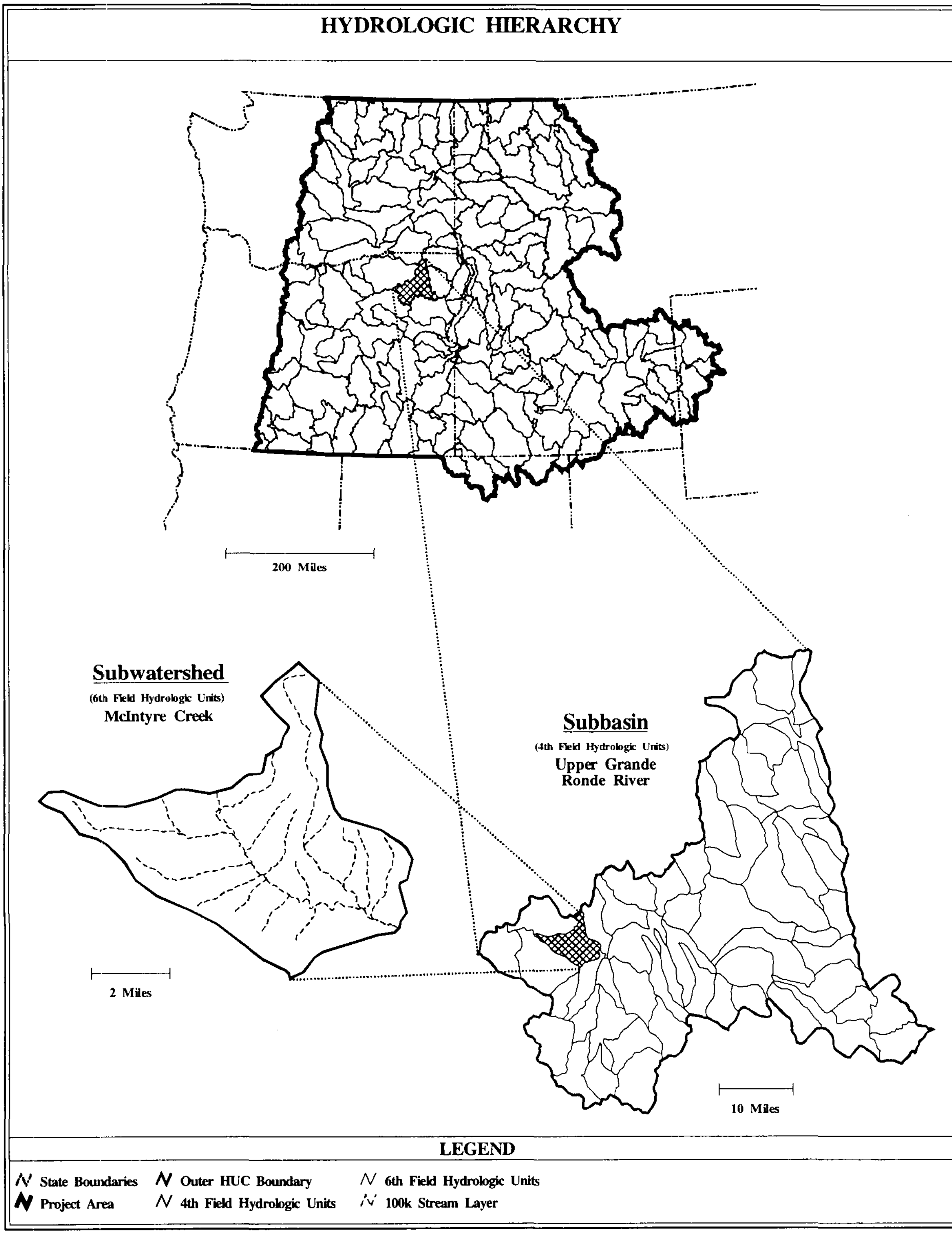
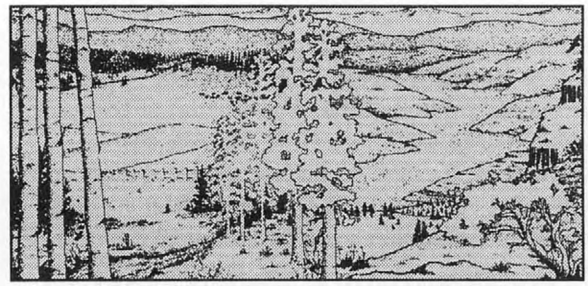


Figure 4.1— Hierarchical division of hydrological drainage areas. A) The entire assessment area (Basin) showing subbasins, with the Upper Grand Ronde subbasin shaded. B) The Upper Grand Ronde subbasin showing subwatersheds. C) The McIntyre Creek subwatershed, showing the 1:100,000 hydrography.

FACTORS INFLUENCING THE INTEGRITY OF AQUATIC ECOSYSTEMS



Geologic and geomorphic processes that formed and continue to affect the assessment area include tectonism, volcanism, glaciation, erosion, sediment transport, and deposition. The first three processes are not influenced by humans. The other processes are dominant forces shaping the landscape but may be influenced by human actions. All of these processes, in concert with the underlying physical environment, establish the template for and constrain the successional pathways for aquatic habitats and their associated communities. Human disturbances shape the status of the abiotic and biotic elements, but the physical setting and natural disturbances ultimately constrain the landscape's potential. In this section we consider the natural processes affecting landscapes and aquatic ecosystems within the Basin and discuss the effects of human disturbance.

Influence of Natural Processes on Aquatic Ecosystems

At the scale of watersheds and valley-bottoms, the physiography of the Columbia River basin is a relict of the Pleistocene (1.6 million - 10,000 years ago). At the local scale of channels and hill slopes, the physiography is influenced by disturbance processes that have occurred over the last few thousand years. Different histories (geologic, tectonic, glacial, recent) and different processes (hydrologic and sediment regimes) within and among subbasins and ERUs contribute to the diversity of aquatic habitats in the Basin (see Biophysical Environments chapter, this document).

During the last 4,000 years, volcanoes of the Cascade Range have erupted about twice per century. Effects of blasts, lava flows, floods, or other volcanic deposits are extensive. Long-term effects associated with erosion of tephra deposits (volcanic ash, pumice, and other debris) are apparent far from the volcanic source. For example, the pyroclastic surge of the 1980 eruption of Mount St. Helens leveled trees as far as 28 km from the crater and affected a 600 square kilometer area (Moore and Sisson 1981).

Probabilities of significant future volcanic episodes and tephra deposition are highest in the Northern Cascades and lowest in the eastern portions of the assessment area [Blue Mountains, Northern Glaciated Mountains, Lower and Upper Clark Fork, Owyhee Uplands, Central Idaho Mountains, Upper Snake, Snake Headwaters ERUs (Biophysical Environments chapter)]. Although the potential for ash deposits at a location in the western part of the assessment area in any single year is very small, it is virtually certain that within 200 years there will be a significant eruption within or adjacent to the assessment area. Actions taken to protect threatened or endangered species may have a longer-term positive impact if such timeframes and occurrences are considered at the onset.

Earthquakes also shape local physiography. Major landslides and permanent and temporary damming of rivers that are associated with such events can have significant effects on aquatic systems.

Earthquakes do not occur uniformly throughout the assessment area, and the areas of highest probability for major earthquakes are found in the Snake River Headwaters. The portion of the Upper Snake River located in Nevada, the Upper Clark Fork, Central Idaho Mountains, Northern Cascades, and Southern Cascades ERUs are also high risk earthquake areas (Biophysical Environments chapter). In these areas, there is a 10 percent probability that large seismic events could occur within a 50-year period.

During the Pleistocene, silt and fine-sand outwash from alpine and continental glaciers and glacial outburst floods were entrained by wind and redeposited as thick blankets of loess. These sequential layers of loess comprise much of the agricultural land in the Columbia Valley, Columbia Plateau, and Snake River Plain (Malde 1991). Because loess deposits are easily eroded by wind or water, large volumes of air and water-borne sediment are frequent byproducts of erosion. Although gravelly alluvial fan and valley-fill complexes in or near mountainous areas were also deposited during the Pleistocene, sediment production from hill slopes was much greater than at present. Because modern streams rarely mobilize the coarsest material that composes these fans, streams are usually constrained to movement within banks of fine-grained Holocene loess and alluvium. This is particularly true for the highly erosive Columbia Basin Plateau and Palouse area in eastern Washington.

Because of the size and complex topographic structure of the Basin, patterns of precipitation are not uniform. Distribution of vegetation in the area reflects the long-term precipitation patterns (Biophysical Environments chapter). For a given year, some areas of the Basin can be wetter than normal, while others are drier. Such anomalies suggest that future shifts in the seasonal weather system could affect portions of the Basin differently.

The hydrologic regime in the assessment area varies both spatially and temporally. Timing, duration, and magnitude of high and low flows, connection with groundwater, water quality and quantity, and temperature are influenced by the diverse physiography found among ERUs and within watersheds. Temporal variability of the hydrologic regime is due to weather and climate cycles as well as geomorphic factors such as channel deposition, changing channel width, and variation in the pattern of riparian vegetation.

Confined high-gradient streams are often shaped by rare large events that flush sediments relatively quickly downstream. The low gradient morphology of unconfined streams is generally more responsive to upstream disturbances and less-frequent disturbance events. The lowest gradient streams are the most responsive to gross channel morphological change from chronic disturbances (Biophysical Environments chapter).

Modern floods do not equal the magnitude of the Pleistocene floods, but they are important events nonetheless. Floods are naturally occurring events which vary in size and frequency depending on climatic factors and, to a lesser extent, on management practices. Wolman and Miller (1960) postulate that most flood-induced geomorphic changes are caused by intermediate (approximately two year recurrence intervals) events. The area affected by any single flood may be limited to a stream reach or extend to cover large river basins (such as the Columbia River basin in 1948).

The variability in climatic and geologic processes within the Basin has resulted in a complex variety of aquatic habitats. In turn, habitat heterogeneity can be key to the expression and maintenance of biological diversity in terrestrial and aquatic environments (Gresswell and others 1994; Schlosser 1991). The maintenance of habitat complexity in the Basin then becomes critical if we are to conserve the natural diversity of aquatic biota in the face of disturbance. Although climatic and geologic processes cannot be managed, human response to them can be planned, and in some cases, human disturbances might be modified to maintain desired habitat complexity in the context of natural disturbance regimes (Reeves and others 1995).

Influence of the Marine Environments on Freshwater Aquatic Ecosystems

Another significant factor beyond human control that affects aquatic species integrity is the natural fluctuations in the marine environment. In the last 15 years, the role of ocean conditions, which are driven by atmospheric circulation patterns, and their effects on anadromous salmonids have received increasing interest (Pearcy 1992). Traditionally, most salmonid research has focused on the freshwater environment and biologists attributed variation in population size solely to conditions in freshwater. Recent work strongly suggests that the abundance of anadromous salmonids and other fishes may be significantly affected by short- and long-term variation in atmospheric and oceanic circulation patterns (Francis and Sibley 1991; Ware and Thomson 1991). The evidence suggests that northeast Pacific Ocean conditions shifted in the mid-1970s, and anadromous salmonid populations along the entire West Coast of North America have responded to these large-scale changes (Francis and Sibley 1991; Pearcy 1992). The role of freshwater habitats must be considered in the context of larger-scale fluctuations in anadromous fish populations brought on by climate and oceanic conditions.

Cycles in marine productivity have the potential to mask the effects of degradation in freshwater habitats. Lawson (1993) presented a conceptual model for considering the combined effects of oceanic cycles and habitat degradation in freshwater. As freshwater habitats are degraded, populations of anadromous fishes do not decline in a linear fashion. Instead, a general downward trend is masked by long-term oscillations in ocean productivity. During periods of unfavorable ocean conditions, the consequences of degradation in freshwater habitats are most evident, and the risk of local extirpation is greatest. However, when periods of favorable ocean conditions are coupled with declining freshwater habitats, anadromous fish populations may appear to be stable or even increasing. Thus, favorable ocean conditions can lead to false beliefs of overall improvement in freshwater habitat quality. Similarly, haphazard restoration strategies may appear to be successful as population numbers increase, even though those increases are merely the fortuitous result of improving oceanic conditions. Once again, it is important to be aware of the larger context when assessing affects of management activities in a naturally fluctuating environment.

Long-term variations in ocean productivity also have a significant bearing on harvest and hatchery management. Harvest projections and limits are typically based on maximum sustained yield models that assume a constant environment. Because such models assume linear relationships between production and yield, they are particularly problematic in a changing environment or in one that is tending in a direction different from that in which the model was developed.

Similarly, the survival and production of hatchery-reared fishes may vary significantly with oceanic conditions (Pearcy 1992). In the 1950s and 1960s, coho salmon hatcheries in Oregon were enthusiastically endorsed by commercial fishers (who tripled in number over a 10-year period) and fishery managers because of early success that was largely the result of favorable oceanic conditions. When environmental conditions shifted in the mid-1970s, survival of hatchery-reared coho decreased, and the over-harvested fishery took an increasing toll on wild stocks (Pearcy 1992). The results were significant economic hardship for coastal fishing communities and precipitous declines in wild coho populations.

Although the amount of time that anadromous salmonids reside in freshwater habitat may be relatively short, condition of the habitat is critical, especially considering the natural fluctuations of conditions during their multi-year residence in the ocean. Freshwater habitats of high quality can ameliorate periods of poor ocean production, and it is becoming increasingly apparent that high-quality freshwater habitat is not a luxury but a necessity to the survival of anadromous salmonids. The oscillating conditions of ocean productivity and decline of Pacific salmon and steelhead emphasize the need for monitoring and restoring habitat elements, such as spawning and rearing habitat throughout the Columbia River basin.

Influence of Wildfire on Freshwater Aquatic Ecosystems

Wildfire has been a common agent of change in the assessment area since the Mesozoic (Cope and Chaloner 1985), and present aquatic systems have evolved in response to, and in concordance with, fire (Meyer and others 1992; Swanson 1981). The effects of fire on aquatic systems may be direct and immediate (for example, increased water temperature, chemical input) or indirect occurring over an extended period, but ultimately fire results in a natural mosaic of habitats and populations. The persistence of species in freshwater aquatic systems is linked to adaptation to periodic perturbations such as those resulting from fire (Warren and Liss 1980). In fact, the metapopulation concept is focused on the periodic loss of habitat patches (local extirpations) and subsequent re-invasion by individuals from neighboring patches (dispersal) (Hanski and Gilpin 1991). In an ecologically functioning stream network which provides sufficient stream connectivity for species refuge, reestablishment of fishes is generally rapid (Rieman and others, in press; Sedell and others 1990).

Long-term effects of fire usually result from erosion. Erosional processes potentially change channel morphology, sediment composition and concentration, food availability, and recruitment and distribution of large woody debris (Minshall and others 1990). The intensity and scale of these effects are related to the size and intensity of fire, geology, topography, and size of the stream system, and amount, intensity, and timing of subsequent precipitation events. Physical properties of soil that influence water retention are altered by heating, and in some cases, soils become water repellent after severe burns (McNab and others 1989). The amount of vegetation remaining in a watershed after a fire directly influences runoff and erosion by physically mediating the force of precipitation on soil surfaces, altering the evapotranspiration cycle, and providing soil stability through root systems. Runoff rate and pattern and subsequent erosion potential are directly affected by the amount of organic debris left in the watershed.

Revegetation of burned areas is influenced by the intensity and duration of a fire (Knight 1987), and the amount and type of new vegetation are related to changes in water yield and nutrient retention in the watershed. Erosional effects of fire generally peak within 10 years following the event (Brown 1989).

Influence of Human Activities on Aquatic Ecosystems

By the late 1800s, human activities had begun to alter the assessment area landscape, including the hydrologic function of rivers and streams and features that served as important habitat for aquatic life. By 1860 livestock grazing had reduced extensive willow coverage along many streams to scattered patches (Elmore and Kaufman 1994; Vavra and others 1994). Virtually every major tributary in the basin that was navigable (by canoe) had been altered by removing waterfalls, boulders, and log jams. Throughout the Basin the story was the same: sloughs and backwaters were isolated; pools were filled; log jams were cleared; and boulders were blasted. Clearing streams and rivers for passage of boats and milling of logs reduced the interaction of the stream system and flood-plain vegetation. Constructing drains, ditches, and dikes in valley bottoms and lowlands also reduced terrestrial-aquatic interaction. Flood control levees diminished or eliminated complex sloughs and side channels that were valuable rearing areas for salmonids and riparian dependent species. Human activities in many Idaho and Montana watersheds were extensive by the 1920s, and some areas (for example, Flathead Lake in the Northern Glaciated Mountains ERU) had been substantially altered by that time.

Marble Creek on the St. Joe River in Idaho provides one example of these activities. In 1911 there were numerous debris jams that had existed for many years (Blake 1971). In a 29-kilometer stretch ending at Homestead Creek, more than 1,180 cubic meters (500,000 board feet) of marketable timber were removed from the stream channel. An additional amount of wood was used to fuel steampowered equipment used in the logging operation. Fishing in the creek prior to 1911 had been described as exceptional, but Blake (1971) noted that large fish vanished after log drives.

In the early 1900s, the construction of large dams began as isolated areas of the Columbia River basin like the Central Idaho Mountains and the Wenatchee River system were gradually settled. Water withdrawals for irrigation were also developed early and rapidly. From the 1860s to 1930s dredges and hydraulic mining followed each new gold or silver discovery. Low-gradient reaches of streams were excavated, and whole valleys were transformed. Large reaches in low-gradient sections were inundated with fine sediments. Vast wetlands were filled and became farms and hay fields. Because many lakes were barren of fish, numerous species were introduced to provide recreational fishing.

In short, the ecological integrity of streams, lakes, and wetlands was significantly compromised by the 1920s. Increasing human population, technological advances (for example, centrifugal pumps), and availability of heavy equipment after World War II greatly accelerated the development of new irrigation projects, timber harvest, dam construction, and road building. Individually and in combination, these activities continued to fragment and compromise the remaining hydrologically connected and vegetated reaches of streams.

Changes in Extent and Type of Riparian Vegetation in the Basin

To determine riparian vegetation changes in the Basin, we conducted an analysis at the mid-scale (subbasins). Our approach was to determine current condition of riparian vegetation and changes in vegetation type and cover from historical potential as well as from the past 40 to 60 years. We used methods appropriate to both the extensive spatial and temporal scales and the limited resolution of the data.

The analysis focused on an analysis of trends in physiognomic types, major vegetation cover types, and major vegetation structural stages. This riparian zone vegetation characterization and analysis of trends are based on the midscale vegetation analysis of the upland vegetation as described in Hessburg and others (1995). Within the assessment area, the 164 subbasins were stratified using broad biophysical environment settings into 16 strata. Using this stratification, a random sample of 43 subbasins was selected for further subsampling. Mid-scale subwatershed characterizations were used to describe changes in subbasins representative of the Basin. Aerial photography was used to compare recent historical conditions (early 1930s to mid-1950s) to current (mid-1980s to present) conditions. Vegetation patches were defined using multiple attributes of vertical and horizontal structure and composition. A complete list of the photo-interpreted attributes can be found in Hessburg and others (1995). The existing vegetation attribute was characterized using cover type classes of forest, woodland, shrubland, herbland, and non-forest/non-range. Significant change ($p \leq 0.20$) from recent historical to current conditions was determined by examining the 80 percent confidence interval around the mean difference for the subbasin. More details of the analysis procedure of the mid-scale vegetation data are outlined in Hessburg and others (1995).

The mid-scale aerial photography used in the vegetation analysis, and the 7.5 minute U.S. Geological Survey (USGS) quadrangle maps (1:24,000 scale) used to map and characterize valley bottom settings and stream channel segments, were used to analyze the riparian vegetation at the mid-scale. The valley bottom setting and channel segment maps were overlaid on the mid-scale vegetation maps to produce a coverage of riparian zone vegetation. The riparian sample area consisted of the strip of vegetation, 20 to 30 meters in width, adjacent to all surface hydrography (that is, around the perimeters of lakes and wetlands greater than 0.5 hectares in area, and on both sides of all mapped perennial and intermittent stream channels).

Not all vegetation cover types and structural stages in this zone are wetland vegetation types; some are more typically associated with upland vegetation.

The analysis was based on sample areas within the Basin (and ERUs). Sample sites have a higher proportion of Forest Service (FS) and Bureau of Land Management (BLM) lands than private, tribal, or other Federal lands as compared to the proportions within the entire Basin. We looked at cover types, structure, and physiognomic types in a comparison of recent historical to current levels. This section highlights some of those attributes that directly influence the aquatic environment and the ERUs where the most significant changes occurred.

The data indicate that many areas in the Basin are showing a reduction in the large tree component in riparian zones. This can affect the amount of shading provided to streams and the potential large woody debris recruitment. Those ERUs showing a loss of large trees are the Northern Cascades, Upper Klamath, Columbia Plateau, Blue Mountains, and Snake Headwaters. Some of these (primarily the Columbia Plateau and the Upper Klamath) did not have a large percentage of large trees either historically or currently but have shown significant losses. ERUs showing some increase in the large tree component of the riparian zone are Southern Cascades, Northern Glaciated Mountains, and Lower Clark Fork. The increase in the large tree component appears to be inconsistent with those results found at the broad scale for the Southern Cascades and Northern Glaciated Mountains ERUs. This inconsistency is most likely attributed to the difference in land ownership at the broad scale (all land ownership) and the ownership at the mid scale where sample areas were predominantly Forest Service- and BLM-administered lands.

In an analysis of mid-scale vegetation classification, significant changes occurred in most ERUs (table 4.3). There was a decline in shrublands in the riparian zones in more than half of the ERUs. Shrublands predominantly shifted to forests and herblands, through succession or disturbance.

The mid-scale results in shrub loss are consistent with the broadscale results with the exception of two ERUs. Both the Lower Clark Fork and Upper Klamath ERUs showed a significant loss in riparian shrub communities in the broad-scale analysis. In the Upper Klamath, those losses occurred primarily on private lands. This was not the case in the Lower Clark Fork. It is likely that the difference in time period played a role in the inconsistent results. The broad scale compared presettlement conditions to current, while the mid-scale analysis looked at changes over the last 40 to 60 years. The more significant losses may have occurred prior to the aerial photography.

Forests (which include cottonwood, aspen, and willow, typically riparian associated species), woodlands and herblands increased in area or stayed approximately the same. There was a decrease in six of the thirteen ERUs of these cover types, with significant decreases in the Snake Headwaters and Columbia Plateau. Significant increases in woodlands occurred in five ERUs. In the Northern Great Basin, Blue Mountains, and Columbia Plateau this increase is attributed to the conversion of shrubland to juniper stands. Nonforest cover types (agriculture, urban, rock, and water) stayed the same in most ERUs with some increase in the Northern Cascades, Upper Clark Fork, and Upper Snake ERUs. The findings in the sample areas at the mid-scale level are similar to those reported by others [McIntosh and others 1994a; U.S. Department of Agriculture (USDA), Forest Service 1993; Wissmar and others 1994b].

Influence of Human Activities on Water Quality

The extent and intensity of land development and land-use activities within the Basin have increased during the past century. Environmental disturbances from non-mechanized, agriculturally based settlements have evolved into perturbations associated with urban and suburban development, industrialization, and intensive large-scale and mechanized agricultural practices. Even areas that have been developed solely for recreational use

Table 4.3— Mid-scale (1:20,000) riparian vegetation changes by Ecological Reporting Unit. Only statistically significant changes are shown ($p \leq 0.20$).

Ecological Reporting Unit	Number of Subsampled Subwatersheds	Forest	Woodland	Herbland	Shrubland
Blue Mountains	46	+	+	*	-
Central Idaho Mountains	43	*	*	+	-
Columbia Plateau	38	+	+	*	-
Lower Clark Fork	5	+	*	*	*
Northern Cascades	48	*	+	*	-
Northern Glaciated Mountains	41	*	*	*	*
Northern Great Basin	4	+	+	+	-
Owyhee Uplands	23	*	+	+	-
Snake Headwaters	16	*	*	*	-
Southern Cascades	16	*	*	+	*
Upper Clark Fork	32	*	*	*	*
Upper Klamath	14	*	*	*	*
Upper Snake	15	*	*	*	-

+ = significant increase from historic condition

- = significant decrease from historic condition

* = no statistically significant change

have not escaped degradation. Aquatic ecosystem perturbations related to these activities include: 1) thermal pollution; 2) toxicity due to the presence of organic compounds (synthetic and natural) and heavy metal ions; 3) introduction of pathogenic organisms; 4) organic wastes that result in potentially catastrophic changes in dissolved oxygen levels; 5) acidification; 6) elevated sedimentation rates; and 7) increased eutrophication (Ellis 1989).

Eutrophication is indicative of deteriorating water quality associated with a buildup of nutrients, especially nitrogen and phosphorus. Increased rates of nutrient loading can be related to changes and/or disturbances within a watershed (Brugam and Vallarino 1989; Dojlido and Best 1993; Stauffer 1991). Development activities that contribute to increased nutrient levels include point sources such as industrial effluents and waterborne sewage systems and nonpoint sources such as agricultural operations, residential development and septic systems, road construction, and forest practices. (Dojlido and Best 1993; Spencer 1991; Thralls 1991).

Nonpoint source pollution may be the most problematic cause of water quality deterioration because the origin of perturbation is often difficult to identify and control. Residential development (often a response to the recreational potential of the area) around or near lakes, reservoirs, and wetlands is directly associated with much of this nonpoint source pollution. Analysis of lakes in the Basin (appendix 4A) concluded that lakes in the Columbia Plateau, Northern Great Basin, and Upper Klamath ERUs have very high total phosphorus values. Although the phosphorus values may be affected by volcanically influenced soils, anthropogenic activities may also influence lake productivity. Development can also result in increases of nitrogen and phosphorus in surface waters resulting from septic system effluents (Scott 1991; Sorrie 1994; Stauffer 1991), runoff from fertilized lawns and agricultural lands (Lewis and others 1984; Power and Schepers 1989), and runoff from highways and roads (Ehrenfeld and Schneider 1991; Lewis and others 1984).

Nonpoint source pollution is extensive in the assessment area. In Montana, for example, nonpoint source pollution caused by anthropogenic land use is the major cause of declining groundwater and surface water quality (Thralls 1991). Spencer (1991) found that settlement and subsequent economic development (timber harvest, forest products production, and road and railroad construction) of Montana's Flathead Basin increased rates of sediment deposition in three lakes. Even in remote areas such as Oregon's Odell Lake, inputs of nutrients from septic systems associated with a resort, nearby campgrounds, and summer homes built along the lake shoreline seriously affect water quality in the lake (Odell Pilot Watershed Analysis 1994). Moderate eutrophication of Pine and Otter lakes in western Washington appeared to coincide with the construction of recreational cabins and suburban homes around the lakes (Brugam 1987; Brugam and Vallarino 1989).

Analysis of lakes in the Basin (appendix 4A) reported the potential effects of land-use activities on lakes in the assessment area (table 4.4). Lakes in the eastern part of the Northern Glaciated Mountains and Lower Clark Fork ERUs may be the most sensitive to disturbance. These lakes have low levels of total phosphorous and are commonly located in terrain that is more accessible to development and recreation. Lakes in the Northern Cascades, Southern Cascades, Upper Clark Fork, Central Idaho Mountains, and Blue Mountains also had low levels of total phosphorus, and are generally sensitive to nutrient additions from any source. Lakes that are most sensitive to atmospheric pollution sources (low-alkalinity lakes) are located in the Central Idaho Mountains, upper Clark Fork, Blue Mountains, Northern Cascades, and Southern Cascades ERUs. Because these lakes are generally sensitive to nutrient additions from any source, natural or prescribed forest fire, particularly low intensity spring and fall burns that release large quantities of smoke and ash and deliver nutrients to surface waters, may affect water quality.

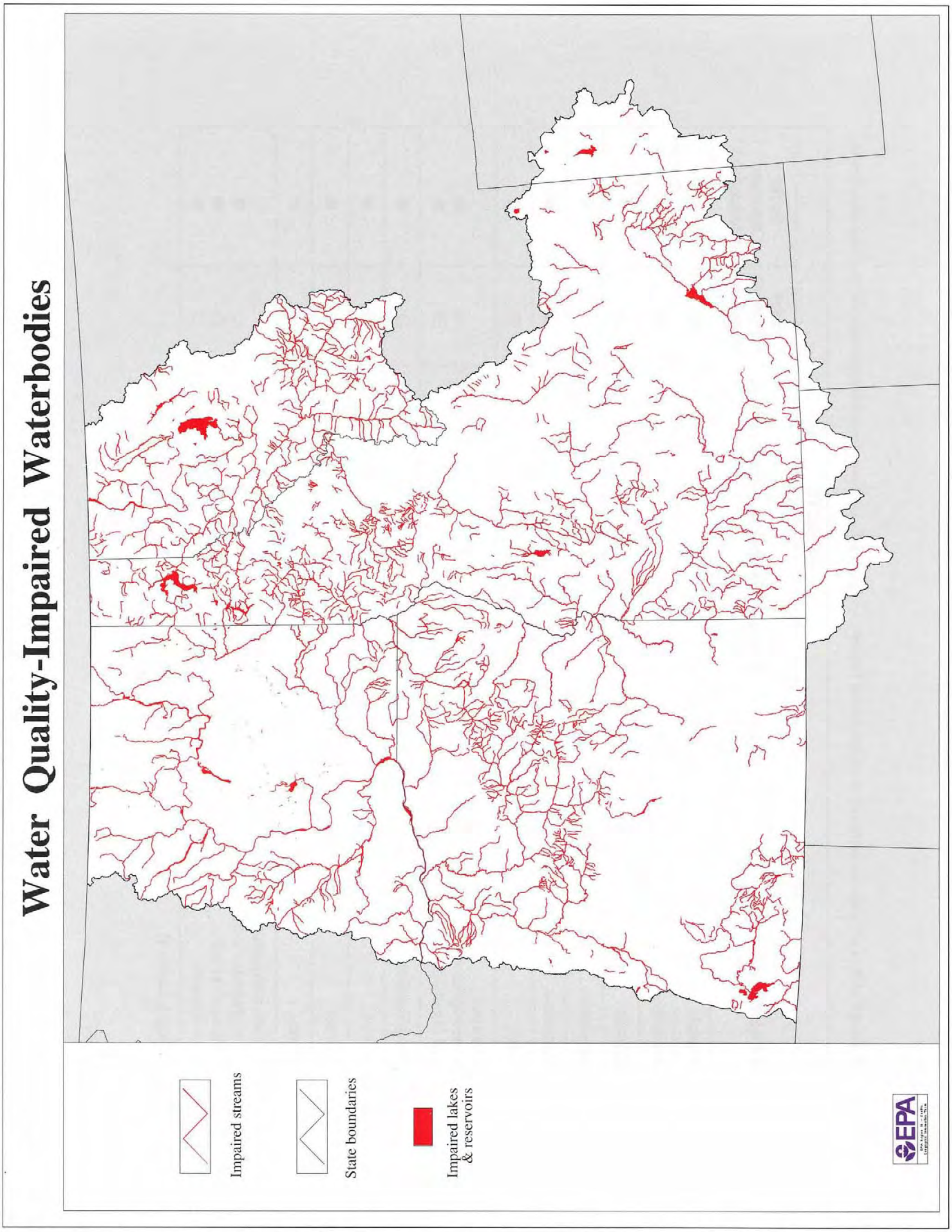
Overall water quality impairment (42,271 stream km; map 4.2) estimated by the Environmental Protection Agency (EPA) appears to be modest in comparison to total length of streams within the assessment area (428,500 stream km; table 4.5). Water temperature, turbidity and sedimentation, nutrients, and streamflow alteration were the most common causes of water-quality impairments reported, but the predominant factors varied among ERUs (fig. 4.2). Because these estimates are based on existing and accessible data from locally specific state and federal monitoring programs, they likely underestimate the real extent and distribution of impairment.

Results of a recent assessment of water temperature extending from the Canadian border to the Oregon/Nevada border also identified areas where conditions have changed substantially from historical condition; however, changes were not uniform across the study area (fig. 4.3; USDA Forest Service 1993). Streams in the Northern Glaciated Mountains and Northern Cascades ERUs displayed little change, but in the Blue Mountains, Southern Cascades, Northern Great Basin, and Upper Klamath ERUs, stream temperatures were elevated and often exceeded state water quality standards for temperature. Because many of the streams with elevated temperatures were not identified by Environmental Protection Agency assessment reports, it appears that water quality concerns within the Basin may be more severe than previously described.

Table 4.4— Potential effects of land-use activities on lakes in the Interior Columbia Basin Ecosystem Management Project assessment area.

	IMPACTS						
Ecological Reporting Unit	Timber Production	Grazing	Mining	Recreation	Development	Water Transfers/ Irrigation	Exotic Species Introductions
Columbia Plateau	NA	●	○	○	○	●	●
Northern Great Basin	NA	●	○	○	○	●	●
Owyhee Uplands	NA	●	○	○	○	●	●
Northern Cascades	●	○	○	❑	❑	○	●
Southern Cascades	●	○	○	❑	❑	○	●
Upper Klamath	●	●	○	○	○	●	●
Snake Headwaters							
Yellowstone Highlands	❑	❑	○	❑	○	○	●
Overthrust Mountains	●	❑	❑	❑	○	○	●
Central Idaho Mountains	●	❑	❑	❑	○	❑	●
Upper Clark Fork	●	❑	❑	❑	○	❑	●
Lower Clark Fork	●	❑	❑	❑	❑	○	●
Blue Mountains	●	❑	❑	❑	○	❑	●
Northern Glaciated Mountains							
Okanogan Highlands - west	●	❑	❑	○	○	○	●
Flathead Valley - central	●	❑	❑	❑	❑	○	●
N. Rockies - eastern	●	❑	❑	❑	○	○	●

Slight - ● (or highly localized)
Moderate - ❑
Major - ○
Insufficient sample - NA



Map 4.2— Impaired water bodies within the ICBEMP assessment area WQ1 reported by the Environmental Protection Agency. Nearly 42,377 km of rivers, streams, and lake shore lines are shown. Impairment status derived from the 1994 State/EPA water body lists required under Section 303(d) of the Clean Water Act (CWA), except for Oregon and Washington. Sec. 303(d) waterbodies are those for which existing pollution controls are deemed inadequate to provide for attainment of State water quality standards & designated beneficial uses. Oregon waterbodies shown are from the States' draft 1994/1996 CWA 303(d) list, Dec. 1995. This list is subject to revision based upon results of public comment, proposed modifications to State water quality standard, and EPA approval of the State's final list. Washington data developed under Section 305(b) were used in lieu of the 303(d) list because impairment data for more waterbodies were available, in GIS-compatible format. These waterbodies are not fully supporting designated beneficial uses.

Table 4.5— Water quality impaired waters reported by the States[1] and the Environmental Protection Agency as kilometers of streams and rivers in the portions of States within the Interior Columbia Basin Ecosystem Management Project assessment area.

	Total Stream Kilometers	Any Impairment	Temperature Impairment	Nutrient Impairment	Sediment/ Siltation/ Turbidity Impairment	Flow Impairment
Oregon (1994/96)	121,000	13,072	11,842	450	1,525	2,032
Forest Service		4,068	3,965	0	515	215
BLM		1,040	917	79	105	126
Washington (305b)	79,100	6,376	4,530	53	No Data	4,106
Forest Service		930	328	0	No Data	717
BLM		42	22	0	No Data	26
Idaho (303d)	159,300	16,132	4,235	5,566	14,182	4,368
Forest Service		4,828	733	493	4,133	477
BLM		2,172	826	630	1,911	815
Montana (303d)	50,400	6,295	1,692	1,831	4,883	2,883
Forest Service		2,188	436	159	1,694	894
BLM		112	32	16	90	73
Wyoming (303d)	7,700	115	0	0	74	0
Forest Service		63	0	0	56	0
BLM		4	0	0	4	0
Nevada (303d)	11,000	281	259	22	0	0
Forest Service		18	2	15	0	0
BLM		77	77	0	0	0
TOTALS						
All States	**428,500**	**42,271**	**22,558**	**7,922**	**20,664**	**13,389**
Forest Service		**12,095**	**5,464**	**667**	**6,398**	**2,303**
BLM		**3,447**	**1,874**	**725**	**2,110**	**1,040**

[1]The Utah Department of Environmental Quality reports no impaired streams or rivers within the project area in Utah.

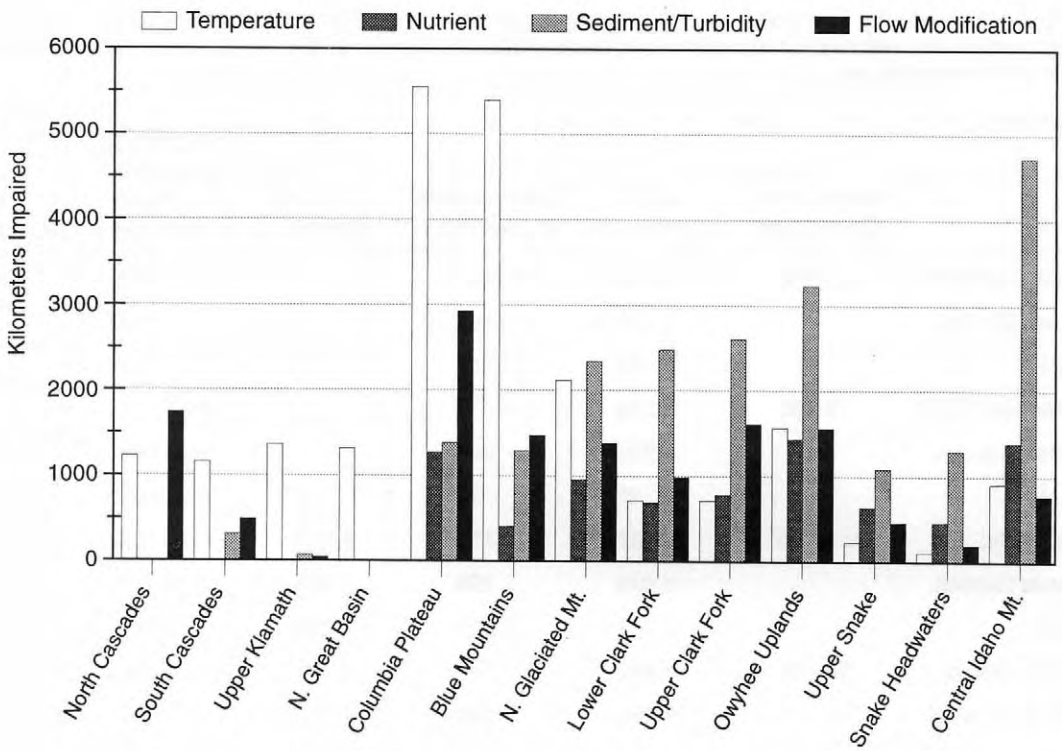


Figure 4.2— Major types of water quality impairment by Ecological Reporting Units within the Basin. Sediment and turbidity were not reported for Washington which lowers the estimate for this type in the Columbia Plateau and Northern Cascades and northern Blue Mountains. Note many streams are reported for multiple impairment types.

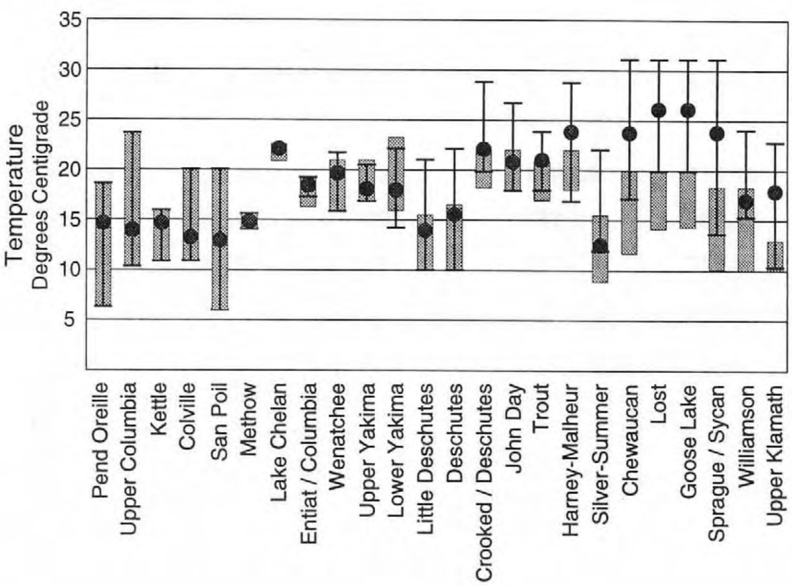


Figure 4.3— Historical range (shaded bars), current range (bars), and current mode (•) of water temperature for streams east of the Cascade Mountains in Washington and Oregon (USDA Forest Service 1993).

Influence of Dams, Water Storage and Withdrawals

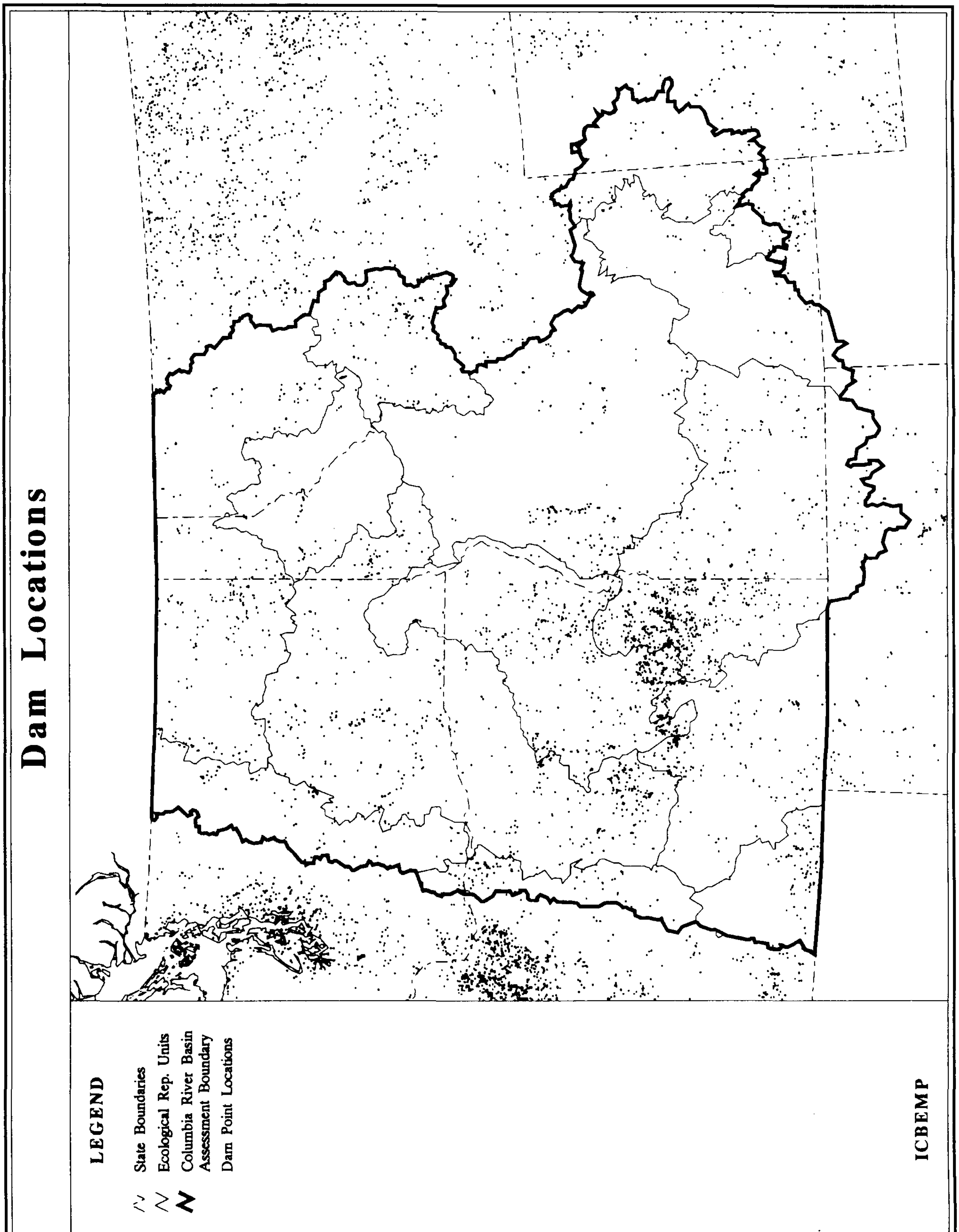
One of the most significant changes in the assessment area is associated with efforts to store, control, and direct water. Thousands of dams, ranging from tiny stock ponds in the driest regions to the largest hydroelectric facilities on the Columbia River (Grand Coulee Dam with a storage capacity of 6.4 billion cubic meters), are presently found in the area. Of these, there are at least 1,239 dams with storage capacity in excess of 62,000 cubic meters within the assessment area. Because federal inventory and inspection are required only for the larger dams (greater than two meters in height with storage capacity greater than 62,000 cubic meters, or greater than 7.6 meters in height with storage capacity greater than 18,500 cubic meters) and those with downstream hazard potential, the total number of dams is unknown. Many states record only those dams required for federal inventory. The actual number of dams in the assessment area could be several times greater than those reported because 74 percent of 1,600 dams identified in eastern Oregon have less than 62,000 cubic meter capacities (map 4.3; information compiled from state inventory of dams databases).

Construction of dams in the Columbia River basin has greatly reduced the accessible range of anadromous fishes and has interrupted migrating patterns of migratory forms of non-anadromous fishes. Spawning and rearing areas for anadromous fishes in the upper Columbia River basin were isolated after the Grand Coulee (1941) and Chief Joseph (1955) dams were completed. Since 1967, Hells Canyon Dam has blocked anadromous fish access to the Snake River and tributaries above the dam. A similar loss of most spawning and rearing habitats followed the construction of Cabinet Gorge Dam on the Clark Fork River above Lake Pend Oreille (Pratt and Huston 1993). Such problems are not limited to large dams. The National Research Council (1995) analyzed data from the state databases in Oregon, Washington, Idaho, and California and found that most small dams do not have fish passage facilities. The extent to which these dams impede migration or affect spawning and rearing habitats of fishes has not been documented.

Even with fish passage facilities, detrimental effects from dams occur as a result of direct mortality of juveniles in turbines and bypass systems. Indirect mortality is caused by delays in migration of adults and juveniles as a result of inability to find routes around dams, slack water, physiological stress, and increased susceptibility to predators. Efforts to mitigate the effects of dams by barging and trucking fish around dams also are not without problems. Collecting, handling, and transporting juvenile chinook salmon past McNary Dam on the Columbia River resulted in cumulative levels of stress, including such physiological parameters as decreases in white blood cell counts, osmoregulatory ability, and swimming performance (Maule and others 1988). Even for those fish that successfully circumvent dams, the combination of physiological stress, confusion, and discharge by tubes into restricted areas below dams renders them vulnerable to predation. From 1983 to 1986, mortality of juvenile salmonids due to predation by squawfish, walleye, and smallmouth bass in John Day Reservoir averaged 2.7 million, or approximately 14 percent of all juvenile salmonids that successfully passed through or around McNary Dam (Rieman and others 1991).

Trends in the number of dams constructed over time (fig. 4.4) and impounded water volumes (fig. 4.5) indicate that many streams and rivers have experienced a rapid and massive change in their hydrology. Even though the rate of increase in storage volume has leveled since the mid-1970s, the total number of dams continues to increase, suggesting that new construction is focused on smaller dams (National Research Council 1995).

The combined storage capacity of all dams for which there is information in state databases exceeds 79 billion cubic meters of water. Although many dams have multiple functions, irrigation (the most common use) accounts for 48 percent of dams and 54 percent of total storage capacity. Recreational use is secondary (19% of all dams). Hydropower accounts for only 6 percent of all dams, but 66 percent of total storage capacity is associated with power generation. Figure 4.6 shows the different proportions of dams uses in the assessment area.



Map 4.3—Locations of dams for which information is maintained by the States (2,103 dams within the Basin). Data were provided by state Dam Inventory databases (Montana and Nevada data was provided by the US Army Corps of Engineers National Inventory of Dams). Data quality and completeness varies by state.

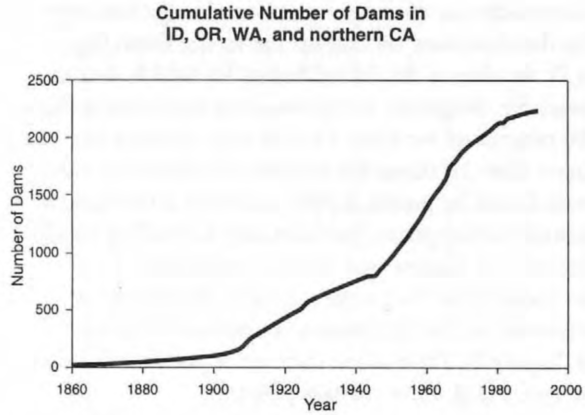


Figure 4.4— Cumulative number of federal and nonfederal dams in the Pacific Northwest (Idaho, Oregon, Washington, and northern California) from 1860 to 1990. (Data from individual State water-resources agencies; minimum size of dams varies by state).

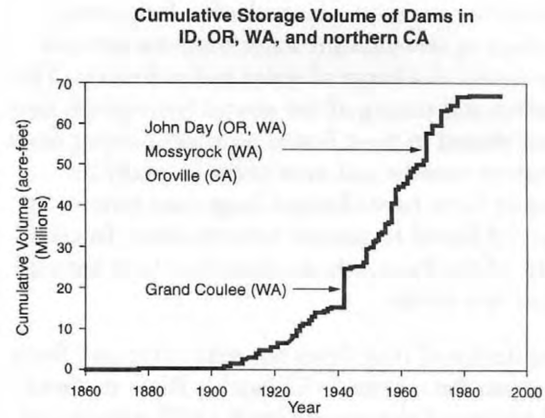


Figure 4.5— Cumulative volume of water impounded by federal and nonfederal dams in Pacific Northwest (Idaho, Oregon, Washington, and northern California) from 1860 to 1990. (Data from individual State water-resources agencies; minimum size of dams varies by state).

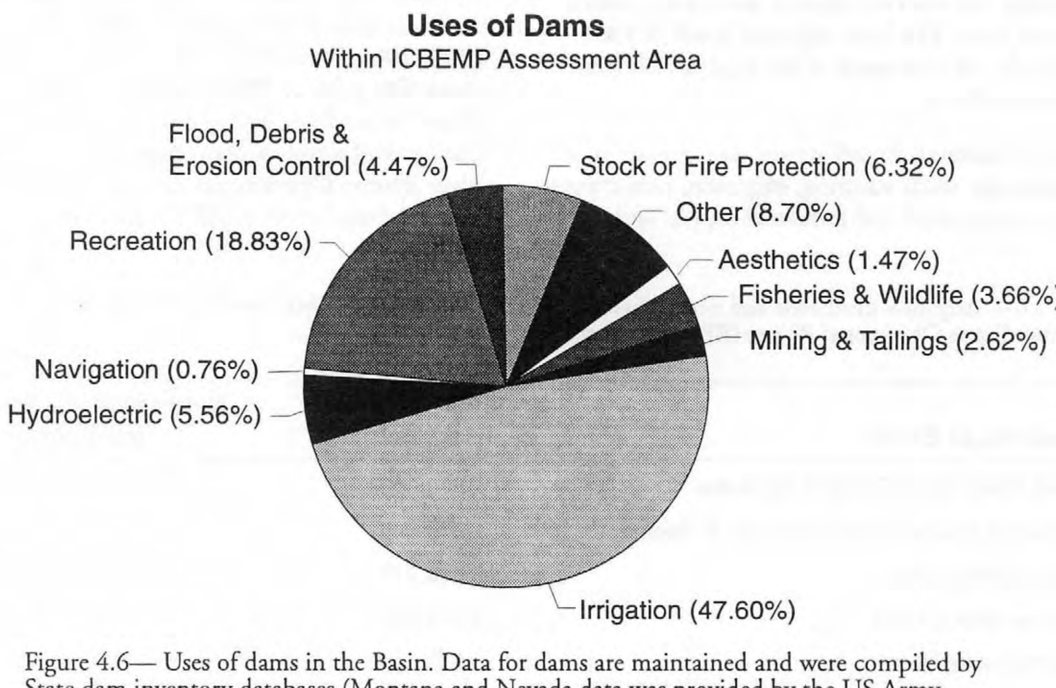


Figure 4.6— Uses of dams in the Basin. Data for dams are maintained and were compiled by State dam inventory databases (Montana and Nevada data was provided by the US Army Corps of Engineers National Inventory of Dams). Not all dam records (2,103) contained "type of use" information. Data quality and completeness varied by State.

Reservoir operation has resulted in long-term changes in downstream water temperatures and the annual discharge of water and sediments. The pattern and timing of the annual hydrograph have been altered in most basins on scales ranging from hours to months and even years. In many instances dams have changed large river systems to isolated fluvial fragments between lakes. In arid parts of the Basin, stream diversions have reduced flows to a trickle.

Regulation of river flows has reduced spring flood flows on the mainstem Columbia River to about 50 percent of the natural levels and has increased fall minimum flows by 10 to 50 percent (Sherwood and others 1990). Prior to regulation, major spring floods (for example, the flood of 1948) occurred when snow melted rapidly and simultaneously throughout the eastern portions of the Basin (Paulson 1949). Although recent spring floods (in 1972 and 1974) produced comparable discharge, the reservoir system prevented natural overland flow. The large regional flood of 1964-1965 only affected parts of the mid- and lower Columbia Basin.

Water withdrawals for off-stream uses include rural domestic use, stock watering, irrigation, public water supply, commercial and industrial supply, and thermoelectric cooling. Agricultural irrigation is by far the dominant off-stream use in the Basin (fig. 4.7). In nine of the 14 subbasins for which data are available, irrigation withdrawals represent more than 80 percent of the total. Overall they account for more than 10 times the combined volumes of water withdrawn by public supply, industry, and thermoelectric power plants (Jackson and Kimerling 1993). Projections for the year 2040 for withdrawal amounts from municipal and irrigation users are discussed in the Economics Assessment chapter (Chapter 6). Projections indicate significant increases in municipal water consumption.

The actual amount of water withdrawn in the Basin is not known because of the large number of irrigation projects and difficulties in assessing return-flow. Estimates for the Columbia River during the early 1980s ranged from seven to 10 percent of the mean annual discharge (Sherwood and others 1990). Simenstad and others (1992) calculated net water consumption in the Columbia River system to be greater than 144 m^3/s in 1900, 200 m^3/s in 1910, 310 m^3/s in 1928, and about 680 m^3/s in 1980. Table 4.6 gives irrigation diversions and net depletions for the 1990-1991 base level of development, from the Columbia River System Operational Review [Bonneville Power Administration (BPA) and others 1995].

Table 4.6— Irrigation diversions and net depletions by hydrologic region. Table reproduced from the *1995 Columbia River System Operational Review* (BPA and others 1995).[1]

Hydrologic Basin	Irrigation Diversion (Hectares)	Net Irrigation Depletion (Hectares)
Clark Fork - Pend Oreille & Spokane	520,849	311,052
Columbia Plateau, East Cascade, & Yakima	2,279,420	1,386,118
Upper Snake River	5,813,718	1,886,331
Central Snake River	3,053,696	1,061,739
Lower Snake River	343,594	215,903
Mid-Columbia	952,101	540,242
Total	12,963,378	5,401,385

[1]Source: USBR/BPA, Columbia River Basin, System Operational Review. Irrigation Depletion Estimate. September 10, 1993, prepared for Bonneville Power Administration by A.G. Crook Company.

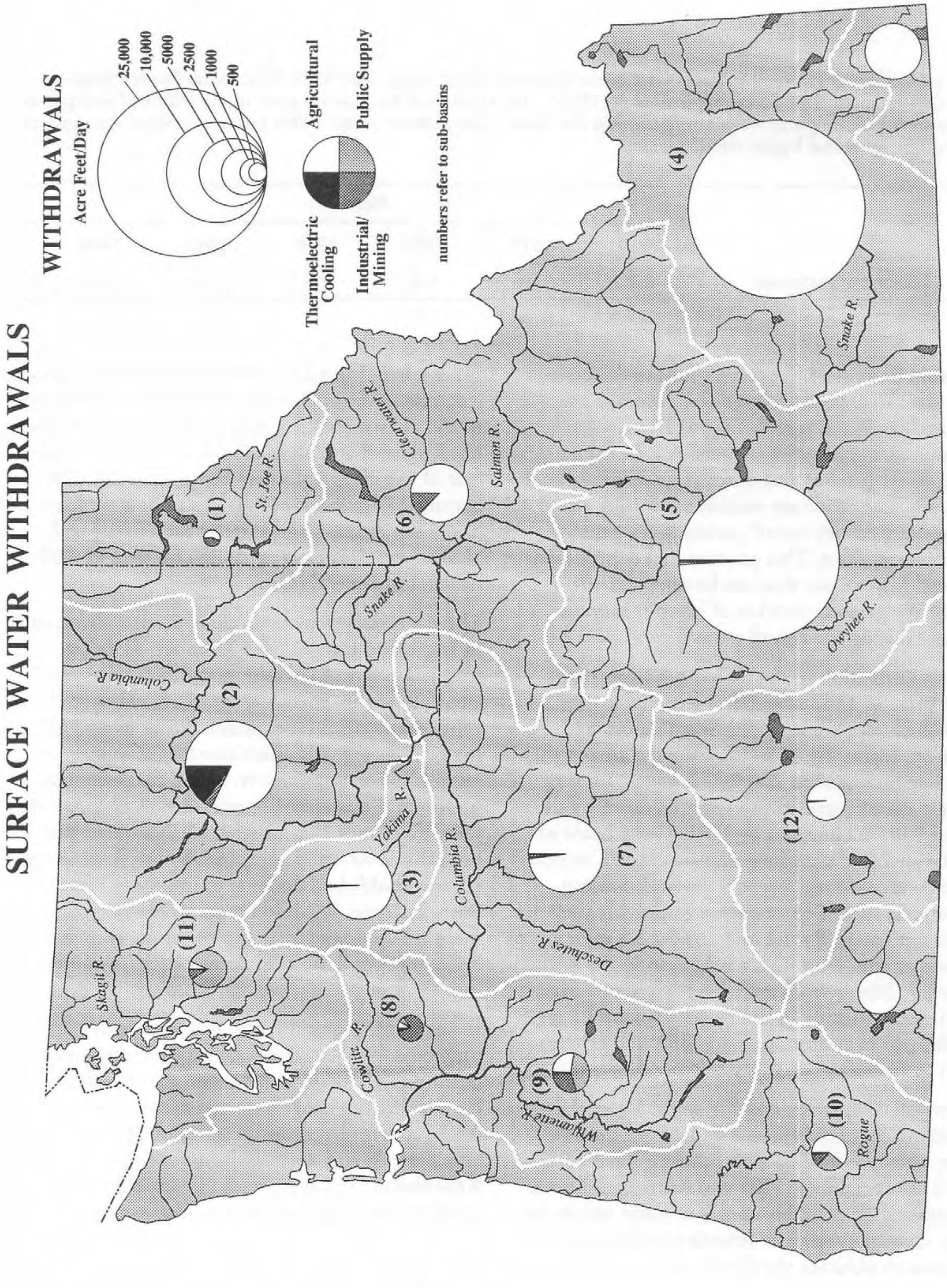


Figure 4.7— Spatial variation of four primary offstream uses of water in the region. From A. Jon Kimerling, Oregon State University, Corvallis, Oregon.

Table 4.7— Changes in total irrigated land in the Columbia River Basin since 1900 (Northwest Power Planning Council-Depletions Task Force, September 30,1993). The increase in hectares irrigated in the 1960s reflects greater groundwater pumping and water storage within the Basin. The decrease in the 1980s probably reflects the response to a drought cycle and higher electrical costs.

	Year					
	1900	1910	1928	1966	1980	1990
Millions of Hectares	0.2	0.9	1.5	2.6	3.0	2.8

Irrigation patterns differ throughout the assessment area. In areas with large water storage reservoirs such as Washington's Yakima River, diversions are greatest in the summer. In the Lemhi River (Idaho) and other areas that use flood irrigation, diversions are maximized during the spring and summer runoff period in the attempt to saturate aquifers. This practice makes it difficult to install fish screens that can be operated efficiently; the greatest number of juvenile salmon migrates during the runoff period.

Intensive irrigation in the Columbia River basin began prior to 1840 at mission settlement sites established near Walla Walla, Washington, and Lewiston, Idaho. By 1890 the irrigated area in the basin had increased to 200,000 hectares. During the next decade, irrigation expanded rapidly and totaled 930,000 hectares by 1910. Great tracts of land located in the arid region east of the Cascade Mountains could not support dryland farming, and farmers turned to irrigation to provide their crop water needs. By the mid 1960s, 2.6 million hectares of land were under irrigation in the Basin (table 4.7; Northwest Power Planning Council 1993). Most streams in the Pacific Northwest are now fully or over-appropriated (BPA and others 1995).

In many areas (particularly the Snake River Plain) irrigation influenced springs return significantly cooler water to streams. This return groundwater flow is beneficial to wildlife and aquatic resources, and typifies the complex nature of water use in the Basin. Cool groundwater returns could be used in arid areas to enhance the distribution of cold water fishes where instream habitat characteristics (pools, instream cover, and under-cut banks) are adequate. In the assessment area, hundreds of reservoirs have resulted from irrigation efforts that put the equivalent of 1.2 meters of water per year on 202,000 hectares. Thousands of hectares of new wetlands have been created, and interest on wildlife and aquatic recreation has expanded on over 81,000 hectares of Federal land.

There are numerous small gravity water diversions in Idaho and Oregon, and although there are fewer diversions in Washington, they often are larger. In Idaho, there are approximately 455 water diversions in the Salmon River basin.[1] Of these, 278 currently affect summer chinook salmon in the Snake River, which are protected under the Endangered Species Act. About 270 of these diversions currently have fish screens, but only about 60 are in compliance with fish passage criteria established under the Columbia River Basin Fish and Wildlife Program. Additionally, there are approximately 250 pump-intakes in the Salmon River Basin that do not have fish protection devices.

In the upper Salmon River, fish are adequately protected from larger diversions on migration corridors, but many small tributaries where spawning and rearing occur have unscreened diversions. These streams historically provided habitat to anadromous and non-anadromous fishes. Keifenheim (1992) indicated that salmon and steelhead no longer use many small streams in the

[1]Personal communication. 1995. Clayton Hawkes, Columbia Basin Fish and Wildlife Authority, Portland, Oregon.

upper Salmon River area because of irrigation withdrawals. Currently instream flow reductions in these streams may result in migration barriers, substantially diminished spawning and rearing habitat, or poor water quality. Although most of these small streams were not likely to have produced large numbers of fish historically, the total loss of spawning and rearing habitat may be significant.

In the current anadromous fish production areas of Oregon above Bonneville Dam, there are approximately 550 water diversions, most of which have fish screens (see footnote 1). However, most (80%) of these screens are several decades old and do not meet current criteria to adequately protect juvenile fish. There are about 55 pump-intakes on the Oregon side of the mainstem Columbia River, most of which were to be screened by the beginning of the 1996 irrigation season, and about 140 screened pump-intakes occur on Columbia River tributaries. Surveys are underway to determine the need for additional pump-intake screens in northeast Oregon.

On the mainstem Columbia and Snake rivers in Washington, there are about 200 pump diversions, most of which were to be screened by the beginning of the 1996 irrigation season. There are approximately 150 gravity diversion fish screens above Bonneville Dam; flows on several of these diversions exceed 28 cubic meters/second. There also are approximately 690 pump-intakes on tributaries in this portion of the Basin, but only about one-fourth have fish screens that currently are in compliance.

Most irrigation diversions on Forest Service and BLM-administered lands are operated by private individuals, but a few of the water rights are held by federal agencies. In subbasins under the jurisdiction of the Wallowa-Whitman National Forest, which include the Hells Canyon National Recreational Area, there are approximately 10 diversions that affect anadromous fish and seven that primarily affect salmonids migrating within fresh waters (Mattson 1993). Less than 10 diversions affecting anadromous fish have been identified on Forest Service lands in mid-Columbia tributaries in Washington.

Irrigation has contributed to the extirpation of salmon and steelhead from many small streams in the Salmon National Forest (Keifenheim 1992). Many streams in the Sawtooth National Recreation Area have inadequate instream flow as a result of irrigation.[2] Seventy-five irrigation diversions occur on Federal lands within the Salmon National Forest, 156 on Federal lands within the Sawtooth National Recreation Area, and 37 on nearby BLM-administered lands. The cumulative loss of spawning and rearing habitat in these tributaries is significant.

Influence of Farming and Grazing

The proportions of land in the Pacific Northwest dedicated to agriculture is relatively small (approximately 16%; see Landscape Dynamics, (Chapter 3). However, agricultural practices can have considerable effects on aquatic resources because the lands are often located on historic flood plains and valley bottoms. The effects of farming on aquatic systems include loss of native vegetation, bank instability, loss of floodplain function, removal of large woody debris sources, changes in sediment supply, changes in hydrology, increases in water temperature, changes in nutrient supply, chemical pollution, channel modification, and habitat simplification (Spence and others 1995). Nutrient and pesticide runoff pollutes many tributaries of the Columbia River, Upper Klamath Basin, and Great Basin. Agricultural herbicides and fertilizers are commonly found in eastern Oregon groundwater samples, and nitrogen is found in concentrations at or above state health advisory levels in five eastern Oregon counties (Vomocil and Hart 1993). The loss of native vegetation extends very near to the stream channel as farmers try to extend the amount of tillable land. Because the landscape alterations are permanent and soil is disturbed several

[2]Personal communication. 1995. M. Moulten, Sawtooth National Recreation Area, Stanley, Idaho.

times a year, negative effects of agricultural development may be more severe than other land uses.

The effects of livestock grazing on aquatic systems are related, in part, to the biophysical attributes of the site (Archer and Smeins 1991). Environmental factors that contribute to the character of an individual watershed include geology, climate, geomorphology, soils, vegetation, and water runoff patterns (Meehan and Platts 1978). Unstable stream conditions often exist as part of the natural conditions of streams; however, grazing can amplify these unstable conditions. In some cases, livestock use may initiate additional instability within a stream system.

Overgrazing by livestock can lead to a reduction of soil structure, soil compaction, and damage or loss of vegetative cover. All of these processes contribute to an increase in the rate and erosive force of surface runoff (Meehan and Platts 1978; Thurow 1991). Resulting increases in soil erosion lead to a loss of stored nutrients in the soil and a decrease in the level of vegetative productivity (Thurow 1991). The degree of soil erosion associated with livestock grazing is related to slope gradient and aspect of the site being grazed, the condition of the soil, type and density of vegetation, and the accessibility of the site to livestock (Meehan and Platts 1978).

Riparian areas maintain stream structure and function through processes such as water filtration, bank stabilization, water storage, groundwater recharge, nutrient retention, regulation of light and temperature, channel shape and pattern (morphology and micro-topography), and dispersal of plants and animals (Cummins and others 1984; Gregory and others 1991; Minshall 1967, 1994; Sullivan and others 1987). Because of the availability of water, forage, and thermal cover, riparian areas are often overgrazed by livestock. Livestock grazing can alter the species composition of stream-side vegetation (Archer and Smeins 1991; Platts 1978; Stebbins 1981; Thurow 1991; Vollmer and Kozel 1993) and diminish vegetative productivity (Archer and Smeins 1991; Horning 1994; Meehan and Platts 1978; Platts 1978; Thurow 1991; Vollmer and Kozel 1993). Grazing alters riparian vegetation by removing deep rooting plant species and decreasing canopy cover and riparian vegetation height (Platts 1991). Grazing has been implicated in the alteration of species composition of vegetative communities and associated fire regimes (Agee 1993; Leopold 1924).

Grazing is a major nonpoint source of channel sedimentation (Dunne and Leopold 1978; MacDonald and others 1991; Meehan 1991; Platts 1991). Grazed watersheds typically have higher stream sediment levels than ungrazed watersheds (Lusby 1970; Platts 1991; Rich and others 1992; Scully and Petrosky 1991). Increased sedimentation is the result of grazing effects on soils (compaction), vegetation (elimination), hydrology (channel incision, overland flow), and bank erosion (sloughing) (Kauffman and others 1983; MacDonald and others 1991; Parsons 1965; Platts 1981a, 1981b; Rhodes and others 1994). Sediment loads that exceed natural background levels can fill pools, silt spawning gravels, decrease channel stability, modify channel morphology, and reduce survival of emerging salmon fry (Burton and others 1993; Everest and others 1987; MacDonald and others 1991; Meehan 1991; Rhodes and others 1994). In addition, runoff contaminated by livestock wastes can cause an increase in potentially harmful bacteria (for example, *Pseudomonas aeruginosa* and *Aeromonas hydrophila*) (Taylor and others 1989; Hall and Amy 1990; Thurow 1991). Compared to ungrazed sites, aquatic insect communities in stream reaches associated with grazing activities often are composed of organisms more tolerant of increased silt levels, increased levels of total alkalinity and mean conductivity, and elevated water temperatures (Rinne 1988).

Influence of Timber Harvest Activities

Timber harvest activities (felling, yarding, skidding, landings, and silviculture) is one of the major land management activities within the assessment area. Timber harvest occurs in forests in 9 of 13 ERUs within the assessment area (McNab and Avers 1994). These areas include: the yellow pine forests of the Upper Klamath; the lodgepole and ponderosa pine forests of the eastern Cascades ERUs; mixed conifer forests in the Northern Glaciated Mountains, Lower Clark Fork, Blue Mountains, Central Idaho Mountains, and Upper Clark Fork ERUs; and forests within the Snake Headwaters that contain at least 50 percent Douglas-fir.

Anderson (1988), citing a 1986 report of the Montana State Water Quality Bureau, suggested that the single greatest threat to watersheds and aquatic life is timber harvest and associated road building within forests. This threat is due, in part, to the increased level of harvesting timber from steeper, more environmentally sensitive terrain (Anderson 1988; Platts and Megahan 1975). Accelerated surface erosion and increased levels of sedimentation can decrease after initial disturbance but may remain above natural levels for many years (Platts and Megahan 1975; Spencer 1991; Swanson 1981).

The mechanical processes involved in timber harvest and associated road construction, in conjunction with natural conditions, influence the level of disruption or disturbance within watersheds. Negative effects tend to increase when activities occur on environmentally sensitive terrain with steep slopes composed of highly erodible soils that are subject to high climatic stresses (Anderson 1988; Platts and Megahan 1975). Vulnerable watersheds generally have high slope gradients, high levels of potential soil erodibility, soils having moderate to very poor drainage, or soil moisture contents in excess of field capacity for long periods of the year (van Kesteren 1986).

Soil and site disturbance that inevitably occur during timber harvest activities are often responsible for increased rates of erosion and sedimentation (Chamberlain and others 1991; FEMAT 1993; MacDonald and others 1991; Meehan 1991; Reid 1993; Rhodes and others 1994); modification and destruction of terrestrial and aquatic habitats (FEMAT 1993; van Kesteren 1986); changes in water quality and quantity (Bjornn and Reiser 1991; Brooks and others 1992; Chamberlain and others 1991; Rhodes and others 1994); and perturbation of nutrient cycles within aquatic ecosystems (Rowe and others 1992). Physical changes affect runoff events, bank stability, sediment supply, large woody debris retention, and energy relationships involving temperature (Li and Gregory 1995). All of these changes can eventually culminate in the loss of biodiversity within a watershed (FEMAT 1993; Rowe and others 1992).

Increased delivery of sediments, especially fine sediments, is usually associated with timber harvesting and road construction (Eaglin and Hubert 1993; Frissell and Liss 1986; Havis and others 1993; Platts and Megahan 1975). As the deposition of fine sediments in salmonid spawning habitat increases, mortality of embryos, alevins, and fry rises. Erosion potential is greatly increased by reduction in vegetation, compaction of soils, and disruption of natural surface and subsurface drainage patterns (Chamberlain and others 1991; Rhodes and others 1994). Generally, logged slopes contribute sediment to streams based on the amount of bare compacted soils that are exposed to rainfall and runoff. Slope steepness and proximity to channels determine the rate of sediment delivery.

Water quality (for example, water temperature, dissolved oxygen, and nutrients) can be altered by timber harvest activities (Chamberlain and others 1991). Stream temperature is affected by eliminating stream-side shading, disrupted subsurface flows, reduced stream flows, elevated sediments, and morphological shifts toward wider and shallower channels with fewer deep pools (Beschta and others 1987; Chamberlain and others 1991; Everest and others 1985; MacDonald and others 1991; Reid 1993; Rhodes and others 1994). Dissolved oxygen can be reduced by low stream flows, elevated temperatures, increased fine inorganic and organic materials that have infiltrated into stream gravels retarding intergravel flows (Bustard 1986; Chamberlain and others 1991).

Nutrient concentrations may increase following logging but generally return quickly to normal levels (Chamberlain and others 1991).

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 altered by timber harvest (Bisson and others 1987; Sedell and others 1988; Robison and Beschta 1990). Shifts in the composition and size of trees within the riparian area affect the recruitment potential and longevity of large woody debris within the stream channel. Large woody debris influences channel morphology, especially in forming pools and instream cover, retention of nutrients, and storage and buffering of sediment. Any 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 (Rainville and others 1985; Robison and Beschta 1990). Large woody debris increases the quality of pools, provides hiding cover, slow water refuges, shade, and deep water areas (Rhodes and others 1994). Ralph and others (1994) found instream wood to be significantly smaller and pool depths significantly shallower in intensively logged watersheds. The size of woody debris in a logged watershed in Idaho was smaller than that found in a relatively undisturbed watershed (Overton and others 1993).

Because water is often delivered to lakes via stream channels, we can infer that effects to streams related to timber harvest and road construction may eventually be manifested within lakes. For example, stream erosion and the subsequent increase in sediment and nutrient transport due to land use activities in the Flathead Basin has contributed to lake eutrophication (Flathead Basin Forest Practices Water Quality and Fisheries Cooperative Program 1991). Birch and others (1980) reported that timber harvest activities caused increases in lake sedimentation rate and lake productivity in three of four lakes studied in western Washington, accelerating the rate of change in the trophic status of each lake. Timber harvest activities and road construction, including railroad construction, increased sedimentation rates above natural levels in three lakes of the Flathead Basin (Spencer 1991). Road construction appeared to be the greatest cause of disturbance resulting in enhanced fine sediment deposition in lakes downstream from the construction areas.

Influence of Roads

An assessment of federal roads within the Basin reveals that there are at least 204,333 kilometers of roads on Forest Service and BLM lands (data from Forest and BLM Inventoried Road databases; table 4.8). The majority (77%) are classified for basic maintenance only. Since many roads are not included in the transportation "system," substantial road kilometers are not included in the transportation databases. Therefore, based on discussions with Forest Engineers, we estimate that the magnitude of uninventoried road kilometers is as much as 30-50 percent of the inventoried road kilometers.

Roads contribute more sediment to streams than any other land management activity (Gibbons and Salo 1973; Meehan 1991), but most land management activities, such as mining, timber harvest, grazing, recreation, and water diversions are dependent on roads. The majority of sediment from timber harvest activities is related to roads and road construction (Chamberlain and others 1991; Dunne and Leopold 1978; Furniss and others 1991; Megahan and others 1978; MacDonald and Ritland 1989) and associated increased erosion rates (Beschta 1978; Gardner 1979; Meehan 1991; Reid 1993; Reid and Dunne 1984; Rhodes and others 1994; Swanson and Dyrness 1975; Swanston and Swanson 1976). Serious degradation of fish habitat can result from poorly planned, designed, located, constructed, or maintained roads (Furniss and others 1991; MacDonald and others 1991; Rhodes and others 1994). Roads can also affect water quality through applied road chemicals and toxic spills (Furniss and others 1991; Rhodes and others 1994). The likelihood of toxic spills has increased with the large number of roads paralleling streams.

Table 4.8— Kilometers of Forest Service and BLM roads within the ICBEMP assessment area (from Forest and BLM Inventoried Road databases).

Maintenance Level*	FS Region 1	FS Region 4	FS Region 6	BLM	Total
1	17,718	5,770	22,726	677	46,891
2	22,429	16,283	62,616	9,632	110,960
3	17,714	3,947	12,798	2,450	36,909
4	4,053	526	1,797	1,115	7,491
5	898	197	670	102	1,867
0			70	144	214
Total	**62,811**	**26,723**	**100,678**	**14,120**	**204,333**

* Maintenance Level

1 = Basic custodial care. Closed to 4x4 vehicles 40" wide or wider.

2 = Basic drainage only. High Clearance (4x4) vehicles.

3 = Must meet Highway Standards Act. Normal Clearance vehicles (public).

4 = Must meet Highway Standards Act. Moderate degree of comfort to users.

5 = Must meet Highway Standards Act. High degree of safety and comfort.

0 = Unclassified

NOTE: Many roads not considered as part of the transportation "system" are not included in the transportation databases. Based on discussions with Forest Engineers we estimate that the magnitude of uninventoried road kilometers is as much as 30-50% of the inventoried road kilometers.

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. For example, interruption of hill-slope drainage patterns alters the timing and magnitude of peak flows and changes base stream discharge (Furniss and others 1991; Harr and others 1975) and sub-surface flows (Furniss and others 1991; Megahan 1972). Road-related mass soil movements can continue for decades after the roads have been constructed (Furniss and others 1991). Such habitat alterations can adversely affect all life-stages of fishes, including migration, spawning, incubation, emergence, and rearing (Furniss and others 1991; Henjum and others 1994; MacDonald and others 1991; Rhodes and others 1994).

Poor road location, concentration of surface and sub-surface water by cross slope roads, inadequate road maintenance, undersized culverts, and sidecast materials can all lead to road-related mass movements (Lyons and Beschta 1983; Swanston 1971; Swanston and Swanson 1976; Wolf 1982). Sediment production from logging roads in the Idaho batholith was 770 times higher than in undisturbed areas; approximately 71 percent of the increased sediment production was due to mass erosion (Megahan and Kidd 1972) and 29 percent was due to surface erosion.

In granitic landtypes, sedimentation is directly proportional to the amount of road mileage (Jensen and Finn 1966). For instance, 91 percent (48,900 m^3) of the annual sediment production by land use activities (53,500 m^3) in the South Fork of the Salmon River has been attributed to roads and skid trails (Arnold and Lundeen 1968). King (1993) determined that roads in the Idaho batholith increased surface erosion by 220 times the natural rates per unit area. Roaded and logged watersheds in the South Fork of the Salmon River drainage also have significantly higher channel bed substrate embeddedness ratings than undeveloped watersheds (Burns 1984).

Roads greatly increase the frequency of landslides, debris flow, and other mass movements (Dunne and Leopold 1978; Furniss and others 1991; Megahan and others 1992). Mass movement in the western Cascade Range in Oregon was 30 to 300 times greater in roaded than in unroaded watersheds (Sidle and others 1985). Megahan and others (1992) found that 88 percent of landslides within Idaho were associated with roads. Roads were considered to be primary factor in accelerated mass movement activity in the Zena Creek drainage (Idaho batholith) following 1964-65 winter storms (Gonsior and Gardner 1971). Out of 89 landslides examined in the South Fork of the Salmon River, 77 percent originated on road hillslopes (Jensen and Cole 1965). Cederholm and others (1981) found that increases (above natural levels) in the percentage of fine sediment in fish spawning habitat occurred when the area of roads exceeded 3.0 percent of the Clearwater River (Washington) basin area. Increased stream channel sedimentation in Oregon and Washington watersheds east of the Cascade Range has also been associated with road density (Anderson and others 1992; McIntosh and others 1995).

Road/stream crossings can also be a major source of sediment to streams resulting from channel fill around culverts and subsequent road crossing failures (Furniss and others 1991). Plugged culverts and fill slope failures are frequent and often lead to catastrophic increases in stream channel sediment, especially on old abandoned or unmaintained roads (Weaver and others 1987). Unnatural channel widths, slope, and stream bed form occur upstream and downstream of stream crossings (Heede 1980), and these alterations in channel morphology may persist for long periods of time. Channelized stream sections resulting from riprapping of roads adjacent to stream channels are directly affected by sediment from side casting, snow removal, and road grading; such activities can trigger fill slope erosion and failures. Because improper culverts can reduce or eliminate fish passage (Belford and Gould 1989), road crossings are a common migration barrier to fishes (Evans and Johnston 1980; Furniss and others 1991; Clancy and Reichmuth 1990).

Influence of Mining Activities

Mining in the assessment area is focused on metals (such as antimony, copper, gold, iron, lead, mercury, molybdenum, silver, and zinc), industrial minerals (such as asbestos, clay, diatomite, dolomite, feldspar, fluorine, gravel, limestone, perlite, phosphate, pumice, sand, silica, talc, and zeolite), gem stones and abrasives (such as garnet), and fuel minerals (such as uranium) (Bryant and others 1980; Jackson and Kimerling 1993). Aluminum is processed in the Basin, but most of the barite and alumina are imported from other countries (Bryant and others 1980); however, there are potential reserves of barite, vanadium, and cobalt in the Basin (Bryant and others 1980). Mining for these materials occurs either as surface mining or underground mining. Although any mining activity may have negative effects on aquatic ecosystems (according to the Pacific States Marine Fisheries Commission 1994, 14,400 kilometers of rivers and streams in the western United States have been polluted by mining), the largest impacts are generally associated with surface mining.

Mining activities can affect aquatic systems in a number of ways: through the addition of large quantities of sediments, the addition of solutions contaminated with metals or acids, the acceleration of erosion, increased bank and streambed instability, and changes in channel formation and stability. Sediments enter streams through erosion of mine tailings (Besser and Rabeni 1987), by direct discharge of mining wastes to aquatic systems, and through movement of groundwater (Davies-Colley and others 1992). Coarse particles that enter watersheds are likely to settle relatively rapidly (Davies-Colley and others 1992), and therefore, effects on aquatic systems are greatest near mining activities. Fine inorganic particles (like clays) settle slowly and may travel great distances from the point of their introduction and therefore may have a greater effect on water bodies such as lakes further from mining activities. Fine suspended material reduces the amount of light available for benthic algae and plants, and thereby, biomass and primary production are diminished. Fine suspended materials may also reduce the quantity and quality of epilithon (substrate surface biofilm) that serves as food for benthic invertebrates. If suspended sediments damage respiratory structures of benthic invertebrates, their abundance may decline (Davies-Colley and others 1992).

Acidification of surface waters, a process associated with surface mining, mobilizes toxic metals naturally embedded in soils and streambeds. As surface water (including rain) washes through waste piles left (often indefinitely) from mining operations, it is acidified via iron oxidation and then flows into streams where metals are released and converted to forms which are available to aquatic life (Nelson and others 1991). Acidification of surface waters can affect organisms directly, such as salmonids which experience reduced egg viability, fry survival, growth rate, and other ills, or indirectly from toxic metals or substances which can affect growth, reproduction, behavior, and migration of salmonids and production of benthic algae (Spence and others 1995). Ecosystem responses to contaminants are dependent on the chemical, physical, biological, and geological processes at each site (Pascoe and others 1993). Depending on concentration, trace metal toxicity may reduce growth and reproduction or cause death of aquatic organisms (Leland and Kuwabara 1985). Adult stages of mollusks and fish can generally withstand higher concentrations of metals than other organisms (Leland and Kuwabara 1985), but embryonic and larval stages are quite sensitive to heavy metals (Leland and Kuwabara 1985). The combination of some metals may inhibit primary production more than any single metal alone (Wong and others 1978); therefore, when several metals are present, water quality criteria for single metals are insufficient for protecting aquatic life (Borgmann 1980).

Surface mining practices of dredging and placer mining have altered aquatic habitats by destroying riparian vegetation and reworking channels. Gold mining in Idaho's Crooked River forced unnatural meanders in some streams and straightened others (Nelson and others 1991). Some streams, such as the upper reaches of the Grande Ronde River (McIntosh and others 1994a), have been severely altered and now flow underground through rubble dredged from the stream bottoms decades ago.

Common practice for extracting gold today involves heap leach mining, a form of open-pit mining used for low-grade ore deposits. Piles of crushed ore are sprayed with a solution of sodium-cyanide (NaCN) that bonds with gold particles and is deposited in pools from which the gold is recovered. Numerous, small heap leach fields are located in the Basin, primarily in floodplains of rivers or streams which are susceptible to large floods, creating the potential for flood inundation of the toxic leach pools and consequent contamination of river or stream habitats.

Influence of Non-native Fish Species Introductions

The introduction of non-native fishes and aquatic invertebrates has had an important influence on species assemblages and aquatic communities throughout the assessment area. Currently at least 35 species, subspecies, or stocks of fish have been introduced to the Basin or have moved to habitats within the Basin where they did not occur naturally.

Most introductions have been centrarchid (sunfishes and basses), ictalurid (catfishes), cyprinid (minnows), and salmonid fishes, but a few exotic fishes such as the Oriental weatherfish, *Misgurnus anguillicaudatus*, appear to have been introduced through the aquarium trade. At least eight native fish species or subspecies have been transported, some widely within the Basin, outside their historic ranges. This latter group includes Yellowstone, Lahontan, and westslope cutthroat trout; fall chinook salmon; coho salmon; white sturgeon; steelhead; and interior redband (rainbow) trout.

Most introductions have been made with the intent of creating or expanding fishing opportunities and were initiated in earnest as early as the late 1800s (Evermann 1893; Simpson and Wallace 1978). Stocking of mountain lakes with cultured stocks of cutthroat, brook, and rainbow trout has been extensive (Bahls 1992; Liss and others 1995; Rieman and Apperson 1989). Many lakes that were historically barren of fish were capable of sustaining them, but lack of spawning habitat or isolation from colonizing populations prevented natural invasion. A variety of species such as kokanee salmon, chinook salmon, lake trout, brown trout, Atlantic salmon, coho salmon, black bass and other centrarchids, and ictalurids were introduced in these systems to diversify angling opportunities, create trophy fisheries, and to provide forage for potential trophy species. Many ephemeral lakes in the Great Basin have been stocked with crappie, bass, bullheads, and other centrarchids and ictalurids. Cultured strains of rainbow trout have been widely used to sustain put-and-take fisheries in lakes and rivers where angler harvest or habitat degradation is too excessive to rely on natural reproduction. Additional introductions have occurred through illegal transplants, unintentional escape from commercial hatcheries or the aquarium trade, and natural dispersal mechanisms.

Such introductions have led to the elimination of some native populations, while further fragmentation and isolation of other populations have left them more vulnerable to future extirpation. Although introductions have provided increased fishing opportunities and socioeconomic benefits, they have also led to catastrophic failures in some fisheries and expanded costs to management of declining stocks (Bowles and others 1991; Gresswell 1991; Gresswell and Varley 1988; Wydoski and Bennett 1981).

In the assessment area, competition between native and non-native salmonids has resulted in displacement or further isolation of some populations of cutthroat trout (Griffith 1988; Fausch 1988; Rieman and Apperson 1989) and bull trout (Donald and Alger 1992; Rieman and McIntyre 1993). Non-native fishes also threaten native species through hybridization and subsequent loss of the native genome through introgression. Hybridization with introduced trout is one of the most important factors in the decline of Yellowstone and westslope cutthroat trout (Allendorf and Leary 1988; Gresswell 1995; Liknes and Graham 1988; Rieman and Apperson 1989; Varley and Gresswell 1988). Liknes and Graham (1988) reported that genetically unaltered westslope cutthroat trout populations remained in only 2.5 percent of the historic range in Montana, whereas Rieman and Apperson (1989) estimated that less than 4 percent of remaining populations in Idaho were numerically strong and not threatened by hybridization. Hybridization between brook trout and bull trout appears to be common where the species overlap (Adams 1994; Leary and others 1993; Rieman and McIntyre 1993), and elimination or displacement of bull trout can be a common outcome (Leary and others 1993).

Predation by non-native species may have an important influence on some native cyprinids and catostomids (Williams and others 1990), resident trout populations (Griffith 1988; Rieman and Apperson 1989), and on the survival of juvenile anadromous salmonids (Rieman and others 1991). Large numbers of introduced white crappie prey on larval Warner suckers, apparently inhibiting successful recruitment of this threatened species

(Williams and others 1990). Predation by introduced lake trout has been implicated in the dramatic decline of bull trout in the Flathead River Basin,[3] westslope cutthroat trout in Glacier National Park (Marnell 1988), and Yellowstone cutthroat trout in Jackson Lake (Snake River drainage).[4] Predation by introduced fishes is also commonly identified as a major factor in the isolation and decline of native amphibians (Bahls 1992; Bradford and others 1993; Liss and others 1995) and has important effects on local invertebrate faunas as well (Bahls 1992; Liss and others 1995).

Effects of some introductions are unknown and will remain uncertain pending further studies. The Oriental weatherfish, a small species of loach (Cobitidae) native to central and southern Europe, was first collected in the interior Columbia River basin from irrigation canals along the lower Boise River (Courtenay and others 1987). This exotic European fish has now been found in the mainstem Boise River and reported from the lower Powder River Basin in Oregon. The extent of spread and effects associated with this species are unknown.

Consequences of introducing non-native species are not limited to a few interacting species. Effects frequently cascade through entire ecosystems (Winter and Hughes 1995) and compromise structure and ecological function in ways that rarely can be anticipated (Li and Moyle 1981; Magnuson 1976; Moyle and others 1986). The introduction of *Mysis relicta* into Flathead Lake and other large lakes in northern Idaho and Montana, for example, greatly reduced populations of cladocerans (zooplankton), a preferred food of *Mysis* and adult kokanee (Bowles and others 1991). Declines in kokanee abundance and a collapse of recreational fisheries in those systems have followed (Bowles and others 1991; Spencer and others 1991). Ironically, *Mysis* were introduced to provide additional food for kokanee. The shrimp feed on zooplankton in the epilimnion during night and then migrate to deep water (greater than 30 m) during daylight; this vertical migration allows *Mysis* to escape intense predation by kokanee that feed actively only during the day (Bowles and others 1991).

There is growing recognition that biological integrity and not just species diversity (Angermeier 1994; Angermeier and Karr 1994) is an important characteristic of aquatic ecosystem health. The loss or restriction of native species and the dramatic expansion of non-native species leave few systems that are not compromised.

Influence of Hatcheries

Although the cultured stocks of salmonids have been frequently used to mitigate the effects of over-harvest and habitat degradation, there is substantial evidence that this practice has detrimental effects on native populations (Hindar and others 1991; Krueger and May 1991; Marnell 1986; Miller 1954). Offspring of hatchery fish spawning in the wild do not survive as well as the offspring of wild fish (Chilcote and others 1986; Leider and others 1990; Nickelson and others 1986), even if the hatchery stock was developed from wild adults (Reisenbichler and McIntyre 1977). There is unavoidable selection for traits favoring survival in the artificial conditions of egg trays, tanks, raceways, and holding ponds. Hatchery fish thus become genetically distinct from wild fish. If they stray and subsequently spawn with wild fish in natural areas, survival of the offspring is compromised (Chilcote and others 1986).

Despite lower survival, hatchery fish occupy habitat that would otherwise be used by wild fish (Miller 1954). In addition, artificially high densities of fish returning to hatcheries attract intensive fisheries that can over-harvest wild fish (Reisenbichler, in press; Wright 1981, 1993). Increasing harvest rates for Oregon coho salmon, for example, caused a sharp decline in the escape-

[3]Personal communication. 1995. T. Weaver, Montana Department of Fish, Wildlife and Parks, Kalispell, Montana.

[4]Personal communication. 1995. J. Erikson, Wyoming Game and Fish, Jackson, Wyoming.

ment of wild coho in coastal streams and Columbia River tributaries in the late 1960s (Nickelsen and others 1986). To offset the loss, surplus fish at hatcheries were used to supplement spawning in natural habitats, but the problem was exacerbated because the hatchery fish apparently experienced low survival and displaced naturally produced fry.

Many hatcheries located on tributaries of the Columbia River have water intakes upstream of structures designed to divert migrating fish into hatchery ponds. In order to reduce the risk of transmitting diseases to the hatchery via its water intake, adult fish are not passed upstream of the intake barrier at many sites. Protection of hatchery water supplies often prevents natural populations from accessing large tracts of historic spawning and nursery area.

Efforts to protect the investment in hatcheries cause water managers to produce flows for optimizing survival of hatchery fish. Manipulation of flows in the Columbia River to facilitate downstream migration of hatchery fish often precedes the time that naturally produced salmon smolts emigrate (Waples 1991). Where the distribution of hatchery fish overlaps native populations in time and space, the artificially enhanced concentration of fish can aggravate competitive interactions and attract predators (Ames and others, n.d.; Nelson and Soulé 1987; Reisenbichler 1984; Royal 1972; Steward and Bjornn 1990)

Influence of Commercial and Recreational Harvest

Commercial harvest in the Columbia River basin contributed to the decline of spring and summer chinook salmon beginning in the late 1800s (Fulton 1969) and to the decline of fall chinook since 1920 (Lichatowich and Mobrand 1995). Lichatowich and Mobrand (1995) divided the history of the chinook salmon fishery into four phases: initial development (1866-1888), sustained production (1889-1922), resource decline (1923-1958), and maintenance at a depressed level (post 1958). Historical ocean and river harvest rates exceeded 80 percent (Ricker 1959). Prior to 1880, chinook salmon were the primary target of the commercial fishery, but harvest probably shifted to steelhead and other species as chinook salmon runs declined (Mullan and others 1992). Landings of steelhead declined steadily during the 1930s and 1940s as exploitation rates exceeded 60 percent [Northwest Power Planning Council (NWPPC) 1986].

Salmon and steelhead destined for the Columbia River basin are often captured off the coasts of the United States and Canada and in the high seas driftnet fisheries. Ocean harvest may include net, seine, sport, and troll fisheries. The ocean troll fishery began in 1912 (NWPPC 1986). Chapman and others (1994b) reported that 3.5 percent of 3,472 steelhead that were tagged in the mid-Columbia River were recaptured in ocean fisheries off the coasts of Alaska and Canada. Driftnet fisheries for squid operated by Japan, Korea, and Taiwan have contributed to the decline in steelhead by harvesting (illegally or incidentally) from five to 31 percent of the adult steelhead returning to North America (Cooper and Johnson 1992).

Angler harvest directly increases mortality and thereby influences total population abundance, size- and age-structure, and reproductive potential (Ricker 1975). Fishing may lead to substantial declines in abundance, especially in populations that are extremely vulnerable to certain types of gear. Angler harvest of Yellowstone and westslope cutthroat trout, for example, has contributed to substantial declines in abundance of these subspecies (Binns 1977; Gresswell and Varley 1988; Rieman and Apperson 1989; Thurow and others 1988; Varley and Gresswell 1988). Schill and others (1986) estimated that individual Yellowstone cutthroat trout were caught and released an average of 9.7 times during a 108-day angling season, and many were captured two or three times in a single day. Although high catchability may be desirable in sport fisheries, it may lead to substantial declines in abundance and changes in population structure without restrictions (Gresswell 1990; Gresswell and others 1994;

Gresswell and Liss 1995). Chapman and others (1994b) suggested that harvest of steelhead parr could also have significant effects on recruitment of wild steelhead in the mid-Columbia River.

Although management agencies have attempted to reduce or eliminate fishing as a source of mortality, incidental harvest of many sensitive native fish stocks is a problem in the Basin. For example, commercial harvest of steelhead is prohibited, but individuals are incidentally killed in nets set for other species. Chapman and others (1994b) estimated that incidental mortality to A-run steelhead equaled a 4 to 5 percent exploitation rate in 1987-1988 and that this mortality would be additive to harvest rates occurring in tribal fisheries.

Anglers may also affect fish stocks by altering fish habitat through redd trampling and increased bank erosion. Roberts and White (1992) demonstrated that wading on trout redds can cause mortality to eggs and fry. For many years, stream reaches in some states have been closed to angling during salmon spawning season to reduce harassment of spawning fish.

For decades most fish management agencies have supported a maximum sustained yield philosophy that promoted the maximum harvest of surplus fish production (Gresswell 1980; Ricker 1975). Within the past decade, however, many agencies have adopted new philosophies of management that prioritize restoration and management of native fish stocks and their habitats [Idaho Department of Fish and Game (IDFG) 1991] and recognize the non-consumptive values of fish (Botsford 1994; Gresswell 1995). Where habitat for native species remains suitable, fish populations have increased substantially following implementation of restrictive harvest regulations (Gresswell 1990; Varley and Gresswell 1988). For example, the abundance and size structure of Yellowstone and westslope cutthroat trout populations increased dramatically following reductions in angler harvest (Gresswell and Varley 1988; Johnson and Bjornn 1978; Rieman and Apperson 1989; Thurow and Bjornn 1978; Thurow and others 1988). Bull trout numbers and redds also increased in response to decreased harvest (Ratliff 1992).[5] These examples suggest that where populations retain resilience, restoration efforts can be successful.

Influence of General Recreation Activities

Mountain lakes, especially those in national parks and scenic forested areas, may be the most susceptible aquatic systems to the negative effects of recreation. The inherent sensitivity of a lake to pollutants influences its susceptibility to water-quality degradation (Gilliom and others 1980). Sensitivity varies among lake types. Large, deep lakes with a large inflow may be least susceptible to water quality degradation because pollutants are diluted by large volumes of water and settle along with particulate matter (Gilliom and others 1980). Lakes that are small and shallow, or that have a low inflow, are more sensitive to pollutants (Gilliom and others 1980). Likelihood of pollutant-loading increases if soil, geologic, or hydrologic characteristics of a watershed favor the transport of pollutants to the lake (Gilliom and others 1980).

Where visitor use is high, trampling associated with foot traffic can affect vegetation along lakes and streams through direct mechanical action and indirectly through changes in soil (Liddle 1975). Resistance to trampling depends on plant life form; large and broad-leaved plants are most susceptible, and grasses generally are most resistant (Burden and Randerson 1972). Loss of vegetation from shorelines, wetlands, or steep slopes can cause erosion and pollution problems (Burden and Randerson 1972; Gilliom and others 1980).

Effects of recreational use on fish may be more severe during sensitive life history phases. Rafts on the Salmon River were observed to disturb spawning chinook salmon and lead to movement away from redds during low-flow conditions.[6]

[5]Also, personal communication. 1995. J. Stelfox, Alberta Fish and Wildlife Division, Alberta, Canada.

[6]McIntyre, J. 1995. Boise, ID: U.S. Department of Agriculture, Forest Service, Boise, Idaho. Personal communication of unpublished data.

Power boats can have numerous negative effects on lake environments. Resuspension of bed sediments can occur with passage of a single boat (Garrad and Hey 1987). Patterns of suspended sediment have been found to correlate with frequency of boat movements, boat speed, and hull shape. Concomitant high levels of turbidity and reduced light penetration may be a major factor in declining populations of submerged macrophytes. Loss of macrophytes can, in turn, make shorelines more susceptible to erosion from boat wash (Garrad and Hey 1987). Power boats are also associated with the spread of the exotic Eurasian watermilfoil (*Myriophyllum spicatum*). Because it reproduces from seeds, rhizomes, and fragmented stems, this non-native plant is easily transported between water bodies when plant matter becomes entangled on boat propellers or trailers (Reed 1977).

Outboard engines introduce hydrocarbon emissions to the aquatic environment, and emissions have a high phenol content that is quite toxic to aquatic organisms (Wachs and others 1992). Increased lead levels in reservoirs may be attributed to recreational boating and gasoline spills (Cairns and Palmer 1993). Boats, especially larger ones, may also introduce the chemical tributyltin into aquatic environments (Becker and others 1992). Tributyltin is a biocide used in anti-fouling paints, and it is one of the most toxic compounds ever introduced into water (Maguire and others 1986). It has adverse effects on freshwater mollusks at concentrations as low as 0.1 μg/L (Hall and Pinkney 1985).

Effects of off-road recreational vehicle use on aquatic resources are documented only for a few types of natural systems. On sand dunes and shorelines, off-road vehicles can result in significant reductions of vegetation (Anders and Leatherman 1987; Wisheu and Keddy 1991). In dune areas negative effects can occur at rates as low as one vehicle pass per week (Anders and Leatherman 1987). Disturbance associated with off-road vehicle use can alter plant community composition or create openings in cover vegetation on shorelines (Wisheu and Keddy 1991). Partial loss of vegetation from shorelines can result in increased erosion that continues until those shorelines are devoid of vegetation (Wisheu and Keddy 1991). Because seeds tend not to be deeply buried in shoreline wetlands, they may be particularly sensitive to intense disturbance (Wisheu and Keddy 1991), and recovery of disturbed shorelines may be very slow. Use of off-road vehicles may be particularly detrimental in fragile soils or in areas where habitat for sensitive species is limited (Williams 1995). Additionally, off-road vehicle use in streams can result in destruction of redds, eggs, and young.

General Influence of Habitat Fragmentation and Simplification

The physical environment and the natural and human-caused disturbances to that environment profoundly influence the structure, composition, and processes defining aquatic ecosystems. Aquatic habitat fragmentation (impassable obstructions, temperature increases, and water diversion) and simplification (channelization, removal of woody debris, channel bed sedimentation, removal of riparian vegetation, and water flow regulation) have resulted in a loss of diversity within and among native fish populations. The fragmentation of aquatic systems occurs through natural, dynamic processes as well. Over geologic time river basins become connected or isolated. Within the assessment area, river basins have been isolated by geologic processes that influence the distribution of species and subspecies. Natural populations of Yellowstone cutthroat trout, for example, are found almost exclusively within the upper Snake River Basin, and westslope cutthroat trout are restricted to basins outside that area.

Climatic variation and catastrophic events such as fire and flood may change the suitability of, or access to, streams and local habitats, and species distributions expand or contract, often becoming more or less continuous. The distribution of spawning and rearing habitat of bull trout, for example, appears to be strongly influenced by water temperature, and in turn by local climate

(Rieman and McIntyre 1995). Climatic variation will likely lead to changes in the amount of headwater habitat available to bull trout and also to the degree of isolation or connection among those habitats. This natural fragmentation varies with elevation and latitude of streams and tends to be more accentuated on the southern limits of the bull trout range. The fragmentation of habitat available for any species will depend both on these natural processes and on the specific habitat requirements for that species. Bull trout, for example, appear to have a narrower range of suitable habitat than westslope cutthroat trout (Rieman and McIntyre 1993), and even in relatively pristine environments bull trout exhibit a patchy distribution.

The natural patchiness and heterogeneity of habitats represent both problems and benefits for fishes and other aquatic organisms. Theories from population and conservation biology predict that smaller or more isolated populations have an increased risk of extirpation, and that smaller patches of habitat are likely to support less diverse communities (Boyce 1992; Gilpin and Soule 1986; MacArthur and Wilson 1967; Simberloff 1988). There is empirical evidence that these are important issues for many aquatic communities and species (Gilpin and Diamond 1981; Hanksi 1991; Sjogren 1991) including fishes (Rieman and McIntyre 1995; Schlosser 1991; Sheldon 1988). At the same time species and communities that are spatially diverse face lower risks of regional extirpation in highly variable environments (den Boer 1968; Simberloff 1988). Core or source populations that are resistant to disturbance may support populations in other marginal or ephemeral habitats through dispersal (Bowers 1992; Simberloff 1988). The quality and distribution of even a few such key areas may ultimately dominate the dynamics of whole systems (Bowers 1992).

The heterogeneity of habitats for aquatic organisms, and particularly fishes, has been clearly recognized at multiple scales from microhabitat units to entire basins (Sedell and others 1990; Schlosser 1991). This spatial complexity is seen as an important factor influencing species diversity and ecosystem stability (Bowers 1992; Gresswell and others 1994; Schlosser 1991) and results in discontinuous distribution of life stages, populations, metapopulations, or subspecies and species as well. Important habitat types, such as pools or off-channel rearing areas, are discontinuous within stream reaches and influence the distributions and relative abundances of a species or life stages at that scale (Schlosser 1991). At larger watershed scales the distribution among reaches and among streams may be influenced by such things as local climate, stream temperature, stream gradients, the distribution of suitable spawning sites and gravels, and stream size (Fausch and others 1994; McIntyre and Rieman 1995; Rieman and McIntyre 1995). Spawning and rearing of bull trout and westslope and Yellowstone cutthroat trout, for example, may be restricted to smaller, headwater streams both by temperature and stream size even though subadults and adults may move widely throughout entire river basins (Gresswell 1995; McIntyre and Rieman 1995; Rieman and McIntyre 1995). Because spawning salmonids show a strong fidelity to natal habitats (Quinn 1993), even species that disperse widely throughout large river basins are likely structured into a collection of demes or local populations with varying levels of reproductive isolation (Gresswell and others 1994).

Fringe environments that do not support a large abundance of fishes may actually contribute much of the genetic variability to the population and may contribute in a critical way to the persistence of much larger systems (Northcote 1992; Scudder 1989). The connection among spatially diverse and temporally dynamic habitats and populations is likely to be a critical factor to persistence and integrity of aquatic communities.

Fishes, particularly salmonids, exhibit remarkable diversity of life-history strategies (Lichatowich and Mobrand 1995; Rieman and McIntyre 1993; Thorpe 1994) and important dispersal mechanisms for dealing with naturally fragmented and

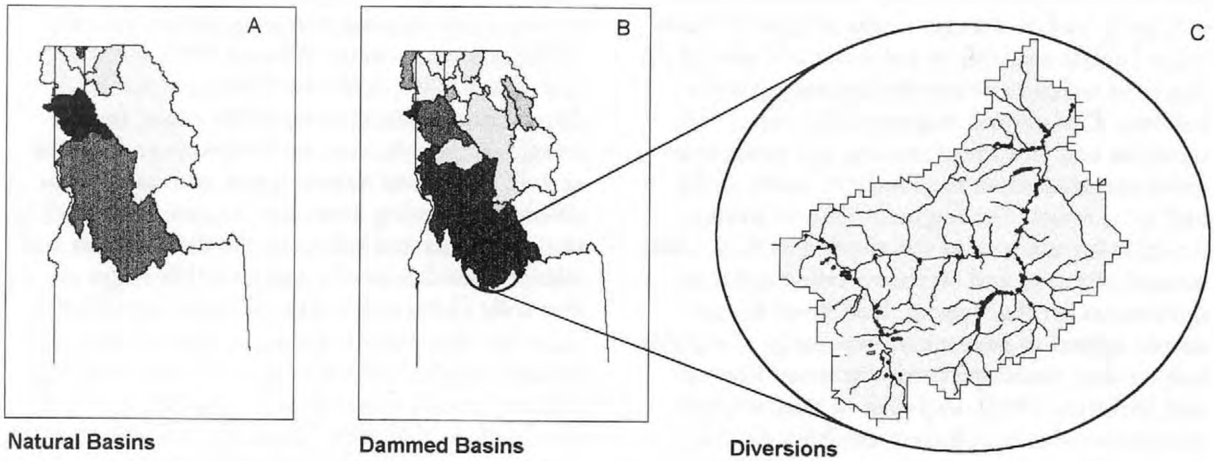


Figure 4.8— Fragmentation of watersheds in northern Idaho and western Montana. 'A' shows basins blocked to upstream movement of fishes by natural barriers. 'B' shows basins blocked to upstream movement by natural barriers and dams. 'C' shows irrigation diversions that may act as partial barriers to upstream and downstream movement within the Upper Salmon River subbasin.

variable environments (Milner and Bailey 1989; Quinn 1993; Thorpe 1994). Migratory life-history forms may be a particularly important mechanism of dispersal and risk aversion in highly variable environments for species like bull and Yellowstone cutthroat trout (Gresswell and others 1994; Rieman and McIntyre 1993).

The loss or degradation of habitats resulting from anthropogenic activities has not occurred in a random or uniformly dispersed fashion. Often lower elevation lands are more accessible, have wider floodplain valleys, and are more easily developed, hence habitat degradation has been greater in lower watersheds or in the lower reaches of larger systems. Dams and water diversions often result in fragmented streams and rivers (fig. 4.8). As a result, watersheds retaining the best remaining habitats are not well dispersed throughout the individual basins; they are often restricted to less productive headwater areas. Small streams in the headwater basins actually represent more extreme or sensitive environments with limited resilience to disturbance, increased synchrony among the populations, and relatively poor potential for dispersal throughout the entire Basin.

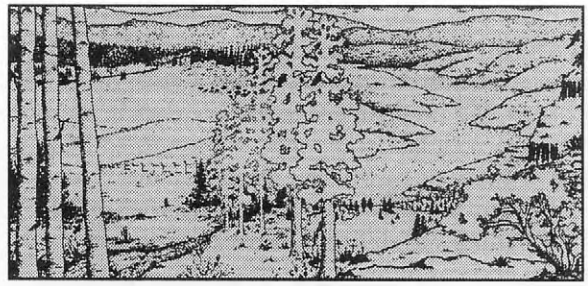
Because life-history stages and forms are also distributed in non-uniform or non-random patterns (Lichatowich and Mobrand 1995; Rieman and Apperson 1989; Schlosser 1991), some have been more likely to disappear than others. Within heavily managed areas, disturbance has often been dispersed among watersheds in an effort to minimize damage in any single area. If most watersheds are compromised, there are few local populations with the resilience to persist in the face of major storm or other catastrophic events that eventually test those populations. When high quality habitats are isolated in a system, the loss of migratory life histories, elimination of connecting corridors, or the poor quality of interspersed habitats that may act as "stepping stones" (Gilpin 1987) for dispersal may seriously limit the connectivity among populations. Eventually the ability of populations to refound or support those that are lost is diminished.

Although the importance of protecting hydrologic connectivity and hydraulic continuity of groundwater with surface water is accepted by state agencies throughout the Basin (for example, Washington State Water Resources Act 1971, RCW Chapter 90.54), it has received relatively little management attention. Nonetheless, maintaining connections between surface and subsurface flows can be critical. Many spring-dwelling species in the assessment area, such as the Borax Lake chub and Foskett speckled dace, are dependent upon the integrity of subsurface aquifers. The high degree of hydraulic continuity between groundwater and surface waters in the Methow Valley causes groundwater flows to reverse direction and well volumes to fall in response to flow changes in the river channel (Geo Engineers 1990; Golder Associates 1991). Extensive valley floodplains with alluvial deposits and groundwater flows like those in the Methow Valley have been shown to be extremely important to the ecology of the river. Floodplains having hydraulic connections with channels can support flowing groundwater (hyporheic) habitats that are used by riverine animals. The abundance of animals and nutrients underscore the extreme importance of these subsurface waters to the biological productivity of the channel and riparian ecosystems (Stanford and Ward 1988). These hyporheic systems also serve as a refugia for macroinvertebrate communities during time of drought and thermal stress.

The anticipated effects of human-induced disturbances on aquatic ecosystems throughout the assessment area include the reduced stability of regional aquatic populations and communities and an increased risk of both local and regional extirpations. Even with no further degradation of habitat, local extinctions driven by demographic and environmental variability could continue (Rieman and McIntyre 1993, 1995). The loss of life history expression influences the connectivity and stability among populations, but it also has restricted the full potential for fish production (Lichatowich and Mobrand 1995). The challenge for aquatic ecosystem management will be the maintenance and restoration of spatially diverse, high quality habitats that minimize the risks of extinction (Frissell and others 1993; Reeves and Sedell 1992) and that provide for the full expression of potential life histories (Healey 1994; Lichatowich and Mobrand 1995).

In the following sections, we attempt to assess the condition of key elements of the aquatic ecosystems throughout the assessment area. It is our intent to add to our understanding of the nature and magnitudes of change that have occurred, and to identify the key components that remain. That information can form the basis for the conservation and restoration of the structure and composition of aquatic ecosystems and for linking our understanding with similar problems in terrestrial ecosystems.

ASSESSMENT OF LAND-USE EFFECTS ON INSTREAM HABITAT CONDITIONS



The effects of land use on aquatic systems are often manifest through substantive changes in hydrology and morphology of streams and rivers. Such changes can have serious ramifications for aquatic organisms. Streams and adjacent environments are generally the most biologically productive areas within watersheds, and are often the sites of greatest conflict in resource management (Thomas 1979). Previous researchers have noted detrimental effects of land management on aquatic habitats (Chamberlin and others 1991; Frissell 1993; Reid 1993), and subsequent effects on aquatic species (Hicks and others 1991; Reeves and others 1993; Williams and others 1989). This has spurred a closer look at the connections between land use, stream-channel characteristics, and habitat conditions.

For decades, various agencies have routinely inventoried streams within their jurisdictions in order to monitor key aspects of stream-channel conditions. These inventories potentially provide a rich source of information for assessing relations between land-management practices and aquatic/riparian habitat conditions—when combined with landscape information not generally measured as part of the stream inventories. Comprehensive comparisons of stream inventory data have been enhanced by the recent adoption of mutually accepted protocols for channel measurements and electronic data storage. Since the late 1980s, standardized stream inventories have been completed within the Basin by the Forest Service, BLM, Oregon Department of Fish and Wildlife, University of Washington Center for Streamside Studies, Oregon State University, and the Confederated Tribes of the Umatilla Reservation.

These extensive inventories allow comparison across broad geophysical settings and among different management regimes, and also permit historical comparisons in some areas. From 1934 to 1945, the Bureau of Commercial Fisheries [now National Marine Fisheries Service (NMFS)] surveyed more than 6,400 kilometers of streams in the Columbia River Basin to determine the condition of freshwater habitat for anadromous salmonids after completion of Grand Coulee and Bonneville Dams (Rich 1948). Many historically surveyed streams throughout the Columbia River Basin were recently resurveyed (McIntosh 1995). Of the 390 streams surveyed by the Bureau of Commercial Fisheries, 120 (30%) were resurveyed, encompassing 2,259 kilometers (35%) of the original survey (6,454 km). The resurveyed streams included a broad range of geologic conditions, land ownerships, and land-use histories (McIntosh 1992, 1995; McIntosh and others 1994b), and were distributed among the Coast Range, North Cascades, West Cascades, Northern Rockies, and the Blue Mountains. McIntosh (1995) examined historical changes in large-pool and deep-pool habitats using the information from the resurveyed streams.

In this section, we summarize the findings of McIntosh (1995), and report on our own analysis of stream inventory data from a wide range of streams. We used stream inventory data in combination with landscape information to address one primary question: can we detect and characterize land-use effects on aquatic habitats, given high natural variability in instream measurements and confounding geophysical, vegetative, and climatic factors? This question is of critical importance to evaluating the potential effects of management and to monitoring the effects of past activities.

Historical Changes in Large-pool and Deep-pool Habitats in the Columbia River Basin

To examine land-use induced changes in fish habitat over time, McIntosh (1995) classified individual streams according to the land management history of the watershed (managed or unmanaged) and looked at the frequency of large (> 20 m^2 area and > 0.8 m depth) and deep (> 20 m^2 area and > 1.6 m depth) pools. Managed watersheds were predominantly multiple-use roaded areas, where timber, grazing, agriculture, or mining production were permitted. Unmanaged watersheds were minimally affected by human disturbance, including wilderness and roadless areas. Although many unmanaged areas historically included grazing and mining, these activities have stopped or been reduced such that they typically affect less of the watershed and are of lower intensity in unmanaged basins relative to managed basins.

Quantifying change within individual streams is problematic (McIntosh 1995). Expressing results (pool increases or decreases) as percentages is confounded by variability in the magnitude of change. For example, streams with infrequent pools show large percentage changes with small actual changes. To overcome this problem, a 95 percent confidence interval around zero was estimated for individual streams in each treatment category (large and deep, managed and unmanaged); only changes (as percentage) that fell outside the respective confidence interval were judged to be significant (fig. 4.9).

One-hundred-four streams in managed areas within the Columbia River Basin were analyzed for changes in large pools. Decreases in the frequency of large pools were observed in 51 percent of the streams; thirty-one percent showed increases, and 18 percent remain unchanged (fig. 4.9b). For unmanaged streams (n = 25), 56 percent of the streams showed an increase in pool frequency, 40 percent experienced no change, and 4 percent decreased (fig. 4.9a).

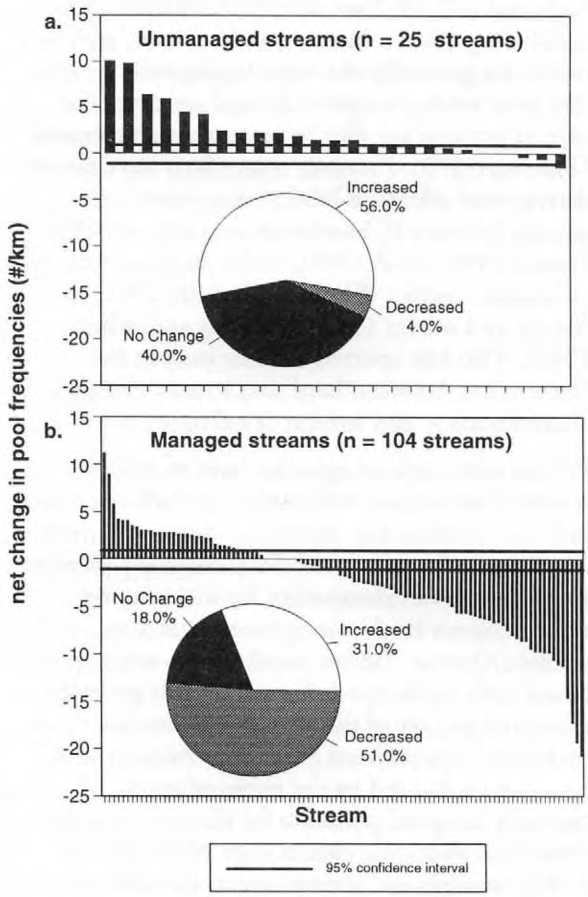


Figure 4.9— Changes in large pools (>20 m^2 and 0.8 m deep) from 1935-1945 to 1991-1994 on Columbia River Basin streams. Stream basins were either (A-upper) in unmanaged condition or (B-lower) under significant active management over the last 50+ years.

Over the last 50 to 60 years, there has been a dramatic loss of deep pools in managed streams throughout the Columbia River Basin. Of managed streams (n = 100), deep pools decreased in 53 percent; only 4 percent increased, and 43 percent measured no change (fig. 4.10b). Approximately 42 percent of the streams which showed no change did not have any deep pools in the historical surveys. Deep pools in about 33 percent of the unmanaged streams (n = 24) increased, 13 percent decreased, and 54 percent showed no change (fig. 4.10a). Of the streams which showed no change, 25 percent had no deep pools in the historic survey.

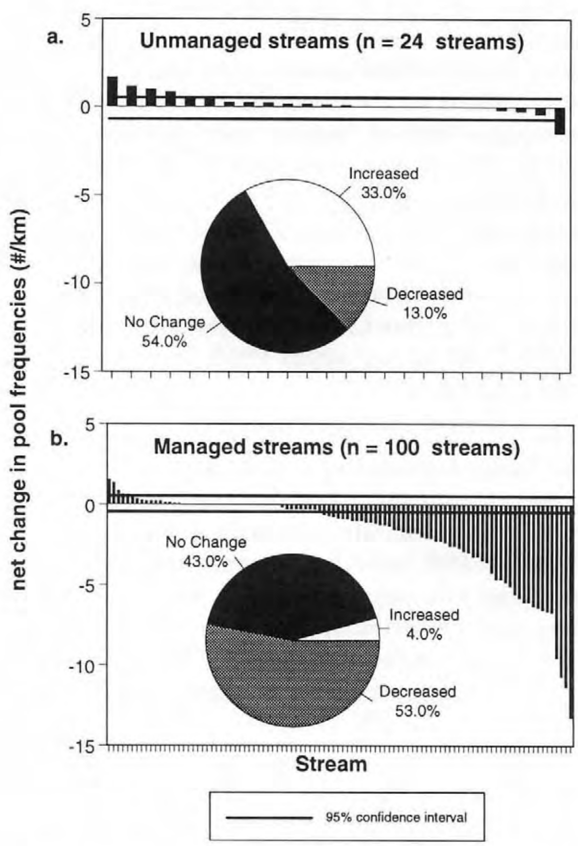


Figure 4.10— Changes in deep pools (>20m^2 and 1.6m deep) from 1935-1945 to 1991-1994 on Columbia River Basin streams. Stream basins were either (A-upper) in unmanaged condition or (B-lower) under significant active management over the last 50+ years.

Streams from three ERUs (Northern Cascades, Central Idaho Mountains, and Blue Mountains) were included in the analysis. Among these, the Northern Cascades had fewer large pools than either the Central Idaho Mountains or the Blue Mountains, but differences between the Central Idaho Mountains and the Blue Mountains were not significant. There were no statistically significant differences in the frequency of deep pools among ERUs.

A total of 285 kilometers in 20 managed watersheds in the Central Idaho Mountains was surveyed (table 4.9). There was a significant decrease ($p < 0.01$) in large pools from 9.4 pools/kilometer to 5.6 large pools/kilometer (40% decrease). Large pools in 11 unmanaged streams (171 km) in the Salmon River subbasin of the Central Idaho Mountains showed no significant change (from 5.3 to 6.6 large pools/km). There was a significant decrease ($p < 0.05$) in the frequency of deep pools (from 1.2 to 0.3 deep pools/km) in 182 kilometers of 18 managed streams in the Central Idaho Mountains. Deep pools in the 11 unmanaged streams increased significantly ($p < 0.05$) from 0.2 to 0.5 deep pools/kilometer. The data also suggest lower elevation and lower gradient streams had more numerous pools and a more diverse habitat structure.

In the Blue Mountains, 15 managed streams (299 km) and two unmanaged streams (18 km) were surveyed (table 4.9). Managed streams yielded a 58 percent decrease ($p < 0.01$) in large pools (from 5.9 to 2.5 large pools/km). The two unmanaged streams showed no significant change (from 0.0 to 1.3 large pools/km). Deep pools were scarce in the Blue Mountains, and the change from 0.7 to 0.1 deep pools/kilometer in managed streams was not significant. Deep pools in the Umatilla River declined from 4.2 deep pools/kilometer to 0.7 deep pools/kilometer. In the two unmanaged stream reaches on the Tucannon River and Panjab Creek, no deep pools were found in either survey.

In the Northern Cascades, 21 managed streams (479 km) and 10 unmanaged streams (187 km) were surveyed (table 4.9; McIntosh 1995). Large pools in managed streams doubled from 1.9 to 3.8 large pools/kilometer ($p < 0.01$). Unmanaged streams showed an even larger increase in large pools (1.7 to 4.6 pools/km; $p < 0.01$). Neither managed streams nor unmanaged steams showed significant change in deep pools during the 55-year interval between surveys (from 0.9 to 0.6 deep pools/km and from 0.6 to 0.7 deep pools/km for managed and unmanaged streams, respectively). These results may reflect intense grazing at the turn of the century and intact floodplain forests interacting with two large floods in 1948 and 1972 (McIntosh and others 1994b; Wissmar and others 1994a, 1994b).

In summary, in managed streams, roads and accompanying human activities including timber harvest, highway construction, grazing, agricultural practices, and loss of riparian vegetation have combined to create major decreases in pool habitat within the Columbia River Basin. The losses appear to have been the greatest in the lower-gradient, biologically productive areas of river basins most disturbed by humans. The unmanaged streams that were resurveyed generally were in steeper and more highly dissected landforms within the Columbia River Basin, and started with fewer large or deep pools. Most unmanaged streams either have retained pools or have improved pool habitat during the last 55 to 60 years.

Basin-wide Analysis of Stream Inventories

For our analysis, we compiled stream inventory data from electronic databases maintained by Regions 1, 4, and 6 of the Forest Service, the BLM, Oregon Department of Fish and Wildlife, and the Confederated Tribes of the Umatilla Indian Reservation. Stream inventory data were collected using protocols derived from the habitat-based inventory methodology of Hankin and Reeves (1988), which followed that of Bisson and others (1982). For details of the different stream inventory procedures, see Oregon Department of Fish and Wildlife (ODFW 1994), Overton and others (1994), and U.S. Department of Agriculture (USDA 1995).

The sampling unit in these inventories was the reach, identified as a geomorphically homogenous section of stream that is several-to-hundreds of channel widths long. A single reach was generally between major tributary junctions and within a setting of uniform valley slope and morphology (Grant and others 1990; Montgomery and Buffington 1993; USDA 1995). Certain measurements such as slope (channel and valley), valley width, and drainage area were made on an overall reach basis. Most measurements, however, were on individual channel units that were then summarized for the entire reach. Channel units, sometimes termed "habitat units" were portions of the reach that were one to several channel widths long and had distinct morphologic and hydraulic characteristics. Channel units were typically described as pools, riffles, cascades, steps, runs, and rapids (Bisson and others 1982; Grant and others 1990; Hawkins and others 1993; Leopold and others 1964; Montgomery and Buffington 1993).

The final database consisted of summaries of 6,352 reaches covering 17,121 kilometers on 1,982 streams (map 4.4). In total, the summarized reaches contain information from more than 600,000 individual channel units. The reach lengths ranged from 0.2 to 45.6 kilometers, and consisted of 1 to 1,367 channel units. The mean reach length was 2.7 kilometers, and the average number of channel units in each reach was 75.

Distribution and Sampling Issues

The vast majority of inventoried reaches are on federally administered lands, and most of those are on National Forest lands. Because stream inventories are generally motivated by concerns regarding fish habitat, the sampled reaches are

Table 4.9— Comparison of changes in large and deep pools in the Coast Range and western Cascades, by Ecological Reporting Unit. Bureau of Fisheries (BOF) surveys are compared to Pacific Northwest Research Station (PNW) and Intermountain Research Station (INT) surveys.

LARGE POOLS (>20 m² and 0.8 m deep)

MANAGED STREAMS

Stream (n)	km Surveyed	BOF (#/km)	PNW/INT (#/km)	Change (#/km)	Percent Change	Significance	
Coast Range (15)	182	8.2	7.2	-1.0	-12	p>0.05	No Change
Western Cascades (34)	636	10.8	6.9	-4	-37	p<0.01	Decrease
Northern Cascades (21)	479	1.9	3.8	1.9	95	p<0.01	Increase
Blue Mountains (15)	299	5.9	2.5	-3.4	-58	p<0.01	Decrease
Central Idaho Mt. (20)	285	9.4	5.6	-3.8	-40	p<0.01	Decrease
Total (104)	**1881**						

UNMANAGED STREAMS

Stream (n)	km Surveyed	BOF (#/km)	PNW/INT (#/km)	Change (#/km)	Percent Change	Significance	
Coast Range	NO DATA						
Western Cascades (1)	2	2.5	12.5	10	400	p<0.01	Increase
Northern Cascades (10)	187	1.7	4.6	2.9	171	p<0.01	Increase
Blue Mountains (2)	18	0	1.3	1.3		p>0.05	No Change
Central Idaho Mt. (11)	171	5.3	6.6	1.3	25	p>0.05	No Change
Total (25)	**378**						

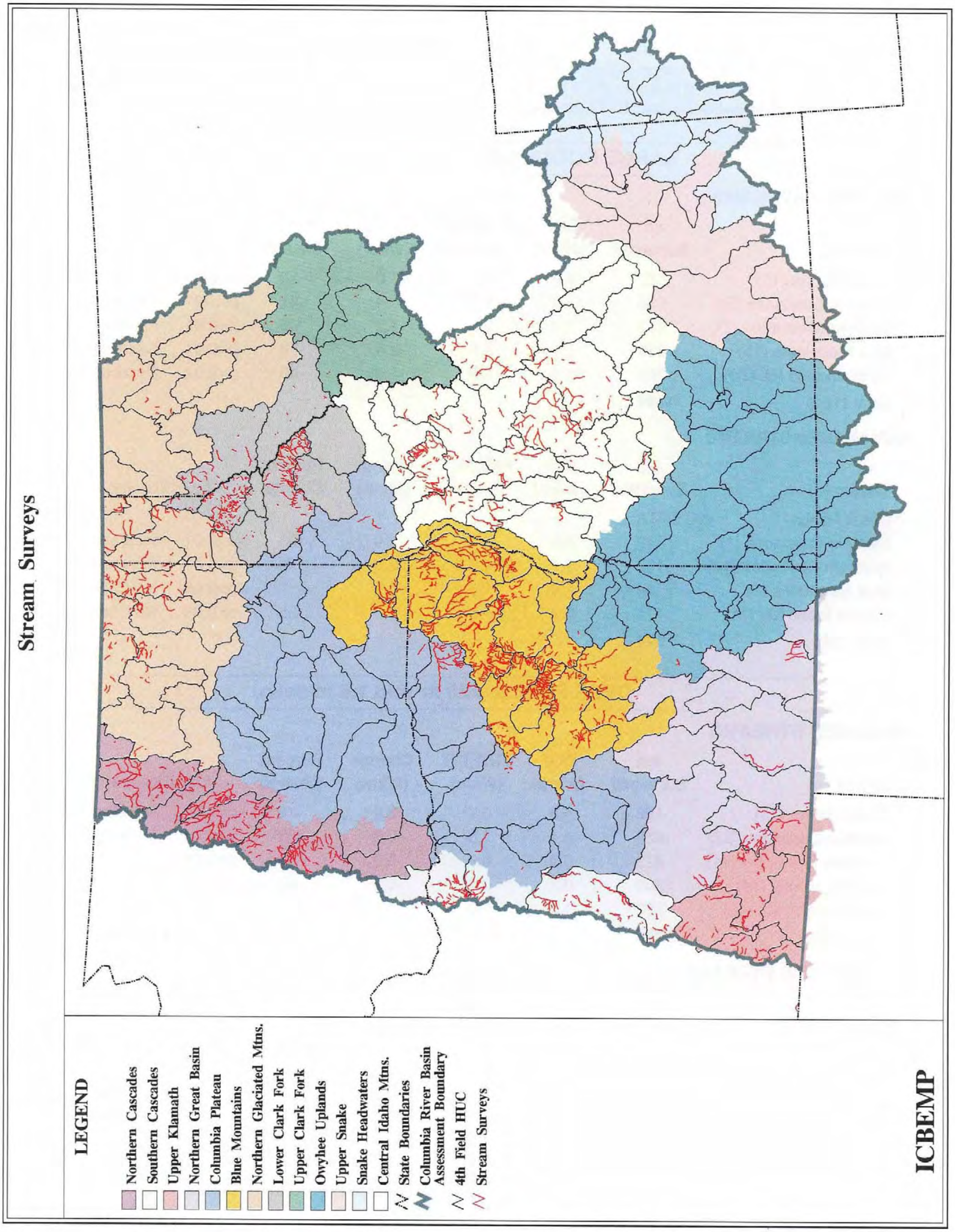
DEEP POOLS (>20 m² and 1.6 m deep)

MANAGED STREAMS

Stream (n)	km Surveyed	BOF (#/km)	PNW/INT (#/km)	Change (#/km)	Percent Change	Significance	
Coast Range (15)	182	4.6	0.7	-3.9	-85	p<0.01	Decrease
Western Cascades (32)	626	4.4	1.6	-2.8	-63	p<0.01	Decrease
Northern Cascades (21)	479	0.9	0.6	-0.3	-33	p>0.05	No Change
Blue Mountains (14)	299	0.7	0.1	-0.6	-86	p>0.05	No Change
Central Idaho Mt. (18)	182	1.2	0.3	-0.9	-75	p<0.05	Decrease
Total (100)	**1788**						

UNMANAGED STREAMS

Stream (n)	km Surveyed	BOF (#/km)	PNW/INT (#/km)	Change (#/km)	Percent Change	Significance	
Coast Range				NO DATA			
Western Cascades				NO DATA			
Northern Cascades (10)	187	0.6	0.7	0.1	17	p>0.05	No Change
Blue Mountains (2)	18	0	0	0	0	p>0.05	No Change
Central Idaho Mt. (11)	171	0.2	0.5	0.3	150	p<0.05	Increase
Total (24)	**377**						



Map 4.4— Inventoried stream reaches, Ecological Reporting Units (ERUs), and subbasin delineations for the assessment area.

Table 4.10— Number of reaches and total length of inventoried stream in each Ecological Reporting Unit.

Ecological Reporting Unit	Number of Reaches Surveyed	Kilometers Surveyed
Northern Cascades	687	2,064
Southern Cascades	324	916
Upper Klamath	216	522
Northern Great Basin	170	454
Columbia Plateau	172	712
Blue Mountains	1,604	5,108
Northern Glaciated Mountains	757	1,667
Lower Clark Fork	946	3,567
Upper Clark Fork	40	24
Owyhee Uplands	11	16
Central Idaho Mountains	1,442	2,044
Total	**6,369**	**17,094**

not evenly or randomly distributed across the assessment area. The inventories are located primarily in mountainous, headwater, and fish-bearing streams. Eleven of the 13 ERUs contain inventoried reaches, but only seven ERUs contain more than 200 inventoried reaches (table 4.10).

The only basin-wide reference with which to evaluate stream length and stream type sampling coverage is the 1:100,000 hydrography (streams, rivers, lakes, reservoirs, and canals portrayed on 1:100,000 USGS topographic maps). The 1:100,000 hydrography under-represents the total stream network by omitting many small headwater streams, but still provides a relative framework for evaluating sampling coverage. The inventoried streams represent 3 percent of the total stream network in the assessment area as portrayed on the 1:100,000 hydrography. Stream order (Strahler 1957) was assigned to each stream on the 1:100,000 hydrography and to the spatially linked inventoried reaches, allowing us to assess stream order sampling. Strahler's (1957) stream order begins with first-order, representing headwater streams without distinguishable tributaries, and increases with the order of contributing tributaries. For example, two first-order streams join to form a second-order stream, two second-order streams form a third-order stream, and so forth. Basin-wide, the most complete sampling is for the third- and fourth-order streams, where about 10 percent of their lengths were surveyed. Sampling coverage is much less for higher-order streams, which are generally outside federally-administered lands, and for lower-order streams, which are more numerous and typically receive less attention by land management agencies, despite being important habitat for many aquatic species (fig. 4.11).

The stream inventory database contains reaches in 1,052 (14%) of the 7,469 subwatersheds within the Basin. In terms of the 1:100,000 hydrography, the percent by length coverage of the inventoried subwatersheds ranges from 0.06

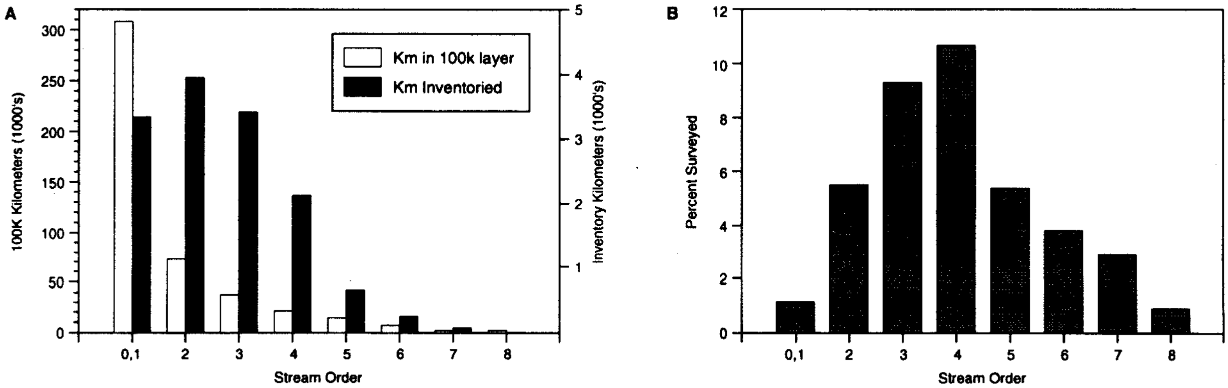


Figure 4.11— Portrayal of stream inventory sampling by stream order. A) The distribution by stream order of all streams and inventoried streams within the assessment area. The lengths are from the 1:100,000 hydrography. The total length of surveyed streams is 13,653 km based on the 1:100,000 hydrography—80 percent of the field-measured length (17,121 km). The difference is the result of channel sinuosity and path complexity not being adequately portrayed on the 1:100,000 hydrography. B) The percent of each stream-order class inventoried, based on lengths from the 1:100,000 hydrography.

percent to over 100 percent. Sampling coverage values greater than 100 percent reflect intensively surveyed streams where there were multiple surveys on individual reaches.

Besides the uneven sampling, there are other considerations and limitations that potentially cloud interpretation and use of the summaries and analyses. During the compilation process, we did not systematically assess the quality of the stream inventory data. Most of the data had undergone some level of prior quality control, but the quality of the data was uneven. Data that had obvious and numerous problems were discarded, but potential erroneous observations and recording errors remain. There also may be errors in spatial location because of the inherent difficulties in precisely locating the inventoried reaches on the 1:100,000 hydrography. Most map locations should be within 0.5 kilometers of the true reach locations. In addition, many stream inventories required channel geometry to be estimated in the field and then corrected by a calibration developed from both measured and estimated channel geometries. The calibration coefficients are generally between 0.9 and 1.1. For this analysis, we used corrected values for all data except for Forest Service Region 6 (eastern Oregon and Washington) stream inventories, where corrections were not applied. Finally, some reaches were surveyed more than once. Multiple inventories of the same reach were not isolated and removed from the database.

Analyzed Channel Features

The choice of stream channel characteristics that can be evaluated for the entire Basin is limited by the different inventory protocols used by the different agencies and administrative regions of the Forest Service. Only the channel geometry measurements (unit width, depth, and length) are directly comparable for most surveys. The choice of summarized variables was further influenced by our understanding of which channel attributes are sensitive to land-use practices and may be important indicators of habitat quality (Overton and others 1993, 1995; Ralph and others 1994; Reid 1993). Moreover, our experience suggests that the channel geometry measurements are least affected by observer error or bias. We identified 13 variables that are suitable for analysis (table 4.11), and chose four variables that have biological implications: *large pool frequency*, *pool frequency*, *R6 wood frequency*, and *surface fines*.

Table 4.11— Thirteen variables and the variable descriptions that were used to characterize stream channels for the entire Columbia River Basin.

Variable	Description
Large pool frequency	Number of pools with maximum depth > 0.8 m and surface area > 20 m^2 per mean reach riffle width.
Pool frequency	Number of pools per mean reach riffle width.
Fraction slow water	Fraction of total reach length consisting of pools and glides.
Mean pool depth/width	The mean of the ratio of maximum depth to width for all pool channel units in a reach. For reaches surveyed with procedures that measured mean depth, but did not measure or report maximum depth, an estimate of the maximum depth was derived by applying the regression equation *maximum pool depth* = 0.19 + 1.504(*mean pool depth*). This regression was developed from a regression relation between mean depth and maximum depth for pools where both measurements were reported (n=3517, r^2=0.59).
Variance pool depth/width	Variance of the ratio of max. depth to width for all pools in a reach. In statistical analysis, standard deviation (s.d. pool depth/width) is evaluated rather than variance.
Mean riffle depth/width	Mean of the depth to width ratio for all riffles in a reach. Mean depth was used to compute the ratio for each riffle except where only max. depth was reported, in which case the max. depth measurement was used without modification.
Variance riffle depth/width	Variance of the ratio of depth to width for all habitat units in a reach. Standard deviation of riffle depth/width was evaluated in the statistical analysis rather than variance.
R6 wood frequency	The number of pieces of wood per average riffle width, surveyed with the Forest Service Region 6 protocol. Tallied wood includes all pieces with diameters greater than 30 cm and lengths greater than 10 m. This includes the "small" and "large" size classifications of the Forest Service Region 6 protocol.
R1/4 wood frequency	The number of single pieces of wood per average riffle width, surveyed with the Forest Sevice Region 1/4 protocol. Tallied wood includes all pieces with diameters greater than 10 cm and lengths that exceed 3 m or two-thirds the channel width.
Wood aggregate frequency	The number of wood aggregates per average riffle width, surveyed with the Forest Service Region 1/4 protocol. Wood aggregates are defined as multiple pieces of wood (>10 cm in diameter, longer than 3 m or two-thirds the channel width) that are touching each other and within the active channel.
Embeddedness	"Yes" or "no" summarization of cobble substrate embeddedness for the reach, surveyed with the Forest Service Region 6 protocol. Substrate is classified as being embedded if 35 percent of the interstices are filled with fine sediment.
Bank stability	Fraction of bank in reach that is estimated as being stable, surveyed with Forest Service Region 1/4 protocol.
Surface fines	Reach mean of the areal fraction of each pool tail and low-gradient riffle covered with sediment < 6 mm in diameter, surveyed with Forest Service Region 1/4 protocol.

Large pool frequency and *pool frequency* measures were available across the Basin. *R6 wood frequency* was measured only within the states of Oregon and Washington. *Surface fines* was measured only within the states of Idaho and Montana.

Landscape Characterization

In order to elucidate relationships between aquatic and terrestrial ecosystems, we developed statistical models that linked landscape features and management activities to stream channel measures (this section) and fish populations (next section). Here, we present our approach to developing a set of variables that describe landscape characteristics and relevant management activities. Many of these variables were taken directly from the data developed by the landscape and spatial teams. The remainder generally were derived from information generated by the landscape team that we modified or summarized, as described below, for our specific purposes. The 30 variables chosen to describe landscape attributes were roughly divided into three broad categories: 1) geophysical variables; 2) terrestrial vegetation; and 3) land ownership and management. The only variable which did not fall within any single category was Ecological Reporting Unit (*eru*), the 13 geographic regions in the Basin that are delineated on the basis of physiographic and vegetative considerations. Variable names for landscape variables used in the analysis of channel morphology and fish populations are identified in table 4.12; *italics* used within the text correspond to variable names identified in the table. Several of these variables, for example, *anadac, dampass,* and *mtemp*, do not describe physical stream features and were only used in addressing the distribution and status of fishes. We include them here for completeness.

Much of the information that went into developing the landscape variables was developed from continuous 90-meter digital elevation data, one-kilometer vegetation data (from satellite imagery), 1:100,000 hydrography, state geologic maps, land ownership and activity maps, and extrapolated landscape characteristics derived from aerial photographic analysis of about 5 percent of the Basin. All variables that describe landscape characteristics were summarized for each of the 7,469 subwatersheds within the Basin. See the Landscape Dynamics and Information System Development chapters in this report for more information regarding derivation of many of these variables.

Physiographic and Geophysical Variables

The physiographic and geophysical variables developed by the Landscape Ecology Team included topographic descriptions such as slope, elevation, and stream length; climatic descriptions such as annual mean temperature, estimated annual precipitation, and average solar radiation; and variables reflecting erosion potential, for example, soil texture, base erosion (susceptibility of bare soil to erosion), and bank erosion. The number and range of variables generated by the Landscape Ecology Team are much broader than the subset that we selected. Our selection was based on an initial screening in which we chose a fairly large number of potential variables that have some plausible relationship with channel morphology or fish. We then calculated Spearman rank correlations (Zar 1984) for all possible pairs of variables and chose a subset which were highly correlated ($|r_s|>0.8$) with the rest of the set, but which were relatively independent of each other. Our objective was to develop a smaller subset of variables that would be easier to manage and interpret, yet still contain much of the information inherent in the larger set.

For the analysis of fish distribution, we developed three additional variables: *anadac*, which indicated whether an area was accessible to anadromous fish; *dampass*, which referred to the number of passable dams between an area and the Columbia River Estuary (used only within the accessible range of anadromous fish); and *hucorder*, which referred to the cumulative number of subwatersheds that contributed hydrologically to a given unit.

Table 4.12— Descriptions of predictor landscape variables used in the stream inventory analysis and the analysis of fish distribution. All values expressed as percents refer to the percent area of the subwatershed.

Variable Name	Description
eru	Ecological Reporting Unit
Physiographic and Geophysical Variables	
slope	area weighted average midslope
slope2	percent of area in slope class 2 (slopes >10%, <30%)
con1	percent weakly consolidated lithologies
con2	percent moderately consolidated lithologies
con3	percent strongly consolidated lithologies
sdt1	percent lithologies that produce coarse-textured weathering products
sdt2	percent lithologies that produce medium-textured weathering products
sdt3	percent lithologies that produce fine-textured weathering products
alsi1	percent felsic lithologies
alsi2	percent intermediate aluminosilicated lithologies
alsi3	percent mafic lithologies
alsi4[1]	percent carbonate lithologies
pprecip	mean annual precipiation (PRISM)
elev	mean elevation (ft)
mtemp[1]	mean annual temperature
solar[1]	mean annual solar radiation
streams[1]	length of 1:100,000 streams in 6th-code hydrologic unit (miles)
drnden	drainage density (mi/mi^2)
anadac[1]	access for anadromous fish (0=no, 1=yes)
dampass[1]	number of intervening dams
hucorder[1]	number of upstream 6th-code hydrologic units
hk	soil texture coefficient
baseero	base erosion index
ero	surface erosion hazard
bank	streambank erosion hazard
Vegetation Indices plus Ownership and Management Variables	
vmf	vegetation amelioration
vegclus	clustered vegetation types
roaddn	road density class
mgclus	management classification

[1]Variable only used in analysis of fish distribution (see section entitled Distribution and Status of Fishes).

Hucorder values ranged from zero for headwater areas that receive no inputs, to several thousand for units spanning the mainstem Columbia River. The majority of subwatersheds had low *hucorder* values; *hucorder* equalled zero for 55 percent of the watersheds, while only 13 percent had values of 20 or more.

We summarized the lithology classifications created by the Landscape Ecology Team according to three different physical qualities. The set of variables we constructed (table 4.13) reflects the degree of consolidation in the parent material (*con1-con3*), the aluminosilicate or carbonate composition (*alsi1-alsi4*), and the size or texture of the weathering products typically produced (*sdt1-sdt3*). Values for each variable were produced for each subwatershed by summing the appropriate lithology variables.

Vegetation Indices

Two measures of current vegetation were used. The first, *vmf*, was the vegetation factor generated by the Landscape Ecology Team to reflect the relative degree to which vegetation ameliorates erosion. It was based on coarse characteristics of the current vegetation. The second, *vegclus*, was derived from a statistical cluster analysis of the current vegetation information provided by the Landscape Ecology Team.

The principal reason for using a cluster analysis was to integrate the relatively complex vegetation information into a simpler metric that would be appropriate for the aquatic analysis. The vegetation data provided by the Landscape Ecology Team identified the amount of area within each subwatershed and classified according to potential vegetation type, structural stage, and cover type. The 13 potential vegetation types, 24 structural stages, and 41 cover types used to characterize vegetation resulted in 205 unique combinations within the Basin. Since many of these 205 vegetative combinations likely have similar effects on aquatic communities, we sought to combine them into a more manageable number of classes. We then used this simpler classification scheme within our cluster analysis.

First, each structural stage and cover class was rated as none, low, medium, or high, depending on the potential of the vegetation within it to grow tall enough to produce shade or large enough to produce woody debris. After rating each structural stage and cover type, a matrix was constructed that combined shade and woody debris potentials into a numerical score (table 4.14). The scores ranged from zero to 10, where zero represented no potential for wood or shade, and 10 represented maximum potential.

These numerical scores were examined for each combination of cover type, structural stage, and potential vegetation type. Structural stages and cover types within potential vegetation groups and with similar ratings were combined. This reduced the 205 combinations to 61 vegetative classes. The new vegetative classes were then ranked according to the amount of the Basin occupied by each class. Vegetative classes covering less than 100,000 hectares (0.12% of total area) were eliminated from further analysis, reducing the number of vegetative classes from 61 to 40 (appendix 4.B). A cluster analysis was performed using the 40 vegetative classes and a disjoint clustering procedure (SAS Institute Inc. 1989) that placed similar subwatersheds into one of 12 clusters. Descriptive names for the clusters were determined by looking at the relative frequency of each vegetation combination within each cluster. Eleven of the twelve clusters were dominated by a small number of combinations, making the descriptions of those clusters straightforward. The twelfth cluster contained a wide variety of combinations, with no dominant theme. This vegetation mix, and the spatial distribution of the twelfth cluster, suggested transitional areas. The 12 clusters were described as: 1) agriculture; 2) moist forest—understory reinitiation; 3) grasslands; 4) desert shrublands; 5) transitions; 6) young, dry forests; 7) aspen stands; 8) young, spruce-fir-lodgepole stands; 9) older, spruce-fir-lodgepole stands; 10) older, dry forest; 11) mountain shrublands; and 12) moist forest—stem exclusion (map 4.5).

Table 4.13— Criteria for classifying mapped lithologies. The 41 mapped lithologies were recast into three sets of variables that reflect sediment texture, aluminosilicate lithology, and degree of consolidation.

Description	Sediment Texture[1]	Aluminosilicate Content[2]	Degree of Consolidation[3]
alluvium	4	5	1
argillite and slate	1	1	2
calc-alkaline meta-volcanic	1	2	3
carbonate		4	4
calc-alkaline volcanoclastic	4	2	3
conglomerate	4	5	2
dune sand	3	1	1
glacial drift	4	5	1
felsic volcanic flow	2	1	2
mafic volcanic flow	2	3	2
granitic gneiss	1	1	3
mafic gneiss	1	3	3
granite	2	1	2
glacial ice	4	5	4
interlayered meta-sedimentary	3	2	2
alkalic intrusive	1	5	2
calc-alkaline intrusive	2	5	2
mafic intrusive	2	3	3
lake sediment and playa	3	5	1
landslide	4	5	1
loess	3	5	1
metamorphosed carbonate and shale	1	4	2
meta-conglomerate	4	5	2
meta-siltstone	1	1	2
mets-sandstone	1	1	3
mafic meta-volcanic	1	3	3
mixed carbonate and shale	1	4	2
mixed eugeosynclinal	1	2	3
mixed miogeosynclinal	1	1	3
felsic pyroclastic	4	1	3
mafic pyroclastic	4	3	3
quartzite	1	1	3
sandstone	4	1	2
mafic schist and greenstone	1	3	3
meta-sedimentary phyllite and schist	3	2	2
shale and mudstone	3	1	2
siltstone	3	1	2
tuff	3	2	2
ultramafic	1	3	3
unclassified	4	5	4
openwater	4	5	4

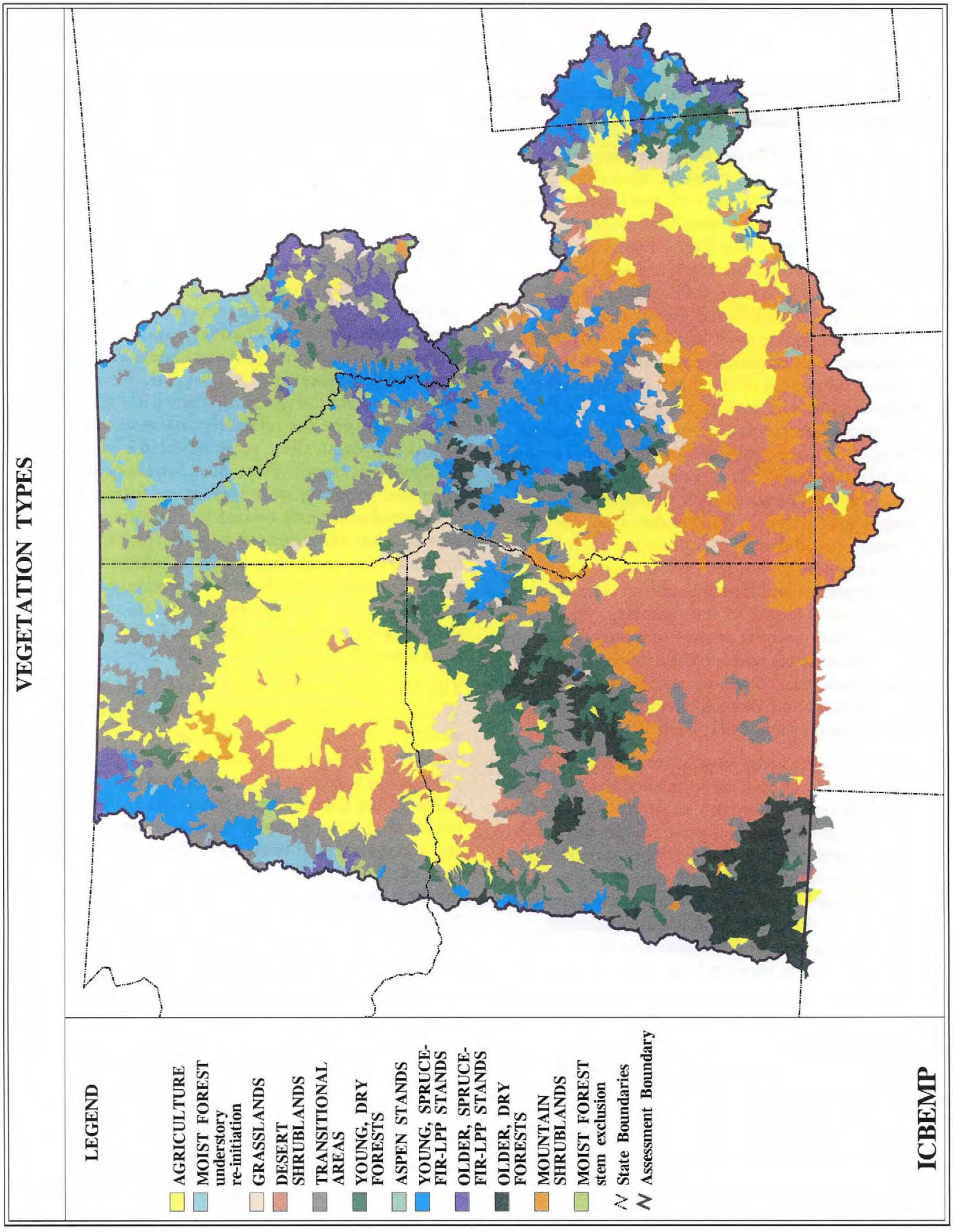
[1] 1 = Coarse; 2 = Medium; 3 = Fine; 4 = Unknown

[2] 1 = Felsic; 2 = Intermediate; 3 = Mafic; 4 = Carbonate; 5 = unknown

[3] 1 = Weak or None; 2 = Moderate; 3 = Strong; 4 = Other

Table 4.14— Potential production of woody debris and stream shading for vegetation cover classes and structural stages. On a scale of 0 to 10, 0 indicates no potential for wood or shading, 10 indicates maximum potential for wood and shading.

	Structural Stage						
Cover Class	**Other**	**Stand Initiation**	**Multi-Strata**	**Stem Exclusion**	**Understory Reinitiation**	**Old Multi- or Single Strata**	**Tall Shrub**
Other	0						
Englemann Spruce / Subalpine Fir		1	5	8	10	10	
Douglas-fir		1	5	7	9	8	
Lodgepole Pine		1	4	7	10	10	
Whitebark Pine / Alpine Larch		1	1	1	1	1	
Whitebark Pine		1	1	2	2	2	
Aspen		1	1	2	2		
Juniper Woodlands			1		2	2	
Mixed Conifer Woodlands			1	3	3	3	
Grand Fir / White Fir		1	5	7	10	8	
Ponderosa Pine (Interior)		1	5	7	9	8	
Ponderosa Pine (Pacific)			5		9	8	
Sierra Nevada Mixed			5	7	9	8	
Western Juniper / Big Sagebrush			1		2		
Mountain Hemlock		1	5	6	10	10	
Pacific Silver / Mountain Hemlock		1	5	6	10	10	
Red Fir		1	5	7	9	8	
Western Larch		1	4	7	9	8	
Western Redcedar / Western Hemlock		1	5	6	10	10	
Western White Pine		1	5	7	9	8	
Seral Shrub - Regeneration							7
Herbaceous Wetland / Shrub							7
Cottonwood/ Willow			5	9	10	10	
Limber Pine				2	2	2	



Map 4.5— Spatial distribution of 12 vegetation types identified using cluster anlaysis.

Human Factors—Ownership, Management Class, and Roads

We characterized ownership and management using two categorical variables, road density (*roaddn*) and management cluster (*mgclus*). Road density is a categorical classification of each subwatershed based on the estimated length of roads per unit area. Six levels of road density were defined: 0 to 0.02, 0.02 to 0.1, 0.1 to 0.7, 0.7 to 1.7, 1.7 to 4.7, and greater than 4.7 miles of road per square mile. Road density estimates were based on statistical relationships derived from a subsample of roughly 3 percent of the subwatersheds in the Basin; they do not reflect actual measurements and are subject to random errors that can be substantial. Such errors tended to be dampened at larger scales, thus, the estimates should be adequate for interpretation of broad-scale patterns. Further details on the derivation of road density estimates and their limitations are provided in the Information System Development and Documentation chapter of this report. Despite its shortcomings, we used road density because of the numerous direct effects that roads and activities dependent on roads have been shown to have on sediment production and hydrologic regimes (for example, Furniss and others 1991; Reid 1993).

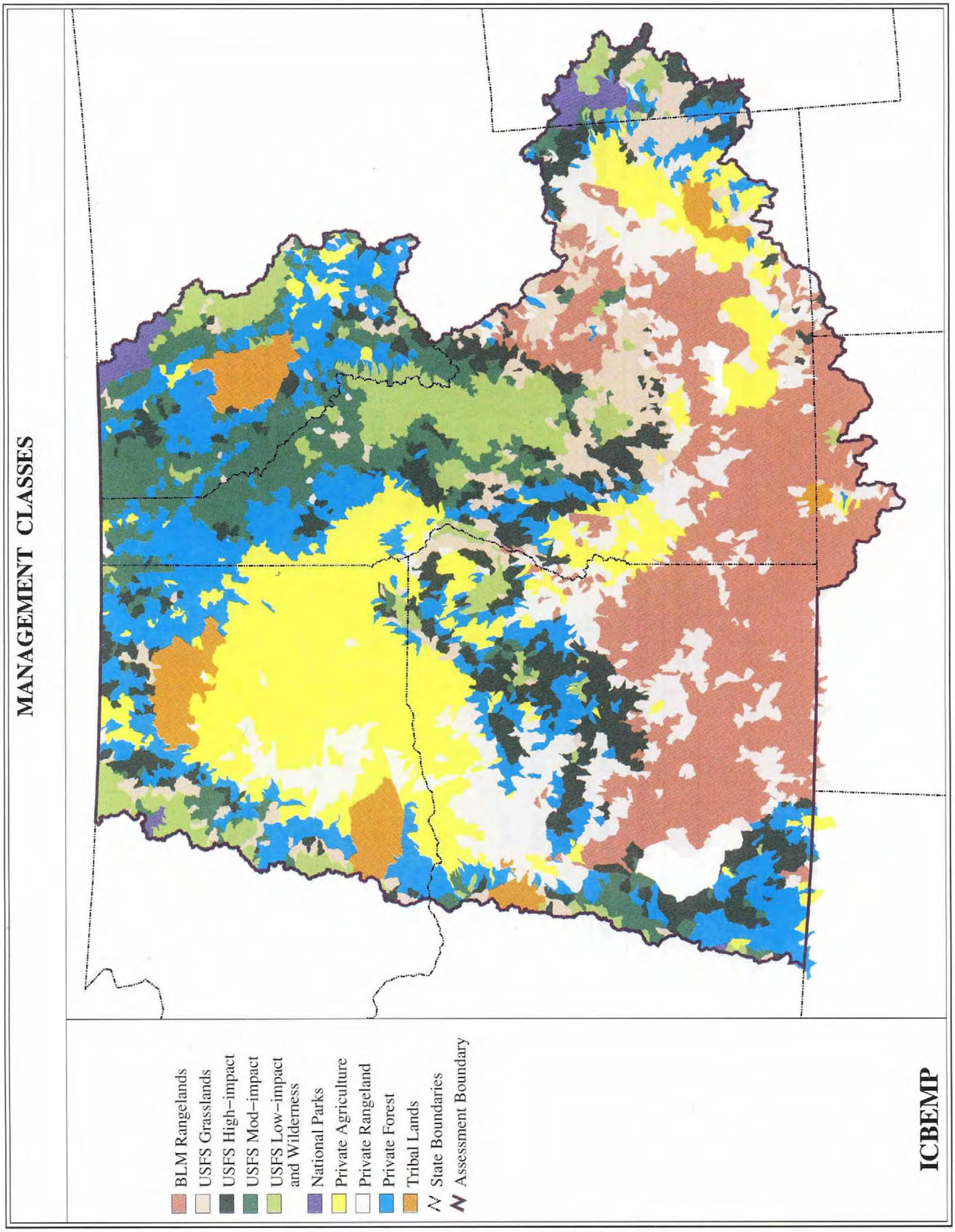
Management cluster (*mgclus*) is a categorical classification of each subwatershed that reflects past and present human use. *Mgclus* was derived from a suite of variables generated by the Landscape Ecology Team that describes various aspects of human activity and ownership. Specifically, these variables describe: 1) life form (categorization of the dominant land use as agriculture, forest, or range); 2) management area classification (unknown, managed grasslands, or forests subject to low, moderate, or high human use); 3) ownership (private, state, tribal, Forest Service, BLM, or National Park Service); 4) percentage grazed; and 5) percentage designated as wilderness. A cluster analysis procedure (SAS Institute Inc. 1989) was used to identify 10 distinctive groupings of subwatersheds with similar properties. A descriptive name and two-letter identifier was attached to each cluster based on the variable means (table 4.15). The distribution of these clusters illustrates the complex pattern of ownership and management found within the Basin (map 4.6).

Results

The simplest way to highlight possible management effects on streams was to array the mean values of stream-channel responses across ownership/management classes and road-density classes (fig. 4.12-4.14). We also constructed similar plots for each ERU (not shown). Despite the relatively coarse nature of this approach, these plots revealed a number of consistent trends. First, *pool frequency* and *large pool frequency* were highest on forested lands predominantly owned by the Forest Service, that are managed as wilderness or moderate use areas (fig. 4.12). The lowest values for pools were found on privately-owned and BLM rangelands. Pools also showed a clear decline with increasing road density.

The effect of road density on wood is less clear. Within the states of Oregon and Washington, where *R6 wood frequency* was measured, *R6 wood frequency* was higher in Forest Service lands managed as wilderness or as moderate use (fig. 4.13). In the Central Idaho Mountains, where *surface fines* was measured, privately owned forests and agricultural lands showed the highest level of surface fines; however, data are limited for these areas (fig. 4.14). The next highest levels were in high-use Forest Service lands. The effect of roads on surface fines is unclear, though the highest mean values were found in the highest road density class.

Examination of spatial patterns in the data identified differences among ERUs. These differences were not specifically related to management effects.



Map 4.6— Spatial distribution of ownership and management classes identified using cluster analysis.

Table 4.15— Management and ownership classification of subwatersheds identified using cluster analysis. Cluster means are for variables used in the analysis; all values refer to percent of area within unit.

Cluster Means																
	Land Type			Management Area Classification					Ownership							
Cluster*	Agric	Forest	Range	Unknown	1+2	3+4	5	6	Private	USFS	BLM	NPS	Tribal	Grazed	Wilderness
TL	20.7	62.9	15.6	98.9	0.2	0.2	0.5	0.2	4.5	0.9	0.3	0.0	93.3	39.8	0.3
PA	77.4	10.0	11.6	95.8	0.4	0.6	1.2	1.9	90.4	1.7	2.4	0.0	1.0	8.2	0.0
PR	14.5	9.0	75.9	76.4	2.4	1.8	5.1	13.3	60.7	5.3	19.4	0.0	1.4	55.8	0.2
FG	8.3	65.1	26.3	6.5	14.5	60.9	8.4	9.6	5.7	88.8	3.4	0.0	0.3	74.6	2.5
NP	1.1	83.4	13.8	95.1	2.6	1.1	1.2	0.0	1.5	5.1	0.0	92.1	0.0	0.9	10.1
PF	14.1	78.9	6.0	77.3	2.2	5.0	14.1	1.2	66.3	20.0	2.7	0.2	0.8	22.3	0.7
FM	2.3	88.1	9.2	9.7	9.2	21.9	58.2	0.2	8.8	88.8	0.4	0.5	0.1	3.7	3.2
BR	4.6	2.6	92.5	15.7	15.2	1.9	3.9	62.5	10.7	4.9	80.0	0.0	0.1	94.1	0.1
FW	2.4	84.2	13.1	1.5	90.9	4.6	2.7	0.1	1.0	98.2	0.1	0.5	0.0	21.6	82.1
FH	6.8	82.9	10.1	13.7	4.2	8.1	72.0	1.9	12.5	82.7	2.6	0.1	0.2	87.1	1.1

*TL = Tribal lands; PA = Private agriculture; PR = Private and BLM rangeland; FG = FS forest and rangeland, moderate impact, grazed; NP = NPS forest land; PF = Private land and FS forest land; FM = FS forest, high-moderate impact, no grazing; BR = BLM rangelands; FW = FS-managed wilderness; FH = Forest Service forest, high impact, grazed.

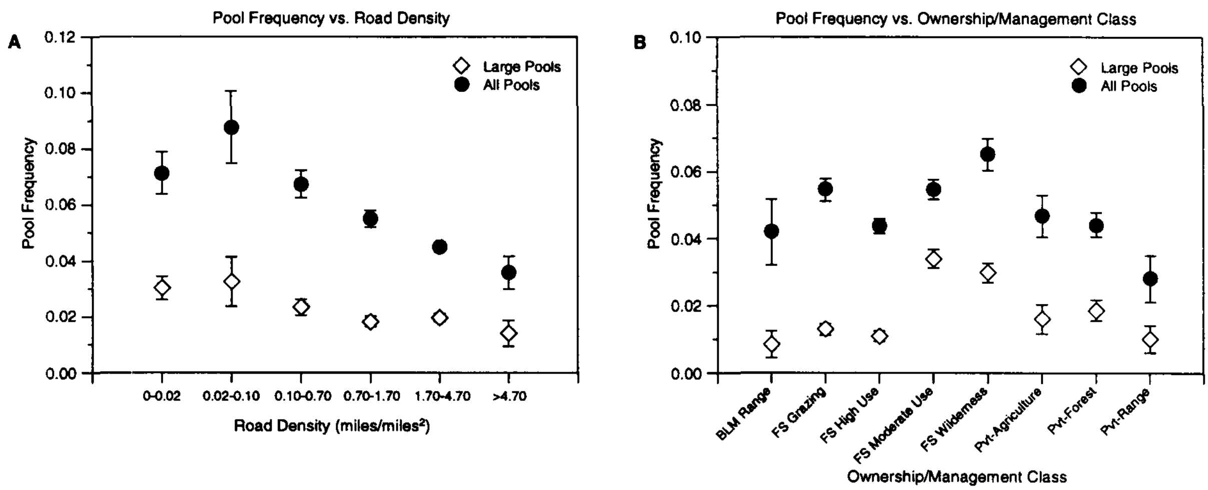


Figure 4.12— Box plots of *pool frequency* and *large pool frequency* for streams with gradients less than 0.02 arrayed by (A) road density classes and (B)ownership/management.

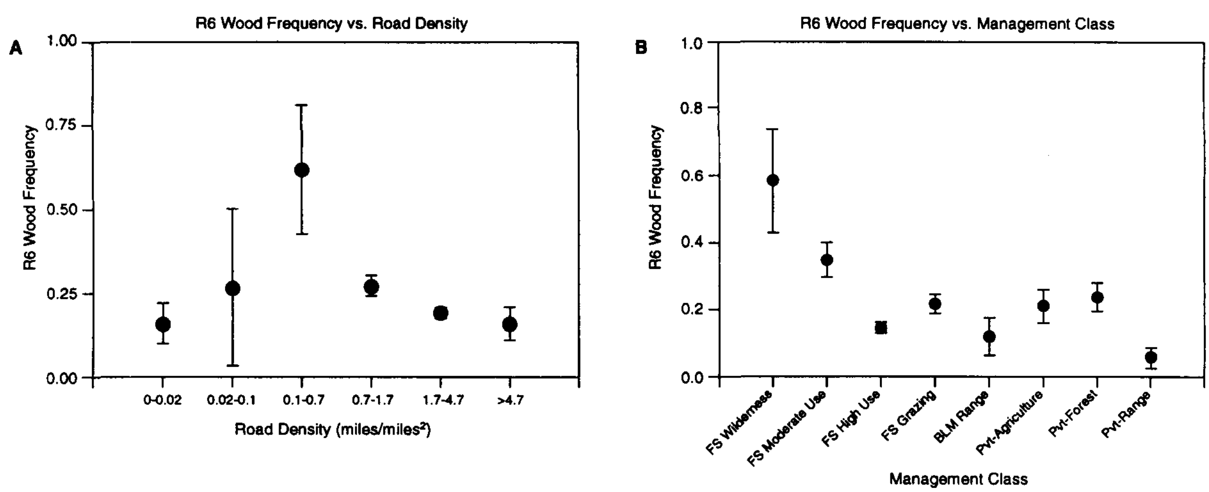


Figure 4.13— Box plots of *R6 wood frequency* for streams with gradients less than 0.02 arrayed by (A) road density classes and (B) ownership/management.

Large pool frequency: The frequency of pools with surface areas greater than 20 square meters and depth greater than 0.8 meters is generally greater in mountainous areas, particularly the Northern Glaciated Mountains, Northern Cascades, and Lower Clark Fork. Compared to other mountainous areas, streams in the Upper Klamath and Blue Mountains ERUs have noticeably fewer large pools.

Pool frequency: The frequency of pools of all sizes is also generally greater in mountainous areas, particularly in the Northern Glaciated Mountains, Upper Clark Fork, and Central Idaho Mountains. Again, *pool frequency* in the Upper Klamath and Blue Mountains ERUs is lower and has values similar to those reported for the Northern Great Basin, Columbia Plateau, and Owyhee Uplands.

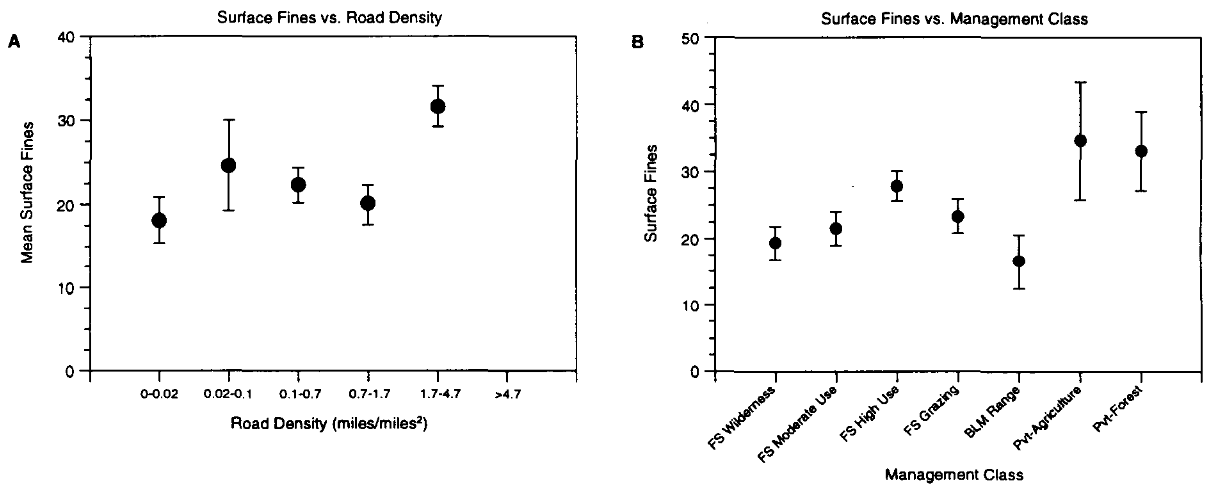


Figure 4.14— Box plots of *surface fines* for streams with gradients less than 0.02 arrayed by (A) road density classes and (B) ownership/management.

R6 wood frequency: Wood is most abundant in Cascade Range streams, where wood frequency is greatest for steep and low-order streams. Median wood frequency values for high-order and low-slope streams in the Northern Cascades and Southern Cascades range from 0.3 to 0.4 pieces per channel width. The lowest values for wood frequency are in the Northern Great Basin, Columbia Plateau, and Blue Mountains, where for high-order and low-slope streams, median values for wood frequency are less than 0.1 pieces per channel width.

R1/4 wood frequency: Streams located in the Central Idaho Mountains apparently have less wood (median values of about 0.2 pieces per channel width) than streams in the Northern Glaciated region, where median values range from 0.3 to 0.7 pieces per channel width. In the Central Idaho Mountains, wood frequency decreases with stream order, a result that contrasts with streams in the Northern and Southern Cascades.

Surface fines: The areal extent of riffles and pool tails covered with fine sediment (clasts with diameters less than 6 mm) was extensively measured only in the Central Idaho Mountains and Lower Clark Fork. For both ERUs, surface-fine coverage was generally greater in the lower slope classes. The Central Idaho Mountains has higher mean values, and median values range from 14 to 35 percent, compared to 5 to 15 percent for the Lower Clark Fork.

Quantitative Analysis

Summarizing the data as we have in figures 4.12-4.14 grossly understates the complexity of interactions between land and streams. We know from examining the data that apparent differences can be found in almost any response variable under a variety of stratification schemes, all of them reasonable. For example, we can show differences in pool frequency due to stream order and slope class. We can also show that responses vary greatly from one ERU to another, or that vegetation and lithology correlate with differences in stream features. None of these results are particularly surprising; each is consistent with a large literature on geomorphological processes. The thorny analytical problem is not distinguishing coarse landscape differences; rather, it is separating the actual response due to management from the variation introduced by a host of confounding factors.

There are several possible approaches for trying to separate management effects. One approach is to stratify the assessment area into more homogeneous units based on geophysical settings. Such a stratification scheme could start with ERUs, and sequentially subdivide smaller and smaller areas based on geomorphic similarities. The problem with this approach is that as the assessment area is divided into progressively smaller units, the amount of data available for use within a given stratum decreases. Since we can make inferences only for those areas where we have data, the areal extent of our inferences would be quite limited. It is not clear how one would reasonably extrapolate to unsampled areas. An alternative approach is to build a statistical model that uses a large set of covariates, both continuous and categorical, to account for the confounding factors. This approach has the usual problems of model specification and selection, and of dealing with potentially large numbers of interactions.

We chose an approach that includes elements of both the stratification scheme and the statistical modeling approach. We used ERU as a broad-scale spatial stratification, and also used stream order and slope class as more physiographic stratifiers to classify all reaches within the Basin. We then constructed a general linear model that allowed us to account for variations induced by geoclimatic and biophysical factors, and test for management effects within a given stratum, depending on the response variable. *Large pool frequency* and *pool frequency* were stratified by slope class and order class, respectively, in two separate, Basin-wide analyses. *Wood frequency* and *surface fines* were examined Basin-wide, and then again by ERU. In each analysis, landscape variables that described geophysical setting and vegetation were entered into the linear model first. Management cluster, then road-density class were entered last, and the procedure was repeated with the order of entry for management cluster and road density reversed. By examining the sequential reduction in unexplained variation, we were able to statistically test the effects of management cluster and road density after adjusting for other factors.

The partial stratification by ERU, stream order, or slope class was necessary because streams from different strata respond differently to environmental conditions and disturbances. Grant and others (1990), Montgomery and Buffington (1993), and Rosgen (1994) proposed various stream classification systems applicable to higher gradient channels. Both the Montgomery and Buffington (1993) and Rosgen (1994) classification systems are essentially considerations of channel confinement. Confined channels are those with boundaries (beds, banks, and floodplains) that are resistant to erosion. They commonly have beds and banks composed of bedrock, colluvium, or relict and relatively immobile alluvium. The bed structure of confined channels generally consists of bedrock outcrops, rapids, cascades, and steps separated by short pools. Confined channels are generally stable, with most introduced sediment flushed rapidly downstream. Bed and bank materials are only mobilized by rare and large events (Grant and others 1990; Lisle 1986). Montgomery and Buffington (1993) referred to these confined channels as "source" or "transport" channels.

Unconfined channels are "authors of their own geometry" (Leopold and Maddock 1953). Unconfined streams commonly have floodplains that are frequently inundated and have alluvial banks and beds that are readily mobilized. Unconfined channels are commonly meandering or braided in planform, and have pool-riffle or plane-bed longitudinal profiles. Unconfined channels, referred to as "response reaches" by Montgomery and Buffington (1993), are more sensitive to disturbance than confined channels and may be particularly sensitive to cumulative effects of upstream processes and disturbances that are rapidly transmitted downstream through confined reaches (Montgomery and Buffington 1993; Rosgen 1994).

Because of the diverse stream inventory protocols within the Basin, it was not possible to use a consistent measure of confinement for all the reaches. Consequently, channel slope was used as an approximate proxy for confinement and we used slope classes of "low" (gradient <0.02), "medium" (0.02 to <0.04), and "high" (≥0.04). These slope classes generally are consistent with process-based classification systems (Montgomery and Buffington 1993; Rosgen 1994) that distinguish streams of distinct morphology and channel forming processes. Streams with gradients less 0.02 are generally unconfined and have pool-riffle or plane-bed morphologies; streams with slopes ≥0.04 are generally confined and have step-pool or cascade bed structures; and streams with slopes between 0.02 and 0.04 have an intermediate character with pool-riffle, plane-bed, or step-pool bed structures, depending on local conditions. While accurately measured slope probably is a good indicator of stream type, the quality of the slope information associated with the stream inventories is inconsistent. Some slopes were calculated from maps, while some were measured in the field. Field measurements were performed by a variety of techniques, ranging from hand-held clinometer sightings of single channel units to reach-long instrument surveys. Of the 6,352 reaches, 144 could not be classified because there was no slope information.

In part because of the unreliability of the slope measurements, we also classified streams by stream order (Horton 1945; Strahler 1957). Stream order serves as a proxy for drainage area; streams with low-order values within a basin generally have smaller drainage areas than those with larger stream orders. Drainage area has been shown to be an important discriminant in evaluating channel form (Dunne and Leopold 1978; Leopold and others 1964) and responses to human disturbance (Overton and others 1993). Classifying streams solely on the basis of stream order for the purpose of evaluating stream processes has been discouraged (Montgomery and Buffington 1993) because actual and artificial differences in mapped drainage network density and pattern affect the calculated stream order in a manner that probably does not relate to channel form and function. Nevertheless, we established stream-order classes to stratify the analysis because: 1) stream order probably serves as a better indicator of drainage area than slope; 2) we are unsure of the reliability of the measurements used to classify the streams by slope class; 3) every inventoried stream reach can be classified; and 4) the use of stream-order classes in conjunction with the 1:100,000 hydrography offers the only means for assessing sample completeness. Streams were grouped into three classes: order 0 to 1, order 2 to 3, and order greater than or equal to 4. Stream orders of zero refer to intermittent streams or unconnected segments on the 1:100,000 hydrography. Inspection of the 1:100,000 hydrography indicates that most of these zero-order streams on federally administered lands are headwater tributaries, and are appropriately grouped with the first-order streams.

Analytical Results

Despite the aforementioned limitations in the data, the channel morphology variables showed strong and consistent patterns of response to several of the landscape variables for all stream stratifications. Moreover, after accounting for the variance explained by landscape variables, the channel morphology variables displayed significant differences among management clusters and/or road density classifications. Some of the patterns that emerged from this analysis are presented in the following paragraphs.

Differences in *large pool frequency* and *pool frequency* were partially explained by slope, lithology, and vegetation. This was consistent with the obvious and well-established tenet that channel morphology is strongly influenced by local physiography and vegetation (Grant 1990; Hack 1957; Montgomery and Buffington 1993; Rosgen 1994). Commonly, the low-slope and high-order streams (generally the larger and unconfined streams) responded differently (in the opposite direction) than high-slope and low-order streams to slope and lithology variables.

For example, for low-slope and high-order streams, as the average slope of the subwatershed increased, *pool frequency* decreased. The response was distinctly different for steeper and low-order streams, where as average watershed slope increased, *pool frequency* increased.

After accounting for other landscape features, management cluster and road density had significant impacts on *pool frequency* and *large pool frequency* for almost all order and slope classifications. The only exceptions were for *large pool frequency* on medium-slope streams. For streams of order 4 and higher, management cluster and road density accounted for almost 5 percent of the total variance in *pool frequency*. For *large pool frequency* and *pool frequency* in low-slope and order 4 and higher streams, human disturbance factors accounted for twice as much of the variance as they did for the pool frequency variables in steeper and low-order streams. This is consistent with the hypothesis that low-gradient channels are more sensitive to human disturbance (Montgomery and Buffington 1993; Rosgen 1994). In general, reversing the order in which management cluster and road density were entered into the model did not substantially affect the resulting significance levels, indicating that there was little interaction between these two predictor variables.

For five of the seven ERUs in which *R6 wood frequency* was summarized, vegetation (*vegclus*) significantly affected wood frequency. In the Blue Mountains, Upper Klamath, and Columbia Plateau, precipitation had a highly significant effect on wood frequency. *R6 wood frequency* was significantly affected by management cluster and road density in the Northern Cascades, Southern Cascades, Columbia Plateau, Northern Glaciated Mountains, and Blue Mountains. *R6 wood frequency* was not significantly associated with management cluster and road density in the Upper Klamath, and there were insufficient data to evaluate the Northern Great Basin.

Only the Lower Clark Fork and Central Idaho Mountains had sufficient data to construct generalized linear models for *surface fines*. In the Lower Clark Fork, *surface fines* was affected by rock type, but was not significantly affected by management cluster or road density. In the Central Idaho Mountains, *surface fines* was significantly affected by topography, lithology, vegetation, and management/ownership.

Summary

The results of the generalized linear modeling reinforce the general trends portrayed in figures 4.12-4.14 and the historical pool analysis of McIntosh (1995). The evidence is strong; streams within the Basin have been significantly affected by human activities. Most notably, pool frequency (large pools and all pools) is inversely correlated with road density and management intensity. This result is similar to that of other studies of smaller regions west of the Cascade Range (Ralph and others 1994) and in Alaska (Smith and Buffington, in press), where pool frequency has been shown to decline in harvested forests. The negative correlation between pool frequency and management intensity is also corroborated by the repeat surveys of Columbia River Basin streams.

Pools are a fundamental aspect of fish habitat (Bjornn and Reiser 1991). Understanding the sensitivity of pool frequency, especially of large pools, to physical factors and human disturbance is critical to developing management strategies designed to maintain or enhance aquatic conditions. Pools provide rearing habitat for juvenile fish, resting places, overwintering areas, and refugia from floods, drought, and extreme temperatures (Sedell and others 1990). The loss of pools, especially deep pools, substantially and adversely affects fish habitat. Loss of pools in the Basin may be especially critical because of the great extremes of temperature and flow compared to streams west of the Cascade crest (Henjum and others 1994; Wissmar and others 1994a).

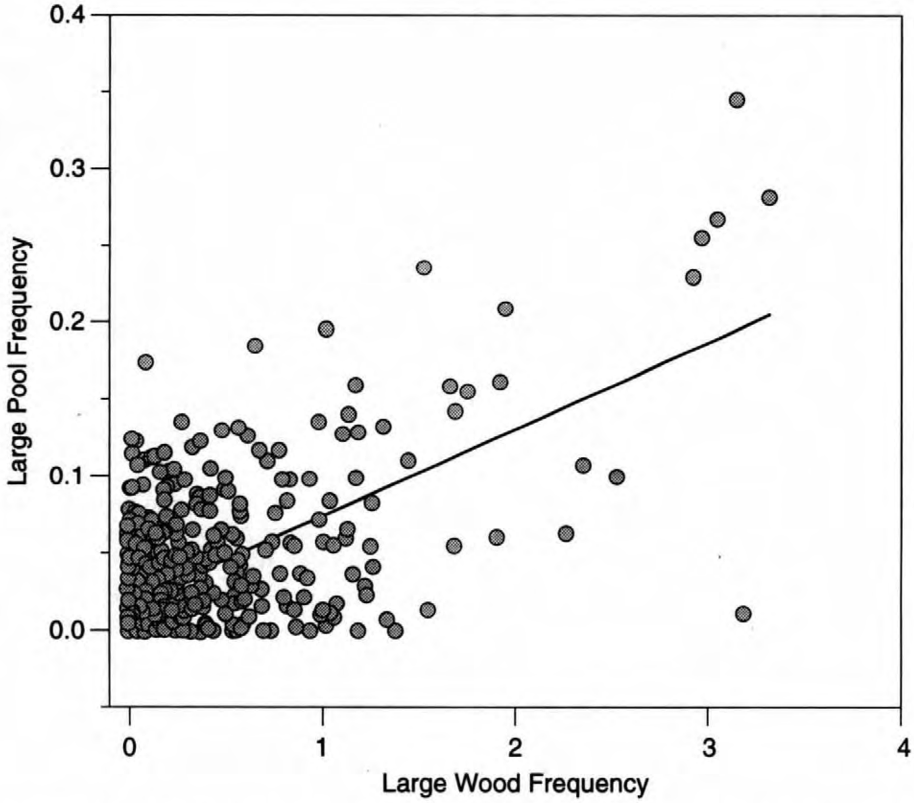


Figure 4.15— Scatter plot of *large-wood frequency* vs. *large pool frequency*. Large wood frequency is the number of large wood pieces measured using the Forest Service Region 6 (Oregon and Washington) protocol per average channel width. Large wood pieces are those with diameters greater than 50 cm and lengths greater than 10.6 m (35 ft). Large pools are those with areas greater than 20 m^2 and depths greater than 0.8 m. The total number of observations (number of reaches) is 646, $r^2 = 0.37$.

A factor likely to be important in controlling pool frequency in the Basin is the abundance of in-stream wood. Wood forms pools by causing hydraulic obstructions and forcing local scour, especially for larger and low-gradient channels (Bilby and Ward 1989; Cordova 1995; Keller and Swanson 1979; Montgomery and others 1995). In the southern portion of the Blue Mountains, 80 percent of large pools in unconstrained reaches are directly associated within stream wood (Cordova 1995). Wood frequency for streams inventoried with the Forest Service Region 6 protocol varies greatly among ERUs, but in mountainous areas, median values range between 0.1 and 0.4, equivalent to 1-2.5 pieces per 10 channel widths, substantially less than wood frequencies in wetter forested regions (Montgomery and others 1995; Ralph and others 1994; Smith and Buffington, in press). Less abundant wood may partly explain why overall large pool frequency is lower in the assessment area relative to streams west of the Cascade crest (McIntosh 1995).

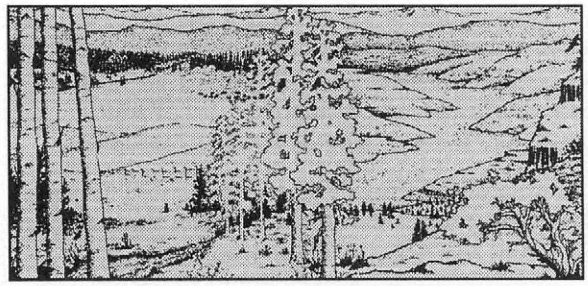
The significant relationship seen in the generalized linear modeling between vegetation factors and pool frequency also points to the importance of wood to pool formation. This observation is further supported by the correlation between wood frequency and pool frequency throughout the assessment area, but most notably between occurrence of large wood and large pools on low-gradient streams (figure 4.15). This is consistent with Cordova's (1995) analysis that showed that in the southern Blue Mountains, wood abundance was important in influencing pool frequency in all types of channels, but especially in unconfined streams.

Wood abundance in streams, in terms of both size and frequency, has been shown to be significantly impacted by many timber harvesting practices (Bilby and Ward 1991; Montgomery and others 1995; Ralph and others 1994). This is also apparently the case in the Columbia River Basin (McIntosh and others 1994b), where wood frequency in high- and moderate-use forests is less than that for wilderness areas, especially in low-gradient streams. Wood is also important for reasons besides pool formation. Wood effectively stabilizes channels, influences sediment routing, and provides a major component of the instream organic matter (Pearsons and others 1992). Wood also provides cover for fish and habitat for some invertebrates, and increases overall channel complexity (Bilby and Ward 1991; Henjum and others 1994). All of these factors call for special attention to protecting sources of instream wood for Columbia basin streams. There is a relatively small amount of wood; it plays a critical role for pool formation and habitat conditions; and wood frequency is sensitive to management practices.

Another important aspect of habitat quality that apparently is influenced by management is the amount of fine sediment (sediment less than 6 mm) on channel beds. Elevated levels of fine sediment adversely affect salmonid embryo survival by impeding intergravel flow and reducing oxygen in redds (Bjornn and Reiser 1991; Chapman 1988; Everest and others 1987). Several studies have shown that forest management practices, especially road building and ground disturbance, can substantially increase sediment flux into streams (reviewed by Everest and others 1991; Furniss and others 1991; Henjum and others 1994; Hicks and others 1991; Reid 1993). Increased sediment loads in streams also cause aggradation, filling of pools, and increased channel widths and width/depth ratios (Madej 1982). Fine sediment has only been consistently measured in the Central Idaho Mountains, where stream sedimentation has been perceived as a problem for several decades (Megahan and Kidd 1972; Platts and Megahan 1975; Platts and others 1989). The results of the generalized linear modeling, where road density significantly affects surface fines, corroborates the link between forest management practices and channel sediment characteristics.

An aspect of channel conditions in the Basin that has not been explicitly evaluated in this analysis is the role of low-frequency disturbance events in controlling channel morphology (Swanston 1991). Low-frequency events such as large floods, mass movements, and fire can profoundly affect stream channels by introducing and/or mobilizing large quantities of sediment, thereby altering bed structure and channel form in manners that can persist for decades or hundreds of years (Baker 1977, Reeves and others 1995). Human activities in watersheds can strongly influence the timing and magnitude of natural events; for example, clearcutting and other watershed disruptions are linked to increased water yields, bedload movement, more frequent flooding and scour events, and channel instability (Rieman and McIntyre 1993). Consequently, streams in similar settings can have very different channel conditions. These differences are the result of varied histories of natural and anthropogenic events. A large portion of the unexplained variance in the general linear models is probably due to the past sequence of natural and human events and their effects on channel morphology. An example is Rapid River, a roadless watershed in central Idaho, where several fires over the last few decades have resulted in rapid wood delivery to the stream—present wood frequency is about 3 pieces per channel width. This wood will continue to affect channel morphology for several decades. We specifically point this out because the dynamic aspect of channels is difficult to evaluate in empirical models such as those presented here. Moreover, potential long-lasting perturbations to channel conditions need to be considered when considering long-term management strategies and evaluating risk to fish and fish habitat on a regional scale.

DISTRIBUTION AND STATUS OF FISHES



Native fish communities are an integral element in the composition, structure, and function of aquatic ecosystems. Fish typically dominate the aquatic vertebrates. They are sensitive to disturbance, and potentially integrate the effects of landscape and watershed processes over large spatial and temporal scales. Fish have influenced the development, status, and success of human social and economic systems. They also can be important pathways for nutrient and energy flows between aquatic and terrestrial systems. Even in waters historically barren of fish, introduced fishes profoundly influence the structure of aquatic communities (Bahls 1992). In this section, we focus on the distribution, structure, and status of fish assemblages and select, or key, fish species across the assessment area.

Many species of fish and other aquatic biota in the Basin are considered imperiled (Frissell and others 1993, 1995; Moyle and Williams 1990; Nehlsen and others 1991; Williams and others 1989). Presently at least 45 of the Basin's 88 native fish taxa are identified as threatened, endangered, sensitive, or of special concern by state or federal agencies or by the American Fisheries Society. Ten of those fishes are formally listed as threatened or endangered under the Endangered Species Act (ESA) of 1973 (PL 93-205, as amended) (table 4.16) by the National Marine Fisheries Service (NMFS) (anadromous salmonids) or by the U.S. Fish and Wildlife Service (USFWS) (non-anadromous fishes). Two others are designated as Category 1 species by the Federal agencies with listing considered warranted but precluded by other priorities. A single species is under review for listing by NMFS.

Status reviews exist for anadromous salmonids [Columbia Basin Fish and Wildlife Authority (CBFWA) 1990; Chapman and others 1994a, 1994b; Didier and Anderson, in preparation; Howell and others 1985a, 1985b; Huntington and others 1994; Idaho Department of Fish and Game (IDFG) and others 1990; Kostow and others 1994; Nehlsen and others 1991; Northwest Power Planning Council (NWPPC) 1986; Washington Department of Fisheries (WDF) and others 1993] and several non-anadromous salmonids (Liknes 1984; Rieman and Apperson 1989; Thomas 1992; Young 1995). These reviews typically focused on high-profile salmonids in specific geographic areas. Many reviews used different methods with varied resolution, making a synthesis across the Basin difficult. Most reviews summarized known presence and abundance, but provided little information on the absence of species. The lack of detailed inventories limits our understanding of factors affecting the distribution or persistence of many species. Frissell (1993) provided the most complete broad-scale analysis of native fish assemblages within the Pacific Northwest and California. Frissell's analysis provided little resolution below the scale of major river subbasins, however, and was based primarily on published records of broad, well-documented extinctions of native fishes. We are aware of no attempt to document the distribution or influence of non-native fishes at scales larger than individual watersheds or subbasins.

Our goal was to provide a consistent evaluation of the status and distribution of all fishes throughout the Basin, organized by three primary objectives:

Objective 1—Evaluate the current condition of aquatic ecosystems by summarizing the current distribution of all fishes and the status of select species. Existing reports within the Basin do not adequately reflect the collective (and generally unpublished) knowledge of agencies and individuals familiar with the status and distribution of fishes. Our intent was to capture as much relevant information and understanding as possible. We addressed both complete assemblages (all native and introduced species) and selected several species as indicators of aquatic ecosystem health and integrity.

Objective 2—Identify unique or important areas in terms of fish species diversity and the integrity and composition of fish communities. Ecosystem management includes goals to conserve the structure, function, and resilience of ecosystems by maintaining or restoring processes that influence those characteristics (Grumbine 1994). A starting point may be the conservation of aquatic systems that retain, sensitive species and biological diversity. Recognition of important subbasins and watersheds and the connecting corridors is a first step.

Objective 3—Identify linkages between the status and distribution of fishes and other components of the landscape, including land management and anthropogenic disturbance. We explored the relationships between fish distributions and status and landscape features using information provided by the landscape ecology group. We used the emergent patterns to predict distributions of fish in unsampled areas of the Basin and to identify influential landscape features. We also summarized current knowledge of the physical, biological, and anthropogenic factors or processes that are known to influence population dynamics and distribution patterns of selected species.

Our analysis focused on the distribution and status of native fishes. Most introductions of non-native species or of species native to the Basin but outside their historical ranges (collectively called "introduced" species) were intended to create or expand sport fishing opportunity. Such efforts began in the 1800s (Evermann 1893; Simpson and Wallace 1978). Introduced salmonids, centrarchids and percids now support much, if not most, of the sport-fishing opportunities throughout the Basin. Introduced species are important, and likely permanent, components of the aquatic ecosystems. Their role in sport fisheries implies a large social and economic influence. The widespread occurrence of many introduced species suggests that they are also well adapted to the altered conditions of aquatic environments prevalent throughout the Basin. They tend to be less sensitive to disturbance associated with land management than native fishes, and are more prevalent in waters further removed from the direct effects of change on Federal lands (Marcot and others 1994). Therefore, native fish species provide the more sensitive measure of aquatic ecosystem condition. Management and conservation efforts on Federal lands that maintain or restore the distribution and integrity of native fishes, however, can also benefit desirable non-native species.

A total of 143 fish taxa was reported within the Basin. Table 4.16 lists common and scientific names for each species, subspecies, race or life-history form. In the remainder of this document, we use common names.

Table 4.16—Relative abundance of native and introduced species. The columns "Freq" and "Pct" refer to the frequency reported by watershed and percentage of reported watersheds. Common and scientific names of fishes follow accepted nomenclature of the American Fisheries Society (Robins and others 1991).

ID	Rank	Group rank	Asc[1]	Conservation[2] status	Common name	Freq	Pct	Family name	Scientific name
Native Species									
1	36	24	Y	1,2,3	White sturgeon	141	6.3%	Acipenseridae	*Acipenser transmontanus*
2	50	34	Y		Utah sucker	79	3.6%	Catostomidae	*Catostomus ardens*
3	23	17	Y		Longnose sucker	292	13.1%	Catostomidae	*Catostomus catostomus*
4	14	12	Y		Bridgelip sucker	548	24.7%	Catostomidae	*Catostomus columbianus*
4A[3]	—	—	E		Wood River bridgelip sucker	—	—	Catostomidae	*Catostomus columbianus hubbsi*
5	73	50			Bluehead sucker	22	0.:%	Catostomidae	*Catostomus discobolus*
6	8	6	Y		Largescale sucker	776	34.9%	Catostomidae	*Catostomus macrocheilus*
7	96	69			Sacramento sucker	7	0.3%	Catostomidae	*Catostomus occidentalis*
8	97	70	E	3	Goose Lake sucker	7	0.3%	Catostomidae	*C. occidentalis lacusanserinus*
9	37	25	Y		Mountain sucker	133	5.:%	Catostomidae	*Catostomus platyrhynchus*
10	92	66			Klamath smallscale sucker	8	0.4%	Catostomidae	*Catostomus rimiculus*
11	64	42		3	Klamath largescale sucker	34	1.5%	Catostomidae	*Catostomus snyderi*
12	30	21			Sucker, generic	219	9.9%	Catostomidae	*Catostomus sp.*
13	121	86			Tahoe sucker	1	0.0%	Catostomidae	*Catostomus tahoensis*
14	86	61	E	1,2	Warner sucker	10	0.4%	Catostomidae	*Catostomus warnerensis*
15	74	51	E	1,2	Shortnose sucker	20	0.9%	Catostomidae	*Chasmistes brevirostris*
16	87	62	E	1,2	Lost River sucker	10	0.4%	Catostomidae	*Deltistes luxatus*
17	93	67			Coastrange sculpin	8	0.4%	Cottidae	*Cottus aleuticus*
18	69	46			Prickly sculpin	26	1.2%	Cottidae	*Cottus asper*
19	34	23	Y		Mottled sculpin	154	6.9%	Cottidae	*Cottus bairdi*
20	66	44	E	3	Malheur sculpin	33	1.5%	Cottidae	*Cottus bairdi ssp.*
21	41	29	Y		Paiute sculpin	102	4.6%	Cottidae	*Cottus beldingi*
22	29	20	Y		Slimy sculpin	227	10.2%	Cottidae	*Cottus cognatus*
23	54	37	3		Shorthead sculpin	72	3.2%	Cottidae	*Cottus confusus*
24	111	79	E	3	Shoshone sculpin	3	0.1%	Cottidae	*Cottus greenei*
25	65	43			Riffle sculpin	34	1.5%	Cottidae	*Cottus gulosus*
26	79	55			Marbled sculpin	14	0.6%	Cottidae	*Cottus klamathensis*
27	68	45	E	3	Wood River sculpin	27	1.2%	Cottidae	*Cottus leiopomus*
28	75	52	E	3	Margined sculpin	19	0.9%	Cottidae	*Cottus marginatus*
29	70	47			Reticulate sculpin	24	1.1%	Cottidae	*Cottus perplexus*
30	108	76	E	3	Pit sculpin	4	0.2%	Cottidae	*Cottus pitensis*
31	98	71	E		Klamath Lake sculpin	7	0.3%	Cottidae	*Cottus princeps*
32	46	32	Y	3	Torrent sculpin	90	4.0%	Cottidae	*Cottus rhotheus*
33	89	64	E	3	Slender sculpin	9	0.4%	Cottidae	*Cottus tenuis*
34	2	1			Sculpin, generic	1299	58.4%	Cottidae	*Cottus sp.*
35	76	53			Pacific staghorn sculpin	19	0.9%	Cottidae	*Leptocottus armutus*
36	18	15	Y		Chiselmouth	379	17.0%	Cyprinidae	*Acrocheilus alutaceus*
37	82	58	E	3	Alvord chub	12	0.5%	Cyprinidae	*Gila alvordensis*
38	57	38			Utah chub	62	2.8%	Cyprinidae	*Gila atraria*
39	47	33	Y		Tui chub	88	3.:%	Cyprinidae	*Gila bicolor*
40	106	75	E	3	Sheldon tui chub	5	0.2%	Cyprinidae	*Gila bicolor eurysoma*
41	112	80	E	3	Oregon Lakes tui chub	3	0.1%	Cyprinidae	*Gila bicolor oregonensis*
42	90	65	E	3	Catlow tui chub	9	0.4%	Cyprinidae	*Gila bicolor ssp.*
43	122	87	E	1,2	Hutton tui chub	1	0.0%	Cyprinidae	*Gila bicolor ssp.*
44	113	81	E	3	Summer Basin tui chub	3	0.1%	Cyprinidae	*Gila bicolor ssp.*
45	103	73	E	3	Warner Basin tui chub	6	0.3%	Cyprinidae	*Gila bicolor ssp.*
46	123	88	E	3	XL Spring tui chub	1	0.0%	Cyprinidae	*Gila bicolor ssp.*
47	104	74	E	3	Goose Lake tui chub	6	0.3%	Cyprinidae	*Gila bicolor thallassina*
48	124	89	E	1,2	Borax Lake chub	1	0.0%	Cyprinidae	*Gila boraxobius*
49	99	72			Blue chub	7	0.3%	Cyprinidae	*Gila coerulea*
50	81	57	3		Leatherside chub	13	0.6%	Cyprinidae	*Gila copei*
51	83	59	3		Northern roach	12	0.5%	Cyprinidae	*Hesperoleucus symmetricus mitrulus*
52	24	18	Y		Peamouth	291	13.1%	Cyprinidae	*Mylocheilus caurinus*
53	7	5	Y		Northern squawfish	801	36.0%	Cyprinidae	*Ptychocheilus oregonensis*
54	116	82			Umpqua squawfish	2	0.1%	Cyprinidae	*Ptychocheilus umpquae*

Table 4.16 (continued).

ID	Rank	Group rank	Asc[1]	Conservation[2] status	Common name	Freq	Pct	Family name	Scientific name
55	15	13	Y		Longnose dace	531	23.9%	Cyprinidae	*Rhinichthys cataractae*
56	71	48			Leopard dace	23	1.0%	Cyprinidae	*Rhinichthys falcatus*
57	11	9	Y		Speckled dace	665	29.9%	Cyprinidae	*Rhinichthys osculus*
58	109	77	E		Klamath speckled dace	4	0.2%	Cyprinidae	*Rhinichthys osculus klamathensis*
59	125	90	E	1,2	Foskett speckled dace	1	0.0%	Cyprinidae	*Rhinichthys osculus ssp.*
60	12	10			Dace, generic	663	29.8%	Cyprinidae	*Rhinichthys sp.*
61	6	4	YD		Redside shiner	962	43.3%	Cyprinidae	*Richardsonius balteatus*
62	117	83			Lahontan redside	2	0.1%	Cyprinidae	*Richardsonius egregius*
63	40	28	Y		Shiner perch	107	4.8%	Embiotocidae	*Cymatogaster aggregata*
64	53	36		2	Burbot	74	3.3%	Gadidae	*Lota lota*
65	61	40			Threespine stickleback	45	2.0%	Gasterosteidae	*Gasterosteus aculeatus*
66	77	54	E	3	Sand roller	17	0.8%	Percopsidae	*Percopsis transmontana*
67	126	91			River lamprey	1	0.0%	Petromyzontidae	*Lampetra ayresi*
68	80	56			Pit-Klamath brook lamprey	14	0.6%	Petromyzontidae	*Lampetra lethophaga*
69	118	84	E		Miller Lake lamprey	2	0.1%	Petromyzontidae	*Lampetra minima*
70	63	41			Western brook lamprey	42	1.9%	Petromyzontidae	*Lampetra richardsoni*
71	85	60	E		Klamath lamprey	11	0.5%	Petromyzontidae	*Lampetra similis*
72	38	26	Y	2	Pacific lamprey	128	5.8%	Petromyzontidae	*Lampetra tridentata*
73	119	85	E	3	Goose Lake lamprey	2	0.1%	Petromyzontidae	*Lampetra tridentata ssp.*
74	32	22	YD	3	Yellowstone cutthroat trout	188	8.5%	Salmonidae	*Oncorhynchus clarki bouvieri*
75	60	39		3	Coastal cutthroat trout	48	2.2%	Salmonidae	*Oncorhynchus clarki clarki*
76	39	27	Y	1,2	Lahontan cutthroat trout	111	4.:%	Salmonidae	*Oncorhynchus clarki henshawi*
77	10	8	YD	3	Westslope cutthroat trout	737	33.2%	Salmonidae	*Oncorhynchus clarki lewisi*
78	28	19			Cutthroat trout, generic	242	10.9%	Salmonidae	*Oncorhynchus clarki spp.*
79	94	68		3	Chum salmon	8	0.4%	Salmonidae	*Oncorhynchus keta*
80	51	35	Y	3	Coho salmon	79	3.6%	Salmonidae	*Oncorhynchus kisutch*
81	3	2	Y	3	Interior redband trout	1244	55.:%	Salmonidae	*Oncorhynchus mykiss gibbsi*
82	13	11	Y	3	Summer steelhead	649	29.2%	Salmonidae	*Oncorhynchus mykiss mykiss*
83	42	30	Y	3	Winter steelhead	102	4.6%	Salmonidae	*Oncorhynchus mykiss mykiss*
84	110	78		3	Catlow Valley redband trout	4	0.2%	Salmonidae	*Oncorhynchus mykiss ssp.*
85	88	63		3	Warner Valley redband trout	10	0.4%	Salmonidae	*Oncorhynchus mykiss ssp.*
86	22	16	Y	1,2	Sockeye (kokanee) salmon	294	13.2%	Salmonidae	*Oncorhynchus nerka*
87	43	31	Y	1,2	Ocean-type chinook salmon	100	4.5%	Salmonidae	*Oncorhynchus tshawytscha*
88	17	14	Y	1,2	Stream-type chinook salmon	444	19.:%	Salmonidae	*Oncorhynchus tshawytscha*
89	72	49		3	Pygmy whitefish	23	1.0%	Salmonidae	*Prosopium coulteri*
90	5	3	YD		Mountain whitefish	1063	47.8%	Salmonidae	*Prosopium williamsoni*
91	9	7	Y	3	Bull trout	752	33.8%	Salmonidae	*Salvelinus confluentus*
Introduced Species									
92	107	32			White sucker	5	0.2%	Catostomidae	*Catostomus commersoni*
93	84	25			Green sunfish	12	0.5%	Centrarchidae	*Lepomis cyanellus*
94	26	8	Y		Pumpkinseed	261	11.7%	Centrarchidae	*Lepomis gibbosus*
95	105	31			Warmouth	6	0.3%	Centrarchidae	*Lepomis gulosus*
96	33	11	Y		Bluegill	186	8.4%	Centrarchidae	*Lepomis macrochirus*
97	19	4	Y		Smallmouth bass	379	17.0%	Centrarchidae	*Micropterus dolomieu*
98	20	5	Y		Largemouth bass	359	16.1%	Centrarchidae	*Micropterus salmoides*
99	49	16	Y		White crappie	84	3.8%	Centrarchidae	*Pomoxis annularis*
100	31	10	Y		Black crappie	207	9.3%	Centrarchidae	*Pomoxis nigromaculatus*
101	52	17			American shad	76	3.4%	Clupeidae	*Alosa sapidissima*
102	62	22			Goldfish	45	2.0%	Cyprinidae	*Carassius auratus*
103	120	35			Finescale dace	2	0.1%	Cyprinidae	*Phoxinus neogaeus*
104	27	9	Y		Carp	259	11.7%	Cyprinidae	*Cyprinus carpio*
105	114	33			Spottail shiner	3	0.1%	Cyprinidae	*Notropis hudsonius*
106	95	27			Fathead minnow	8	0.4%	Cyprinidae	*Pimephales promelas*
107	59	21			Tench	54	2.4%	Cyprinidae	*Tinca tinca*
108	48	15	Y		Northern pike	87	3.9%	Esocidae	*Esox lucius*
109	45	14	Y		Black bullhead	95	4.3%	Ictaluridae	*Ameiurus melas*
110	58	20			Yellow bullhead	62	2.8%	Ictaluridae	*Ameiurus natalis*
111	25	7	Y		Brown bullhead	280	12.6%	Ictaluridae	*Ameiurus nebulosus*
112	35	12	Y		Channel catfish	142	6.4%	Ictaluridae	*Ictalurus punctatus*

Table 4.16 (continued).

ID	Rank	Group rank	Asc[1]	Conservation[2] status	Common name	Freq	Pct	Family name	Scientific name
113	100	28			Tadpole madtom	7	0.3%	Ictaluridae	*Noturus gyrinus*
114	91	26			Flathead catfish	9	0.4%	Ictaluridae	*Pylodictis olivaris*
115	21	6	Y		Yellow perch	340	15.3%	Percidae	*Perca flavescens*
116	55	18			Walleye	71	3.2%	Percidae	*Stizostedion vitreum vitreum*
117	115	34			Southern platyfish	3	0.1%	Poeciliidae	*Xiphophorus maculatus*
118	56	19			Lake whitefish	71	3.2%	Salmonidae	*Coregonus clupeaformis*
119	101	29			Golden trout	7	0.3%	Salmonidae	*Oncorhynchus aguabonita*
120	1	1	Y		Rainbow trout	1726	77.6%	Salmonidae	*Oncorhynchus mykiss*
121	102	30			Kamloops trout	7	0.3%	Salmonidae	*Oncorhynchus mykiss mykiss*
122	78	24			Atlantic salmon	17	0.8%	Salmonidae	*Salmo salar*
123	16	3	Y		Brown trout	517	23.3%	Salmonidae	*Salmo trutta*
124	127	36			Sunapee char	1	0.0%	Salmonidae	*Salvelinus alpinus oquassa*
125	4	2	Y		Brook trout	1106	49.8%	Salmonidae	*Salvelinus fontinalis*
126	44	13	Y		Lake trout	96	4.3%	Salmonidae	*Salvelinus namaycush*
127	67	23			Arctic grayling	28	1.3%	Salmonidae	*Thymallus arcticus*
The following introduced species were reported by subbasin.									
128					Shortnose sucker			Catostomidae	*Chasmistes stomias*
129					Rock bass			Centrarchidae	*Ambloplites rupestris*
130					Sacramento perch			Centrarchidae	*Archoplites interruptus*
131					Redear sunfish			Centrarchidae	*Lepomis microlophus*
132					Tambaqui			Characidae	*Colossoma macropomum*
133					Convict cichlid			Cichlidae	*Cichlasoma nigrofasciatum*
134					Tilapia			Cichlidae	*Cichlidae sp.*
135					Oriental weatherfish			Cobitidae	*Misgurnus anguillicaudatus*
136					Loach			Cobitidae	*Misgurnus mizolepis*
137					Grass carp			Cyprinidae	*Ctenopharyngodon idella*
138					Tiger barb			Cyprinidae	*Puntius tetrazona*
139					Grass pickerel			Esocidae	*Esox americanus*
140					Gar			Lepisosteidae	*Lepisosteus sp.*
141					Striped bass			Percichthyidae	*Morone saxatilis*
142					Rainbow smelt			Osmeridae	*Osmerus mordax*
143					Saddleback gunnel			Pholidae	*Pholis ornata*
144					Mosquitofish			Poeciliidae	*Gambusia affinis*
145					Green swordtail			Poeciliidae	*Xiphophorus helleri*
146					Shortfin molly			Poeciliidae	*Poecilia mexicana*
147					Guppy			Poeciliidae	*Poecilia reticulata*
148					Arctic char			Salmonidae	*Salvelinus alpinus*
The following composite species groups were used in the assemblage analysis:									
			Y		Suckers	1324	59.6%		
			YD		Dace	1392	62.6%		
			Y		Sculpins	1676	75.4%		
			Y		Shiners	963	43.3%		
			Y		Chubs	188	8.5%		
			Y		Crappie	224	10.1%		
			YD		Bullheads	339	15.2%		
			Y		Lampreys	170	7.6%		
			Y		Cutthroat	1029	46.3%		
			Y		Trout	2167	97.5%		
			Y		Whitefish	1075	48.4%		
			YD		Steelhead	659	29.6%		
			YD		Rainbow (all)	1972	88.7%		
			YD		Chinook	454	20.4%		
			YD		Sunfish	331	14.9%		

[1] Asc codes: Y=species used in the association analysis; YD=divisor species; E=narrowly distributed endemics.

[2] Conservation status codes: 1=Federally listed as endangered or threatened; 2=State or American Fisheries Society listed as endangered, threatened, sensitive, or of special concern; 3= Federal candidate and/or sensitive species.

[3] The Wood River, Idaho supports a distinct subspecies of bridgelip sucker. We found the taxonomic description of this subspecies after the association analysis was completed.

Overview of Approach and Data Sources

We considered the fishes at three levels of detail:

1) **Fish Species Assemblages**—We summarized the known occurrence of all fish taxa, native and introduced, across the Basin. Species assemblages were defined by species composition and mapped by watershed across the Basin. Species richness and diversity indices were calculated for each species assemblage.

2) **Sensitive Native Species**—We compiled information for 38 taxa considered sensitive, threatened, endangered, or of special concern. We summarized current knowledge regarding fish status relative to the known historical range, biology and life history, and important threats to persistence. We used composite distributions of the most sensitive or narrowly distributed of these species to indicate sensitive or otherwise important watersheds.

3) **Key Salmonids**—We considered seven select salmonids in the greatest detail: bull, westslope cutthroat, Yellowstone cutthroat, and redband trout; steelhead; and ocean-type (age-0 migrant) and stream-type (age-1 migrant) chinook salmon. We summarized the presence, absence, and status in subwatersheds that supported spawning and rearing habitats for each species or life history. Because existing information describing the occurrence and status of these salmonids is incomplete, we used classification trees to analyze patterns in the known distributions and to predict the occurrence and status in watersheds lacking such information. We also briefly considered the distribution of several introduced salmonids.

There are several reasons for focusing on these salmonids, other than their obvious social and cultural values. First, we know more about them, and are more likely to discern important environmental relationships. Second, they are or were widely distributed, which allows for broad-scale comparisons. Third, salmonids act as predators, competitors, and prey for a variety of other aquatic and terrestrial taxa. Thus, they are likely to influence the structure and function of aquatic ecosystems and may serve as critical links with energy and nutrient flows to terrestrial systems (Henjum and others 1994; Willson and Halupka 1995). Finally, the salmonids potentially are more sensitive to disturbance than other groups (Marcot and others 1994). Different species and life stages often use widely divergent habitats, exposing individual populations and native assemblages to a wide variety of environmental and habitat disturbances. Salmonids may therefore integrate the cumulative effects of environmental change over broad areas.

We developed two primary databases, one that summarizes the current status of the selected "key" salmonids, and a second that summarizes the reported presence of all native or introduced fish taxa. As part of each database, we also summarized the historical distributions for each key salmonid and species recognized as sensitive, threatened, endangered, or of special concern by state or federal agencies.

Key Salmonids Current-Status Database

The seven key salmonids include both distinct species and life-history forms. Bull, westslope cutthroat, and Yellowstone cutthroat trout are taxonomically and geographically distinct. Redband trout and chinook salmon, however, are each represented by two distinct life-history forms. Non-anadromous redband trout and anadromous steelhead are commonly recognized as interior rainbow trout, and occur both together in sympatry and in isolation from each other (allopatry) throughout a major portion of the Basin. For purposes of this analysis, redband trout designates the non-anadromous form. We adopted Gilbert (1912 cited by Matthews and Waples 1991) and Healeys' (1991) characterization of chinook salmon that migrate seaward primarily at age-1 as "stream-type" and that migrate primarily at age-0 as "ocean-type."

The current-status database was developed through the classification of subwatersheds by private, agency, and tribal fishery biologists working throughout the Basin. The data were augmented with information from existing electronic databases maintained by state and federal agencies, and other sources (appendix 4C).

Biologists were asked to summarize the status of each of the key salmonids from existing data. We used a series of workshops to train facilitators who then trained or worked with other biologists from the study area. Each participant was responsible for an area familiar to them, where they either generated or maintained information or had access to the available data and expertise. Biologists worked in teams and incorporated information from other federal, state, private and tribal biologists and inventories. More than 150 biologists contributed to the project (appendix 4C).

Our sample units were 6th-field watersheds, also termed subwatersheds. Biologists classified the status of naturally reproducing populations only. If populations were supported solely by hatchery-reared fish, naturally spawning fish were considered absent. Biologists classified units where fish were present as follows: spawning and rearing habitat; overwintering and migratory-corridor habitat; or present but of unknown status. Subwatersheds supporting spawning and rearing habitat were further classified as strong or depressed. We asked biologists to judge strong or depressed status based on population characteristics, including life-history forms, recent trends in abundance, and current abundance. Status should not have been inferred from surrounding landscape characteristics or the occurrence or other species (for example, habitat condition or presence of introduced fishes). The following criteria guided classification:

Present, strong—Strong subwatersheds include those with the following characteristics: 1) all major life histories (for example: stream resident or migratory) that historically occurred within the watershed are present; 2) numbers are stable or increasing, and the local population is likely to be at half or more of its historical size or density; and 3) the population or metapopulation within the subwatershed, or within a larger region of which the subwatershed is a part, probably contains at least 5,000 individuals or 500 adults.

Present, depressed—Depressed watersheds include those with at least one of the following characteristics: 1) a major life-history component (for example, a migratory or resident form of westslope cutthroat trout) has been eliminated; 2) numbers within the subwatershed are declining, or the salmonid occurs in less than half of its historical habitat, or numbers are less than half of what the watershed supported historically; if historical information is unavailable, densities are less than half of what is found in comparable streams in an undisturbed condition where the species is well distributed; in cases with relatively strong numbers, but where a population was seriously hybridized with an introduced species or subspecies, the population representing the pure native species/subspecies was considered depressed; or 3) total abundance for the population or metapopulation within the subwatershed, or the larger region of which this subwatershed is a part, is less than 5,000 total fish or 500 adults; the fish within the subwatershed are isolated by distance or natural barriers from other populations that would collectively exceed these numbers.

Absent—The key salmonid is not present in this subwatershed. It is either extinct or it never occupied the subwatershed. The subwatershed is located within the natural range of the species and colonization of the subwatershed was historically possible even though habitat or other environmental conditions might make the habitat unsuitable.

Present, unknown—The key salmonid is present, but there is no reliable information to determine current status.

Present, migration corridor—Migration corridors are habitats that do not support spawning or rearing, and function solely as routes or staging and wintering areas for migrating fish. If spawning or rearing are known to occur, the population status is judged as present-depressed, present-strong, or present-unknown. Many salmonids may disperse widely from natal habitats prior to maturation. Where a distinction could be made between areas that serve as initial rearing areas prior to migration and those that support transient or subadult fish (for example, mainstem rivers or lakes for bull trout), we considered the latter to be corridor habitat.

Unknown—No information exists regarding the current presence or absence of the species. Subwatersheds that were unclassified were considered unknown.

All status classifications were done by color coding maps of the subwatersheds. Data were entered into the current-status database and proofed against the original maps and against preexisting databases (appendix 4C). We accepted classifications of absent from the preexisting databases only when sampling was documented. Subwatersheds where the current and preexisting databases conflicted regarding the known presence or absence of a species were classified as unknown. Rules employed for merging data sets and treatment of conflicting information are detailed in appendix 4C.

Species-assemblage Database

The species-assemblage database was developed at the same time as the current-status database. For this database the sample units were 5th-field watersheds. The biologists recorded the known presence of all potential fish species in each watershed. They used a master list of all taxa (that is, native and introduced) expected within the Basin, with a unique numeric code for each taxa. Biologists characterized watersheds where they were knowledgable about all species, but they were also asked to use professional judgement to provide as complete a classification as possible. For example, where available data indicated a common species complex in some watersheds but data were limited in other watersheds, biologists were encouraged to extrapolate species occurrence to similar, adjacent waters. Where sampling, records, or personal experience where too limited to include all likely species, watersheds were classified as unknown even though some species were known present. The assemblage database was proofed in the same manner as the current-status database. We compared our database with the preexisting databases maintained by each state and the Oregon State University collection records. If any database included a species as present, it was included as present in the final assemblage database. Additional records of introduced species were added at the subbasin level using data provided by the National Biological Service's Southeastern Biological Science Center.

Historical Ranges

Historical ranges were defined for the seven key salmonids and, whenever possible, for all species, subspecies, or races recognized as sensitive, threatened, endangered or otherwise of special concern. We defined historical as prior to European settlement whenever such inference was possible. Ranges were characterized from the historical distributions in the preexisting databases and augmented through published and anecdotal accounts. We did not accept all accounts as correct, but reviewed and revised distributions whenever possible. The Wilderness Society (1993) distribution of several salmon species, for example, was based on the range of contiguous waters that were historically accessible and not on specific species accounts.

For many fish species, the historical ranges remain speculative and more accurately represent the potential range rather than an actual historical distribution. Few records are available with which to ascertain the historical range of steelhead and salmon throughout the Bruneau River subbasin, for example, because access was blocked early in the century and prior to any detailed population survey. Some species like bull trout are also unlikely to occupy all reaches or all accessible streams within the watersheds of the historical range. Distributions may be restricted by elevation, temperature, and local channel features. Because we mapped historical ranges at the watershed scale, the entire watershed was included in the historical range, even where species may have occurred in only part of a watershed. In some cases, the historical range may be overestimated because of the loss of resolution.

The historical ranges of redband trout and steelhead are ambiguous where the two co-occur (see species narrative for redband trout). Although all native non-anadromous rainbow trout were considered redband trout in our summary, we believe that populations isolated from steelhead in recent geologic history may represent evolutionarily distinct subgroups of the species. Such isolation may result in substantial genetic divergence between groups, representing important genetic variability for the species. For this reason, we identified two historical ranges for redband trout, those isolated from (allopatric) and those contiguous with (sympatric) the historical distribution of steelhead.

Historical ranges were merged with the current-status and species-assemblage databases. A summary and documentation of the defined historical ranges are presented in appendix 4D.

Current Fishes

Like many portions of western North America, the Basin includes a moderately sized, but locally diverse fish fauna. The native fish fauna of the Columbia River Basin is unusual in that it clearly is not a single faunal unit, but is rather comprised of several subbasin faunas with limited species overlap among subbasins.

Ichthyological Provinces

Based on local endemism and common relationships to nearby basins, McPhail and Lindsey (1986) described five ichthyological provinces within the Columbia River Basin: Upper Snake, Wood River, Glaciated Columbia, Middle Columbia, and Lower Columbia. The Snake River was an independent drainage until the Early Pleistocene, and the Upper Snake remains isolated by Shoshone Falls in south central Idaho. The fishes of the Upper Snake are more closely related to those of the Bonneville Basin than to the remainder of the Columbia Basin.[1] Eighteen fish species present in the Snake River below Shoshone Falls are not found above; however, 10 of 12 species that occur in the Snake River above and below the Falls also occur in the Bonneville system. Exchanges of fish faunas between the Upper Snake and Bonneville systems probably occurred during Pleistocene spillovers of pluvial Lake Bonneville into the Upper Snake (McPhail and Lindsey 1986; Wheeler and Cook 1954). Recognition of the Wood River drainage as a separate province is supported by the presence of Wood River sculpins and a unique subspecies of bridgelip sucker (McPhail and Lindsey 1986), both of which are endemic to the Wood River, and the Shoshone sculpin, which is endemic to the Snake River near the confluence of the Wood River.

The glaciated portion of the Basin includes roughly the northern third, which is characterized as mountainous, heavily forested, and with cold, high-gradient streams. Pygmy whitefish, lake chub, longnose sucker, burbot, and slimy sculpin are confined to glaciated areas and the southern edges of their ranges coincide with the limits of glaciation. One species, the margined sculpin, is restricted to the Middle Columbia, which flows through an arid plateau region. In contrast, streams in the Lower Columbia are well connected and contain 12 species; nine of which are euryhaline, tolerant of higher salinities, or anadromous species (McPhail and Lindsey 1986). Some of these latter species are not included within the Basin because waters most influenced by ocean conditions are located to the west of the crest of the Cascades.

Portions of the Klamath and Great Basin drainages also occur in the area considered in this assessment. The Upper Klamath Basin is dominated by Upper Klamath and Agency lakes, which harbor a

[1]Personal communication. 1995. W. Minckley, Arizona State University, Tempe, Arizona.

diverse assemblage of specialized catostomid fishes (Andreasen 1975b). The Great Basin is characterized by a series of isolated subbasins, each with a largely or wholly closed drainage pattern and resulting depauperate, but highly endemic, fish assemblages (Sigler and Sigler 1987). The distinctive native fishes of both the Upper Klamath and Great Basin bear little resemblance to those of the Columbia River Basin.

With some notable exceptions, the Ecological Reporting Units (ERUs) correspond well to the ichthyological provinces described above (see map 4.1). The Upper Snake province of McPhail and Lindsey (1986) includes the Upper Snake and Snake Headwaters. The Wood River province is problematic in that it is rather evenly split between the Central Idaho Mountains and Owyhee Uplands ERUs. The Glaciated Columbia province includes three ERUs, as follows: the Northern Glaciated Mountains; the Lower Clark Fork; and the Upper Clark Fork. The Middle Columbia province includes four ERUs, as follows: first, the Columbia Plateau; second, the Blue Mountains; third, the Owyhee Uplands; and fourth, the Central Idaho Mountains. The Blue Mountains ERU includes the Silvies subbasin, which is part of the Harney Basin (Bisson and Bond 1971) and is therefore more correctly associated with the Great Basin than the Columbia River system. The Goose Lake Basin is placed with the Upper Klamath, but more appropriately should be associated with the Northern Great Basin or treated separately in addition to the Great Basin. Except for the ubiquitous speckled dace and tui chub, there is no overlap of species between the fish faunas of the Goose Lake subbasin and the Upper Klamath basin (Moyle and Daniels 1982).

Species List and Relative Abundance

The fish communities found in the Basin reflect broad-scale differences in ichthylogical provinces, environmental gradients from headwater streams to major rivers, and the myriad changes that post-European settlement has wrought through altered habitat, harvest, and introduced species. To understand the joint distributions of numerous species, both native and introduced, we summarized the available information using community analysis techniques, specifically, diversity indices and association analysis.

As explained in the subsection on "Overview of Approach and Data Sources," we were restricted to presence or absence information. This is misleading, in that the information referred not so much to "presence" or "absence" as to "reported" or "not reported." Strictly speaking, "not reported" does not necessarily mean "absent." Based on verification of the data, we believe that more errors of omission (not reporting species that were present), than commission (reporting species that were not present) were committed. We use the term relative abundance to refer to the frequency with which a species is reported as present, rather than as a measure of numerical abundance *per se*. Thus, bull trout will have a higher relative abundance than speckled dace because they were more frequently reported, not because they are more numerically abundant.

A total of 143 recognized species, subspecies, or races was reported within the Basin (table 4.16). There were 88 native taxa (61%) reported, and 55 taxa (39%) that have been introduced from outside areas. Of the total, 124 were reported by watershed. The remaining 19 taxa were introduced species that could only be recorded by subbasins. We also recorded the occurrence of four "generic" taxa that were used when species or subspecies designations were unknown (cutthroat, dace, sculpins, and suckers) at the watershed level and two generic taxa (gar and tilapia) that were used at the subbasin level.

A brief examination of the species list in table 4.16 provides some perspective. Although our list of 88 native fishes may seem like high species richness, it includes only 10 families, 21 genera, and 66 species. By comparison, the Rio Grande River system has 154 native species in 29 families (Smith and Miller 1986), and the western Mississippi River system has 235 native species in 33 families (Cross and others 1986). Compared with these systems, species richness within the Basin is low. The Basin more closely resembles the Sacramento-San Joaquin system, with 55 native species in 13 families (Moyle 1976). The lower diversity of western rivers likely reflects a relatively young geologic age, and greater zoogeographic isolation. Also, fewer marine species invade the Columbia River than the Mississippi River (Cross and others 1986).

Major disruptive events that have shaped a dynamic Basin landscape have undoubtedly influenced fish communities as well (Li and others 1987). The native species fall into two groups. The first group is comprised of 15 to 20 species that are widely distributed throughout the Basin and/or reported in 20 percent or more of the watersheds. The second, larger group of roughly 60 species is the narrowly distributed species that have restricted ranges or are infrequently reported (less than 5% of the watersheds).

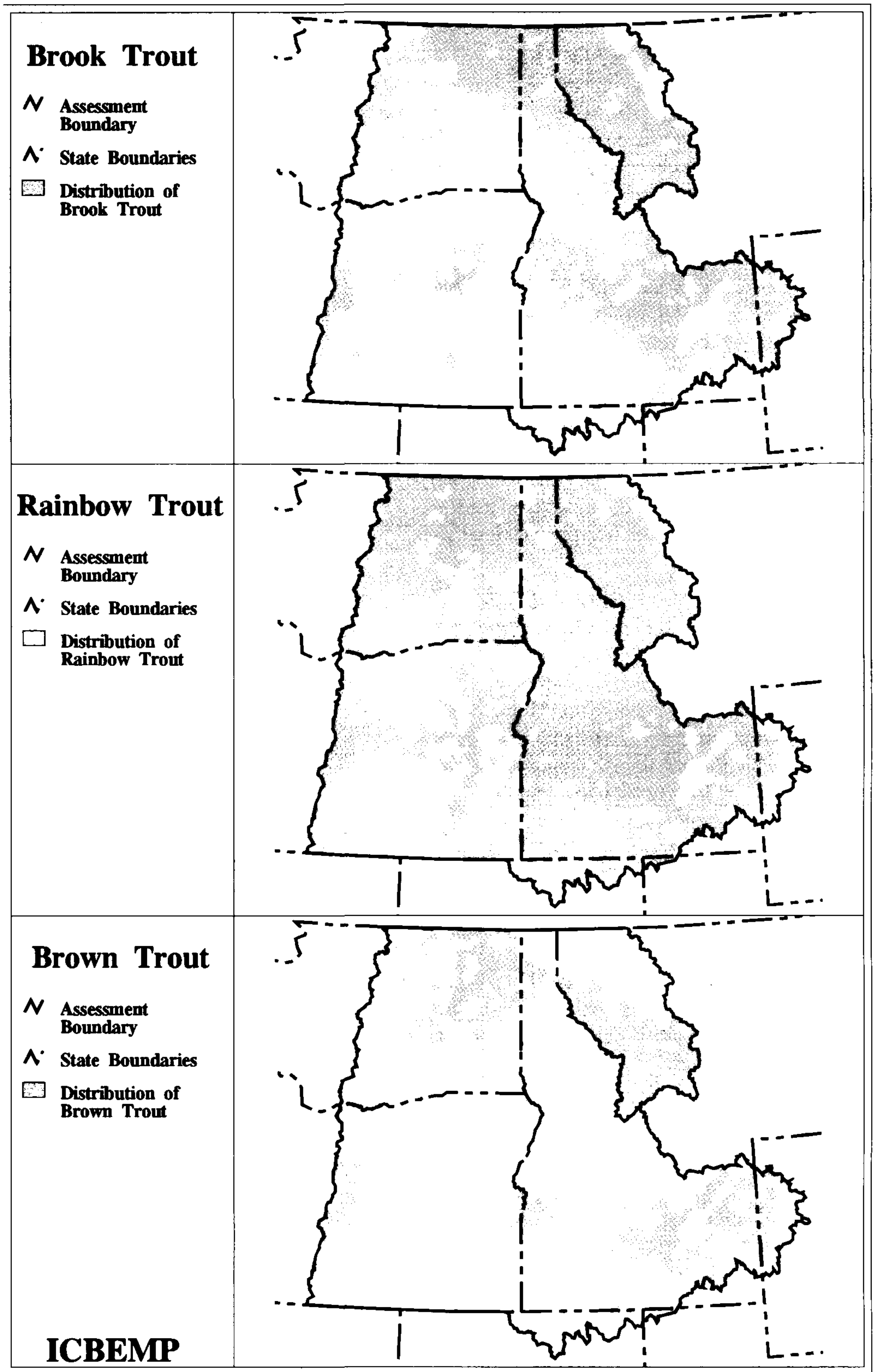
Clearly, introduced species are a major component of the current icthyofauna. The most frequently reported species was introduced rainbow trout, occurring in 78 percent of the Basin (map 4.7). Introduced brook trout were reported in 50 percent of the watersheds (map 4.7). The majority of introduced species are game fishes that were purposefully introduced by fishery management agencies and private individuals to provide fishing opportunities; sixteen (32%) of the 50 most-reported species are introduced game fishes. In addition, fishing practices likely contributed a number of bait-bucket introductions of non-native fish, as well. More unusual occurrences likely resulted from aquaria releases. For example, Amazonian tambaqui were reported in the Snake River, and oriental weatherfish were reported in the Boise River.

Species Assemblages

Species lists and individual species distributions provide limited information. A more complete understanding of ecological processes can be gained from analysis of aquatic communities or assemblages. Following Fauth and others (1996) we define a community to include all species occurring together at the same time, and use the more restrictive term, assemblage, to refer to phylogenetically related species within a given community. There are several ecological principles at play within an aquatic community or assemblage of fishes. First, for any given species there generally are several other species that have similar habitat or resource requirements (Liem 1984). Second, there may be complex predator-prey relationships or competitive interactions where the presence of one species influences the presence of another (Chesson and Rosenzweig 1991; Sredl and Collins 1992; Winston 1995). Finally, the distribution of fish assemblages within stream networks is determined not only by environmental gradients in habitat features (Beecher and others 1988; Li and others 1987; Schlosser 1990, 1991), but also by temporal variability (Poff and Allan 1995; Schlosser 1990). Collectively, these interacting principles make analysis of fish assemblages both interesting and challenging.

Analytical Approach

The first step in our analysis was to identify distinct fish assemblages using the species presence/absence data collected at the watershed level. Association analysis was used, which is a monothetic, divisive, hierarchial scheme (Ludwig and Reynolds 1988). Association analysis is monothetic in that the presence/absence of single species is used to separate adjacent groups, divisive in that all units begin as a single group that is successively partitioned, and hierarchical in that lower level groups are exclusive subsets of parent groups at higher levels. Association analysis arose from the vegetation classification work of Goodall (1953). In the conventional application, one implicitly assumes that certain species are more sensitive to environmental factors controlling community structure. These species form the basis for the classification scheme (Coetzee and Werger 1975). We did not make this assumption. Instead, we looked for those species that had the highest level of association with other species, both positive



Map 4.7—Current distribution of brook, introduced rainbow, and brown trout by watershed, based on the assemblages database.

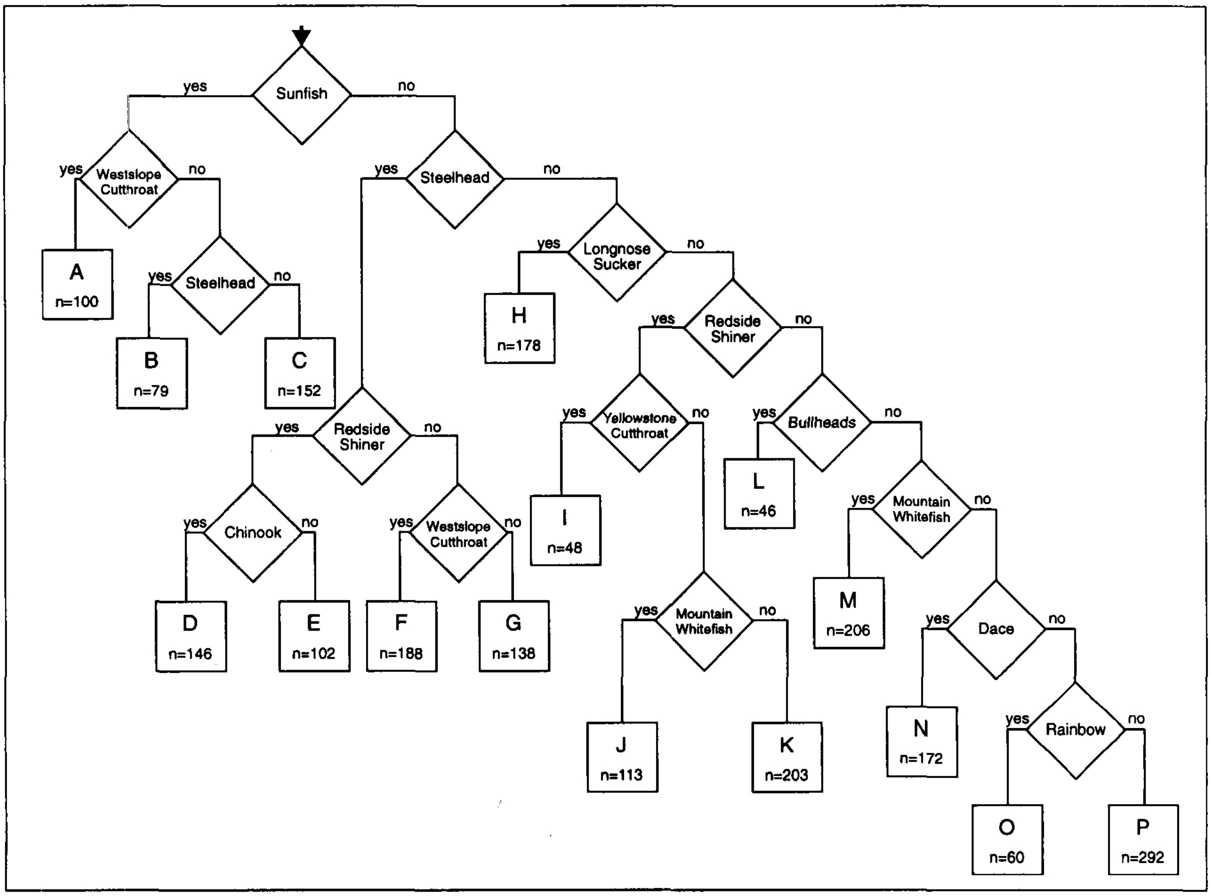


Figure 4.16. Dichotomous rule set for grouping observations based on the presence/absence of select taxa (diamonds) with the highest degree of association with other taxa. Assemblages (squares) are identified by alphabetic notation and described further in the text.

and negative, and used these associations to group watersheds. Our hope was that the resultant groupings of watersheds would suggest meaningful ecological patterns.

The 47 most frequently reported species and 15 composite groups composed of combinations of species (table 4.16) were used in the association analysis. We followed Ludwig and Reynolds (1988), except that we terminated the subdivisions at levels considerably above any of the alternative stopping rules proposed by Ludwig and Reynolds. Our intent was to keep a manageable number of species assemblages that would highlight important differences in species composition and distribution of assemblages. Thus, we did not subdivide groups of less than 150 observations, or where the maximum chi-square statistic for two potential groups was less than 500. All watersheds with at least two species reported were used, giving an initial sample size of 2,223 watersheds. The association analysis grouped the observations into 16 classes, which we identified with the letters A through P, using the dichotomous rule set illustrated in fig. 4.16.

Table 4.17—Summary statistics for 16 species assemblages identified in the association analysis

	Assemblage:															
Sample statistics	**A**	**B**	**C**	**D**	**E**	**F**	**G**	**H**	**I**	**J**	**K**	**L**	**M**	**N**	**O**	**P**
sample size	100	79	152	146	102	188	138	178	48	113	203	46	206	172	60	292
mean number of taxa	19.9	26.2	16.6	14.9	10.5	9.0	8.1	12.1	11.9	11.0	7.4	10.5	7.0	5.0	2.8	3.6
mean number of groups	9.3	11.1	8.0	9.4	7.5	6.8	6.1	6.9	8.0	7.2	5.3	6.0	5.2	4.1	2.4	2.9
mean number of native tx.	10.5	13.7	7.6	11.7	8.8	6.9	6.0	9.0	8.3	7.8	5.7	6.2	4.8	3.6	2.2	1.6
mean number of exotic tx.	9.4	12.5	9.0	3.2	1.7	2.1	2.1	3.1	3.5	3.2	1.7	4.4	2.2	1.5	0.6	2.0
min taxa	8	12	2	7	4	3	2	7	7	5	2	2	2	2	2	2
max taxa	46	46	31	25	20	16	20	22	17	20	21	20	17	13	7	9
min groups	5	7	1	3	4	3	2	6	6	5	2	2	3	2	1	2
max groups	14	14	12	11	10	9	10	9	11	10	10	9	8	7	4	5
min native	1	4	0.0	5	4	2	2	4	4	5	1	0.0	2	1	1	0.0
max native	21	27	16	18	16	14	14	16	11	12	12	14	9	9	6	6
min exotic	3	4	1	0.0	0.0	0.0	0.0	1	1	0.0	0.0	1	0.0	0.0	0.0	0.0
max exotic	25	25	17	10	8	6	13	7	9	9	9	11	9	4	2	6
Diversity and Evenness*:																
total taxa	69	70	92	54	45	37	53	50	35	58	58	65	55	70	46	57
abundant taxa (T1)	36	48	41	29	23	16	25	23	21	27	23	40	21	24	25	14
very abundant taxa (T2)	30	42	32	23	18	13	18	19	17	20	16	27	14	15	16	8
taxa evenness	0.82	0.86	0.79	0.79	0.77	0.78	0.69	0.80	0.83	0.73	0.68	0.68	0.66	0.58	0.62	0.52
total native species	42	45	64	39	31	29	36	32	22	41	43	51	41	60	42	42
abundant native (N1)	20	30	24	23	18	14	19	17	13	19	17	34	17	21	24	13
very abundant native (N2)	17	25	18	19	15	11	14	14	12	14	12	25	11	12	15	7
native ratio (N1/T1)	0.56	0.62	0.59	0.80	0.81	0.85	0.75	0.73	0.64	0.69	0.72	0.85	0.79	0.85	0.96	0.94
native species evenness	0.81	0.84	0.73	0.81	0.80	0.76	0.73	0.82	0.86	0.74	0.69	0.71	0.65	0.53	0.62	0.47
composite index (Z)	0.71	0.81	0.66	1.00	1.0	1.0	0.85	0.92	0.85	0.79	0.77	0.94	0.79	0.70	0.93	0.68

* Reference: Hill, M.O. 1973. Diversity and evenness: a unifying notation and its consequences. Ecology. 54: 427-432.

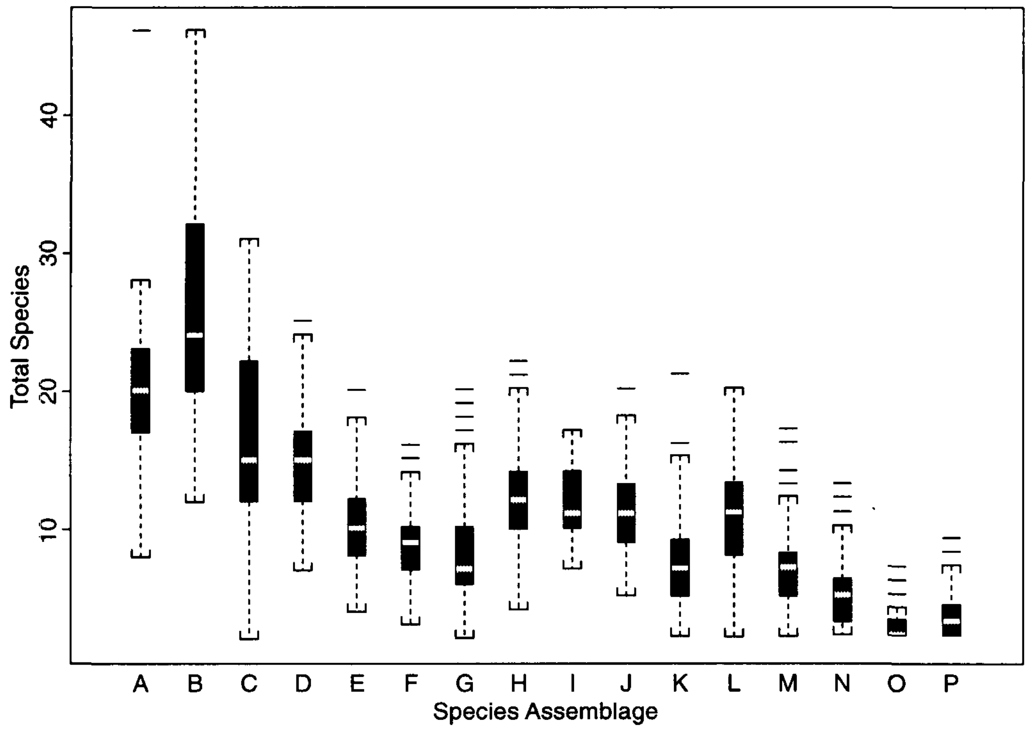


Figure 4.17—Box and whisker plots illustrating the distribution of total species across watersheds grouped according to the rules developed in the association analysis. The horizontal line in the interior of the box represents the median, the box height equals the interquartile distance, and the dotted lines extend to the extreme values of the data. Data points outside the dotted lines may be outliers and are indicated by horizontal lines.

The full complement of species information was summarized for each fish assemblage in a number of ways. First, we calculated basic summary statistics for all species, composite groups, native species, and introduced species (table 4.17), and plotted the distribution of total species per sample within each assemblage (fig. 4.17). Diversity indices were calculated by treating the total sample for each assemblage as a single observation and using the frequency counts as abundance measures, adapting Hill's (1973) approach as presented in Ludwig and Reynolds (1988) (see also Alatalo 1981; Peet 1974). Though unconventional, this approach was consistent with the view of diversity indices as simple, comparative measures that attempt to incorporate both richness and evenness into a single value. For comparative purposes, we also devised a single composite index from the measures of abundant taxa (T1), abundant natives (N1), and very abundant natives (N2). Our index (Z) is defined as

$$Z' = (N1/T1) * (N2-1)/(N1-1),$$

$$\text{then} \quad Z = Z' / Z'max,$$

which is the ratio of abundant native species to abundant taxa, times native evenness, scaled by the maximum observed value (Z'max) of the intermediate product (Z'). This index incorporates elements of native species abundance relative to non-natives, and evenness, that is, uniformity in distribution, among native species. The intermediate product, Z', approaches one as T1, N1, and N2 become more similar. Dominant species and groups, that is, those reported in 80 percent or

Table 4.18—Dominant taxa found within 16 species assemblages identified in the association analysis.

Assemblage	A	B	C	D	E	F	G	H
Species:	Largescale sucker	Chiselmouth	Rainbow	Sculpin, generic	Redside shiner	Sculpin, generic	Interior redband	Longnose sucker
	Northern squawfish	Northern squawfish	Northern squawfish	Interior redband	Westslope cutthroat	Summer steelhead	Largescale sucker	
	Redside shiner	Summer steelhead	Redside shiner	Summer steelhead	Rainbow	Westslope cutthroat		
	Westslope cutthroat	Mountain whitefish	Interior redband	Summer steelhead	Rainbow			
	Rainbow	Bluegill	Summer steelhead	Stream-type chinook	Mountain whitefish			
	Mountain whitefish	Smallmouth bass	Stream-type chinook	Bull trout	Brook trout			
	Pumpkinseed	Largemouth bass	Mountain whitefish					
	Largemouth bass	Carp						
	Yellow perch							
	Brook trout							
Groups:	Suckers	Suckers	Suckers	Suckers	Suckers	Sculpins	Trout	Suckers
	Sculpins	Dace	Sculpins	Dace	Dace	Cutthroat	Steelhead	Sculpins
	Shiners	Sculpins	Bullheads	Sculpins	Sculpins	Trout	Rainbow (all)	Cutthroat
	Cutthroat	Trout	Trout	Shiners	Shiners	Steelhead	Trout	
	Trout	*Whitefish*	*Rainbow (all)*	*Trout*	*Trout*	*Rainbow (all)*	*Whitefish*	
	Whitefish	Steelhead	Sunfish	Whitefish	Steelhead	Chinook	Rainbow (all)	
	Rainbow (all)	Rainbow (all)	Steelhead	Rainbow (all)				
	Sunfish	Chinook	Rainbow (all)					
	Sunfish	Chinook						

Assemblage	I	J	K	L	M	N	O	P
Species:	Utah sucker	Redside shiner	Redside shiner	Rainbow	Mountain whitefish	none	none	Rainbow
	Speckled dace	Rainbow	Brown bullhead					
	Redside shiner	Mountain whitefish						
	Yellowstone cutthroat							
	Brook trout							
Groups:	Suckers	Suckers	Dace	Bullheads	Sculpins	Dace	none	Trout
	Dace	Dace	Shiners	Trout	Trout	Trout		Rainbow (all)
	Sculpins	Sculpins	Trout	Rainbow (all)	Whitefish			
	Shiners	Shiners						
	Cutthroat	Trout						
	Trout	Whitefish						
	Rainbow (all)							

more watersheds, were also identified for each assemblage (table 4.18). Species assemblages were spatially mapped (map 4.8) and cross-referenced with ERU.

One of the more interesting summaries generated was a plot of native species versus non-native species for each assemblage (fig. 4.18). Each point in figure 4.18 corresponds to an individual watershed. The diagonal line corresponds to points where the number of native species equals the number of non-native species. Points above the diagonal correspond to areas where native species outnumber non-native species; points below the line are areas where the reverse is true. Assemblages exhibiting a majority of points well above the diagonal indicate areas of relatively high native-species integrity.

Characterizations

Data generated by the association analysis provide a rather formidable array of information (table 4.18). To help summarize this information, we provide a brief characterizations of each assemblage.

Assemblage A—These watersheds are found primarily in the Northern Glaciated Mountains, the Lower Clark Fork and Upper Clark Fork, outside the range of anadromous fishes. The watersheds generally contain a high number of fish species, many of them non-native. Species composition consistently includes fish with a wide range of temperature tolerances, suggesting a mix of larger rivers and reservoirs with smaller, cold-water streams.

Assemblage B—These watersheds are found primarily in the Columbia Plateau, Blue Mountains, and Northern Glaciated Mountains, within the range of anadromous fish. The watersheds display the highest taxa diversity and evenness and generally contain many species—many of which are non-native. Dominant species include anadromous steelhead and chinook salmon, several warm-water gamefish, and carp, suggesting that these are larger rivers, and perhaps migration corridors for anadromous fish.

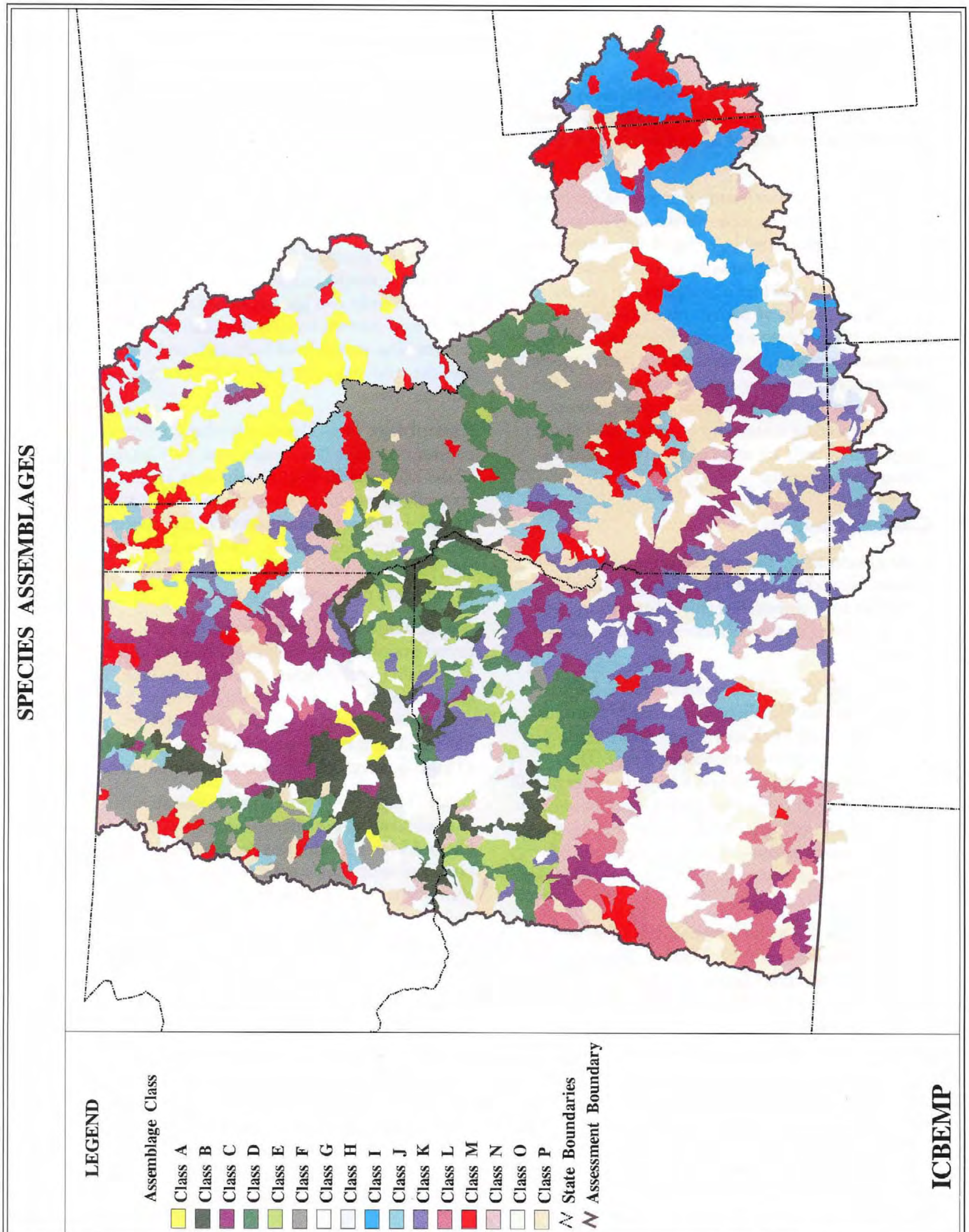
Assemblage C —These watersheds are scattered throughout the Basin, but are most common in the Columbia Plateau, Northern Glaciated Mountains, and the Owyhee Uplands generally outside the range of anadromous fish. The watersheds include the highest total taxa and show high taxa diversity, yet have only one dominant species (introduced rainbow trout) and relatively few dominant groups. In addition, these watersheds are one of only two groups where the mean number of non-natives exceeds the mean number of natives. The presence of bullheads and sunfish, and the relative absence of native trouts suggests warmer rivers.

Assemblage D—These watersheds are most common in Blue Mountains and the Central Idaho Mountains, and contain both steelhead and chinook salmon. The watersheds exhibit high diversity with high numbers of native species and relatively few non-natives. The species' composition suggests a mix of high-quality, cold-water streams and cool-water rivers.

Assemblage E—These watersheds are found mainly in the Columbia Plateau and Blue Mountains, and contain steelhead but lack chinook salmon. The watersheds tend to have moderate numbers of species, with very few non-natives. The species' composition suggests a mix of high-quality, cold- and cool-water habitats.

Assemblage F—These watersheds are most common in the Northern Cascades and the Central Idaho Mountains, within the overlapping ranges of westslope cutthroat trout, steelhead, chinook salmon, and bull trout. The watersheds include predominately native species, mostly salmonids and sculpins that are typical of cold-water habitats, with relatively low diversity.

Assemblage G—These watersheds are scattered through the Northern Cascades, Southern Cascades, Columbia Plateau, Blue Mountains, and Central Idaho Mountains. The watersheds include the fewest total species and highest percentage of non-natives among the cooler-water assemblages that contain steelhead. Redband trout and steelhead are the only dominant species.



Map 4.8—Spatial distribution of the 16 species assemblages identified in the association analysis by watershed.

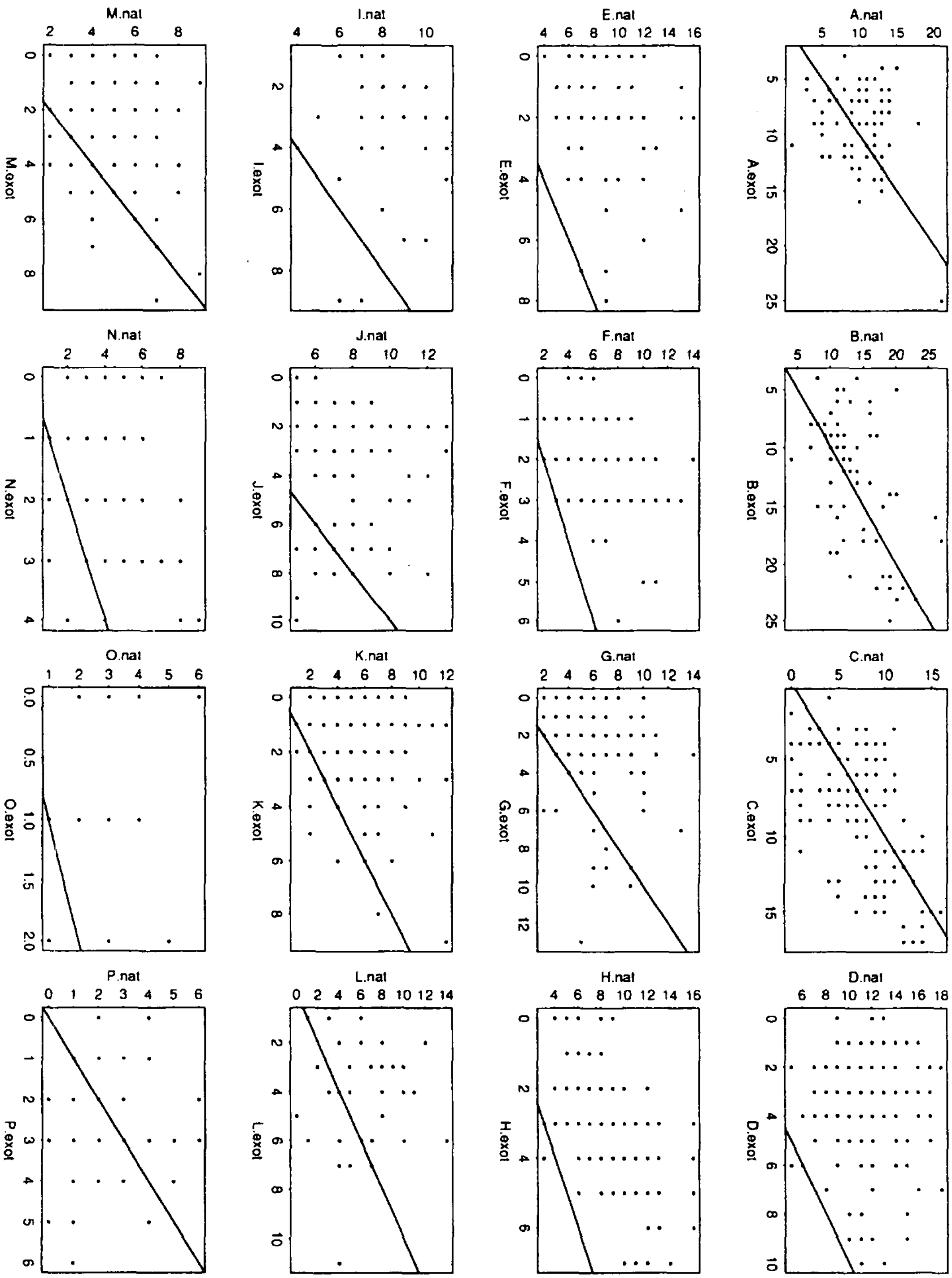


Figure 4.18—Plots of native species versus introduced species for each watershed within each of the 16 species assemblages. The diagonal line corresponds to points where the number of natives equals the number of introduced species.

Assemblage H—These watersheds are found primarily in the Northern Glaciated Mountains, the Lower Clark Fork, and the Upper Clark Fork, outside the range of anadromous fish. They are distinguished by the presence of longnose suckers. They exhibit moderate numbers of species, predominately natives, though introduced rainbow and brook trout are common. The species mix and spatial distribution suggest mid- to higher-elevation, cold- and cool-water streams.

Assemblage I—These watersheds are found in the Upper Snake and Snake Headwaters, within the range of Yellowstone cutthroat trout. The watersheds contain moderate numbers of species, mostly natives, but a relatively high ratio of non-natives for the given species mix.

Assemblage J—These watersheds are scattered throughout the Basin, excluded only from the Southern Cascades and Upper Klamath. The watersheds exhibit moderate numbers of species and diversity, with a fair number of introduced fishes. Dominant species include redside shiners, mountain whitefish, and introduced rainbow trout, suggesting cool-water rivers or transitional areas.

Assemblage K—These watersheds are found most commonly in the Owyhee Upland, and scattered throughout the rest of the Basin. The watersheds exhibit high variability in species counts that are lower than average. Numbers of non-natives are low, but occasionally exceed native counts. Assemblage K is distinguished from Assemblage J by lack of mountain whitefish.

Assemblage L—These watersheds are found in the Southern Cascades, Upper Klamath, Northern Great Basin, and Columbia Plateau. The two dominant species are non-native bullhead and introduced rainbow trout. Non-native species often outnumber native species. Despite this apparent contradiction, the watersheds are very high in native species diversity and native ratio, suggesting a relatively diverse native fauna and fewer, but widespread non-native species.

Assemblage M- These watersheds are found mainly in the Northern Glaciated Mountains, the Snake Headwaters, and the Central Idaho Mountains, but are scattered throughout other ERUs. The watersheds have low species counts, low diversity, and widespread non-natives. Mountain whitefish is the only dominant species, and is generally found in combination with trout and sculpins.

Assemblage N—These watersheds are scattered throughout the Basin, most commonly in the Columbia Plateau, and are excluded only from the Upper Clark Fork. Collectively, the watersheds contain a high total number of species, most of which occur only rarely. Mean counts and diversity are low. Trout and dace are the dominant groups, suggesting smaller, cold-water streams.

Assemblage O—These watersheds are scattered throughout the Basin and have very few species, averaging less than three per watershed. Given the distribution of this assemblage, it probably reflects areas that were incompletely sampled.

Assemblage P—The most abundant and widespread of all assemblages, other than unclassified, these are areas where introduced rainbow trout are known present but, in general, few other species were reported. Reported non-native species generally outnumber native species, though the ratio of abundant natives to abundant taxa is high. Low evenness suggests unequal distribution of species.

Rare and Sensitive Fishes

We prepared summary narratives for 39 rare or sensitive fish species in the Basin (next section). These fishes included federally listed endangered or threatened species, Category 1 Candidate species for Federal protection (the U.S. Fish and Wildlife Service deleted all Category 2 Candidate species from candidates for listing under the Endangered Species Act), taxa recognized for special protection by the states of Oregon, Washington, Idaho, Nevada, or Montana, taxa managed as sensitive species by the Forest Service and/or Bureau of Land Management, and taxa recognized by the American Fisheries Society (Williams and others 1989) as warranting special management status because of rarity. Also included are coastal cutthroat trout and chum and coho salmon, which are regionally important but occupy only a small portion of the Basin, and Sunapee char, an introduced species that is extinct within its native range.

Our special status list includes white sturgeon (Acipenseridae), five lampreys (Petromyzontidae), sockeye, chum and coho salmon (Salmonidae), coastal and Lahontan cutthroat trout (Salmonidae), pygmy whitefish (Salmonidae), burbot (Gadidae), 11 minnows (Cyprinidae), six suckers (Catostomidae), and eight sculpins (Cottidae). Twenty-two of these taxa occur in the Great Basin and Klamath Basin portions of the assessment area. The Great Basin harbors 11 of these taxa, six occur in the Klamath Basin (excluding the Goose Lake Basin), and four in the Goose Lake Basin. Within the interior Columbia River Basin, eight taxa occur entirely or primarily in the mainstem Columbia River, four are restricted to the upper Snake River (including the Wood River), two are restricted to the Northern Glaciated Mountains, two occupy streams in the middle and upper Columbia River Basin, and one is restricted to the Blue Mountains of the middle Columbia River.

The problems facing these special-status fishes are similar to those facing most native fishes, but may be more acute because of their limited distributions and abundances (table 4.19). Hydroelectric development has disrupted movements of migratory forms. Irrigation diversions, water withdrawal, and the loss of wetlands, marshes, and interconnected waterways have radically altered habitats for many species, especially in arid regions. Timber harvest, mining, grazing, and urbanization have degraded the habitat of others through siltation, and increased temperatures, and eutrophication. Especially threatened are those species dependent on springs, such as the Foskett speckled dace, Hutton tui chub, and Shoshone sculpin. Introduced species are a prevalent threat, due to competition, predation, or hybridization.

Management of many special-status fishes may be hindered by a lack of even basic information on life history and habitat requirements. In many cases, we cannot be certain of the species' distribution, much less appreciate life history and habitat characteristics. Our most complete information is for the salmonids, or for a few select species that have attracted the attention of researchers.

Key Salmonids

In this section we summarize the current distribution, status, and biology of selected salmonids. We also consider the factors known to influence each species or form. Our knowledge of these species is necessarily limited to areas that have been sampled and recorded by biologists working throughout the basin. In an attempt to generate the most complete possible picture of the current distribution of these fishes we used the emergent patterns in distribution associated with landscape features to predict occurrence and status for unsampled areas.

Analysis Using Classification Trees

The relationships between the status and distribution of the seven key salmonids, the biophysical environment, and land management were explored quantitatively. We produced a set of predictions that reflect the likelihood of a species presence, or alternatively, the likely status of the population within a subwatershed. The predictions arose from statistical models, called classification trees, that elucidate the relationship between a set of predictor variables and a single response variable. In this section, we discuss the following: 1) the nature of the classification problem and classification trees 2) an example classification analysis of stream-type chinook salmon; 3) application and results for all seven species; and 4) a brief validation of results based on new information not used in the original analysis. We conclude this section with a general discussion of insights gained from the analysis.

Table 4.19—Key factors influencing status for rare and sensitive fish in the Basin.

	Key Risk Factors									
Species	Dams	Water quality	Water quantity	Harvest	Mining	Livestock	Forestry practices	Hatchery	Non-native interactions	Limited distribution
White sturgeon	X			X						
Klamath River lamprey	X		X							
River lamprey	X									
Pacific lamprey	X	X		X			X			
Goose Lake lamprey			X			X				
Pit-Klamath brook lamprey		X				X	X			
Sockeye salmon	X		X	X						
Chum salmon	X	X		X			X			
Coho salmon	X	X	X	X			X	X		
Coastal cutthroat trout				X			X	X	X	
Lahontan cutthroat trout		X	X			X			X	
Pygmy whitefish		X					X			
Burbot	X									
Sand roller		X							X	
Pit roach		X	X						X	
Alvord chub			X			X			X	
Borax Lake chub		X	X			X				X
Catlow tui chub			X			X				X
Oregon Lakes tui chub			X			X			X	
Summer Basin tui chub			X			X			X	X
Sheldon tui chub						X				X
Hutton tui chub		X				X				X
Leatherside chub			X		X	X	X			
Foskett speckled dace						X				X
Lost River sucker	X	X				X	X		X	
Wood River bridgelip sucker	X	X							X	
Warner sucker	X		X			X			X	
Goose Lake sucker	X	X	X			X	X			
Shortnose sucker	X	X				X	X		X	
Klamath largescale sucker	X	X	X			X	X		X	
Torrent sculpin		X	X			X	X			
Shorthead sculpin		X				X	X			
Pit sculpin		X				X	X			X
Slender sculpin		X							X	X
Margined sculpin		X				X	X			
Wood River sculpin		X	X			X			X	
Shoshone sculpin		X	X							X
Malheur sculpin		X				X	X		X	

Our primary objective in this analysis was to accurately predict fish presence or status using landscape features and management history. Ideally, this prediction should be based on a formal model that was developed in a rigorous and repeatable manner and that is accessible for review and validation. Better still, the model should rely on empirical data and provide a quantitative measure of the confidence that can be placed in our predictions.

We began by stating the problem in statistical terms, and for simplicity, considered the problem simply as one of predicting the presence of a species based on a set of predictor variables. We let y denote the response variable, presence, which could assume the values, P for present, or A for absent. Let $x_1, x_2, \ldots, x_k$ denote the set of k predictor variables, which are undefined for now, but which may include both continuous and discrete measures. Thus, we sought a model that can be expressed as:

$$\mathrm{Prob}(y = \mathrm{P}) = \mathrm{fn}(x_1, x_2, , \ldots, x_k),$$

which reads as, "the probability that y equals P is a function, fn, of predictor variables x_1, x_2, through x_k. By definition, $\mathrm{Prob}(y = \mathrm{A}) = 1 - \mathrm{Prob}(y = \mathrm{P})$." The function, fn(x), might assume a variety of forms, for example, simple constants, linear or nonlinear functions, or complex rule sets.

There are a variety of statistical approaches that might be suitable. Logistic regression is perhaps the more familiar approach (for example, Hosmer and Lemeshow 1989), but assumes a specific functional form of the model that requires the predictor variables to enter the model in a linear, or additive fashion. Furthermore, finding the best subset of predictors from a large set of potential predictor variables or incorporating multiple interactions is problematic. The limitations of logistic regression are more acute as the number of levels in the response variable increases. Logistic regression and other log-linear models have the advantage, however, of a large literature on the procedures and statistical inference for such models.

Tree-based models are one alternative to linear models that offer certain advantages (see Breiman and others 1984, Clark and Pregibon 1992, Crawford and Fung 1992, and Taylor and Silverman 1993; the following explanation is based principally on Clark and Pregibon 1992). First, where the set of predictor variables includes both continuous and discrete variables, tree-based models may be easier to interpret. Second, tree-based models are insensitive to monotonic transformations of the predictor variables, relying solely on the rank ordering of variables. Third, tree-based models are more adept at capturing non-additive behavior. The disadvantage of classification trees is that they are of limited utility for drawing statistical inferences. Thus, their primary uses are for building predictive models or data exploration, rather than testing of specific hypotheses.

For categorical response variables, tree-based models result in classification trees, so named because of the branching diagrams used to display the models which resemble inverted trees. To build a classification tree, one generates a dichotomous rule set through a process of recursive partitioning. Recursive partitioning involves sequentially splitting the data set into more homogeneous units, relative to the response variable, until a predefined measure of homogeneity is reached or no further subdivision is desired or feasible. Data are split at each juncture based on a single predictor variable that produces the greatest differences between the two resultant groups of observations. Predictor variables can be reused again at subsequent splits. The objective of the classification algorithm is to derive a terminal set of nodes, each containing a subset of the original data, where the distribution of the response variable is independent of the predictor variables to the greatest extent possible.

Details of the algorithm used to build the classification trees can be found in Clark and Pregibon (1992) and Statistical Sciences (1993).

Example Classification Tree Analysis—The process of developing a suitable classification tree requires several iterative steps. Some steps require professional judgement, beginning with the selection of the variables to include in the analysis. For previous analysis the derivation of meaningful response measures and a useful set of predictor variables was presented. Here, we describe our classification tree analysis using data on stream-type chinook salmon to illustrate the general procedure that we followed for each species.

Building a Classification Tree—All analyses involving classification trees were performed using the Splus®[2] programming language, following the procedures outlined in Clark and Pregibon (1992). Status of stream-type chinook salmon *(stc)*, was used as the response. Four possible values indicated whether the species was absent (A), present at depressed levels (D), present at strong levels (S), or used the subwatershed as a migration corridor (M). Twenty-nine of the potential predictor variables were made available to the model; *anadac* was excluded since the analysis was restricted to only those subwatersheds accessible to anadromous fish (see table 4.12).

The tree-fitting routine produced a full model that used 22 of the 29 potential predictors available to it. Excluded from the model were *alsi2*, *alsi4*, *bank*, *baseero*, *con2*, *drnden*, *hk*, *roaddn*, and *slope*. The model had 64 terminal nodes and an overall misclassification rate of 11.7 percent; predicted values did not match observed values for 147 of the 1,262 observations used to fit the model. While misclassification rate makes intuitive sense, statisticians generally prefer deviance as a measure of model performance and as a criterion for splitting nodes. Simply stated, deviance refers to the degree of homogeneity within a node. A set of identical observations, for example, would have zero deviance.

Cross-validation and Pruning—Many nodes near the termini of the full tree added little to change the predictions or reduce the deviance. This is typical of an over-fitted tree, which is produced intentionally. The next step was to prune the tree by removing the least important nodes. One consideration in tree pruning was to make a fairly robust tree, that is, one that performed well in making predictions using data not used to fit the tree. This posed something of a dilemma. We had limited data, and we wanted our tree to reflect the full expression of relationships found in all available data. Thus by holding out a subset of data for testing the model, we risked missing an important relationship that might exist in the subset, but is not reflected in the sample used for fitting.

To address this problem, Breiman and others (1984) developed a process of cross-validation which uses the full set of observations to build the full model, and mutually exclusive subsets of the data iteratively to prune and test the tree. Cross-validation produces a plot of the residual deviance of a successive set of trees of different sizes. The optimal tree size (minimum deviance) is suggested by the low point on the graph. This is the tree size that shows the highest out-of-sample prediction rate. For our example using stream-type chinook salmon, a tree with 26 nodes seemed appropriate. A smaller tree may predict responses equally well, but we chose to keep the additional nodes for the insight they provide. Sometimes, a higher residual deviance may be preferable if the larger tree adds to interpretation, or better discriminates between particular responses of interest such as distinguishing strong and depressed populations.

The reduced or pruned tree is presented in table 4.20. Figure 4.19 displays the rule set as an inverted tree. The letter within each node reflects the predicted value at that node. Part A of table 4.20 lists the variables used in the reduced tree and summarizes overall performance. Note that decreasing the size of the tree increased the overall misclassification rate to 16.7 percent. Part B of table 4.20 provides a complete listing of the tree structure and identifies each node, the variables and ranges defining each split, the number of observations, the deviance, the modal response, and the relative frequencies of each response level

[2]Splus is a registered trademark of Statistical Sciences Inc.

Table 4.20—The pruned classification tree relating status of stream-type chinook salmon to landscape variables.

A. Summary

Variables actually used in tree construction:

hucorder	pprecip	dampass	streams	mtemp	eru
mngclus	alsi1	solar	alsi3	con3	

Number of terminal nodes: 26

Residual mean deviance: 0.8282 Misclassification error rate: 0.1672 = 211 / 1262

B. Tree structure

Node) Split criterion	Sample size	Deviance	Mode	Relative Frequencies Absent	Depressed	Migration	Strong
1) root	1262	2718.00	A	0.43900	0.36450	0.190200	0.006339
2) hucorder < 30	991	1652.00	A	0.54990	0.41370	0.028250	0.008073
4) hucorder = 0	500	622.40	A	0.75800	0.22200	0.020000	0.000000
8) pprecip < 601	145	93.04	A	0.92410	0.03448	0.041380	0.000000 *
9) pprecip > 601	355	473.80	A	0.69010	0.29860	0.011270	0.000000
18) dampass < 4	23	13.59	D	0.08696	0.91300	0.000000	0.000000 *
19) dampass > 3	332	418.60	A	0.73190	0.25600	0.012050	0.000000
38) streams < 23	145	112.40	A	0.88280	0.11030	0.006897	0.000000 *
39) streams > 23	187	274.20	A	0.61500	0.36900	0.016040	0.000000
78) mtemp < 6.9	158	242.70	A	0.54430	0.43670	0.018990	0.000000
156) streams < 43.6	122	177.00	A	0.63110	0.35250	0.016390	0.000000
312) pprecip < 960.7	52	50.68	A	0.84620	0.13460	0.019230	0.000000 *
313) pprecip > 960.7	70	106.00	D	0.47140	0.51430	0.014290	0.000000 *
157) streams > 43.6	36	49.04	D	0.25000	0.72220	0.027780	0.000000
314) streams < 64.1	31	32.40	D	0.12900	0.83870	0.032260	0.000000 *
315) streams > 64.1	5	0.00	A	1.00000	0.00000	0.000000	0.000000 *
79) mtemp > 6.9	29	0.00	A	1.00000	0.00000	0.000000	0.000000 *
5) hucorder > 0	491	841.50	D	0.33810	0.60900	0.036660	0.016290
10) eru: 5,7	116	117.80	A	0.81900	0.17240	0.008621	0.000000
20) mngclus: BR,PA,PF,PR	106	85.95	A	0.87740	0.11320	0.009434	0.000000
40) alsi1 < 1.56	77	18.55	A	0.97400	0.02597	0.000000	0.000000 *
41) alsi1 > 1.56	29	45.20	A	0.62070	0.34480	0.034480	0.000000 *
21) mngclus: FG,FH,FM,TL	10	10.01	D	0.20000	0.80000	0.000000	0.000000 *
11) eru: 1,2,6,13	375	568.10	D	0.18930	0.74400	0.045330	0.021330
22) mngclus :BR,PA,PR	68	129.60	A	0.45590	0.44120	0.102900	0.000000
44) mtemp < 10.54	62	94.79	A	0.50000	0.48390	0.016130	0.000000 *
45) mtemp > 10.54	6	0.00	M	0.00000	0.00000	1.000000	0.000000 *
23) mngclus: FG,FH,FM,FW,PF,TL	307	394.20	D	0.13030	0.81110	0.032570	0.026060
46) dampass < 4	56	82.10	D	0.10710	0.75000	0.000000	0.142900
92) solar< 330.6	30	8.77	D	0.03333	0.96670	0.000000	0.000000 *
93) solar > 330.6	26	53.37	D	0.19230	0.50000	0.000000	0.307700
186) pprecip < 535.2	10	13.86	A	0.50000	0.50000	0.000000	0.000000 *
187) pprecip > 535.2	16	22.18	D	0.00000	0.50000	0.000000	0.500000 *
47) dampass > 4	251	280.20	D	0.13550	0.82470	0.039840	0.000000
94) hucorder <4	130	149.20	D	0.22310	0.76920	0.007692	0.000000 *
95) hucorder > 3	121	105.00	D	0.04132	0.88430	0.074380	0.000000
190) mngclus: FG,FM	51	0.00	D	0.00000	1.00000	0.000000	0.000000*
191) mngclus: FH,FW,PF	70	88.31	D	0.07143	0.80000	0.128600	0.000000*
3) hucorder > 29	271	334.40	M	0.03321	0.18450	0.782300	0.000000
6) hucorder < 58	74	138.00	M	0.09459	0.39190	0.513500	0.000000
12) eru: 5,7	10	12.22	A	0.70000	0.00000	0.300000	0.000000 *
13) eru: 1,6,13	64	88.16	M	0.00000	0.45310	0.546900	0.000000
26) alsi3 < 59.7	52	65.73	M	0.00000	0.32690	0.673100	0.000000
52) alsi1 < 0.37	9	6.28	D	0.00000	0.88890	0.111100	0.000000 *
53) alsi1 > 0.37	43	44.12	M	0.00000	0.20930	0.790700	0.000000 *
27) alsi3 > 59.7	12	0.00	D	0.00000	1.00000	0.000000	0.000000 *
7) hucorder > 57	197	155.60	M	0.01015	0.10660	0.883200	0.000000
14) dampass < 3	16	19.87	D	0.00000	0.68750	0.312500	0.000000*
15) dampass > 2	181	99.13	M	0.01105	0.05525	0.933700	0.000000
30) con3 < 0.49	90	19.18	M	0.02222	0.00000	0.977800	0.000000*
31) con3 > 0.49	91	63.00	M	0.00000	0.10990	0.890100	0.000000*

* denotes terminal node

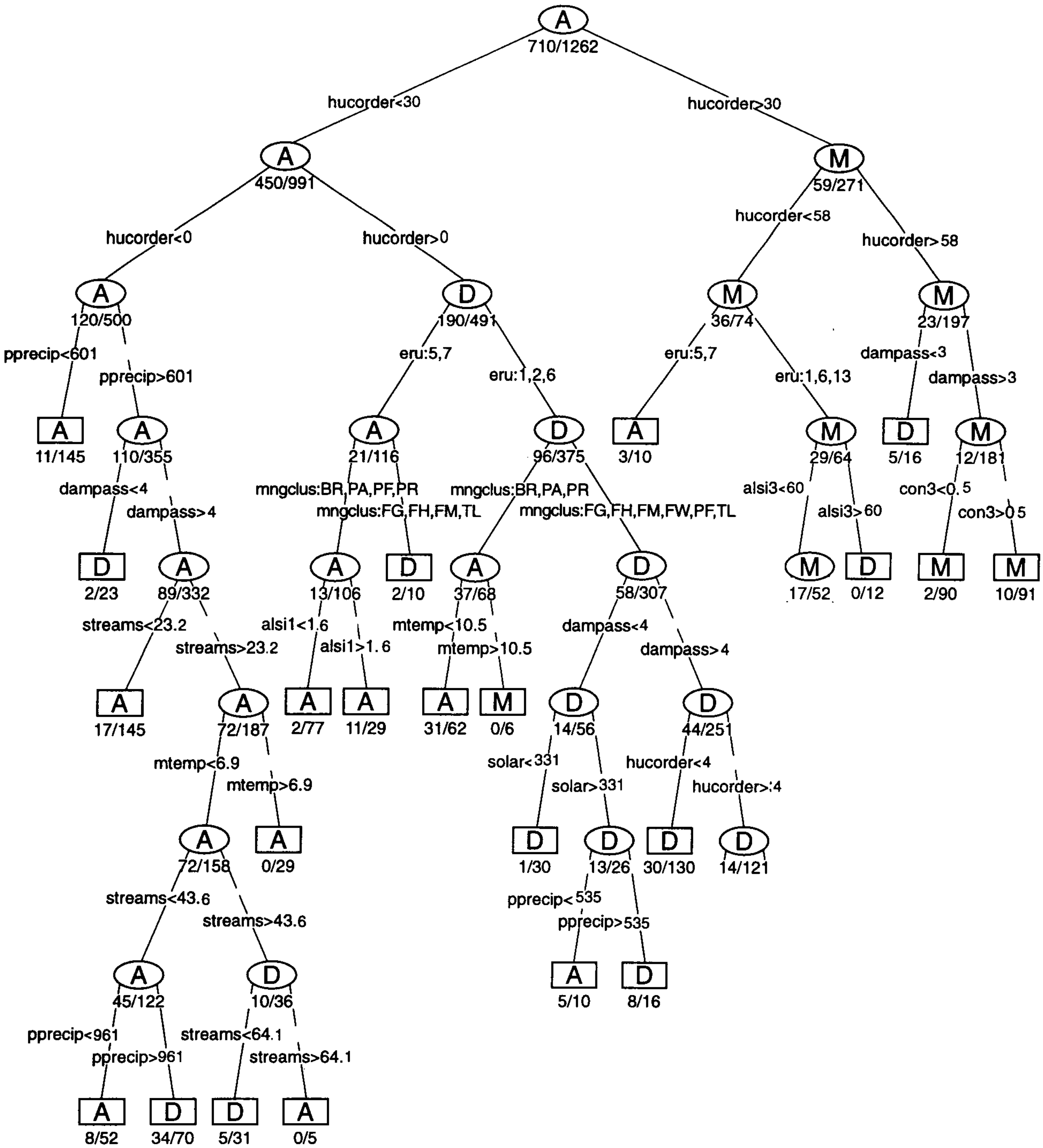


Figure 4.19—Graphical representation of pruned classification tree for stream-type chinook salmon. Each split is labeled with the appropriate predictor variable and splitting values. Letter designations within each node indicate the modal value at that node. The misclassification rate is shown under each node.

at that node. When used in a predictive fashion, these frequencies are equivalent to a probability of a given response. Normally, the modal response is equivalent to the predicted value. With the status data, however, three of the responses indicate different levels of presence, in contrast to a single absent value. Thus, we choose to predict absence only when the relative frequency was greater than or equal to 0.5. Otherwise, the predicted response was M, D, or S, whichever had the highest probability. The probability of presence was defined as one minus the probability of absence.

Prediction and Accuracy—Table 4.21 presents a cross-classification table of our predicted responses versus the observed response for all 1,694 observations in the data set. (Note that the 432 observations judged present-unknown or unknown were not used to build the tree.) The table shows both the accuracy of the predictions and the direction of the errors. Of primary interest are the diagonal cells of the top four rows, beginning in the upper left corner. For the diagonals, the row percentage indicates what percentage of the true values that were correctly predicted. Similarly, the column percentages indicate what fraction of our predictions was actually true. Row and column percentages can be quite different within a given cell. Which of these is more important depends on how the model is used. For predictive purposes, the column percentages perhaps are more mean-

Table 4.21—Cross-classification table of predicted versus observed values for stream-type chinook salmon. There are two or three values in each cell. The top value is the frequency, the second value is the row percentage, and the third is the column percentage for the top four rows.

	Frequency Row pct Column pct	Predicted Values: Absent	Migration Corridor	Present - Depressed	Present - Strong	Total
Observed Values	**Absent**	440 79.4% 89.4%	2 0.4% 0.9%	112 20.2% 20.7%	0 0.0% .	554 43.9%
	Migration Corridor	12 5.0% 2.4%	209 87.1% 90.9%	19 7.9% 3.5%	0 0.0% .	240 19.0%
	Present - Depressed	40 8.7% 8.1%	19 4.1% 8.3%	401 87.2% 74.3%	0 0.0% .	460 36.5%
	Present - Strong	0.0 0.0% 0.0%	0.0 0.0% 0.0%	8 100.0% 1.5%	0 0.0% .	8 0.6%
	Present - Unknown	198 68.0%	3 1.0%	90 30.9%	0 0.0%	291
	Unknown	67 47.5%	1 0.7%	73 51.8%	0 0.0%	141
	Total	757 44.7%	234 13.8%	703 41.5%	0 0.0%	1694

ingful because they indicate the probability that predictions are correct. For identifying relationships, row percentages may be more useful because they show the percentage of known observations that were correctly predicted by the model.

In the example using stream-type chinook salmon, the model does an excellent job of predicting all but present-strong responses within the data used to build the tree. Given that there were only eight (0.6%) present-strong responses in the complete data set, we would not expect the model to distinguish them easily. Yet, looking closely at node 187 in table 4.20 B reveals that all eight strong responses fell within a single node. For node 187, the probability of strong is 0.5. These observations are predicted to be depressed because the rules were set to choose "depressed" in case of a tie between strong and depressed. A second alternative would be to add a further split at this node that separates depressed from strong.

Application to the Seven Key Salmonids— The model building exercise outlined above for stream-type chinook salmon was repeated for ocean-type chinook salmon and steelhead. For the non-anadromous species, several modifications were made. *Dampass* was replaced in the list of potential predictors by *anadac*, indicating if an area was accessible to anadromous fish (which could have positive or negative consequences for resident fish) (see table 4.12). Two separate classification trees were built for each non-anadromous species. In the first analysis, migration corridors were combined with absent calls and a model built to predict status (A, D, or S) in only spawning and rearing areas. In the second analysis, migration corridors, present-depressed, and present-strong were combined into a single prediction, present. A model was then built using the binomial response, present or absent. In neither case were the present-unknown calls used in model building.

The resulting trees, following cross-validation and pruning, are summarized in table 4.22; further details are provided in appendix 4E. The trees ranged in size from 57 nodes for bull trout status, to 9 nodes for predicting Yellowstone cutthroat trout presence only. The size of the trees was roughly proportional to the areal extent of the species range, and thus to the degree of complexity expressed throughout the range. Yellowstone cutthroat trout and ocean-type chinook salmon had the most limited ranges and showed excellent fits with rather small trees. Redband and bull trout, which had much larger ranges, were more difficult to fit accurately, and showed the highest misclassification rates and residual mean deviances.

Cross-classification tables of status calls versus predicted status provide further insight into both the accuracy of the models, and landscape-fish relationships. For the anadromous species (table 4.23), the models did a good job of distinguishing absence, migration corridors, present-depressed, and present-strong. Errors followed a reasonable pattern. For example, absent was often confused with depressed and rarely confused with strong; strong was confused primarily with depressed. Note that the models recognized no ordinal relationships among the responses, *a priori*. Thus, to see reasonable patterns in the errors reinforces the conclusion that the models identified meaningful relationships.

Similar patterns were observed in the status predictions for non-anadromous species (table 4.24). Since migration corridors were grouped with the absent calls in model fitting, migration corridors should have been identified as absent. This pattern held true where significant numbers of migration calls existed, namely for bull and redband trout. While the overall success rates were favorable, increased inaccuracies occurred with the non-anadromous predictions. Localized factors such as hatchery stocking, fishing, or predation, and factors outside the watershed were not reflected in the predictor variables.

We also evaluated the predictive success by examining the mean probability of presence resulting from each model, calculated for each response level (table 4.25). For the anadromous species, the probabilities of presence were the sums of probabilities for migration corridor, present-depressed,

Table 4.22—Summaries of the classification trees developed for seven key salmonids.

Model	Bull trout			Westslope cutthroat			Yellowstone cutthroat			Redband trout		
Status												
Predictor Variables	*mtemp*	*con3*	*con1*	*eru*	*mtemp*	*solar*	*eru*	*solar*	*drnden*	*mngclus*	*mtemp*	*streams*
	roaddn	*elev*	*streams*	*mngclus*	*alsi1*	*pprecip*	*mtemp*	*hk*	*pprecip*	*slope*	*alsi3*	*elev*
	hucorder	*slope*	*alsi1*	*roaddn*	*elev*	*vmf*	*hucorder*	*ero*	*sdt1*	*bank*	*anadac*	*solar*
	vegclus	*bank*	*mngclus*	*hk*	*drnden*	*vegclus*				*vmf*	*vegclus*	*baseero*
	solar	*eru*	*slope2x*	*bank*	*sdt3*	*sdt1*				*hucorder*	*con3*	*ero*
	alsi3	*baseero*	*con2*	*alsi4*	*con2*	*hucorder*				*eru*	*pprecip*	*drnden*
	ero	*pprecip*	*vmf*	*sdt2*	*slope*						*con1*	
Terminal Nodes		57			37			11			35	
Residual Mean Deviance		0.83			1.00			0.79			1.14	
Misclassification Error Rate		0.184 = 500 / 2717			0.197 = 323 / 1640			0.1501 = 59 / 393			0.2426 = 435 / 1793	
Presence/Absence												
Predictor Variables	*slope*	*baseero*	*vegclus*	*eru*	*con2*	*bank*	*mtemp*	*hk*	*hucorder*	*pprecip*	*hucorder*	*anadac*
	hucorder	*bank*	*con3*	*slope2x*	*vegclus*	*drnden*	*solar*	*ero*	*con1*	*baseero*	*slope*	*drnden*
	elev	*alsi2*	*streams*	*sdt3*	*slope*	*streams*			*alsi3*	*eru*	*solar*	*sdt2*
	pprecip	*slope2x*	*drnden*	*mngclus*	*hk*	*hucorder*				*mtemp*	*alsi3*	*slope2x*
	eru	*mtemp*	*hk*	*elev*	*anadac*	*mtemp*				*bank*	*mngclus*	*streams*
	ero	*con1*	*anadac*	*ero*	*pprecip*					*hk*		
		con2										
Terminal Nodes		42			24			9			20	
Residual Mean Deviance		0.74			0.42			0.17			0.78	
Misclassification Error Rate		0.167 = 454 / 2717			0.075 = 123 / 1640			0.0204 = 8 / 393			0.086 = 155 / 1793	

Model	Steelhead			Ocean-type chinook			Stream-type chinook		
Status									
Predictor Variables	*hucorder*	*pprecip*	*sdt2*	*hucorder*	*ero*	*pprecip*	*hucorder*	*solar*	*mtemp*
	eru	*drnden*	*streams*	*vmf*	*elev*	*slope*	*pprecip*	*alsi3*	*eru*
	vmf	*vegclus*	*elev*	*dampass*			*dampass*	*con3*	*mngclus*
	slope	*alsi3*	*bank*				*streams*		*alsi1*
	baseero	*mngclus*	*con1*						
	solar	*hk*	*dampass*						
Terminal Nodes		35			12			26	
Residual Mean Deviance		0.58			0.43			0.83	
Misclassification Error Rate		0.114 = 154 / 1355			0.076 = 17 / 224			0.167 = 211 / 1262	

Table 4.23—Cross-classification of the observed status calls with the predicted values for three anadromous taxa. There are two or three values in each cell. The top value is the frequency, the second value is the row percentage, and the third is the column percentage for the top four rows.

	Frequency	Steelhead Predicted Values					Ocean-type chinook Predicted Values					Stream-type chinook Predicted Values				
	Row pct															
	Col pct	A	M	D	S	Total	A	M	D	S	Total	A	M	D	S	Total
Observed	**Absent**	124	1	53	0	178	89	1	2	1	93	440	2	112	0	554
Values	**(A)**	70%	1%	30%	0%		96%	1%	2%	1%		79%	0%	20%	0%	
		79%	0%	6%	0%	13.1%	93%	2%	4%	5%	41.5%	89%	1%	21%	.	44%
	Migration	2	196	28	0	226	2	54	1	2	59	12	209	19	0	240
	Corridor	1%	87%	12%	0%		3%	92%	2%	3%		5%	87%	8%	0%	
	(M)	1%	87%	3%	0%	16.7%	2%	96%	2%	10%	26.3%	2%	91%	4%	.	19%
	Present -	30	29	870	2	931	5	0	47	0	52	40	19	401	0	460
	Depressed	3%	3%	93%	0%		10%	0%	90%	0%		9%	4%	87%	0%	
	(D)	19%	13%	91%	15%	68.7%	5%	0%	90%	0%	23.2%	8%	8%	74%	.	36%
	Present -	0	0	9	11	20	0	1	2	17	20	0	0	8	0	8
	Strong	0%	0%	45%	55%		0%	5%	10%	85%		0%	0%	100%	0%	
	(S)	0%	0%	1%	85%	1.5%	0%	2%	4%	85%	8.9%	0%	0%	1%	.	1%
	Present -	61	8	210	0	279	9	3	3	1	16	67	1	73	0	141
	Unknown	22%	3%	75%	0%		56%	19%	19%	6%		48%	1%	52%	0%	
	Unknown	68	8	191	0	267	44	2	2	3	51	198	3	90	0	291
		25%	3%	72%	0%		86%	4%	4%	6%		68%	1%	31%	0%	
	Total	285	242	1361	13	1901	149	61	57	24	291	757	234	703	0	1694
		15%	13%	72%	1%		51%	21%	20%	8%		45%	14%	41%	0%	

Table 4.24—Cross-classification of the observed status calls with the predicted values in spawning and rearing areas for four non-anadromous salmonids. Migration corridors were grouped with absent calls in building the models, but are separated here to demonstrate classification patterns. (For example, most migration corridors are predicted as absent for spawning and rearing bull trout.)

	Frequency Row pct Col pct	Bull trout Predicted Values A	D	S	Total	Westslope cutthroat Predicted Values A	D	S	Total	Yellowstone cutthroat Predicted Values A	D	S	Total	Redband trout Predicted Values A	D	S	Total
Observed Values	Absent (A)	1445	81	28	1554	165	84	22	271	97	3	1	101	685	132	8	825
		93%	5%	2%		61%	31%	8%		96%	3%	1%		83%	16%	1%	
		69%	17%	19%	57.2%	67%	8%	7%	16.5%	92%	2%	1%	25.7%	82%	18%	4%	46%
	Present - Depressed (D)	267	335	22	624	33	896	59	988	9	96	11	116	117	525	70	712
		43%	54%	4%		3%	91%	6%		8%	83%	9%		16%	74%	10%	
		13%	70%	15%	23.0%	13%	82%	20%	60.2%	8%	73%	7%	29.5%	14%	72%	31%	40%
	Present - Strong (S)	31	48	90	169	9	103	218	330	0	29	141	170	32	73	145	250
		18%	28%	53%		3%	31%	66%		0%	17%	83%		13%	29%	58%	
		1%	10%	61%	6.2%	4%	9%	73%	20.1%	0%	22%	91%	43.3%	4%	10%	65%	14%
	Migration Corridor (M)	344	18	8	370	38	12	1	51	0	4	2	6	3	2	1	6
		93%	5%	2%		75%	24%	2%		0%	67%	33%		50%	33%	17%	
		16%	4%	5%	13.6%	16%	1%	0%	3.1%	0%	3%	1%	1.5%	0%	0%	0%	0%
	Present - Unknown	322	71	45	438	106	272	193	571	45	26	12	83	807	761	449	2017
		74%	16%	10%		19%	48%	34%		54%	31%	14%		40%	38%	22%	
	Unknown	1065	116	56	1237	140	222	62	424	76	29	22	127	864	586	116	1566
		86%	9%	5%		33%	52%	15%		60%	23%	17%		55%	37%	7%	
	Total	3474	669	249	4392	491	1589	555	2635	227	187	189	603	2508	2079	789	5376
		79%	15%	6%		19%	60%	21%		38%	31%	31%		47%	39%	15%	

Table 4.25—Sample size (N) and estimated probability of presence for seven key salmonids, cross-referenced with observed status.

	Bull trout		Westslope cutthroat		Yellowstone cutthroat		Redband trout		Steelhead		Ocean-type chinook		Stream-type chinook	
Status call	N	Prob.	N	Prob.	N	Prob.	N	Prob.	N	Prob.	N	Prob.	N	Prob.
absent	1554	0.202	271	0.074	101	0.356	825	0.276	178	0.341	93	0.098	554	0.208
migration corrider	370	0.775	51	0.992	6	0.890	6	0.851	226	0.987	59	0.943	240	0.942
present - depressed	624	0.691	988	0.957	116	0.931	712	0.750	931	0.939	52	0.897	460	0.780
present - strong	169	0.771	330	0.986	170	0.933	250	0.806	20	0.980	20	0.980	8	1.000
present - unknown	438	0.554	571	0.363	83	0.809	2017	0.637	279	0.755	16	0.462	141	0.475
unknown	1237	0.303	424	0.413	127	0.660	1566	0.467	267	0.697	51	0.196	291	0.290

and present-strong from the status model. For non-anadromous species, the probabilities of presence were derived from the present/absent models. Again, the model results were consistent; the highest probabilities generally occurred with the present-strong calls and there was strong separation between the absent calls and any of the present calls. Migration corridors were also well defined, exhibiting the highest probability of presence for four of the seven species.

The predicted status and probability of presence where the status calls were present-unknown or unknown both have a lower overall likelihood of supporting populations than areas where we know that the species are present and their status. The fact that the mean probabilities of presence in present-unknown areas are consistently low across non-anadromous species may have more onerous implications. Model results suggest that these areas generally are less likely to support populations than areas where better information is available. Stated another way, unidentified, major population strongholds within the Basin are unlikely; if fish are there in abundance, we generally know of their presence. The predicted probabilities of presence are even less for the totally unknown areas. Other than for steelhead and Yellowstone cutthroat trout, unknown areas tend to resemble areas that do not support fish populations. Naturally, there are exceptions. Where unknown areas are expected to have strong populations of some species (for example, westslope cutthroat trout within the northern Cascades), these areas tend to be in proximity to areas where known populations are strong.

The classification trees were used to predict the status for each of the key salmonids in watersheds where the biologists' classifications were unknown or present-unknown. In the following section we summarize both the known and predicted occurrence and status for each of the seven salmonids. Some watersheds within the historical range of each fish do not have predictions, either because of missing landscape information or corrections in the original status call based on information obtained after predictions were made. The omissions include 14 watersheds for bull trout, 50 for Yellowstone cutthroat trout, five for westslope cutthroat trout, 127 for redband trout, 22 for steelhead, 8 for stream-type chinook salmon and 4 for ocean-type chinook salmon.

Interpretation of Classification Trees—We leave specific interpretations of tree structures to the species narratives that follow. There are two general issues to consider when examining the models. First, the models were not designed to draw strong inferences about linkages between specific watershed characteristics and species status. Our intent was to develop predictive models that would allow us fully to map the distribution and status of the key salmonids and to identify apparent relationships within the data, not test specific hypotheses. Other methods more conducive to testing hypotheses may be pursued in the future.

Second, although predictive variables were limited to a subset not strongly correlated throughout the Basin, spatially driven correlations were still present and may have been strengthened as the classification routine limited observations to smaller regions. Exploration of individual models demonstrated that several variables could often provide nearly equal power in discrimination. The first split in the bull trout model, for example, was based on mean temperature, although elevation, management cluster, and vegetation cluster performed nearly as well. Correlation in the predictive variables is likely because all tend to describe colder, higher-elevation, and steeper settings. Which variable or set of variables provide the more meaningful ecological interpretation is a subjective judgement. Rather than select the more interpretable variables from a subset of likely candidates at each node, we used the statistically optimal choice. This means that the relationships were not always easy to interpret.

The suite of predictor variables that emerged in the models can be broadly grouped into three categories as follows: 1) causal variables, such as solar radiation and road density, where one might postulate a direct causal link between the variables and fish response; 2) correlative variables, such as the erosion index, that likely are responding to similar environmental drivers as fish; and, 3) serendipitous variables, those which show up unexpectedly as good predictors without an apparent ecological mechanism. The serendipitous relationships may represent a fruitful area for hypothesis generation and future research.

Overall, the patterns suggested by the classification trees were consistent with our understanding of salmonid biology and habitat use. Variables related to temperature, stream size, slope, vegetative cover and solar radiation commonly provided important discrimination in the analyses. Cooler temperatures, for example, generally suggested a higher likelihood of finding salmonids; the differences in splitting temperatures associated with bull trout (5.1° C), Yellowstone cutthroat trout (6.1° C), and redband trout (8°-10° C) are also consistent with differences in thermal preferences or tolerances suggested for these species from other work (for example, Mullan and others 1992). As a second example, the classification trees associated the distributions of steelhead and redband trout with steeper slopes, higher elevation, and higher solar radiation. Although redband trout are a cold-water salmonid, they exhibit higher temperature tolerances than others and, within limits, have shown increased production and abundance associated with canopy openings and increased solar radiation. Redband trout have also been found to use steeper gradient channels (Kunkel 1976).

The preponderance of physiographic and geophysical predictor variables within the models suggest that biophysical setting is an important determinant of species distributions and habitat suitability. Variables potentially reflecting the degree of human disturbance within watersheds (roads, management cluster, dams) were important in the distributions of bull trout, westslope cutthroat trout, steelhead, and both chinook salmon. *In no instances could increased disturbance of natural landscapes be interpreted as a positive effect.* The number of mainstem dams in the migratory corridor showed a consistent, negative impact on chinook salmon. Management cluster and road density, however, did not always produce better predictions for all species and no species were excluded from more intensively managed landscapes. However, the relative difficulty in distinguishing status, particularly between strong and depressed, indicates that other effects acting at different scales are important as well. For example, sampling and classification error potentially influenced model results, and many potentially important effects simply could not be represented in the analysis. We could not incorporate the potential effects of fishing, poaching, introduced species, or pollutants other than as surrogates tied to the intensity of land use. We could not incorporate information reflecting the size or spatial patterns in habitat patches that might influence the degree of fragmentation or connectivity among local populations.

The lack of a consistently strong influence of disturbance related variables in the classification trees cannot be construed as evidence that human related activities are not important for some species. A more refined analysis is necessary to explore the relative significance and covariance among the potentially important variables. The resolution in the summary of data may also mask some relationships. The loss of resolution in summarizing information at a subwatershed scale may mask important processes which occur primarily at finer scales characteristic of individual streams or stream reaches. Roughly 30% of all the subwatersheds defined in the Basin are not true watersheds, but represent composites including small tributaries and higher-order reaches of a mainstem stream or river. Because land disturbing activities are likely to be integrated across true watersheds (that is, the entire area drained above a point of interest), the characterization of disturbance within the higher-order watersheds that do not include the headwaters may not reflect the conditions that directly influence habitats and fish.

Validation—Additional fish-survey data became available after the analysis and predictions were completed. Rather than update the databases and redo the analysis, we used the new information as a simple check of the status calls and classification-tree predictions. The new data were made available from fish surveys conducted by the Plum Creek Timber Co. on streams draining their lands in Washington, Idaho, and Montana; from recent updates of the Washington Streams and Lakes database maintained by the Washington Department of Fish and Wildlife; and from fish surveys by the Oregon Department of Fish and Wildlife in the Klamath River basin. We summarized observations by subwatershed as in the original classification and we compared the summary directly with the model predictions for each subwatershed. Observations were available for 1,377 combinations of species and subwatersheds where we had existing status classifications, and 203 species-subwatershed combinations with unknown status. New information was available for all of the key salmonids except Yellowstone cutthroat trout.

Classification success can be judged by the relative frequency with which new data agreed with the status calls or predictions (table 4.26). Three patterns emerged. First, the new data agreed with the status calls more frequently than with the predictions. That pattern is expected simply from the error in predictions resulting from the inherent variability in distributions. Second, the agreement between the new data and status calls or predictions were consistently better for known absent or predicted absent than for known present or predicted present. That is, our models did a better job of anticipating where fish would not be found than where they would be found. Third, the lowest agreement rates occurred with known or predicted presence of the more rare taxa, specifically the anadromous species and bull trout. There are several possible interpretations. One is that these fish are simply less abundant or widespread than we estimate. Alternatively, it may be an artifact of sampling that was inadequate to detect species that were present, but in low abundance.

This comparison of our status classifications and predictions does not meet the standards of a rigorous validation, in that it was not applied systematically across the Basin. Nor are we confident that the new data fully reflect the species status of each subwatershed. Regardless, the general conclusion from this limited exercise is that there are likely errors in our distribution maps of key salmonids, but these errors are not so large as to compromise our general assessment. Errors are most likely in the model predictions, as with any model. Because predictions are probabilistic and based on the broad patterns of occurrence for each species, analysis of broad scale patterns summarized by ERU or subbasin will be little influenced by the errors. Results in subwatersheds, however, will vary. We emphasize our caution that local validation should always accompany the application of any of the distribution and status information in watershed or subwatershed-scale analysis, planning, and management.

Discussion of Key Salmonids Status, Distribution and Management

In this section we summarize the distribution and status of the seven key salmonids. Although we briefly review the biology and life history of each species, we focus on information relevant to the current condition of remaining populations and the factors influencing those populations.

Each species may exhibit a complex set of life-history patterns that include varying age-at-maturity, frequency and timing of spawning, age and seasonal timing and pattern of migration, longevity, habitat selection and a host of other characteristics. We have already noted major differences associated with the life histories of redband trout and steelhead, and chinook salmon. In the following discussion of each species we highlight important life-history characteristics but repeatedly recognize the variation in migratory patterns as a particularly important and well-known phenomenon. In the remaining portion of this chapter we use the nomenclature of anadromous, non-anadromous to refer to salmonids that do and do

Table 4.26—Cross-classification table of status calls (known present and known absent) and model predictions (predicted present and predicted absent) versus recently obtained data from three sources. Numbers refer to subwatersheds where fish were found or not found during sampling. Percentages refer to the agreement or disagreement between the new data and each staus call or prediction (for example, bull trout were found in 64.8% of sampled subwatersheds with previously known bull trout presence). Shaded cells highlight points of agreement.

Species		Data Source[1]	Known Present	Predicted Present	Known Absent	Predicted Absent
Bull Trout	found	WSL	78	0	2	7
		PC	21	1	0	1
		OK	54	0	0	0
		total	153	1	2	8
			64.8%	12.5%	2.6%	11.8%
	not found	WSL	28	1	17	29
		PC	15	2	16	11
		OK	40	4	41	20
		total	83	7	74	60
			35.2%	87.5%	97.4%	88.2%
Westslope Cutthroat Trout	found	WSL	67	3	4	2
		PC	50	4	1	0
		OK	17	0	0	0
		total	134	7	5	2
			71.3%	46.7%	50.0%	28.6%
	not found	WSL	43	6	5	5
		PC	11	1	0	0
		OK	0	1	0	0
		total	54	8	5	5
			28.7%	53.3%	50.0%	71.4%
Redband Trout	found	WSL	99	6	2	11
		PC	9	0	0	2
		OK	218	16	3	2
		total	326	22	5	15
			79.5%	56.4%	50.0%	53.6%
	not found	WSL	38	2	2	4
		PC	6	4	2	5
		OK	40	11	1	4
		total	84	17	5	13
			20.5%	43.6%	50.0%	46.4%
Steelhead	found	WSL	12	1	0	0
			12.8%	9.1%	0.0%	0.0%
	not found	WSL	82	10	36	2
			87.2%	90.9%	100.0%	100.0%
Ocean-type Chinook Salmon	found	WSL	4	0	2	2
			25.0%		10.0%	33.3%
	not found	WSL	12	0	18	4
			75.0%		90.0%	66.7%
Stream-type Chinook Salmon	found	WSL	23	0	2	2
		OK	38	0	1	0
		total	61	0	3	2
			40.1%	0.0%	2.3%	15.4%
	not found	WSL	48	4	45	3
		OK	43	2	83	8
		total	91	6	128	11
			59.9%	100.0%	97.7%	84.6%

[1]WSL=Washington streams and lakes database; PC=Plum Creek Timber Co. surveys for Washington, Idaho, and Montana; OK=Oregon Department of Fish and Wildlife surveys for the Klamath River Basin.

not migrate to and from the ocean, respectively. We recognize non-anadromous forms as resident or migratory.

Migratory populations include fluvial fish that migrate from larger streams into smaller tributaries to spawn and adfluvial fish that migrate from lentic waters to spawn in inlet or outlet tributaries.

Bull Trout (*Salvelinus confluentus*)—Bull trout are recognized as a species of special concern by state management agencies (Rieman and McIntyre 1993) and the American Fisheries Society (Williams and others 1989), and as a sensitive species by the Forest Service. Concern regarding the species' status is evident with their classification as a Category 1, Candidate Species, under the Endangered Species Act (listing warranted but precluded because of other species management priorities) [Federal Register June 12, 1995 (U.S. Government 1995)].

Currently bull trout are found in many major river systems within the Basin, but spawning and juvenile rearing are believed to be restricted to cold and relatively pristine waters. Headwaters of most basins still support populations. The fish species assemblages in higher elevation streams of the Basin are typically simple. Because bull trout are the only native char and often the only important piscivore, the loss of a bull trout population can represent a major loss of species diversity and ecological function.

Historical Distribution—The historical range of bull trout was restricted to North America (Cavendar 1978; Haas and McPhail 1991). Bull trout have been recorded from the McCloud River in northern California, the Klamath River basin in Oregon and throughout much of interior Oregon, Washington, Idaho, western Montana, and British Columbia. They occur east of the Continental divide in headwaters of rivers entering Hudson Bay, and in headwaters of rivers entering the Arctic Ocean. Except in the Little Lost and Big Lost rivers in Idaho, bull trout are not known to occur in the Snake River subbasin above Shoshone Falls.

Bull trout are believed to be a glacial relict (McPhail and Lindsey 1986), and their broad distribution has probably contracted and expanded periodically with natural climate change (Williams and others, in press). Genetic variation suggests an extended and evolutionarily important isolation between populations in the Klamath and Malheur Basins and those in the Columbia River basin (Leary and others 1993). Populations within the Columbia River basin are more closely allied and are thought to have expanded from common glacial refugia or to have maintained higher levels of gene flow among populations in recent geologic time (Williams and others, in press).

Our evaluation of the probable historical range of bull trout within the Basin is summarized in appendix 4D. We estimate the historical range for bull trout as about 60 percent of the Basin (table 4.27). It is unlikely that bull trout occupied all of the accessible streams at any one time. Distribution of existing populations is often patchy even where numbers are still strong and habitat is in good condition (Rieman and McIntyre 1993; Rieman and McIntyre 1995). Habitat preferences or selection is likely important (Dambacher and others, in press; Goetz 1994; Rieman and McIntyre 1995;) but more stochastic extirpation and colonization processes may influence distributions even within suitable habitats (Rieman and McIntyre 1995).

Even though bull trout may move throughout whole river basins seasonally, spawning and juvenile rearing appear to be limited to the coldest streams or stream reaches. The lower limits of habitat used by bull trout are strongly associated with gradients in elevation, longitude, and latitude, that likely approximate a gradient in climate across the Basin (Goetz 1994). The patterns indicate that spatial and temporal variation in climate may strongly influence habitat available to bull trout (see Meisner 1990 for an example with brook trout). While temperatures are probably suitable throughout much of the northern portion of the range, predicted spawning and rearing habitat are restricted to increasingly isolated high elevation or headwater "islands" toward the south (Goetz 1994; Rieman and McIntyre 1995)(map 4.9).

<u>Current Status and Distribution</u>—A total of 2,717 observations was available for the classification analysis. Overall classification success was 82 percent and ranged from 53 to 93 percent for each status class (see tables 4.22 and 4.24). Overall classification success was about 83 percent when limited to presence or absence (table 4.22). The classification success rates reflect the dominance of absence in the observations. Prediction success ranged from 61 to 70 percent. The models tended to over-predict absence and under-predict strong and depressed. Watershed characteristics most useful in the classifications included mean annual air temperature, vegetation cluster, road density, *hucorder*, mean annual precipitation, elevation, slope and several erosion and lithology variables (table 4.22, appendix 4E.). We used the classification models to estimate the probability of occurrence within 1,247 unclassified watersheds (28% of the range). We predicted spawning and rearing status in the unclassified watersheds and an additional 442 watersheds where bull trout were known present but status was unknown (total predictions represented about 38% of the range)(table 4.28).

Based on our known and predicted status and distribution, bull trout are widely distributed throughout the Basin. The largest contiguous areas supporting populations, however, are associated with the mountains of north central Idaho and northwestern Montana (map 4.9). The current distribution corresponds well with our predictions of potential spawning and rearing habitat (map 4.10 and 4.11).

Bull trout are now extinct in California and only remnant populations are found in much of Oregon (Ratliff and Howell 1992). A small population still exists in the headwaters of the Jarbidge River, Nevada which represents the present southern limit of the species range. Bull trout are known or predicted to occur in 45 percent of watersheds in the historical range and to be absent in 55 percent (map 4.10 and 4.11; tables 4.27 and 4.28). The historical range associated with the Owyhee Uplands, the Columbia Plateau, and the Upper Klamath are poorly represented with both the lowest number of subwatersheds and the lowest proportion of subwatersheds present or predicted within the potential historical range (table 4.27).

We found bull trout less widely distributed within the potential historical range than the other non-anadromous salmonids. Because much of the historical distribution is speculative, we cannot quantify distributional losses across the range. Current information indicates, however, that despite the relatively broad distribution, this species is in widespread decline (Howell and Buchanan 1992; Montana Bull Trout Scientific Committee, in preparation; Pratt and Huston 1993; Ratliff and Howell 1992; Thomas 1992; USDA 1996). Watersheds believed or predicted to be strong spawning and initial rearing areas represented only 6 percent of the historical range; if the sample is restricted to only those watersheds classified as strong or depressed in spawning and rearing habitat, about 25 percent were believed or predicted to be strong (table 4.28).

Migratory life histories have been lost or limited throughout the range (for example, Goetz 1994; Jakober 1995; Montana Bull Trout Scientific Committee, in preparation; Pratt and Huston 1993; Ratliff and Howell 1992; Rieman and McIntyre 1993, 1995). There is evidence of declining trends in some populations (Mauser and others 1988; Pratt and Huston 1993; Schill 1992; Weaver 1992) and extirpations of local populations are reportedly widespread. Goetz (1994) found that bull trout still exist in about 56 percent of the historically used habitat throughout the Deschutes River Basin, but only 0.5 percent of that in the upper basin. Brown (1992) reported

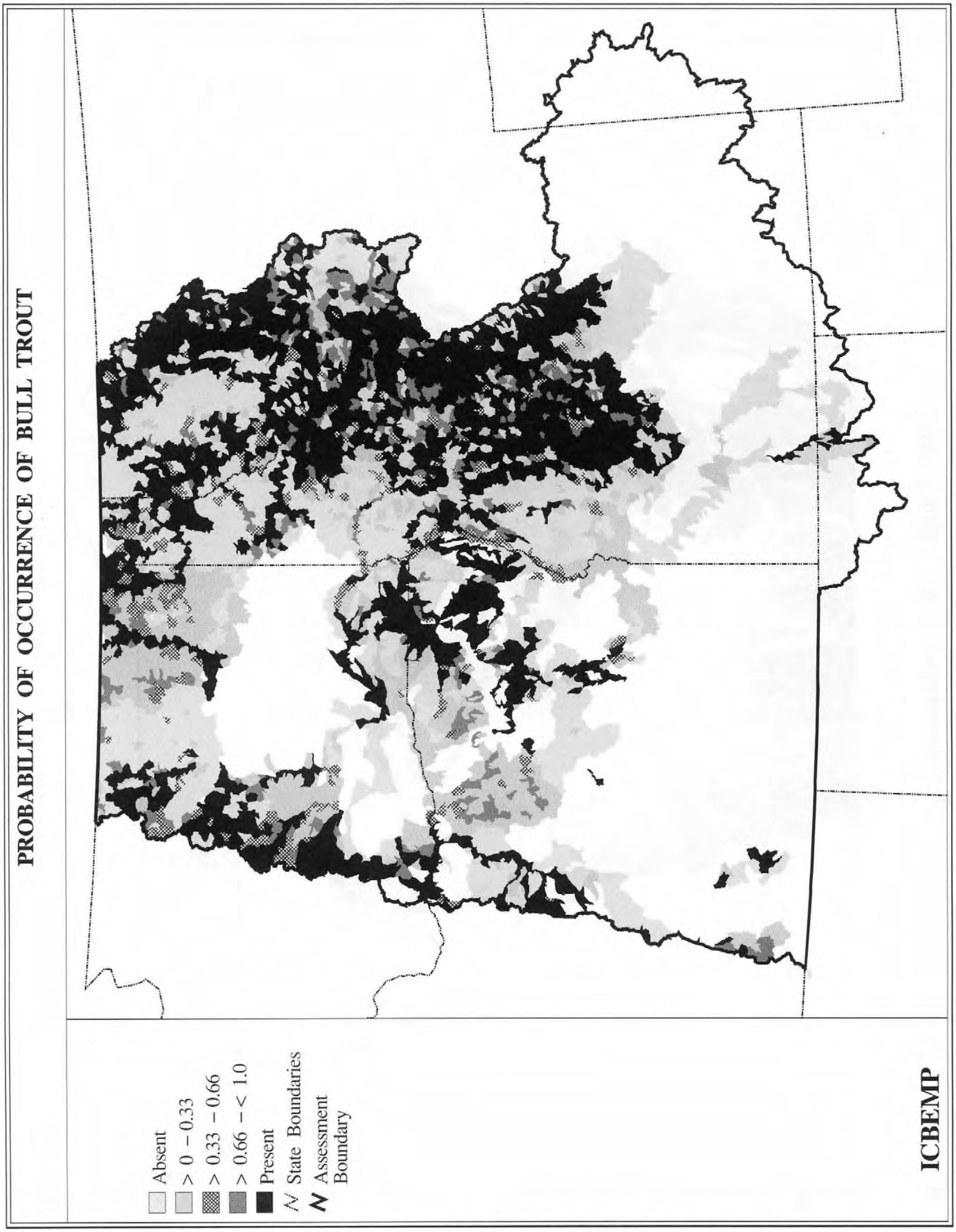
Table 4.27—Summary of the current status and distribution classifications (number of subwatersheds) for bull trout throughout the Ecological Reporting Units of the Basin. Potential spawning and rearing was predicted as in map 4.10.

				Status Where Present							
Ecological Reporting Unit	Total	Historical Range	Total Present	Strong*	Depressed*	Status Unknown	Corridor	Introduced	Absent	No Classification	Potential Spawning and Rearing
Northern Cascades	340	319	144	10	22	90	22	0	84	91	310
Southern Cascades	141	80	32	10	6	9	7	0	40	8	65
Upper Klamath	175	42	9	0	9	0	0	0	17	16	40
Northern Great Basin	506	0	0								
Columbia Plateau	1089	551	59	3	19	17	20	0	217	275	204
Blue Mountains	695	442	163	23	72	31	37	0	177	102	358
Northern Glaciated Mountains	955	955	316	26	125	77	88	0	335	304	924
Lower Clark Fork	415	415	166	4	75	46	41	0	160	89	408
Upper Clark Fork	306	306	145	21	109	5	10	0	84	77	306
Owyhee Uplands	956	174	7	0	2	1	4	0	102	65	43
Upper Snake	301	0	0								
Snake Headwaters	387	0	0								
Central Idaho Mountains	1232	1130	567	72	186	167	142	0	343	220	1065
Entire Assessment Area	7498	4414	1608	169	625	443	371	0	1559	1247	3723

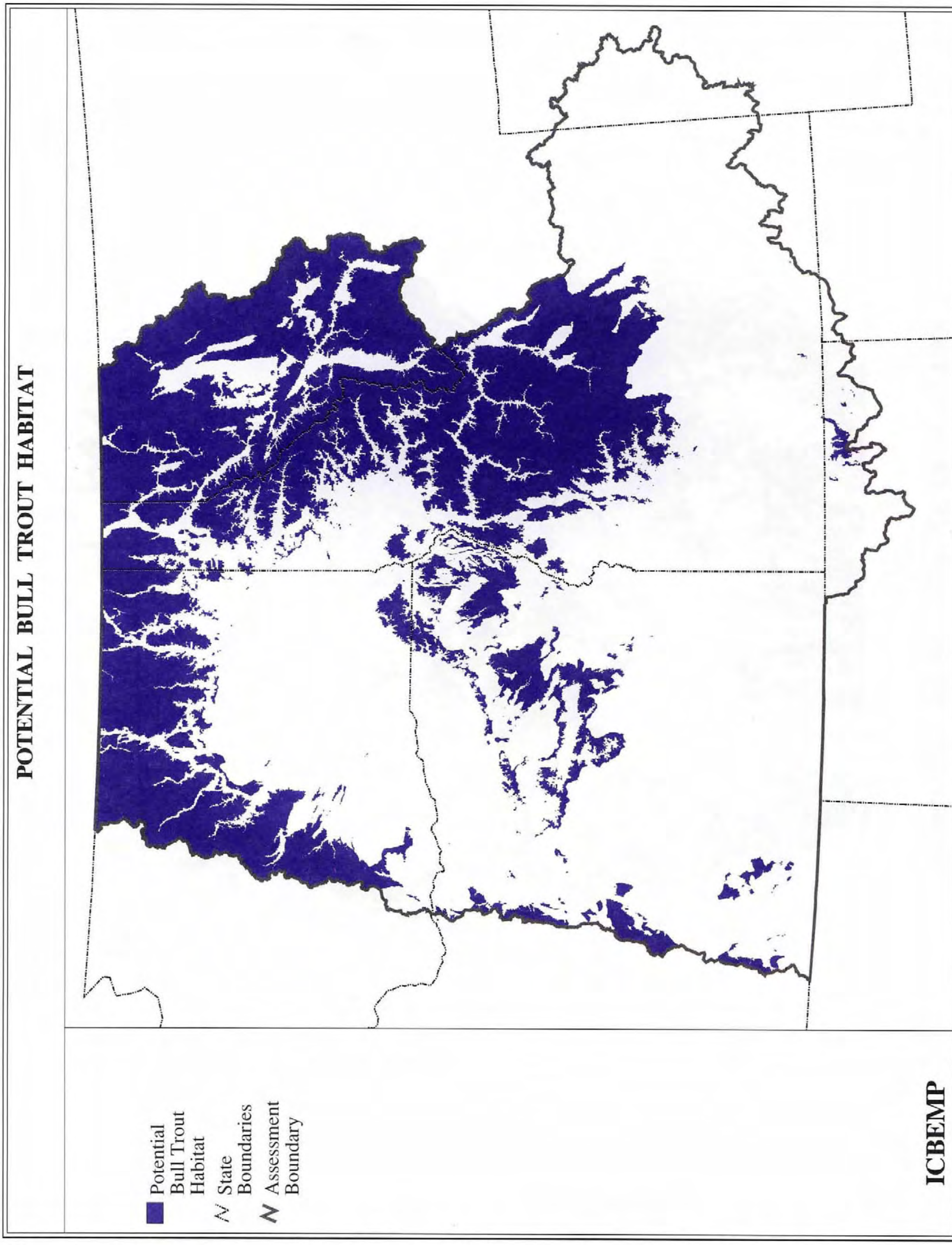
* = Refers to spawning and rearing areas only

Table 4.28—Summary of total known and predicted classifications (number of subwatersheds) for status of bull trout. The classifications are from the current-status database. The numbers predicted are based on the classification trees for bull trout and are shown in parentheses. Thirteen subwatersheds classified as unknown did not have a prediction

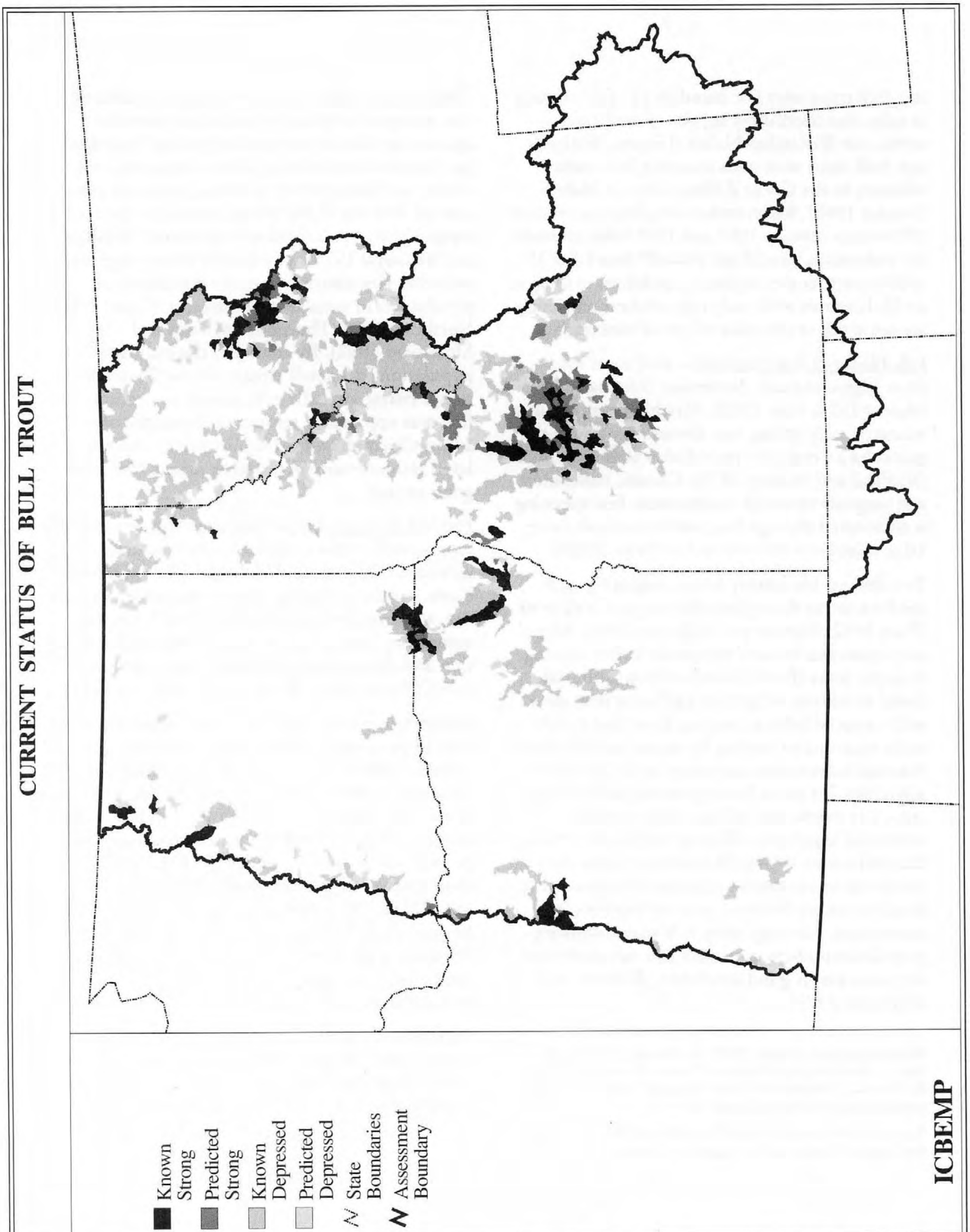
Ecological Reporting Unit	Total	Historical Range	Present	Strong	Depressed	Absent
Northern Cascades	340	319	172 (28)	10 (0)	34 (12)	144 (60)
Southern Cascades	141	80	33 (1)	14 (4)	6 (0)	47 (7)
Upper Klamath	175	42	16 (7)		14 (5)	26 (9)
Northern Great Basin	506	0				
Columbia Plateau	1089	551	124 (65)	3 (0)	22 (2)	426 (209)
Blue Mountains	695	442	200 (37)	28 (5)	83 (11)	242 (65)
Northern Glaciated Mountains	955	955	365 (49)	38 (12)	156 (31)	582 (247)
Lower Clark Fork	415	415	197 (31)	8 (4)	107 (32)	218 (58)
Upper Clark Fork	306	306	197 (52)	26 (5)	128 (19)	109 (25)
Owyhee Uplands	956	174	7 (0)		2 (0)	167 (65)
Upper Snake	301	0				
Snake Headwaters	387	0				
Central Idaho Mountains	1232	1130	654 (87)	143 (71)	261 (75)	475 (132)
Entire Assessment Area	7498	4414	1965 (357)	270 (101)	813 (187)	2436 (877)



Map 4.9—Current presence, absence, and estimated probability of occurrence within the historical range for bull trout by subwatershed. Current distributions are based on the current-status database. Probabilities of occurrence are based on the classification tree for presence or absence.



Map 4.10—Distribution of areas predicted to be suitable for bull trout spawning and initial rearing. Models were based on empirical relationships of occurrence with elevation, latitude, and longitude after Rieman and others (in preparation).



Map 4.11—Current distribution of spawning and rearing areas classified or predicted as strong or depressed for bull trout by subwatershed. Current distributions are based on the current-status database. Predictions are based on the full classification tree for spawning and rearing areas.

that bull trout were not found in 17 of 67 streams or lakes that historically supported bull trout within the Wenatchee National Forest, Washington. Bull trout were once common in streams tributary to the Coeur d'Alene River in Idaho (Maclay 1940), but extensive sampling in more than 100 streams between 1993 and 1995 failed to locate any individuals.[3] Ratliff and Howell[4] found that 15 of 69 known local or regional populations in Oregon are likely extinct while only eight of those remaining are not at risk or otherwise of special concern.

Life-History Characteristics—Bull trout spawn from August through November (McPhail and Murray 1979; Pratt 1992). Hatching may occur in winter or early spring, but alevins may stay in the gravel for an extended period after yolk absorption (McPhail and Murray 1979). Growth, maturation, and longevity vary with environment, first spawning is often noted after age four, with individuals living 10 or more years (Rieman and McIntyre 1993).

Two distinct life-history forms, migratory and resident, occur throughout the range of bull trout (Pratt 1992; Rieman and McIntyre 1993). Migratory forms rear in natal tributaries before moving to larger rivers (fluvial form) or lakes (adfluvial form) to mature. Migratory bull trout may use a wide range of habitats ranging from 2nd to 6th order streams and varying by season and life stage. Seasonal movements may range up to 300 km as migratory fish move from spawning and rearing areas into overwinter habitat in downstream reaches of large basins (Bjornn and Mallet 1964; Elle and others 1994). The resident form may be restricted to headwater streams throughout life. Both forms are believed to exist together in some areas, but migratory fish may dominate populations where corridors and subadult rearing areas are in good condition (Rieman and McIntyre 1993).

[3]Personal communication. 1995. B. Rieman, U.S. Forest Service, Intermountain Research Station, Boise, Idaho, and D. Bennett, University of Idaho, Moscow, Idaho. Personal communication of unpublished data.

[4]Unpublished update of Ratliff and Howell 1992; provided to the Oregon Chapter of the American Fisheries Society (AFS).

The divergent life histories are viewed as alternative strategies weighing reproductive potential against the risks of extensive migration. Variation in migratory form, timing of out-migration, and timing and frequency of spawning represent substantial diversity in life history across the species range and within individual populations (Rieman and McIntyre 1993). Life-history theory suggests such diversity contributes to the persistence of populations in variable environments (Gross 1991; Northcote 1992; Thorpe 1994; Titus and Mosegaard 1992). By example, the bull trout population in a small stream on the Boise National Forest virtually eliminated in a recent fire, was apparently refounded by migratory adults (Rieman and others, in press). An isolated resident-only population might well have gone extinct.

Habitat Relationships—Bull trout appear to have more specific habitat requirements than other salmonids (Rieman and McIntyre 1993). Habitat characteristics including water temperature, stream size, substrate composition, cover and hydraulic complexity have been associated with the distribution and abundance (Dambacher and others, in press; Jakober 1995; Rieman and McIntyre 1993).

Stream temperature and substrate composition may be particularly important characteristics of suitable habitats. Bull trout have repeatedly been associated with the coldest stream reaches within basins. The lower limits of bull trout distributions in map 4.9 correspond to a mean annual air temperature of about 4° C, (range 3° - 6° C) and should equate to ground water temperatures of about 5° to 7° C (Meisner 1990). Temperature may be strongly influenced by land management (Henjum and others 1994) and climate change; both effects may play an important role in the persistence of bull trout.

Bull trout are more strongly tied to the stream bottom and substrate than other salmonids (Pratt 1992). Substrate composition has repeatedly been correlated with the occurrence and abundance of juvenile bull trout (Dambacher and others, in press; Rieman and McIntyre 1993) and spawning site selection by adults (Graham and others 1981; McPhail and Murray 1979). Fine sediments can influence incubation survival and emergence success (Weaver and White 1985), but might also limit access to substrate interstices that are important cover during rearing and overwintering (Goetz 1994; Jakober 1995).

Interpretation of causality in the classification trees is speculative, but the patterns in the current bull trout distributions are consistent with existing knowledge and speculation on bull trout habitat relationships. Mean annual air temperature was important in discrimination of status; watersheds less than 5.1° C were roughly four times more likely to have bull trout present and 7 times more likely to support strong populations than warmer areas. Given the loss of resolution in averaging information across whole watersheds, the splitting temperature is remarkably similar to the mean of about 4° C (range of about 3° to 6° C) associated with the lower limit of bull trout distributions throughout the basin. Other variables in the model suggest that bull trout are more likely to be found in areas with lower road densities, forested rather than unforested areas, mid-size streams, on steeper, wetter, higher elevation and more erosive lands. These results are consistent with the view that bull trout are associated with cold, relatively pristine environments, but they do not exclude bull trout from landscapes influenced by human disturbance.

Key Factors—Angling is a factor influencing the current status of bull trout. Bull trout may be vulnerable to over-harvest (Ratliff and Howell 1992; Rieman and Lukens 1979). Legal harvest of bull trout has been eliminated or substantially restricted by all state management agencies except Nevada. Protection of the remaining bull trout in Nevada may be of particular concern. The remaining fish in the Jarbidge River subbasin appear to be very restricted in number and distribution, but also provide the only representation for this species on the extreme southern fringe of the range. Direct harvest has been limited in most other areas, but incidental catch in expanding lake trout fisheries could pose some problems for populations in the large lakes of northern Idaho and northwestern Montana. Poaching is viewed as an important cause of mortality, especially in accessible streams that support large migratory fish.[5,6]

Watershed disruption is a second factor that has played a role in the decline of bull trout. Changes in or disruptions of watershed processes likely to influence characteristics of stream channels are also likely to influence the dynamics and persistence of bull trout populations. Bull trout have been more strongly associated with pristine or only lightly disturbed basins (Brown 1992; Clancy 1993; Cross and Everest 1995; Dambacher and others, in press; Huntington 1995; Ratliff and Howell 1992).

Patterns of stream flow and the frequency of extreme flow events that influence substrates are anticipated to be important factors in population dynamics (Rieman and McIntyre 1993). With overwinter incubation and a close tie to the substrate, embryos and juveniles may be particularly vulnerable to flooding and channel scour associated with the rain-on-snow events common in some parts of the range within the belt geology of northern Idaho and northwestern Montana (Rieman and McIntyre 1993). Channel dewatering tied to low flows and bed agradation has also blocked access for spawning fish resulting in year class failures (Weaver 1992). These effects could explain the apparent extirpation of bull trout in much of the heavily disturbed upper Coeur d'Alene River Basin.[7]

Changes in sediment delivery, agradation and scour, wood loading, riparian canopy and shading or other factors influencing stream temperatures,

[5]Personal communication. 1995. N. Horner, Idaho Department of Fish and Game, Coeur d'Alene, Idaho.

[6]Personal communication. 1995. J. Vashro, Montana Department of Fish, Wildlife, and Parks, Kalispell, Montana.

[7]Personal communication. 1995. D. Cross, Panhandle National Forest, Coeur d'Alene, Idaho.

and the hydrologic regime (winter flooding and summer low flow) are all likely to affect some, if not most, populations. Although we cannot build precise models predicting population responses, we can conclude that significant long-term changes in any of these characteristics or processes represent important risks for many remaining bull trout populations. Populations are likely to be most sensitive to changes that occur in headwater areas encompassing critical spawning and rearing habitat and remnant resident populations.

Introduced species are a third factor influencing bull trout. More than 30 introduced species occur within the present distribution of bull trout. Some introductions like kokanee may benefit bull trout by providing forage (Bowles and others 1991). Others such as brown, brook, and lake trout are thought to have depressed or replaced bull trout populations (Dambacher and others, in press; Donald and Alger 1992; Howell and Buchanan 1992; Kanda and others, in press; Leary and others 1993; Ratliff and Howell 1992). Brook trout are seen as an especially important problem (Kanda and others, in press; Leary and others 1993) and may progressively displace bull trout through hybridization and higher reproductive potential (Leary and others 1993). Brook trout now occur in the majority of the watersheds representing the current range of bull trout (see map 4.7). Introduced species may pose greater risks to native species where habitat disturbance has occurred (Hobbs and Huenneke 1992).

Isolation and fragmentation are the fourth factor we identified as likely to influence the status of bull trout. Historically bull trout populations were well connected throughout the Basin. Habitat available to bull trout has been fragmented, and in many cases populations have been isolated entirely. Dams have isolated whole subbasins throughout the Basin (see for example, Brown 1992; Kanda and others, in press; Pratt and Huston 1993; Rieman and McIntyre 1995). Irrigation diversions, culverts, and degraded mainstem habitats have eliminated or seriously depressed migratory life histories effectively isolating resident populations in headwater tributaries (Brown 1992; Montana Bull Trout Scientific Committee, in preparation; Ratliff and Howell 1992; Rieman and McIntyre 1993). Introduced species like brook trout may displace bull trout in lower stream reaches further reducing the habitat available in many remaining headwater areas (Adams 1994; Leary and others 1993). Loss of suitable habitat through watershed disturbance may also increase the distance between good or refuge habitats and strong populations thus reducing the likelihood of effective dispersal (Frissell and others 1993).

Risk of Extinction or Extirpation—Life-history and metapopulation theory suggests dispersal capabilities may be particularly important for the local and regional persistence of many species (Rieman and McIntyre 1995; Thorpe 1994). Recent work suggests bull trout are no exception. Rieman and McIntyre (1993) used analytic models and redd count data to evaluate extinction risks. They predicted that few monitored populations were likely to persist above critical thresholds if they were isolated from other populations. Rieman and McIntyre (1995) also found that the occurrence of bull trout in naturally fragmented habitat patches was significantly associated with the size of the patch. The available information, then, supports the predictions of metapopulation theory with important implications for the conservation of bull trout. Further isolation of populations in shrinking habitat will probably lead to increasing rates of extirpation not proportional to the simple loss of habitat area. Even with no further habitat loss, extirpations may be likely for many remaining isolated populations. Global climate change can be expected to exacerbate the problem (Rieman and McIntyre 1993). Long-term conservation of bull trout throughout much of the Columbia River basin may well depend on maintenance or restoration of networks of high quality habitats, and of migratory corridors in larger river subbasins.

Summary—Bull trout are still distributed throughout the Basin, but local extirpations are reportedly widespread and declining populations are believed to be common. Many remaining populations are isolated in headwater streams. Migratory life histories may be severely limited or absent in many systems. Even with no further loss of habitat the likelihood of extirpations seems high. The core of the remaining bull trout distribution is in the Central Idaho Mountains, with important strongholds still evident or likely within the Upper Clark Fork, Northern Glaciated Mountains, Lower Clark Fork, and Blue Mountains. This latter collection of known or likely watersheds appears to be more strongly fragmented or restricted in size than the central Idaho core. These remaining strongholds represent the best opportunities for the long term persistence of bull trout within the Basin. Bull trout also remain within the Northern Cascades, Southern Cascades, Upper Klamath, and Owyhee Uplands. In each of these ERUs the remaining populations appear to be very narrowly distributed and often isolated in small, discontinuous watersheds. Because these fringe areas tend to be quite distinct, both geographically and environmentally, from the core of the distribution, further loss of bull trout would probably represent a disproportionately important loss of genetic diversity and evolutionary legacy. Conservation management of bull trout may therefore be most critical in the fringe areas.

Yellowstone Cutthroat Trout (*Oncorhynchus clarki bouvieri*)—The Yellowstone cutthroat trout is more abundant and inhabits a larger geographical range in the western United States than any other non-anadromous subspecies of cutthroat trout (Varley and Gresswell 1988). Individual populations of the Yellowstone subspecies have evolved numerous life-history characteristics in response to the diverse environments in which they have been isolated since the last glacial retreat. Anthropogenic activities have resulted in a substantial reduction in the historical distribution of this subspecies, however, and many unique local populations have been extirpated. As a result, the Yellowstone cutthroat trout has been designated as a species of special concern - class A by the American Fisheries Society (Johnson 1987). This status has been officially recognized by the Montana Department of Fish, Wildlife and Parks [Yellowstone Cutthroat Trout Working Group (YCTWG) 1994], and the subspecies has been recognized as a species of special concern in Idaho. Both the Northern and Rocky Mountain regions of the Forest Service consider the Yellowstone cutthroat trout a sensitive species (YCTWG 1994). Although the Yellowstone cutthroat trout has not been given formal status in Wyoming, the uniqueness of the subspecies has influenced management (YCTWG 1994).[8]

Historical Distribution—Yellowstone cutthroat trout were historically found in the Yellowstone River drainage in Montana and Wyoming and in the Snake River Drainage in Wyoming, Idaho, Utah, Nevada, and probably Washington (Behnke 1992; Varley and Gresswell 1988). In the Basin, it is the only native trout in the Snake River above Shoshone Falls (map 4.12). The fine-spotted Snake River cutthroat trout, a putative cutthroat trout subspecies, originally existed within the range of the Yellowstone subspecies in the upper Snake River (Behnke 1992), and the two groups are genetically similar (Loudenslager and Gall 1980; Loudenslager and Kitchin 1979). Because the Snake River form has not received formal taxonomic recognition, we considered both groups as Yellowstone cutthroat trout.

The subspecies was the most narrowly distributed of the seven key salmonids in the Basin, occurring in about 9 percent of the watersheds (table 4.29). The original range included primarily the Upper Snake and Snake Headwater where 74 and 98 percent, respectively, of the watersheds supported Yellowstone cutthroat trout. Yellowstone cutthroat trout were also present in a small number of watersheds in the Columbia Plateau and Owyhee Uplands (Behnke 1992)(map 4.12).

Current Status and Distribution—A total of 393 observations was available for the classification analysis (see table 4.22). The overall classification success rate was 85 percent for the three classes

[8]Also, personal communication. 1995. R. Wiley, Wyoming Game and Fish, Laramie, Wyoming.

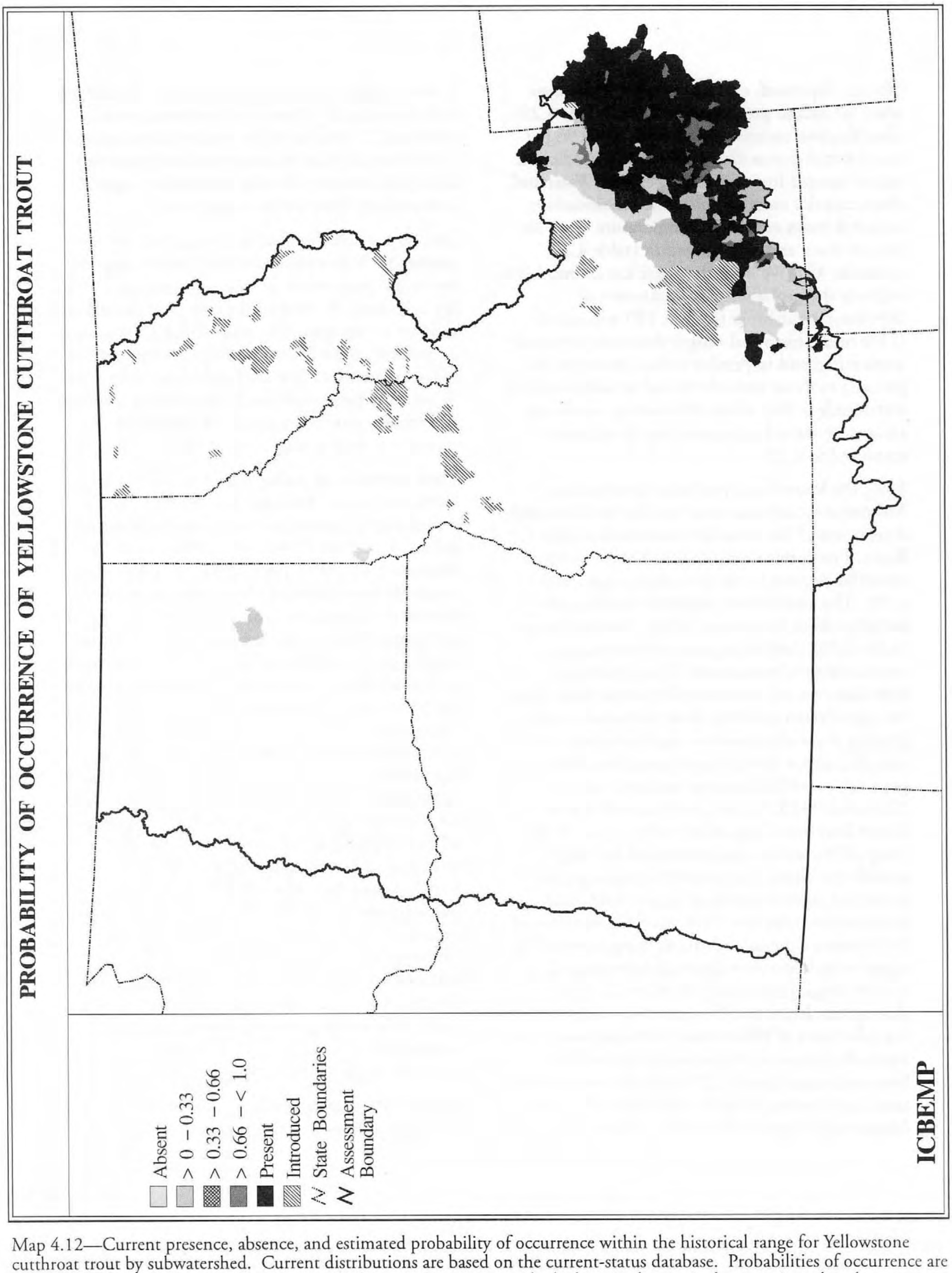
Table 4.29—Summary of the current status and distribution classifications (number of subwatersheds) for Yellowstone cutthroat trout throughout the Ecological Reporting Units of the Basin.

				Status Where Present						
Ecological Reporting Unit	Total	Historical Range	Total Present	Strong*	Depressed*	Status Unknown	Corridor	Introduced	Absent	No Classification
Northern Cascades	340	0	0							
Southern Cascades	141	0	0							
Upper Klamath	175	0	0							
Northern Great Basin	506	0	0							
Columbia Plateau	1089	16	0	0	0	0	0	0	5	11
Blue Mountains	695	0	0							
Northern Glaciated Mountains	955	0	31	0	0	0	0	31	0	0
Lower Clark Fork	415	0	7	0	0	0	0	7	0	0
Upper Clark Fork	306	0	15	0	0	0	0	15	0	0
Owyhee Uplands	956	5	2	0	1	1	0	0	0	3
Upper Snake	301	261	100	4	28	47	4	17	73	105
Snake Headwaters	387	380	303	171	89	41	1	1	26	52
Central Idaho Mountains	1232	0	69	0	0	0	0	69	0	0
Entire Assessment Area	7498	662	527	175	118	89	5	140	104	171

* = Refers to spawning and rearing areas only

Table 4.30—Summary of total known and predicted classifications (number of subwatersheds) for status of Yellowstone cutthroat trout. The classifications are from the current-status database. The numbers predicted are based on the classification trees for Yellowstone cutthroat trout and are shown in parentheses. Forty-four subwatersheds classified as unknown did not have a prediction.

Ecological Reporting Unit	Total	Historical Range	Present	Strong	Depressed	Absent
Northern Cascades	340	0				
Southern Cascades	141	0				
Upper Klamath	175	0				
Northern Great Basin	506	0				
Columbia Plateau	1089	16				16 (11)
Blue Mountains	695	0				
Northern Glaciated Mountains	955	0				
Lower Clark Fork	415	0				
Upper Clark Fork	306	0				
Owyhee Uplands	956	5	2 (0)		1 (0)	
Upper Snake	301	261	105 (22)	4 (0)	34 (6)	118 (45)
Snake Headwaters	387	380	329 (27)	205 (34)	138 (49)	48 (22)
Central Idaho Mountains	1232	0				
Entire Assessment Area	7498	662	436 (49)	209 (34)	173 (55)	182 (78)



Map 4.12—Current presence, absence, and estimated probability of occurrence within the historical range for Yellowstone cutthroat trout by subwatershed. Current distributions are based on the current-status database. Probabilities of occurrence are based on the classification tree for presence or absence. Presence outside the historical range is shown as introduced.

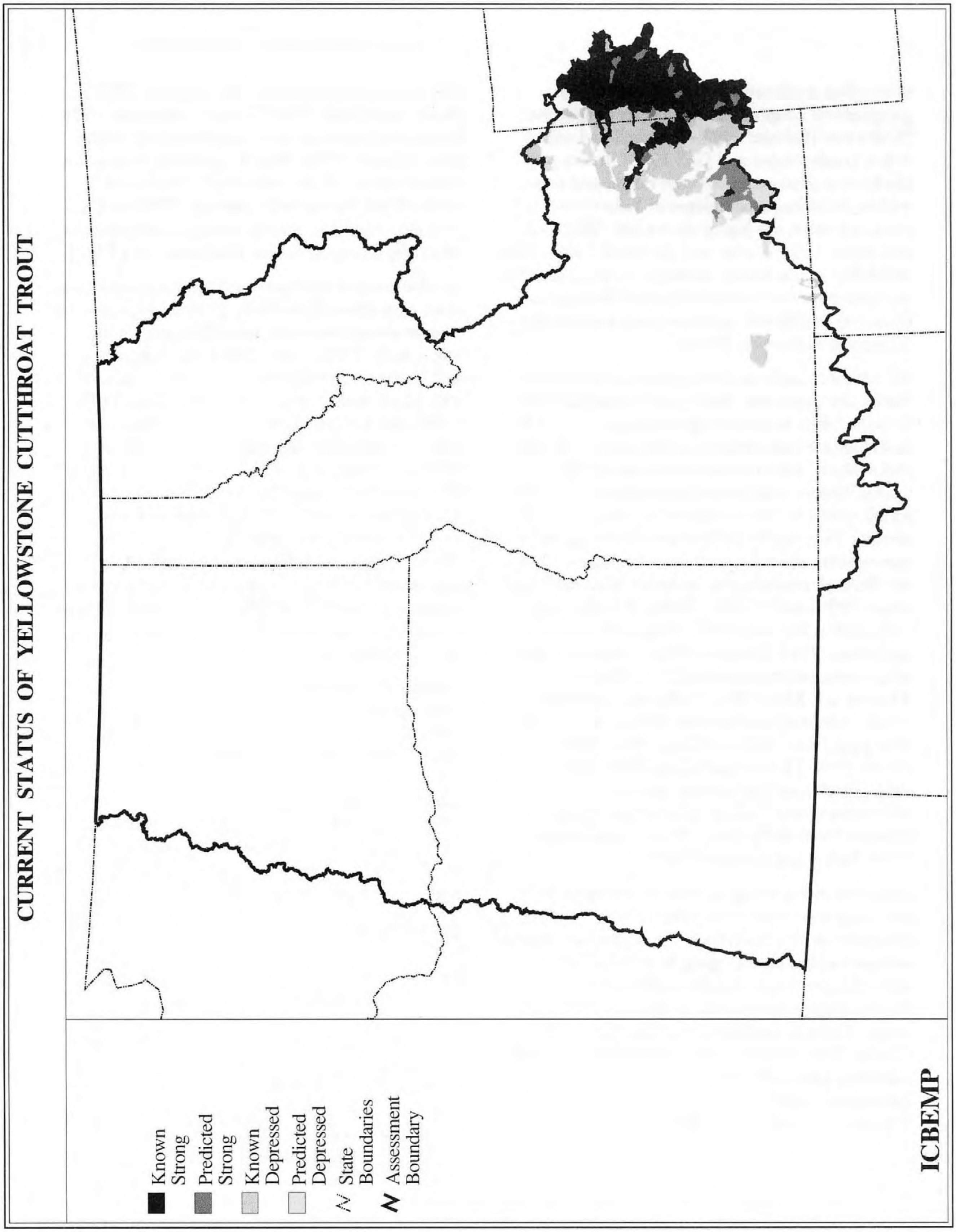
(absent, depressed, and strong) and 98 percent when limited to presence or absence (table 4.22). Classification success ranged from 83 to 96 percent for each status class (table 4.24). Prediction success ranged from 67 to 82 percent. Watershed characteristics most useful in the classifications included mean annual air temperature, *hucorder* (stream size), and solar radiation (table 4.22, appendix 4E). We used the classification models to estimate the probability of occurrence of Yellowstone cutthroat trout in 130 watersheds (21% of the historical range) that were previously unclassified and to predict status (strong or depressed) in those watersheds and an additional 88 watersheds (14%) where Yellowstone cutthroat trout were classed as present but of unknown status (table 4.29).

From the known and predicted distributions, Yellowstone cutthroat trout are the most narrowly distributed of the seven key salmonids within the Basin. The current known distribution includes about 61 percent of the historical range (table 4.29). The known and predicted distribution includes about 66 percent of the historical range (table 4.30). Anthropogenic activities such as introduction of non-native fishes resulting in hybridization and interspecific competition, habitat degradation resulting from water diversions, grazing, mineral extraction, timber harvest activities, and angler harvest have caused extirpation of populations of Yellowstone cutthroat trout (Gresswell 1995; Varley and Gresswell 1988). Losses have been important in the Upper Snake (map 4.13), and in that portion of the range outside the Basin. Large-river populations, in particular, have declined or disappeared. Concomitant with declines in natural distributions of Yellowstone cutthroat trout, stocking activities by agencies and private individuals have expanded the species range, particularly in mountain lakes throughout Idaho and Montana (see map 4.12). Introductions of Yellowstone cutthroat trout have established them in 140 watersheds outside the historical range (table 4.29). Yellowstone cutthroat trout are now found in the Northern Glaciated Mountains, Upper and Lower Clark Fork, and Central Idaho Mountains (map 4.13). Watersheds with introduced Yellowstone cutthroat trout account for 27 percent of the present distribution. The effects of these introductions on genetic or biological integrity of other species and aquatic communities have not been assessed.

Despite the changes that have occurred, we estimated that Yellowstone cutthroat trout support the largest proportion of strong populations of any key salmonid. Watersheds known or predicted to support strong spawning and rearing represented 32 percent of the historical range and 48 percent of the present distribution (table 4.30; map 4.13). When only those watersheds supporting spawning and rearing were considered, 55 percent of the watersheds were strong (table 4.30).

These estimates of strong populations may be overly optimistic, however, because of a high probability of genetic introgression in most populations. Varley and Gresswell (1988) estimated that genetically unaltered populations of Yellowstone cutthroat trout occur in about 10 percent of the historical stream habitat and about 85 percent of the historical lacustrine habitat. They suggested that 91 percent of the present range of genetically unaltered Yellowstone cutthroat trout lies within the boundary of Yellowstone National Park (Gresswell and Liss 1995). Hadley (1984) suggested that only 8 percent of the original range of fluvial Yellowstone cutthroat trout in Montana was inhabited by genetically unaltered populations of Yellowstone cutthroat trout. In Wyoming, nearly all areas supporting Yellowstone cutthroat trout were judged by biologists to be strong (map 4.13), but there may have been little information to judge genetic purity (YCTWG 1994). There was no information to judge presence or absence of Yellowstone cutthroat trout in 171 watersheds within the Upper Snake, Snake Headwaters, Owyhee Uplands, and Columbia Plateau (table 4.29). Within the present known range, nearly 17 percent of the watersheds supporting Yellowstone cutthroat trout were of unknown status.

<u>Life-history Characteristics</u>—Low genetic diversity among populations of Yellowstone cutthroat trout



Map 4.13—Current distribution of spawning and rearing areas classified or predicted as strong or depressed for Yellowstone cutthroat trout by subwatershed. Current distributions are based on the current-status database. Predictions are based on the full classification tree for spawning and rearing areas.

may reflect a substantial compression of the geographical range of the subspecies during the Pleistocene (Behnke 1992; Gresswell and others 1994; Loudenslager and Gall 1980). In contrast, life-history strategies across the range, and even within individual assemblages of Yellowstone cutthroat trout, are highly diversified (Gresswell and others 1994; Varley and Gresswell 1988). The variability in life-history strategies may represent a complex response to environmental fluctuations operating at different temporal and spatial scales (Gresswell and others 1994).

Yellowstone cutthroat trout spawn exclusively in fluvial environments. Varley and Gresswell (1988) described four migratory-spawning patterns. The first three are consistent with the resident, fluvial, and adfluvial patterns mentioned earlier. The fourth, termed allacustrine, is represented by fish which spawn in lake outlets rather than tributary streams. Homing by Yellowstone cutthroat trout spawners is believed to influence life-history diversity through reproductive isolation (Gresswell and others 1994) and straying during the spawning migration is low (Ball 1955; Cope 1957a; Jones and others 1985; Thurow 1982). Migrations begin when temperatures approach 5° C (Byorth 1990; Thurow and King 1994; Varley and Gresswell 1988). Adults spawn between March and August after peak flows (Ball and Cope 1961; Jones and others 1990; Thurow and King 1994). Migration timing is affected by latitude, altitude, water temperature, and hydrographic relationships (Clancy 1988; Kelly 1993; Thurow and others 1988; Varley and Gresswell 1988).

Sex ratios in spawning migrations appear to vary with migratory-spawning pattern. Mean age of spawners varies across the range of the Yellowstone subspecies, typically ranging from three to five years (Clancy 1988; Thurow and others 1988). Few migratory fish mature at less than 250 millimeters (Benson and Bulkley 1963; Byorth 1990; Clancy 1988; Thurow and others 1988). In small subalpine lakes and streams, non-migratory Yellowstone cutthroat trout may mature between 100 and 130 millimeters. Fecundity is related to length, weight, and age of fish (Bagenal 1978; Moore and Schill 1984; Thurow and others 1988). Iteroparity is common in Yellowstone cutthroat trout (Clancy 1988). Repeat spawning may occur in consecutive or alternate years (Thurow and others 1988; Varley and Gresswell 1988) and is probably related to growth, parasitic infection, and other physiological factors (Ball and Cope 1961).

Incubation and emergence are temperature dependent; eggs typically hatch in 25 to 49 days, and fry emerge about two weeks later (Ball and Cope 1961; Kelly 1993; Mills 1966;). Juveniles often seek shallow, slow-flowing areas, and migratory individuals soon begin to emigrate (Kelly 1993; Varley and Gresswell 1988). Some Yellowstone cutthroat trout fry migrate downstream shortly following emergence while others do not emigrate for one to three years (Benson 1960; Byorth 1990; Thurow and others 1988; Gresswell and others 1994). Distance from redd to stream mouth (Welsh 1952) and habitat availability (Thurow and others 1988) may influence the timing of fry migrations. Growth of Yellowstone cutthroat trout is variable and dependent on population and environmental conditions.

<u>Habitat Relationships</u>—Yellowstone cutthroat trout occupy diverse habitats. Lacustrine populations inhabit waters ranging in size from small beaver ponds to large lakes [for example, Yellowstone Lake, 35,400 hectares (Varley and Gresswell 1988)]. Fluvial populations were historically present in streams ranging in size from large rivers such as the Snake River above Shoshone Falls (Varley and Gresswell 1988) to first-order tributaries with mean widths of one meter and less.

The subspecies is well adapted to relatively cold, harsh environments (Carlander 1969; Dwyer and Kramer 1975; Jones and others 1979; Varley and Gresswell 1988). Although Yellowstone cutthroat trout are associated with cold-water habitats, Varley and Gresswell (1988) reported that water temperatures within portions of the historical range exceeded 26° C. Most large-river, warm-water populations have been extirpated; however, several populations have been documented in geothermally heated streams in Yellowstone National Park with an ambient water temperature of 27° C (Varley and Gresswell 1988). Yellowstone cutthroat trout probably survive in warm water by locating thermal refugia.

Yellowstone cutthroat trout tolerate a broad range of chemical conditions. Varley and Gresswell (1988) reported that the subspecies has been collected from waters in Yellowstone National Park with total dissolved solids ranging from about 10 to 700 milligrams/liter. Reported alkalinities in waters supporting Yellowstone cutthroat trout in Montana and Idaho ranged from 46 to 378 milligrams $CaCO_3$/liter (Byorth 1990; Thurow and others 1988). Although Yellowstone cutthroat trout have been found in waters with pH values ranging from 5.6 to over 10.0, none have been observed where pH is below 5.0 (Varley and Gresswell 1988).

Spawning streams are most commonly perennial with groundwater and snow-fed water sources; gradient is usually below 3 percent (Varley and Gresswell 1988). Although use of intermittent streams for spawning is not well documented, where it occurs there may be some reproductive advantage over non-indigenous fall-spawning salmonids (Varley and Gresswell 1988). Thurow and King (1994) emphasized the importance of physical cues, such as depth and velocity, for locating redds in areas with a high probability of hatching success and fry survival. Cope (1957b) reported that forest cover had little effect on the distribution of redds and spawners did not always congregate in areas with the greatest concentration of spawning gravel.

In streams, Yellowstone cutthroat trout fry generally seek areas of low velocity (Varley and Gresswell 1988) and shallow depth (Byorth 1990). As fish become larger, juveniles rear in areas with a larger range of depths and velocities. Yellowstone cutthroat trout occupy different habitats in winter than in summer. In late winter (March - April) at water temperatures ranging from 4° to 7° C, Griffith and Smith (1993) found juvenile Yellowstone cutthroat trout concealed in water shallower than 0.5 meters within 1 meter of the wetted perimeter of the stream. Density of age-0 Yellowstone cutthroat trout increased as the substrate size of non-embedded cover increased.

Adfluvial fry usually begin a downstream migration soon after emergence, but some may overwinter in the natal stream. After emigration, fry congregate in shallow water along the shoreline prior to movement into deeper water (Gresswell and Varley 1988). As lacustrine fish grow, most move back into the littoral zone where a shift in food preference occurs (Benson 1961; Gresswell and Varley 1988).

In the classification analysis variables including ERU, mean annual air temperature, number of upstream 6th code hydrologic units, and solar radiation were useful in discriminating watersheds supporting this subspecies (see table 4.22; appendix 4E). Populations rarely occurred in subwatersheds with mean annual air temperatures higher than 6.1° C. Variables related to land management were not useful in the classification. The core of the remaining range, however, is clearly centered in the Upper Snake which is represented in a large part by National Park and roadless lands. A more refined analysis will be necessary to evaluate the potential influence of disturbance on the status and distribution of Yellowstone cutthroat trout at this scale.

<u>Key Factors</u>— Introgression with introduced salmonids is clearly a key factor in the status of Yellowstone cutthroat trout. Hybridization resulting from introductions of rainbow trout and non-native cutthroat trout is believed to be a primary cause in the decline of this form (Allendorf and Leary 1988; Varley and Gresswell 1988). Hybrids are developmentally successful, and progeny may appear as morphological and meristic intermediates between parental types or virtually identical to a single parental type (Ferguson and others 1985). Consequently, verifying genetic integrity with morphological data alone is virtually impossible. Nuclear allozymes and mitochondrial DNA (mtDNA) haplotypes have proved to be useful for detecting hybridization (Forbes and Allendorf 1991; Leary and others 1987).

Non-salmonid species are commonly indicted as competitors of salmonid species; however, it does not appear that introductions of longnose suckers, redside shiners, and lake chub into Yellowstone Lake have had negative effects on that population (Gresswell and Varley 1988). Although competition may be important in different environments, it would probably be greatest with species exhibiting similar niche requirements (for example, between salmonids; Marrin and Erman 1982). Introduction of brown and rainbow trout to the Madison River in Yellowstone National Park was followed by the extirpation of indigenous westslope cutthroat trout and fluvial Arctic grayling (Jones and others 1981), but Yellowstone cutthroat trout are still abundant in sections of the Yellowstone River where they are sympatric with these two non-native salmonids (Clancy 1988). Introductions of rainbow trout in the Henry's Fork of the Snake River, Idaho resulted in extirpation of Yellowstone cutthroat trout (Thurow and others 1988). In other Idaho streams, Yellowstone cutthroat trout are sympatric with introduced brown and brook trout, and Yellowstone cutthroat trout persist if habitat has not been degraded and angler harvest is not extreme (Thurow and others 1988). Thurow and others (1988) reported that angler harvest may limit Yellowstone cutthroat trout in sympatry with brown trout because cutthroat trout are more vulnerable to angler harvest. Griffith (1988) reported that cutthroat trout are less likely to coexist with brook trout than other non-native salmonids, and Yellowstone cutthroat trout have been extirpated from most areas in Yellowstone National Park where brook trout have been introduced (Gresswell 1991). Large predators such as lake trout may also pose a substantial threat to Yellowstone cutthroat trout (Varley and Schullery 1995).

Habitat degradation is a second factor important in the decline of this trout. Activities such as dam construction, water diversions, grazing, mineral extraction, road construction, and timber harvest have substantially degraded lotic environments throughout the historical range of Yellowstone cutthroat trout (Binns 1977; Clancy 1988; Thurow and others 1988). Recreational use can also be a significant source of disturbance (Gresswell and Liss 1995; Roberts and White 1992). Anthropogenic activities such as road construction have resulted in barriers to migration (Belford and Gould 1989), reduced flows, sediment deposition, groundwater depletion, streambank instability, erosion, and pollution. Efforts to curtail human activities and restore degraded stream segments are increasing, but habitat degradation continues.

Effects of livestock grazing on riparian habitats are well documented (for example, Gresswell and others 1989; Platts 1991). In the range of the Yellowstone cutthroat trout, Thurow and others (1988) reported that intensive livestock grazing has caused degradation of riparian areas and subsequent stream bank sloughing, channel instability, erosion, and siltation. Alterations are broadly distributed on private and public lands throughout the upper Snake River basin in Idaho and Wyoming (Thurow and others 1988). Degraded water quality and unscreened irrigation ditches contribute to the problems associated with water diversions throughout the range of the Yellowstone cutthroat trout (Byorth 1990; Clancy 1988; Johnson 1964; Thurow and others 1988;). Mineral extraction has negatively effected Yellowstone cutthroat trout in the Blackfoot River drainage, Idaho (Thurow and others 1988) and Soda Butte Creek, Montana (Arnold and Sharpe 1967; Jones and others 1982). Future expansions of mining operations may pose renewed threats to the Yellowstone cutthroat trout.

Angling is a third factor that may play an important role in the status of remaining Yellowstone cutthroat trout populations. Yellowstone cutthroat trout are extremely vulnerable to angling, and angler harvest has contributed to substantial declines in population abundance throughout the historical range of the subspecies (Binns 1977; Gresswell and Varley 1988; Hadley 1984; Thurow and others 1988). Schill and others (1986) estimated that individual Yellowstone cutthroat trout were captured an average of 9.7 times during a single angling season; many fish were captured two or three times in a single day. Although high catchability contributes to recreational value, it may lead to substantial declines in abundance (Gresswell 1990; Gresswell and Liss 1995). Angler harvest can reduce mean age and length of spawners (Gresswell 1995; Gresswell and others 1994; Gresswell and Varley 1988) and the proportion of repeat spawners (Varley and Gresswell 1988). Harvest of native Yellowstone cutthroat trout has been restricted in most waters by state management agencies. Harvest of introduced populations, especially in alpine lakes, is less restricted.

Summary—Yellowstone cutthroat trout are the most narrowly distributed of the seven key salmonids, occurring naturally in only 8 percent of the Basin and only within the Upper Snake and Snake Headwaters ERUs. They have been widely introduced and now occur in about 2 percent of the Basin outside the historical range. Yellowstone cutthroat trout appear to be one of the most widely distributed key species within their historical range. Status of populations is uncertain, however, because many populations have hybridized with non-native trout. The range of genetically unaltered populations of Yellowstone cutthroat trout has probably been reduced. The core of remaining strong populations is the Snake Headwaters. Populations remain widespread in the Upper Snake but most are depressed and many appear isolated to small areas.

Population declines and extirpations have been most common in larger low-elevation, less than third-order streams (Hanzel 1959), as illustrated by the current distribution and status of Yellowstone cutthroat trout in the Upper Snake. Low elevation areas have historically been the focus of most agricultural development and resource extraction and access has encouraged angler harvest and non-native species introductions. Remoteness of portions of the native range probably contributed to the preservation of remaining populations, and in much of this area, public ownership in the form of parks and reserves has provided habitat protection that is lacking in low-elevation portions of the range (Varley and Gresswell 1988). Approximately 98 percent of the current strong watersheds in the Basin are in the Snake Headwaters.

Westslope Cutthroat Trout (*Oncorhynchus clarki lewisi*) —Westslope cutthroat trout were once abundant throughout much of the north and central portions of the Basin. Although this subspecies is still widely distributed, remaining populations may be seriously compromised by habitat loss and genetic introgression through hybridization (McIntyre and Rieman 1995; Rieman and Apperson 1989). Westslope cutthroat trout were listed in the U.S. Fish and Wildlife Service "Red Book" of endangered and threatened species from 1966 to 1972. The subspecies was subsequently dropped from the list due to confusion over its classification (Roscoe 1974). Westslope cutthroat trout are presently considered a sensitive or vulnerable species by management agencies in Washington, Oregon, Idaho, and Montana.

Historical Distribution—Westslope cutthroat trout were probably the earliest salmonid to populate the headwaters of the Basin and now exist above barrier falls that have limited the distributions of other species throughout the range (Behnke 1992). Westslope cutthroat trout have a much larger historical range (about 35% of the Basin) than Yellowstone cutthroat trout. Populations in the John Day drainage of Oregon and along the eastern slope of the Cascades in Washington may have been established from the core distribution in what is now Idaho and Montana, through glacial Lake Missoula floods 15,000 to 12,000 years ago (Behnke 1992). Headwater transfers are believed to account for distributions in the upper Missouri and South Saskatchewan river basins east of the Continental Divide (Schultz 1941), and perhaps those in Central Idaho as well (Behnke 1992).

Westslope cutthroat trout were first recorded in 1805 by the Lewis and Clark expedition (Behnke 1992). Early explorers' journals suggest that westslope cutthroat trout were extremely abundant and widely distributed (Trotter and Bisson 1988).

Where habitat remains in relatively good condition westslope cutthroat trout are often found in most streams accessible to them (Rieman and Apperson 1989; Rieman and McIntyre 1993). They probably also occupied many natural lakes within the range as well. We estimate the historical range for westslope cutthroat trout represented about 35 percent of the Basin (table 4.31), and summarize our description in appendix 4D.

Current Status and Distribution—A total of 1,640 watersheds was available for the classification analysis. Overall classification success was 80 percent and increased to 93 percent when restricted to presence and absence (see table 4.22). Classification success rates ranged from 61 to 91 percent across the status classes. The models tended to under-predict absent and strong although the prediction success rates were still good for each class. For example, if a watershed was predicted to be absent or strong, that classification was correct 82 and 73 percent of the time, respectively (table 4.24),

We used the classification models for westslope cutthroat trout to estimate the probability of occurrence in 426 watersheds (16% of the historical range) that were unclassified (table 4.31). We also predicted status in both the unclassified watersheds and 574 watersheds (22% of the historical range) where westslope cutthroat trout were believed to be present but status was unknown.

Watershed characteristics most useful in classifications included general location (ERU), management cluster, road density, and several of the biophysical variables (see table 4.22; appendix 4E).

The known and predicted status information indicated that westslope cutthroat trout remain widely distributed within their historical range. Some extension of the natural distribution has also occurred through hatchery introductions. From the current status classification and extrapolations from the classification tree we estimate that westslope cutthroat trout are still present in at least 85 percent of the historical range (map 4.14).

Despite the broad distribution there appear to be few remaining healthy populations outside the Central Idaho Mountains and potentially the Northern Cascades. We estimated that 22 percent of the historical range was classified or predicted as strong (table 4.32; map 4.15). If the sample is limited to watersheds classified as spawning and rearing areas, 28 percent were believed to be strong. Other status reviews suggest a less healthy condition. Rieman and Apperson (1989) estimated that strong westslope cutthroat trout numbers persisted in 11 percent of the historical range in Idaho, and populations that were both numerically strong and genetically pure existed in 4 percent of the historical range. Liknes and Graham (1988) estimated that the subspecies still occupied 27 percent of the historical range in Montana, and were genetically pure in only 2.5 percent. The largest extirpations may have occurred outside the Basin. We estimate that westslope cutthroat trout still occupy nearly 80 percent of that portion of historical range within the Basin in Montana. The discrepancies may result from the resolution of the summary of data. Because our summary was based on subwatersheds rather than on stream reaches some discrepancies are inevitable. For example, we might show westslope cutthroat trout as present in 100 percent of the watersheds considered when they may have occurred in only a fraction of the streams. In any case, clearly, few strong populations are left in Montana, continuing a decline first documented over 25 years ago (Hanzel 1959). In Oregon, Kostow and others (1994) estimated westslope cutthroat trout are limited to 41 percent of the historically used stream miles in the John Day River subbasins. In contrast, we estimated 64 percent of the subwatersheds were still occupied.

Life-History Characteristics—Similar to Yellowstone cutthroat trout, three life-history forms are commonly identified as adfluvial, fluvial, or resident. All three may occur in a single basin. Resident forms may predominate in headwater areas while migratory forms are more common in mid- and lower basin habitats (Averett and MacPhee 1971; Rieman and Apperson 1989; McIntyre and Rieman 1995). Westslope cutthroat

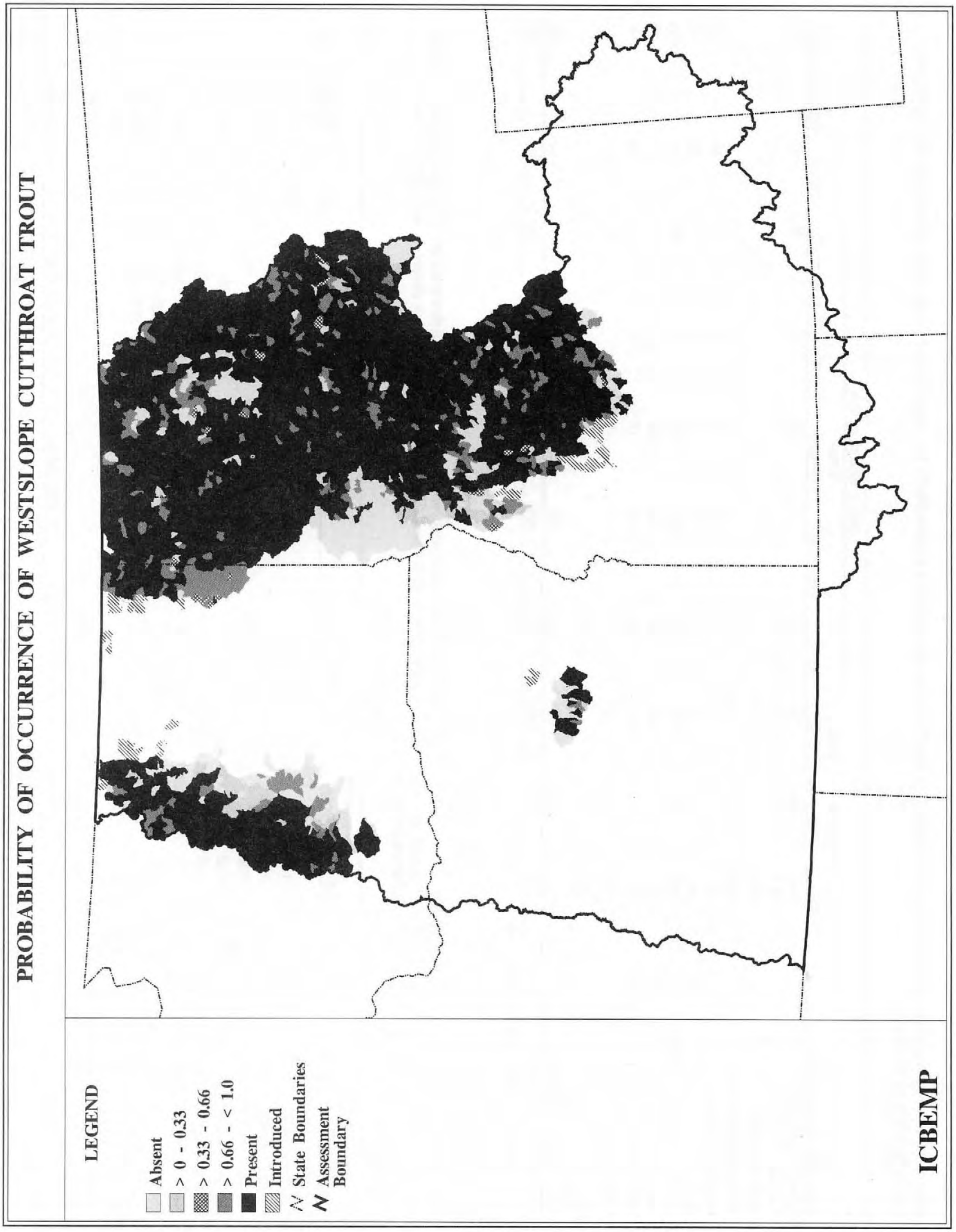
Table 4.31—Summary of the current status and distribution classifications (number of subwatersheds) for westslope cutthroat trout throughout the Ecological Reporting Units of the Basin.

				Status Where Present						
Ecological Reporting Unit	Total	Historical Range	Total Present	Strong*	Depressed*	Status Unknown	Corridor	Introduced	Absent	No Classification
Northern Cascades	340	265	202	33	11	152	0	6	23	46
Southern Cascades	141	0	0							
Upper Klamath	175	0	0							
Northern Great Basin	506	0	0							
Columbia Plateau	1089	174	18	1	5	10	2	0	79	77
Blue Mountains	695	33	21	0	17	0	0	4	13	3
Northern Glaciated Mountains	955	664	524	65	362	81	0	16	53	103
Lower Clark Fork	415	415	383	53	266	63	1	0	5	27
Upper Clark Fork	306	306	244	52	170	17	5	0	18	44
Owyhee Uplands	956	0	0							
Upper Snake	301	0	0							
Snake Headwaters	387	0	0							
Central Idaho Mountains	1232	789	599	128	160	251	43	17	81	126
Entire Assessment Area	7498	2646	1991	332	991	574	51	43	272	426

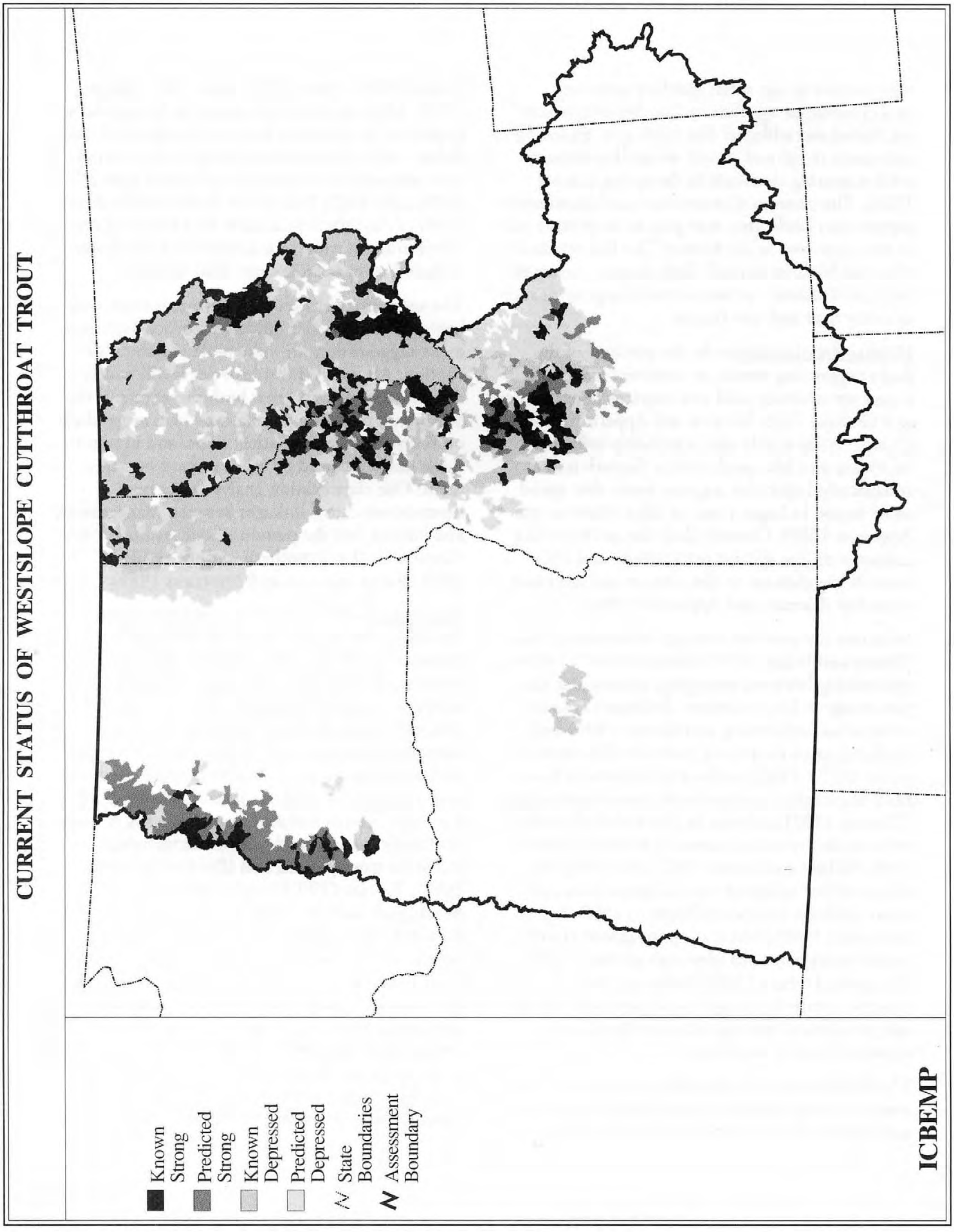
* = Refers to spawning and rearing areas only

Table 4.32—Summary of total known and predicted classifications (number of subwatersheds) for status of westslope cutthroat trout. The classifications are from the current-status database. The numbers predicted are based on the classification trees for westslope cutthroat trout and are shown in parentheses. Two subwatersheds classified as unknown did not have a prediction.

Ecological Reporting Unit	Total	Historical Range	Present	Strong	Depressed	Absent
Northern Cascades	340	265	216 (20)	144 (111)	24 (13)	49 (26)
Southern Cascades	141	0				
Upper Klamath	175	0				
Northern Great Basin	506	0				
Columbia Plateau	1089	174	37 (19)	7 (6)	8 (3)	137 (58)
Blue Mountains	695	33	19 (2)		20 (3)	14 (1)
Northern Glaciated Mountains	955	664	608 (100)	78 (13)	526 (164)	54 (1)
Lower Clark Fork	415	415	410 (27)	70 (17)	339 (73)	5 (0)
Upper Clark Fork	306	306	283 (39)	53 (1)	215 (45)	23 (5)
Owyhee Uplands	956	0				
Upper Snake	301	0				
Snake Headwaters	387	0				
Central Idaho Mountains	1232	789	678 (96)	235 (107)	353 (193)	111 (30)
Entire Assessment Area	7498	2646	2251 (303)	587 (255)	1485 (494)	393 (121)



Map 4.14—Current presence, absence, and estimated probability of occurrence within the historical range for westslope cutthroat trout by subwatershed. Current distributions are based on the current-status database. Probabilities of occurrence are based on the classification tree for presence or absence. Presence outside the historical range is shown as introduced.



Map 4.15—Current distribution of spawning and rearing areas classified or predicted as strong or depressed for westslope cutthroat trout by subwatershed. Current distributions are based on the current-status database. Predictions are based on the full classification tree for spawning and rearing areas.

trout mature at age three, but first spawning occurs mostly at age four or five. Sexually maturing fluvial and adfluvial fish move near spawning tributaries in fall and winter where they remain until migrating upstream in the spring (Liknes 1984). The patterns of movement and distribution suggest that landscapes may play an important role in the expression of life history. The full expression of varied life histories will likely require the maintenance of habitats across relatively large areas such as entire river and lake basins.

Habitat Relationships—In the portions of the Basin supporting westslope cutthroat trout most waters are relatively cold and nutrient poor (Liknes and Graham 1988; Rieman and Apperson 1989). Growth varies widely and is probably influenced by stream and lake productivity. Growth is also consistently higher for migrant forms that spend some period in larger rivers or lakes (Rieman and Apperson 1989). Growth likely has an important influence on the relative productivity and resilience of populations to disturbance and increased mortality (Rieman and Apperson 1989).

Substrate composition strongly influences survival. Weaver and Fraley (1991) demonstrated a negative relationship between emergence success and the percentage of fine sediments. Sediment reduces embryo survival (Irving and Bjornn 1984) and food and space for rearing juveniles (Bjornn and others 1977). Highly embedded substrates have been negatively correlated with juvenile abundance (Thurow 1987) and may be particularly harmful to trout that enter the substrate in winter (Peters 1988; Wilson and others 1987). Predicting the effects of fine sediment on wild populations remains difficult however (Chapman 1988; Everest and others 1987), and some populations clearly persist in systems with very high sediment levels (Magee and others 1996). Increased fines in substrates must be viewed as an increased risk for any population, but precise quantification of expected losses is unrealistic.

The distribution and abundance of larger (>150 mm) westslope cutthroat trout has been strongly associated with the number and quality of pools (Ireland 1993; Peters 1988; Pratt 1984; Shepard 1983). High quality pools appear to be especially important as wintering habitats (Lewynsky 1986; Peters 1988). Habitats that provide some form of cover also seem to be strongly preferred (Griffith 1970; Lider 1985; Pratt 1984). Fraley and Graham (1981) found the best models for predicting the distribution of trout in the Flathead River Basin included cover as an independent variable.

The association of westslope cutthroat trout with habitat characteristics influenced by land management suggests they are vulnerable to habitat disruption. Although this subspecies is still widely distributed, we found that both presence and the occurrence of strong populations were more likely on Forest Service than other lands, and in watersheds less influenced by roads or land management. Our classification analysis does not demonstrate causal linkages between management and habitat, but the pattern of our results is consistent with that hypothesis and the results of other studies (Rieman and Apperson 1989).

Key Factors—Introduced species have played a key role in the current status of westslope cutthroat trout. Non-native salmonids have been introduced throughout the range of westslope cutthroat trout including Glacier National Park (Marnell 1988). Behnke (1992) speculated that non-native kokanee, lake trout, and lake whitefish are important causes of decline through predation and competition in lakes. Mysids (*Mysis relicta*) have also been introduced in several lakes in Idaho and Montana and might influence westslope cutthroat trout populations (Bowles and others 1991). Behnke (1992) concluded that brown, brook, and rainbow trout along with changes in flow and water quality, were responsible for the demise of some westslope cutthroat trout populations in the Spokane and Clark Fork river drainages. Fausch (1988, 1989) suggested that the persistence of cutthroat trout is jeopardized in streams also supporting brook or brown trout. Brook trout are thought to have replaced many westslope cutthroat trout populations in headwater streams (Behnke 1992), but the mechanism of interaction is not clear (Fausch 1988; Griffith 1988). When the two species coexist, cutthroat trout seem to predominate in the higher gradient reaches (Griffith 1988), while brook trout may prevail in lower gradients (Fausch 1989).

Although closely related, cutthroat and rainbow trout have remained reproductively distinct (Behnke 1992) where they evolved in sympatry. Where non-native rainbow trout have been introduced, hybridization is widespread (Behnke and Zarn 1976; Rieman and Apperson 1989). Yellowstone cutthroat trout have also been widely introduced into the westslope cutthroat trout range and hybridization between these two forms is common (Liknes 1984; Rieman and Apperson 1989). Genetic introgression was believed to be the most important cause for decline of westslope cutthroat trout populations in Montana (Liknes and Graham 1988), and may compromise populations throughout the range.

Angling is a second factor important in the status of these fish. Westslope cutthroat trout are highly susceptible to angling (Behnke 1992; Lewynsky 1986; MacPhee 1966). Many populations have increased in response to harvest restrictions (Johnson and Bjornn 1978; Peters 1988; Rieman and Apperson 1989; Thurow and Bjornn 1978). Rieman and Apperson (1989) found evidence of a depensatory effect in fishing (mortality increased with decline in population size) and speculated that harvest could lead to the elimination of some small populations. Others believe that angler harvest led to the virtual elimination of fluvial fish in some river systems.[9] Special harvest restrictions may be necessary to maintain most westslope cutthroat trout populations (Rieman and Apperson 1989). Most state management agencies limit harvest of native westslope cutthroat trout with restrictive angling regulations.

Habitat disruption is a third factor consistently identified in the decline in the status of westslope cutthroat trout. Habitat loss and degradation are primary concerns for persistence of westslope cutthroat trout (Liknes 1984; Liknes and Graham 1988; Rieman and Apperson 1989). Accumulation of fine sediments has been a primary concern for biologists dealing with fish habitat relations (for example, Rieman and Apperson 1989; Stowell and others 1983). In watersheds within the belt geologies of northern Idaho and western Montana, excessive bedload transport and scour are problems associated with watershed disruption and increased peak flow events (Cross and Everest 1995). In low-gradient channels bed aggradation may result both in the loss of pools or pool volume, and in channel dewatering during low flow periods. The relatively simple, and unstable channels that result from intensive management of these basins were overlooked as problems in earlier concerns focused on fine sediments (Gamblin 1988; Rieman and Apperson 1989). Intensive management may lead to habitat disruption through a variety of mechanisms.

<u>Summary</u>—The current abundance of westslope cutthroat trout appears to be restricted from historical conditions (Behnke 1992; Bjornn and Liknes 1986; Liknes and Graham 1988; Rieman and Apperson 1989). Local extirpations are also evident in portions of the range. Construction of dams, irrigation diversions, or other migration barriers has isolated or eliminated westslope cutthroat trout habitats that were once available to migratory populations (Rieman and Apperson 1989). Resident forms may persist in isolated segments of streams but the loss of the migratory life-history and the connection with other populations potentially important to gene flow or metapopulation dynamics, may seriously compromise the potential for long term persistence (McIntyre and Rieman 1995). Climate change might be important in further restriction of westslope cutthroat trout populations (Mullan and others 1992; Rieman and McIntyre 1995). Although small and often isolated populations appear to persist throughout the range, the long term outlook for many of these is poor.

[9]Personal communication. 1995. T.C. Bjornn, University of Idaho, Moscow, Idaho.

There is little chance that populations strongly influenced by introgressive hybridization can ever be recovered as genetically pure. Because most of the genetic variation in the subspecies occurs among rather than within populations (Allendorf and Leary 1988) maintenance of the few remaining strong populations could be critical to preservation of remaining genetic diversity.

Westslope cutthroat trout are still widely distributed throughout the Basin. Among the salmonids, they are second in number of occupied subwatersheds to sympatric redband trout. They still occur in the largest part of the historical range of any of the seven key salmonids. The broad distribution suggests that westslope cutthroat trout are secure. This may be misleading, however, because of questions about the genetic integrity of remaining populations. Most populations are depressed, and genetic introgression, fragmentation and the loss of migratory life-history forms may limit truly healthy populations to a smaller proportion of the historical range than is evident here. The causes for decline of westslope cutthroat trout are varied. Many strong populations in Idaho and Montana occurred largely in roadless and wilderness areas or national parks (Liknes 1984; Liknes and Graham 1988; Marnell 1988; Rieman and Apperson 1989) suggesting that human intervention has been important. In general, strong populations are thought to be primarily associated with areas of limited human influence and the potential effects of fishing, watershed disturbance and non-native introductions (Rieman and Apperson 1989). The association of current distributions with areas of lower road density and with wilderness or low management impact in the classification-tree analysis is consistent with this view.

The core of the distribution for strong populations is clearly associated with the Central Idaho Mountains, and many populations there appear secure. Other important regions of known or likely habitat are in Idaho and Montana within the Upper Clark Fork and Northern Glaciated Mountains. These latter areas, however, are more fragmented and restricted to a relatively smaller portion of the historical distribution than the core of subwatersheds associated with central Idaho. Further erosion of remaining strongholds could influence both the broad representation of and the potential for long-term persistence of westslope cutthroat trout. The Northern Cascades was predicted to support a larger collection of subwatersheds, geographically distinct from the main distribution. The distribution and status were poorly known in this region, however, requiring extrapolation from relatively few subwatersheds to a much broader area. The origin of cutthroat trout throughout the Northern Cascades is also still in some question. More work will be necessary to better understand the status and distribution of native westslope cutthroat trout populations in the Northern Cascades. Westslope cutthroat trout were probably never widely distributed within the Blue Mountain or Columbia Plateau and only remnant or strongly isolated populations are found there now. Conservation of the remaining populations within the latter three regions may be more difficult but particularly important to conservation of the historical distribution and evolutionary legacy of westslope cutthroat trout.

Redband Trout (*Oncorhynchus mykiss*)—The rainbow trout is a widely distributed western North America native salmonid. Rainbow trout have been segregated into three forms (Behnke 1992): 1) Coastal rainbow trout west of the Cascade/Sierra mountain divide; 2) Interior Columbia River redband trout upstream of Celilo Falls, including the Fraser and Athabasca rivers in Canada, the upper Klamath River Basin, and the isolated interior basins of Oregon; and 3) the Sacramento-San Joaquin redband trout (Behnke, 1992). Although the systematics are incomplete, physical characteristics and genetic studies support the view that these three rainbow trout forms warrant subspecific recognition (Allendorf 1975; Allendorf and Utter 1979; Allison and Bond 1983; Berg 1987; Stearley and Smith 1993; Utter and Allendorf 1977).

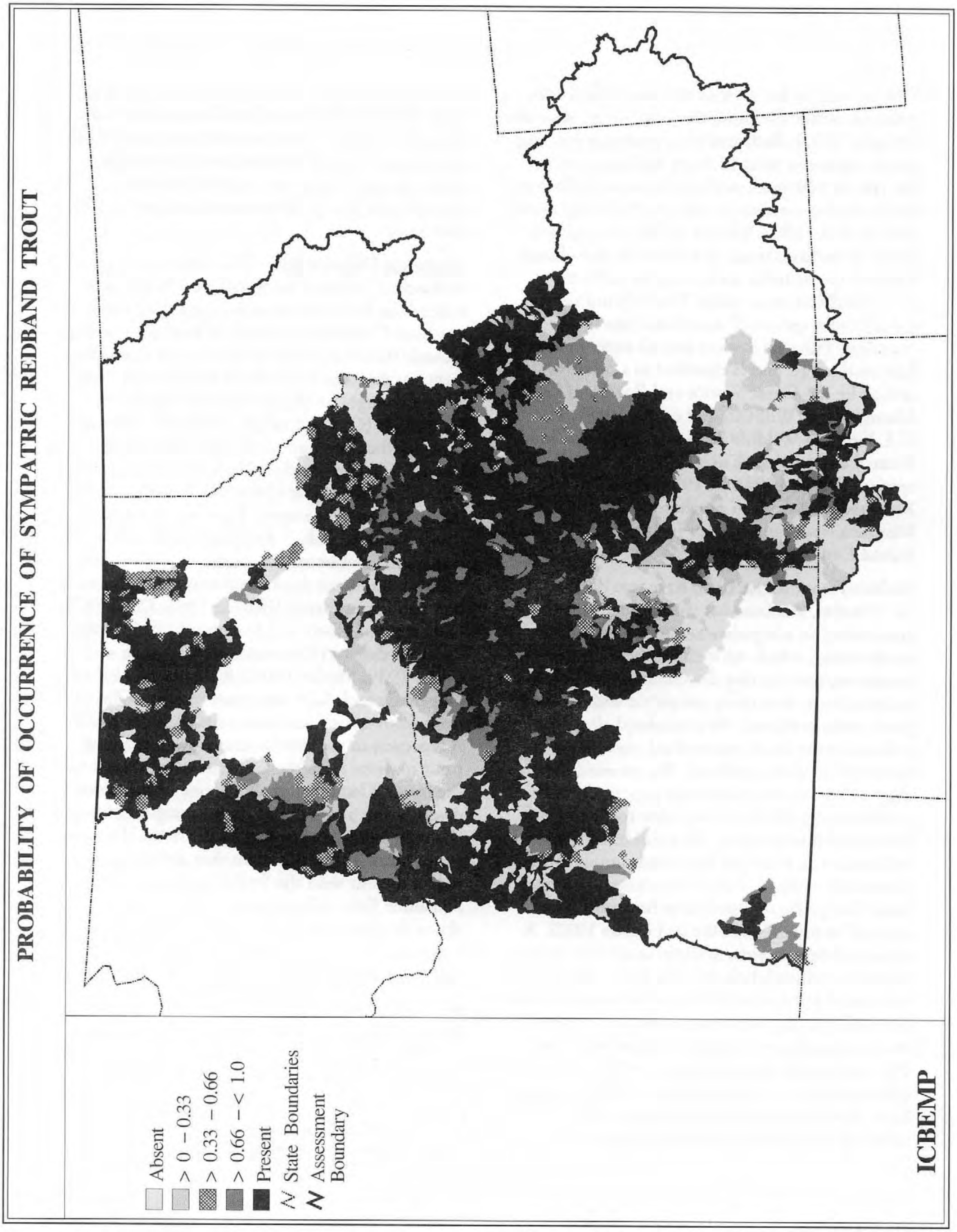
The interior redband trout was more widely distributed within the Basin than any other salmonid (Behnke 1992). Redband trout probably replaced native cutthroat trout in many subbasins within the last 30,000 years, perhaps favored by climatic and hydrologic events during and following glaciation (Behnke 1992; McKee 1972). Introgressed forms of redband trout, hybrids with introduced rainbow or cutthroat trout, may be replacing native redband trout today. The redband trout is considered a species of special concern by the American Fisheries Society and all states in the historical range, and is classified as a sensitive species by the Forest Service and Bureau of Land Management (Williams and others 1989). The U.S. Fish and Wildlife Service lists the McCloud River, California stock as a Category 1 Candidate species (Beattie 1994). In 1994, the Kootenai River redband trout in northern Idaho and Montana was petitioned for listing under the federal Endangered Species Act.

Redband trout in the Basin have two distinct life histories, anadromous (steelhead), which we considered in a separate section, and non-anadromous, which we focus on here. In this section we have further divided non-anadromous redband trout into those sympatric with or allopatric with steelhead. We considered allopatric redband trout those that evolved outside the historical range of steelhead. We assumed the allopatric form was potentially genetically and evolutionarily distinct from other redband trout because of this isolation. We considered sympatric redband trout to be the non-anadromous form historically derived from or associated with steelhead. Sympatric redband trout have also been termed "residuals" (Mullan and others 1992). A non-anadromous form is likely to exist in sympatry with steelhead, however, the level of genetic or behavioral segregation between forms is unknown. Morphologically, anadromous and non-anadromous redband trout juveniles are indistinguishable, and one might anticipate that the life history differences have evolved repeatedly from a single form throughout the Basin (Behnke 1995). We relied on knowledge of established barriers to anadromy to define the range for the allopatric form. The distribution of small populations of allopatric redband trout isolated from, but within the general range of the steelhead (for example, above natural barriers in 2nd and 3rd order streams) was poorly documented and not considered here.

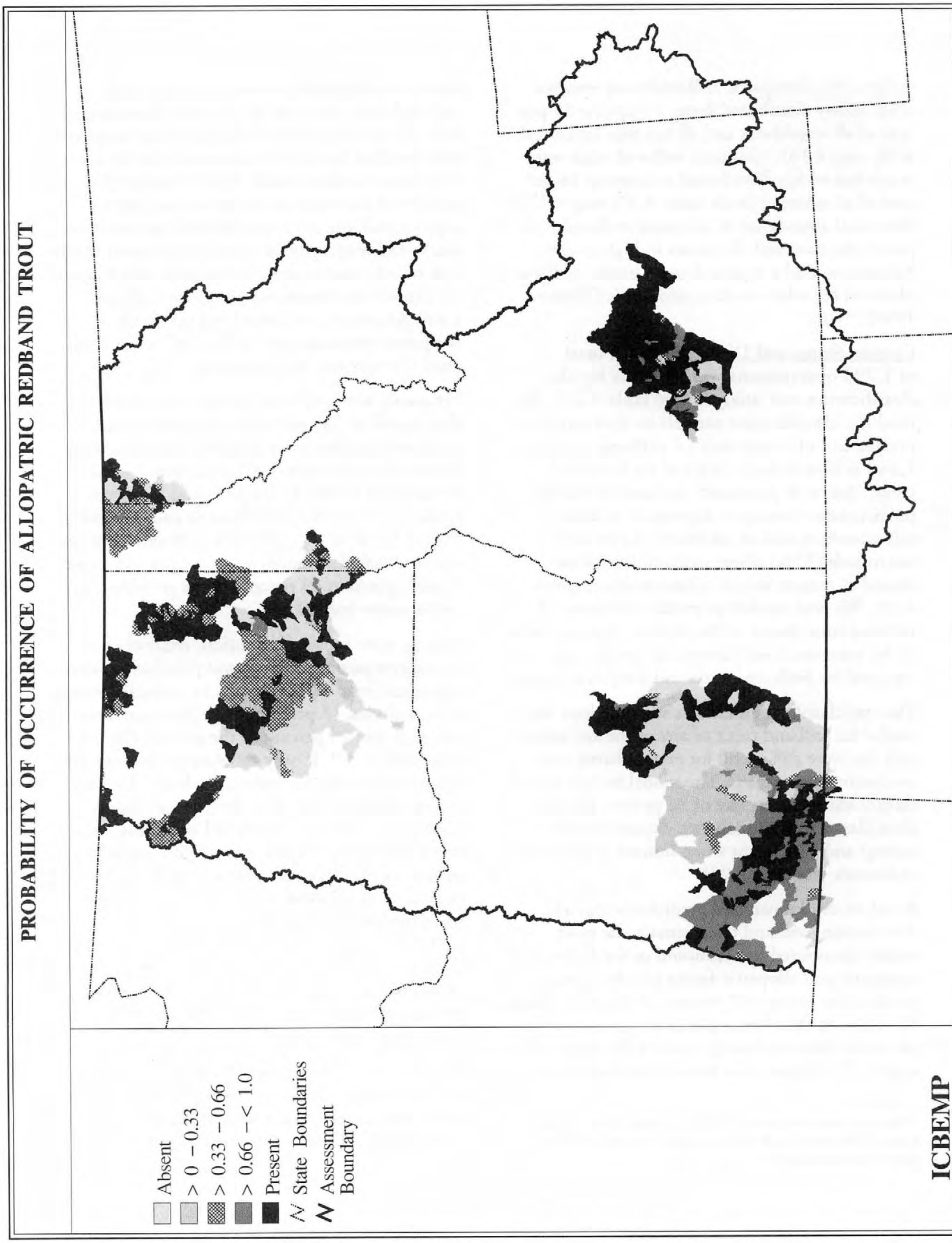
Historical Distribution—The historical range of all forms of redband trout included freshwaters west of the Rocky Mountains, extending from northern California to northern British Columbia, Canada (Behnke 1992). What may be a primitive form has been found in the Athabasca and Peace River drainages on the east flank of the Rocky Mountains (Carl and others 1994). We did not consider the distribution of redband trout in Canada but they are thought to have been present throughout the upper Columbia River basin with the exception of the upper Kootenai River basin above Kootenai Falls.[10] Redband trout were widely distributed and occupied most accessible waters from the southern desert basins to the high mountain coniferous forests (Behnke 1992; Cope 1879; Cope 1889; Gilbert and Evermann 1895; Jordan 1892; Jordan and Evermann 1896; Jordan and others 1930; Snyder 1908). A barrier falls below Upper Klamath Lake separated interior redband trout from coastal rainbow trout (Behnke 1992). A wet cycle in the Pleistocene allowed redband trout to move from the Columbia River basin to the upper Klamath and all but one of the closed desert basins along the southern margin of Oregon (Behnke 1992; Hubbs and Miller 1948). The only major areas within the Basin that did not support redband trout were the Snake River upstream from Shoshone Falls, tributaries to the Spokane River above Spokane Falls, Eastern Rocky Mountain basins in Montana, and portions of the northern Great Basin in Oregon (maps 4.16 and 4.17).

Redband trout were the most widely distributed key salmonid in the Basin, with both forms historically occupying 77 percent of the ERUs and 73 percent of the subwatersheds within the Basin

[10]Personal communication. 1995. E. Parkinson, British Columbia Ministry of Environment, British Columbia, Canada.



Map 4.16—Current presence, absence, and estimated probability of occurrence within the historical range for sympatric redband trout (within the range of steelhead) by subwatershed. Current distributions are based on the current-status database. Probabilities of occurrence are based on the classification tree for presence or absence.



Map 4.17—Current presence, absence, and estimated probability of occurrence within the historical range for allopatric redband trout (outside the range of steelhead) by subwatershed. Current distributions are based on the current-status database. Probabilities of occurrence are based on the classification tree for presence or absence.

(table 4.33). Sympatric redband trout were the most widely distributed form, occupying 56 percent of all watersheds and all but four ERUs (table 4.34; map 4.16). Allopatric redband trout were much less widely distributed, occupying 18 percent of all subwatersheds (table 4.35; map 4.17). Historical abundance of allopatric redband trout is poorly documented. Densities in high quality habitats would be expected to be similar to those observed for other resident salmonids (Thurow 1990).[11]

Current Status and Distribution—A total of 1,793 observations was available for the classification-tree analysis (see table 4.22). We used the classification models to estimate the probability of occurrence of redband trout in 1,648 subwatersheds (30% of the historical range) that were previously unclassified and to predict status (strong or depressed) in those subwatersheds and an additional 2,034 subwatersheds (37%) where redband trout were classed as present but of unknown status (table 4.33). We used models to predict the status of redband trout based on biophysical characteristics of the watershed and patterns of distribution reported for both sympatric and allopatric forms.

The tree classification models were the least successful for redband trout of any of the key salmonids but were still useful for extrapolating into unclassified watersheds. The model had an overall classification success rate of 76 percent for the three classes of status (absent, depressed, and strong) and 91 percent when limited to presence or absence (see table 4.22).

Based on our known and predicted status and distribution, redband trout remain the most widely distributed key salmonid in the Basin, with sympatric and allopatric forms jointly known or predicted to occupy 47 percent of the entire Basin. We estimate that they occur in 64 percent of their combined historical range (table 4.36; map 4.16 and 4.17). Despite their broad distribution, we know less about the current distribution of redband trout than any of the other key salmonids. About 30 percent of the historical range was not classified (unknown occurrence) in the current-status database (table 4.33). Another 37 percent of the historical range was judged to support redband trout but too little information was available to evaluate status. One reason for the lack of information was our inability to differentiate juvenile steelhead and sympatric redband trout. As a result, we considered the status of sympatric redband trout "unknown" when steelhead were present in a watershed.

Sympatric redband trout are the most widely distributed of the two forms, the known and predicted distribution includes 69 percent of the historical range (table 4.37). The largest areas of unoccupied habitat in the potential historical range are in the Owyhee Uplands and Columbia Plateau (table 4.37). Allopatric redband trout are not as widely distributed and are currently found or anticipated in 49 percent of the potential historical range (table 4.38).

Despite their broad distribution, relatively few strong sympatric redband trout populations were identified (map 4.18). Known or predicted strong areas included 17 percent of the potential historical range and 24 percent of the present distribution (table 4.37). If only subwatersheds supporting spawning and rearing were considered, 30 percent of the subwatersheds were classified as strong (table 4.37). Allopatric redband trout had even fewer strong populations (map 4.19), including 9 percent of the potential historical range and 18 percent of the present distribution (table 4.38). Allopatric redband trout populations are least well distributed in the Northern Great Basin and Columbia Plateau where they are believed absent in 72 percent of the potential historical range and few strong populations were known or predicted within the present distribution.

The classification tree for redband trout tended to under-predict the occurrence of strong populations. One might anticipate that this species is more persistent and widely distributed than

[11]Personal communication. 1995. S. Yundt, Idaho Department of Fish and Game, Boise, Idaho. Personal communication of unpublished data.

Table 4.33—Summary of the current status and distribution classifications (number of subwatersheds) for redband trout (sympatric and allopatric) throughout the Ecological Reporting Units of the Basin.

				Status Where Present						
Ecological Reporting Unit	Total	Historical Range	Total Present	Strong*	Depressed*	Status Unknown	Corridor	Introduced	Absent	No Classification
Northern Cascades	340	340	253	5	4	244	0	0	9	78
Southern Cascades	141	125	107	25	38	44	0	0	14	4
Upper Klamath	175	175	44	0	10	34	0	0	0	131
Northern Great Basin	506	405	99	2	49	48	0	0	165	141
Columbia Plateau	1089	1050	465	24	156	282	3	0	210	375
Blue Mountains	695	695	574	48	95	428	3	0	33	88
Northern Glaciated Mountains	955	456	281	25	13	243	0	0	52	123
Lower Clark Fork	415	99	68	3	17	48	0	0	8	23
Upper Clark Fork	306	0	0							
Owyhee Uplands	956	956	311	40	167	104	0	0	287	358
Upper Snake	301	0	0	0	0	0	0	0	0	0
Snake Headwaters	387	0	0							
Central Idaho Mountains	1232	1157	786	79	148	559	0	0	44	327
Entire Assessment Area	7498	5458	2988	251	697	2034	6	0	822	1648

* = Refers to spawning and rearing areas only

Table 4.34—Summary of the current status and distribution classifications (number of subwatersheds) for sympatric redband trout throughout the Ecological Reporting Units of the Basin,

				Status Where Present						
Ecological Reporting Unit	Total	Historical Range	Total Present	Strong*	Depressed*	Status Unknown	Corridor	Introduced	Absent	No Classification
Northern Cascades	340	292	222	5	4	213	0	0	7	63
Southern Cascades	141	125	107	25	38	44	0	0	14	4
Upper Klamath	175	35	8	0	0	8	0	0	0	27
Northern Great Basin	506	0	0							
Columbia Plateau	1089	796	398	23	94	278	3	0	148	250
Blue Mountains	695	643	534	35	69	427	3	0	23	86
Northern Glaciated Mountains	955	258	180	21	3	156	0	0	31	47
Lower Clark Fork	415	98	67	3	17	47	0	0	8	23
Upper Clark Fork	306	0	0							
Owyhee Uplands	956	898	279	38	154	87	0	0	283	336
Upper Snake	301	0	0							
Snake Headwaters	387	0	0							
Central Idaho Mountains	1232	1051	693	66	146	481	0	0	43	315
Entire Assessment Area	7498	4196	2488	216	525	1741	6	0	557	1151

* = Refers to spawning and rearing areas only

Table 4.35—Summary of the current status and distribution classifications (number of subwatersheds) for allopatric redband trout throughout the Ecological Reporting Units of the Basin.

				Status Where Present						
Ecological Reporting Unit	Total	Historical Range	Total Present	Strong*	Depressed*	Status Unknown	Corridor	Introduced	Absent	No Classification
Northern Cascades	340	48	31	0	0	31	0	0	2	15
Southern Cascades	141	0	0							
Upper Klamath	175	140	36	0	10	26	0	0	0	104
Northern Great Basin	506	405	99	2	49	48	0	0	165	141
Columbia Plateau	1089	254	67	1	62	4	0	0	62	125
Blue Mountains	695	52	40	13	26	1	0	0	10	2
Northern Glaciated Mountains	955	198	101	4	10	87	0	0	21	76
Lower Clark Fork	415	1	1	0	0	1	0	0	0	0
Upper Clark Fork	306	0	0							
Owyhee Uplands	956	58	32	2	13	17	0	0	4	22
Upper Snake	301	0	0				0	0		
Snake Headwaters	387	0	0							
Central Idaho Mountains	1232	106	93	13	2	78	0	0	1	12
Entire Assessment Area	7498	1262	500	35	172	293	0	0	265	497

* = Refers to spawning and rearing areas only

Table 4.36—Summary of total known and predicted classifications (number of subwatersheds) for status of redband trout (sympatric and allopatric) pooled. The classifications are from the current-status database. The numbers predicted are based on the classification trees for all redband trout and are shown in parentheses. One hundred four subwatersheds classified as unknown did not have a prediction.

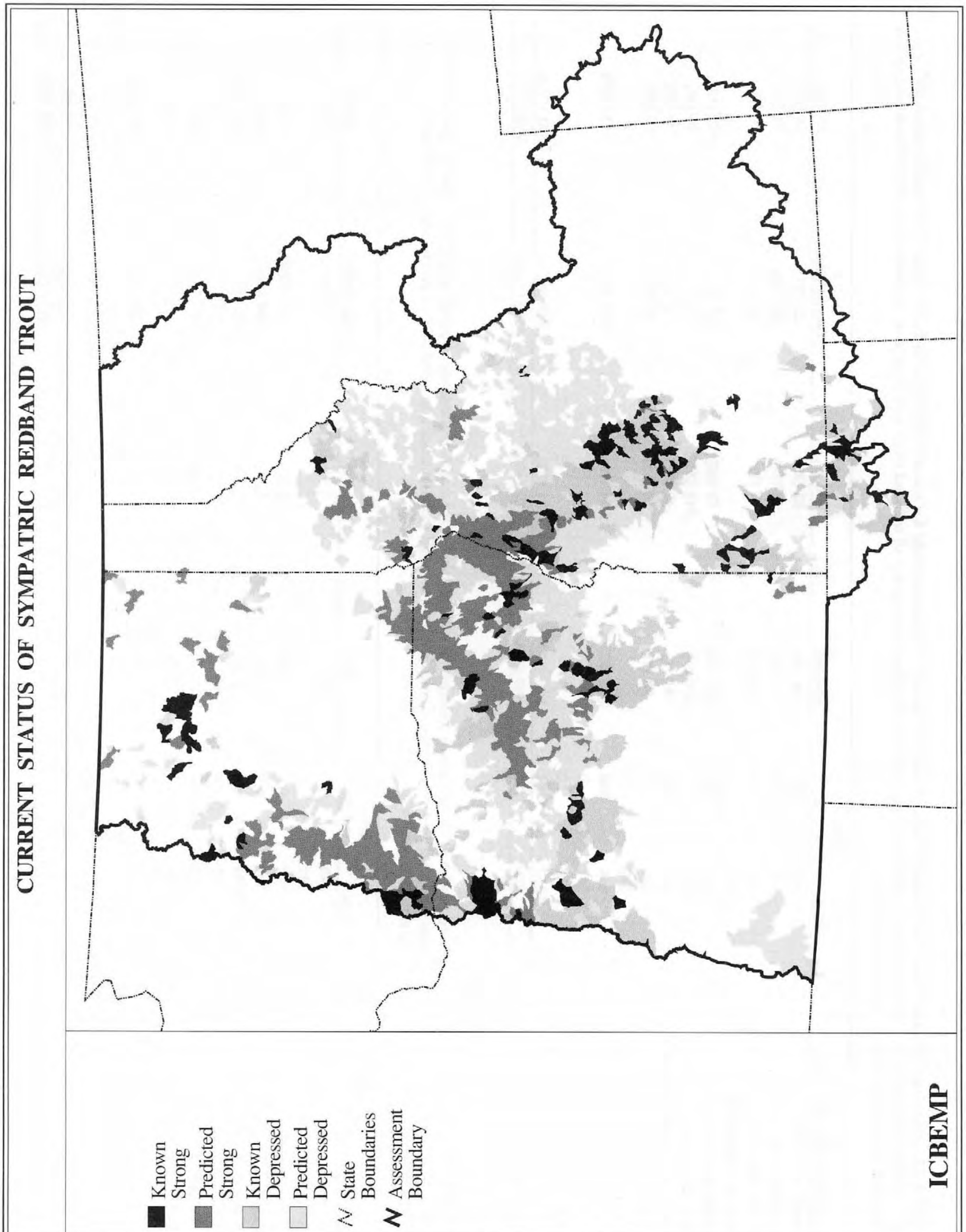
Ecological Reporting Unit	Total	Historical Range	Present	Strong	Depressed	Absent
Northern Cascades	340	340	289 (36)	94 (89)	47 (43)	50 (41)
Southern Cascades	141	125	111 (4)	46 (21)	56 (18)	14 (0)
Upper Klamath	175	175	134 (90)		143 (133)	23 (23)
Northern Great Basin	506	405	117 (18)	5 (3)	89 (40)	258 (93)
Columbia Plateau	1089	1050	529 (64)	74 (50)	425 (269)	521 (311)
Blue Mountains	695	695	621 (47)	264 (216)	268 (173)	74 (41)
Northern Glaciated Mountains	955	456	283 (2)	86 (61)	39 (26)	167 (115)
Lower Clark Fork	415	99	69 (1)	14 (11)	51 (34)	30 (22)
Upper Clark Fork	306	0				
Owyhee Uplands	956	956	375 (62)	56 (16)	236 (69)	532 (245)
Upper Snake	301	0				
Snake Headwaters	387	0				
Central Idaho Mountains	1232	1157	972 (186)	177 (98)	667 (519)	185 (141)
Entire Assessment Area	7498	5458	3500 (510)	816 (565)	2021 (1324)	1854 (1032)

Table 4.37—Summary of total known and predicted classifications (number of subwatersheds) for status of sympatric redband trout. The classifications are from the current-status database. The numbers predicted are based on the classification trees for all redband trout and are shown in parentheses. Fifty-five subwatersheds classified as unknown did not have a prediction.

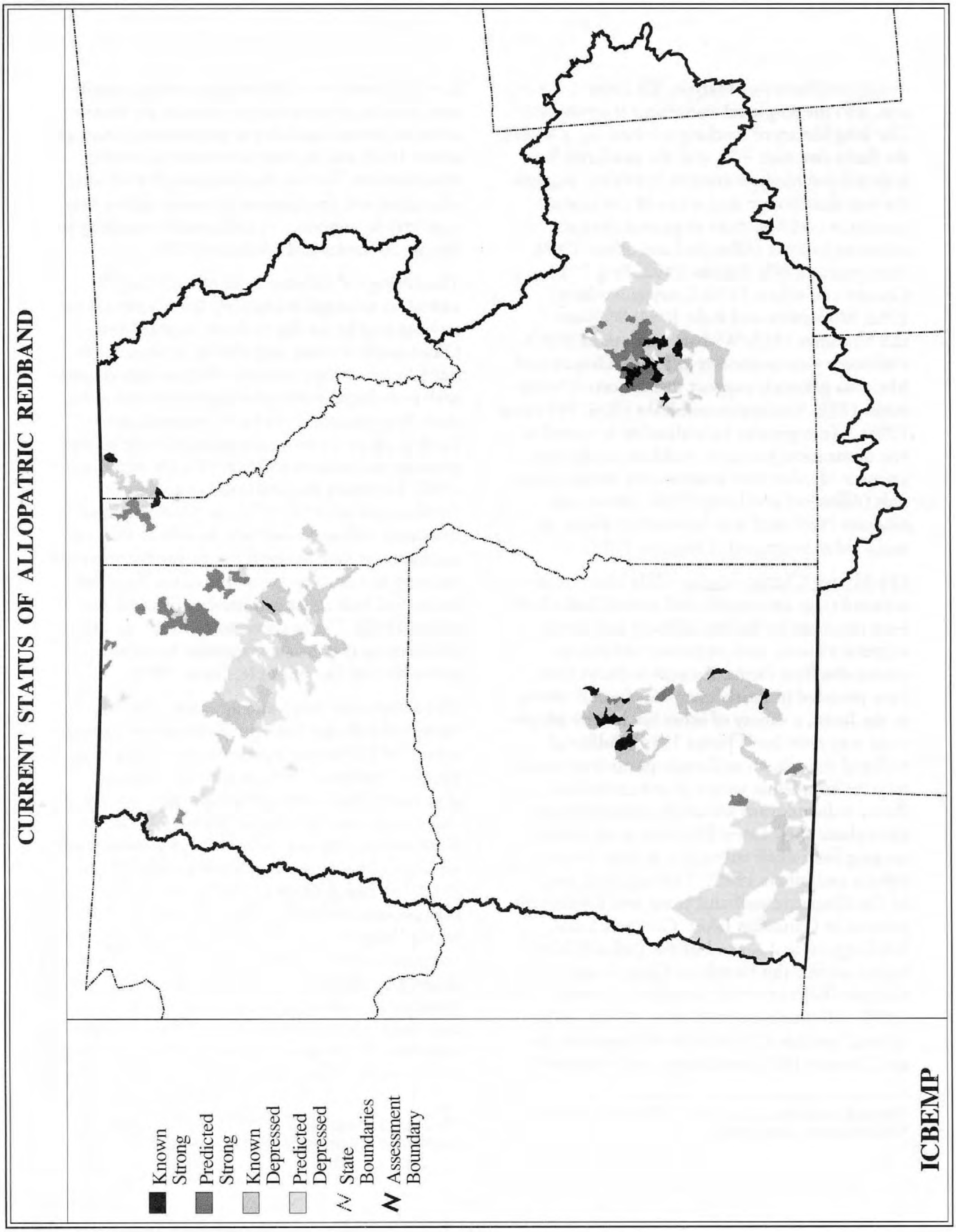
Ecological Reporting Unit	Total	Historical Range	Present	Strong	Depressed	Absent
Northern Cascades	340	292	258 (36)	93 (88)	35 (31)	34 (27)
Southern Cascades	141	125	111 (4)	46 (21)	56 (18)	14 (0)
Upper Klamath	175	35	23 (15)	0 (0)	31 (31)	8 (8)
Northern Great Basin	506	0				
Columbia Plateau	1089	796	457 (59)	73 (50)	305 (211)	339 (191)
Blue Mountains	695	643	579 (45)	249 (214)	241 (172)	64 (41)
Northern Glaciated Mountains	955	258	181 (1)	42 (21)	12 (9)	75 (44)
Lower Clark Fork	415	98	68 (1)	14 (11)	50 (33)	30 (22)
Upper Clark Fork	306	0				
Owyhee Uplands	956	898	336 (57)	54 (16)	213 (59)	513 (230)
Upper Snake	301	0				
Snake Headwaters	387	0				
Central Idaho Mountains	1232	1051	870 (177)	131 (65)	613 (467)	181 (138)
Entire Assessment Area	7498	4196	2883 (395)	702 (486)	1556 (1031)	1258 (701)

Table 4.38—Summary of total known and predicted classifications (number of subwatersheds) for status of allopatric redband trout. The classifications are from the current-status database. The numbers predicted are based on the classification trees for all redband trout and are shown in parentheses. Forty-nine subwatersheds classified as unknown did not have predictions.

Ecological Reporting Unit	Total	Historical Range	Present	Strong	Depressed	Absent
Northern Cascades	340	48	31 (0)	1 (0)	12 (12)	16 (14)
Southern Cascades	141	0				
Upper Klamath	175	140	111 (75)	0 (0)	112 (102)	15 (15)
Northern Great Basin	506	405	117 (18)	5 (3)	89 (40)	258 (93)
Columbia Plateau	1089	254	72 (5)	1 (0)	120 (58)	182 (120)
Blue Mountains	695	52	42 (2)	15 (2)	27 (1)	10 (0)
Northern Glaciated Mountains	955	198	102 (1)	44 (40)	27 (17)	92 (71)
Lower Clark Fork	415	1	1 (0)	0 (0)	1 (1)	0 (0)
Upper Clark Fork	306	0				
Owyhee Uplands	956	58	39 (7)	2 (0)	23 (10)	19 (15)
Upper Snake	301	0				
Snake Headwaters	387	0				
Central Idaho Mountains	1232	106	102 (9)	46 (33)	54 (52)	4 (3)
Entire Assessment Area	7498	1262	617 (117)	114 (79)	465 (293)	596 (331)



Map 4.18—Current distribution of spawning and rearing areas classified or predicted as strong or depressed for sympatric redband trout by subwatershed. Current distributions are based on the current-status database. Predictions are based on the full classification tree for spawning and rearing areas.



Map 4.19—Current distribution of spawning and rearing areas classified or predicted as strong or depressed for allopatric redband trout by subwatershed. Current distributions are based on the current-status database. Predictions are based on the full classification tree for spawning and rearing areas.

anticipated from our analysis. We suspect, however, that the empirical assessment is optimistic. The long history of stocking rainbow trout within the Basin (see map 4.7), and the proclivity for redband and rainbow trout to hybridize, suggests the true distribution and status of the original genotypes could be more depressed than our estimates indicate (Allendorf and others 1980; Anonymous 1995; Behnke 1992; Berg 1987; Currens and others 1990; Leary and others 1992; Moskowitz and Rahr 1994; Williams and Shiozawa 1993; Wishard and others 1984). Preliminary status reviews in Idaho, Oregon and Montana generally support this concern (Anonymous 1995; Moskowitz and Rahr 1994; Perkinson 1995).[12] Introgressive hybridization is viewed as one of the most pervasive problems in the management of other non-anadromous native salmonids (Allendorf and Leary 1988; Liknes and Graham 1988) and may be a serious threat to many fishes in general (Campton 1987).

Life-history Characteristics—Life histories of redband trout are variable and several forms have been described including adfluvial and fluvial migratory forms, non-migratory resident or stream-dwelling forms. Because redband trout have persisted in nearly every biophysical setting in the Basin, a variety of other life history adaptations may exist (see Thorpe 1994). Adfluvial redband trout (such as Kamloops rainbow trout) migrate from lentic waters to tributaries and fluvial redband trout remain in flowing waters throughout their entire life cycle, using streams ranging from small tributaries to large rivers (Moyle and others 1989). The adfluvial form of the allopatric redband trout was historically present in Canadian lakes, Crescent Lake, Washington and several of the isolated lake basins within the Northern Great Basin in Oregon (Behnke 1992; Moyle and others 1989). Allopatric redband trout are also isolated in small patches of habitat above migration barriers (Thurow 1987) and thus persist with only minimal movements. Movement among populations may be an important mechanism for maintenance of genetic variability in populations (Leary and others 1992) and for their persistence in variable environments. The role that isolation plays in local adaptation and development of unique alleles, however, may be important to total genetic variability in the species (Lesica and Allendorf 1995).

The ecology of redband trout remains largely unknown although many early life-history characteristics may be similar to those for steelhead. Other rainbow trout may also be representative, but because of the potential effects of introgression with introduced forms, that representation is not clear. It appears that steelhead confined above barriers adopt a non-anadromous lifestyle appropriate to the habitats available (Moffit and Bjornn 1984) but retain the potential for anadromy (Mullan and others 1992). The possibility that sympatric redband trout may be able to refound anadromous runs has importance for the potential recovery of unique populations of steelhead lost because of human-caused barriers (Mullan and others 1992). The maintenance of such distinct life histories may be an adaptation to variable environments (for example, Gross 1991).

The exceptional range and apparently broad adaptations of redband trout preclude narrow generalizations of life history but some observations are possible. Redband trout are primarily spring spawners (March-June) although they may reproduce at any time of the year (Kunkel 1976).[13] Redband trout spawn exclusively in flowing waters and typically migrate to spawning areas. Thurow (1990) observed allopatric redband trout migrating upstream to suitable spawning locations in spring. Migration timing is likely effected by water temperature and stream flow. Following spawning allopatric redband trout may maintain restricted home ranges until migrating to overwintering areas in the fall (Thurow 1990). Migratory juveniles typically move downstream to their ancestral lake or river after one to three years in natal areas.

[13]Personal communication. 1995. T. Klahr, U.S. Fish and Wildlife Service, Boise, Idaho.

[13]Personal communication. 1995. D. Buchanan, Oregon Department of Fish and Wildlife, Corvallis, Oregon.

Sexual maturity typically occurs at three to five years except in very cold (Mullan and others 1992) or hot climates. Growth is variable but likely dependent on genetic and environmental conditions.

Little is known about the interaction of redband trout with other species. Where redband and westslope cutthroat trout are naturally sympatric, the two species appear to have evolved segregative strategies (Rieman and Apperson 1989).[14] Populations that evolved in lakes have adopted piscivorous food habits. It is generally assumed that populations associated with streams and rivers rely heavily on aquatic and terrestrial invertebrates but foraging strategies are not well known. Redband trout are part of a native community that includes cottids, catostomids, cyprinids, and salmonids including westslope cutthroat trout, bull trout, mountain whitefish, steelhead, and chinook salmon (Hosford and Pribyl 1991; Kunkel 1976).[15]

Habitat Relationships—Redband trout occupy a wide array of habitats (Scott and Crossman 1973) illustrated by the broad distribution within the Basin (see map 4.16 and 4.17). Judged from the distribution among ERUs, redband trout occupy the widest range of biophysical settings of any of the key salmonids we considered. In the classification-tree analysis, variables including management cluster, ERU, slope, mean temperature, *hucorder*, vegetation cluster, solar radiation, and precipitation were useful in discriminating subwatersheds supporting redband trout (see table 4.22). Redband trout were more likely to be present or strong in watersheds less influenced by land management on Forest Service administered land, in mid-size or smaller streams, and in higher gradient streams, with more solar radiation, precipitation greater than about 30 centimeters, and mean air temperatures less than 8°-9° C. Although the classifications performed well, they were generally less successful than for the other salmonids in the analysis.

Other work suggests that redband trout are found in a wide range of conditions, often more extreme than those associated with other species. Populations found in the deserts along the southern margin of the Basin inhabit turbid and alkaline waters that range from near freezing to over 25° C (Johnson and others 1985; Kunkel 1976; Zoellick 1995).[16] Growth has been positively associated with temperature in forested streams (Mullan and others 1992) and redband trout are often found in warmer waters than other salmonids. There is some evidence that removal of canopy may benefit production in colder, high elevation streams (Rieman and others, in press). There are undoubtedly limits to their tolerance, however. In warmer and dryer environments the loss of riparian cover has been associated with reduced numbers and production of fish (Li and others 1994; Tait and others 1994).

There has been relatively little work defining habitat use for this fish, but patterns are generally similar to other salmonids. Thurow (1988) found redband trout most abundant in pool habitats and in association with cover components including undercut banks, large woody debris, and overhanging vegetation. Some have suggested that redband trout, like steelhead, may be associated with higher gradient channels, often in riffles or with substrates dominated by boulders, cobbles and pocket water (Kunkel 1976).[17] Slope was a relatively important variable in the classification models associating the occurrence of fish with steeper landscapes (table 4.22; appendix 4E).

Key Factors—Hybridization and competition are biotic factors influencing redband trout status. At least 35 non-native species have been introduced within the range of redband trout in the Basin. Introduced fishes create risks of genetic introgression, competition for food and space, predation, and increased exposure to disease (Fausch 1988; Reisenbichler 1977). Introduced rainbow trout are

[14]Personal communication. 1995. R. Leary, University of Montana, Missoula, Montana.

[15]Personal communication. 1995. D. Perkinson, Kootenai National Forest, Libby, Montana.

[16]Personal communication. 1995. D. Buchanan, Oregon Department of Fish and Wildlife, Corvallis, Oregon. Personal communication of unpublished data.

[17]Personal communication. 1995. D. Perkinson, Kootenai National Forest, Libby, Montana. Personal communication of unpublished data.

now the most widely distributed fish in the Basin (see map 4.7) and have contributed to losses of the native redband trout genotype through introgression (Behnke 1992; Campton and Johnston 1985). In attempts to reduce introgression some hatchery programs use native brood stocks, however, the practice of selecting for traits that improve fish performance in hatcheries followed by widespread out-planting of a few stocks may also lead to losses of local adaptions (Reisenbichler, in press). Brown trout are widely introduced and represent a potentially important predator and competitor, particularly in the southern range where redband trout are associated with warmer water temperatures (see map 4.7). In attempts to sustain remaining native redband trout, several state agencies have suspended all stocking of non-native species in isolated watersheds.

Fragmentation and isolation of habitats influencing redband trout status. If watershed disturbances result in loss of corridors or connecting habitats, remaining redband trout populations can be progressively isolated into smaller and smaller patches of productive habitats. Corridors that provide habitat for migration, rearing, and over-wintering may be critical to the conservation of species where connections among population are important (Hanski and Gilpin 1991; Rieman and McIntyre 1993). Such effects can be exaggerated by climate change. In the Goose Lake basin, Oregon, adfluvial redband trout find refuge in tributaries when the lake dries and recolonize the lake when it fills.[18] Factors that isolate tributaries from Goose Lake would increase the risk of extirpation during dry cycles. The loss of genetic variability through genetic drift may be a particularly important problem in the more isolated watersheds on the southern portion of the range of redband trout (Berg 1987; Wallace 1981).

Habitat degradation is a third factor influencing redband trout status. Interior redband trout habitats have been altered by a host of land use practices (Anonymous 1995; Moskowitz and Rahr 1994; Perkinson 1995; Williams and others 1989). Diverting water for irrigation threatens many populations in the southern portion of the range. Thurow (1988) reported four principal effects from water diversions: dewatering of stream reaches, loss of fish in unscreened diversions, blockage of migration corridors, and alteration of stream channels by earthmoving equipment. The loss or conversion of riparian cover has been caused by grazing, timber harvest, mining, urbanization and agriculture (Meehan 1991). In desert climates, the loss of riparian canopy has been associated with excessive temperature and reduced redband trout abundance (Li and others 1994; Tait and others 1994). Channel alterations associated with attempts to control flooding, develop floodplains, and construct roads have been extensive. Channel alterations adversely effect stream hydraulics (Bottom and others 1985), nutrient pathways (Schlosser 1982), invertebrate production (Benke and others 1985) and fish production. In Idaho, unaltered stream reaches supported eight to ten times the densities of redband trout observed in altered channels (Irizarry 1969; Thurow 1988). Habitat alterations may reduce the resilience and stability of the entire aquatic assemblage (Pearsons and others 1992; Thurow 1990).

Summary—Although redband trout appear to be widely distributed within the Basin their status is clouded by the uncertainty over taxonomic classification within the species, and by more than a century of stocking non-native rainbow trout and steelhead (Behnke 1992). Habitat degradation, hybridization or competition with introduced species, and a restricted range for some populations are the principal threats to conservation of the remaining redband trout (Williams and others 1989). Redband trout appear to have evolved over a broader range of environmental conditions than the other key salmonids and appear to have less specific habitat requirements. Their apparent persistence even in some heavily disturbed basins suggests they are less strongly influenced by habitat disruption than other salmonids. The loss of a

[18]Personal communication. 1995. J. Williams, Bureau of Land Management, Boise, Idaho.

redband trout population may be an indication of significant disruption in the processes influencing aquatic ecosystems.

Within the historical range of steelhead, sympatric redband trout are known or predicted to be widely distributed in large patches of suitable habitat in the Northern Cascades, Blue Mountains, and Central Idaho Mountains. These watersheds represent the core of the sympatric distribution and appear to be relatively secure. Introgression with introduced rainbow trout is potentially a serious but unevaluated threat. Known or predicted populations in watersheds within the Southern Cascades, Upper Klamath, Owyhee Uplands, and Northern Glaciated Mountains were recently (since 1900) isolated from steelhead by dams. These latter populations appear to be more fragmented in the remaining distribution. Allopatric redband trout within the Northern Great Basin, and portions of the Northern Glaciated Mountains, the Columbia Plateau, Central Idaho Mountains, and the Owyhee Uplands have been isolated from steelhead over geologic time. Remaining populations appear to be severely fragmented and restricted to small patches of known or potential habitat. These areas likely represent a critical element of the evolutionary history for this species and a major challenge in conservation management.

Steelhead (*Oncorhynchus mykiss*)—Steelhead, the anadromous form of rainbow/redband trout, are distributed within the Basin as two genetically distinct subspecies, coastal (*O. m. irideus*) and inland (*O. m. mykiss*) (Utter and others 1980). Coastal steelhead are found only in tributaries to the lower Columbia River west of the Hood River, Oregon (Kostow and others 1994). Each subspecies has two major forms, winter and summer, although coastal steelhead are predominately winter-run and inland steelhead summer-run. Winter-run steelhead enter freshwater three to four months prior to spawning (Withler 1966) and summer-run steelhead enter freshwater nine to ten months prior to spawning. Summer-run steelhead are described as either "A" run or "B" run, based on the time of passage over Bonneville Dam [Idaho Department of Fish and Game (IDFG) and others 1990].

The distribution and abundance of steelhead have declined from historical levels as a result of passage mortality at dams and obstructions, habitat degradation, loss of access to historical habitat, overharvest, and interactions with hatchery-reared and non-native fishes. A majority of the current populations consist of hatchery-reared fish (CBFWA 1990). Numerous state, federal, and provincial management agencies list remaining wild steelhead populations as species of special concern (Johnson 1987). The American Fisheries Society considers all stocks of winter-run steelhead upstream from Bonneville Dam to be at high or moderate risk of extinction and most summer-run steelhead stocks are considered at moderate risk of extinction or of special concern (Nehlsen and others 1991). Concern for the persistence of steelhead stocks culminated in 1994 petitions to the National Marine Fisheries Service for review of the species status under the Endangered Species Act. Steelhead represent a key species because of their broad distribution, value as a sport and commercial fish, and importance as a tribal ceremonial and subsistence resource.

Historical Distribution—The historical range of steelhead was the eastern Pacific Ocean and freshwaters west of the Rocky Mountains, extending from northwest Mexico to the Alaska Peninsula (Scott and Crossman 1973). The broad historical range of steelhead in the Basin is well documented [Howell and others 1985b; Northwest Power Planning Council (NWPPC) 1986] (Appendix 4D). Steelhead were present in most streams, both perennial and intermittent, that were accessible to anadromous fish including all accessible tributaries to the Snake River downstream from Shoshone Falls (Evermann 1896; Parkhurst 1950), and accessible tributaries to the Columbia River (Bakke and Felstner 1990; Fulton 1970; Howell and others 1985b) (map 4.20). Steelhead formerly ascended the Snake River and spawned in reaches of Salmon Falls Creek, Nevada, more than 1,450 kilometers from the ocean. Approximately 16,935 kilometers of stream were accessible to steelhead in the Columbia River basin including Canada (NWPPC 1986).

Steelhead occupied about 50 percent of the subwatersheds in the assessment area (table 4.39) including all ecological units except the Northern Great Basin, Upper Clark Fork, Upper Snake, and Snake Headwaters. It is unlikely that steelhead occupied all reaches of all accessible streams. Water temperature may have restricted their historical distribution. Mullan and others (1992) suggested that rainbow trout/steelhead avoid water temperatures exceeding 22° C (lower limit) and that the distribution of steelhead may be restricted to stream reaches that exceed 1,600 annual temperature units (upper limit). Platts (1974) similarly reported an upper elevational limit in the South Fork Salmon River, rainbow/steelhead populations were not found above 2,075 meters. The resolution of our historical range information was not sufficient to map upper and lower distributional boundaries in individual watersheds, so our maps overestimate the historical range.

Historical steelhead runs were large. Steelhead have been reported in the commercial Columbia River catch since 1889 and 2.23 mega-kilograms of canned steelhead were produced in 1892 (Fulton 1970). Initial estimates of run sizes were derived after the construction of the Bonneville Dam in 1938. In 1940, 423,000 summer-run steelhead passed the dam (NWPPC 1986). Annual sport harvests averaged 117,000 summer-run and 62,000 winter-run fish from 1962 to 1966 (Fulton 1970).

<u>Current Status and Distribution</u>—A total of 1,355 observations was available for the classification analysis (see table 4.22). We used the classification models to estimate the probability of occurrence of steelhead in 279 subwatersheds (7% of the historical range) that were previously unclassified and to predict status (strong, depressed, or corridor) in those subwatersheds and an additional 289 subwatersheds (7.7%) where steelhead were classed as present but of unknown status (table 4.39).

The model was very effective in predicting the distribution of steelhead spawning and rearing areas and migration corridors. Subwatersheds predicted to have high probabilities for spawning and rearing areas and subwatersheds predicted to have high probabilities of migration corridors were very similar to known distributions. The overall classification success rate was 89 percent for the four classes (absent, depressed, strong, and migration corridor) (see table 4.22).

Based on our known and predicted status and distribution, steelhead are the most widely distributed anadromous salmonid, however, steelhead are extinct in large portions of their historical range (map 4.20). The current known distribution of steelhead includes about 41 percent of the historical range (table 4.39). The known and predicted distribution includes 46 percent of the historical range (table 4.40). Steelhead are extinct in the Upper Klamath, Lower Clark Fork, and Owyhee Uplands and are known to be absent in from 21 to 83 percent of the other ERUs (table 4.39). About 12,452 kilometers of the historical range is no longer accessible in the Columbia River basin in the United States and Canada (NWPPC 1986).

Despite their relatively broad distribution, very few healthy steelhead populations exist. Subwatersheds known or predicted to support strong spawning and rearing represented 0.6 percent of the historical range and 1.3 percent of the current range (table 4.40; map 4.21). If only those subwatersheds where fish spawn and rear are considered, 98 percent of the subwatersheds were classified as depressed (table 4.40). Recent status evaluations also suggest that many steelhead stocks are depressed. A recent multi-agency review shows that total escapement of salmon and steelhead to the various Columbia River regions has been in decline since 1986 (Anderson and others 1996). Washington state biologists analyzed 15 summer-run steelhead stocks and three winter-run steelhead stocks in the Basin. Eighty-seven percent of

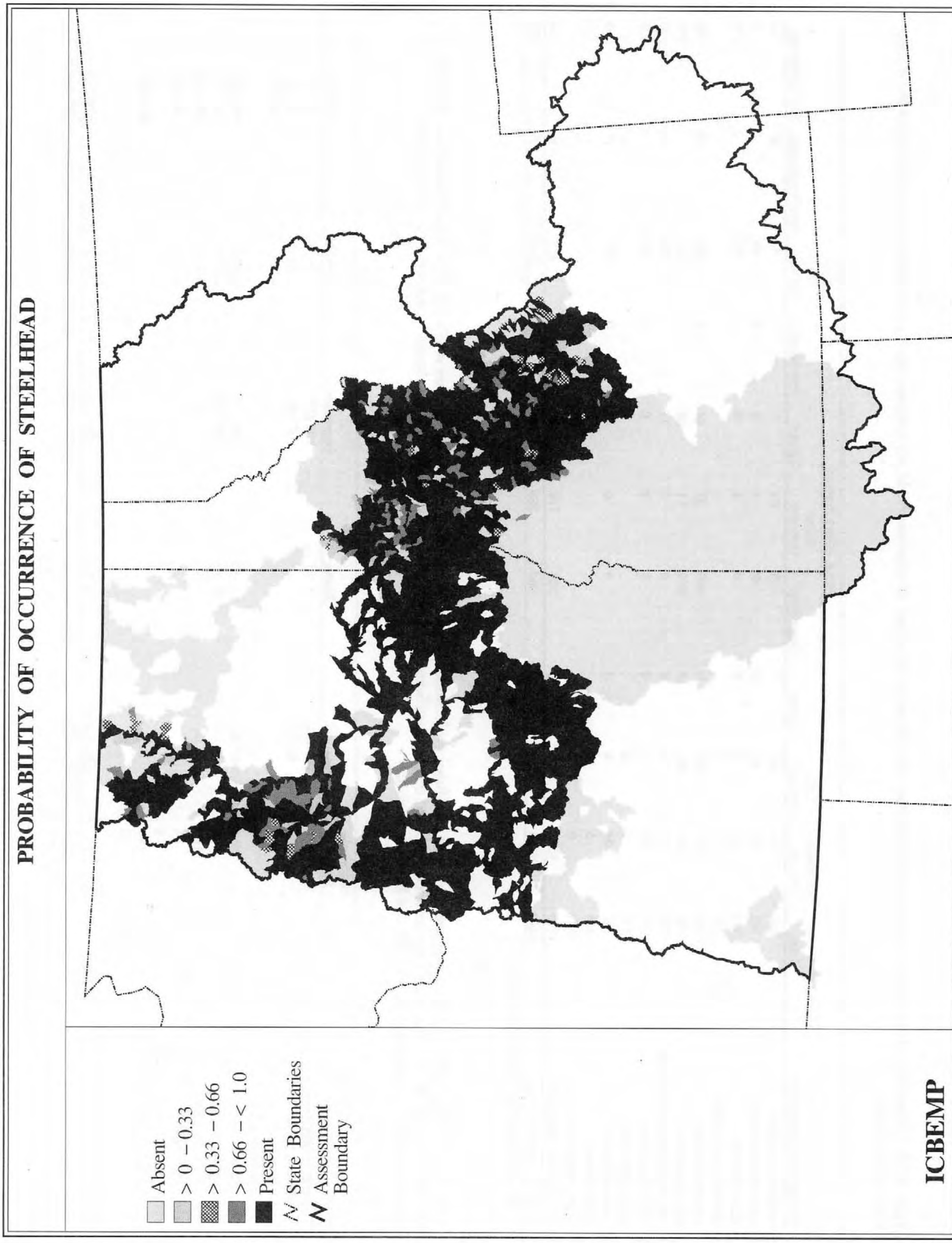
Table 4.39—Summary of the current status and distribution classifications (number of subwatersheds) for summer steelhead throughout the Ecological Reporting Units of the Basin.

				Status Where Present							
Ecological Reporting Unit	Total	Historical Range	Total Present	Strong*	Depressed*	Status Unknown	Corridor	Introduced	Absent	No Classification	Wild Indigenous
Northern Cascades	340	274	186	1	115	63	7	0	58	30	3
Southern Cascades	141	67	42	0	33	4	5	0	25	0	0
Upper Klamath	175	35	0	0	0	0	0	0	35	0	0
Northern Great Basin	506	0	0								
Columbia Plateau	1089	554	332	6	200	37	89	0	138	84	56
Blue Mountains	695	584	375	13	312	11	39	0	204	5	169
Northern Glaciated Mountains	955	193	21	0	6	0	15	0	159	13	0
Lower Clark Fork	415	98	0	0	0	0	0	0	98	0	0
Upper Clark Fork	306	0	0								
Owyhee Uplands	956	898	0	0	0	0	0	0	887	11	0
Upper Snake	301	0	0								
Snake Headwaters	387	0	0								
Central Idaho Mountains	1232	1051	566	3	303	174	86	0	349	136	150
Entire Assessment Area	7498	3754	1522	23	969	289	241	0	1953	279	378

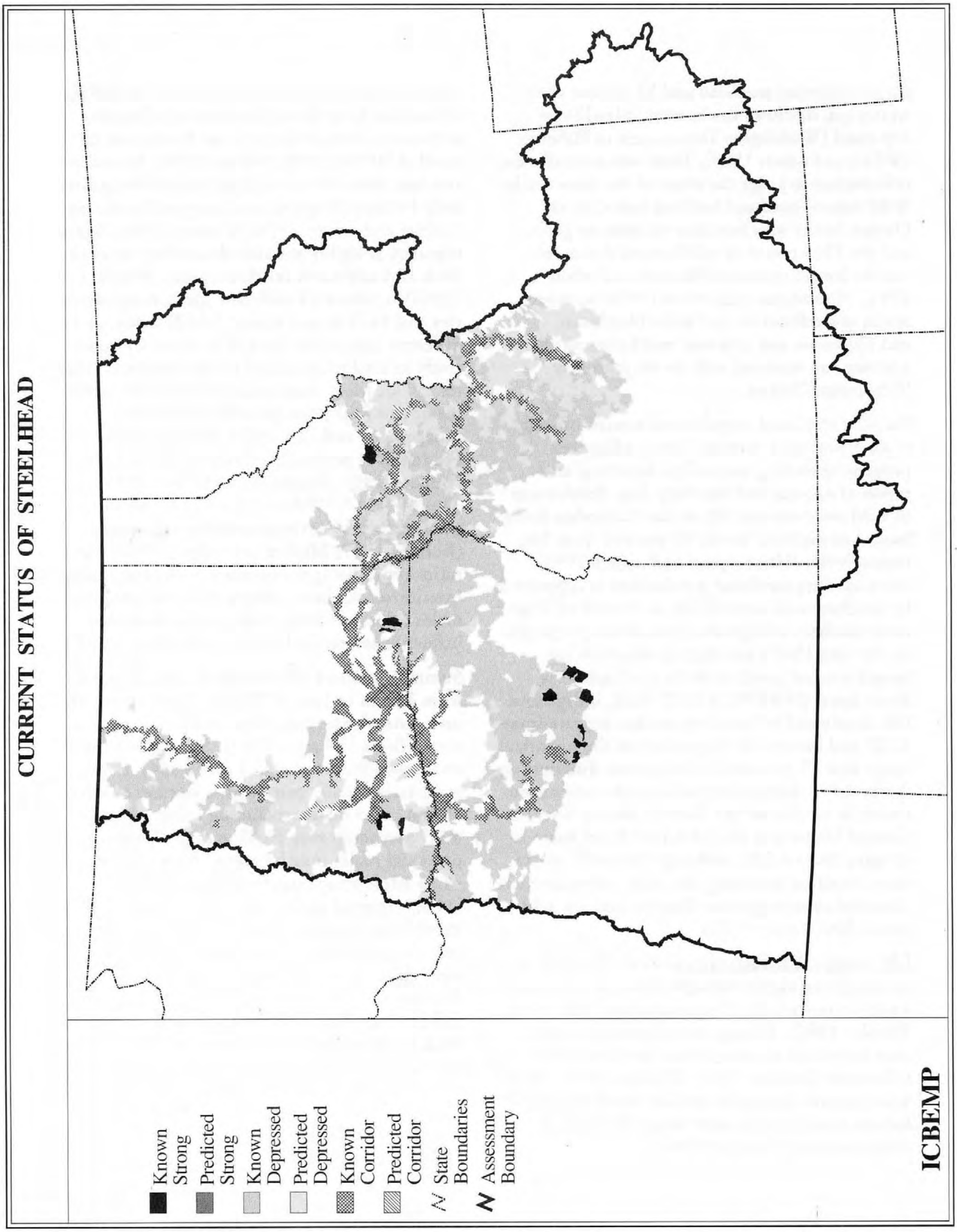
* = Refers to spawning and rearing areas only

Table 4.40—Summary of total known and predicted classifications (number of subwatersheds) for status of summer steelhead. The classifications are from the current-status database. The numbers predicted are based on the classification trees for summer steelhead and are shown in parentheses. Twenty-two subwatersheds classified as unknown did not have predictions.

Ecological Reporting Unit	Total	Historical Range	Present	Strong	Depressed	Corridor	Absent
Northern Cascades	340	274	201 (78)	1 (0)	192 (77)	8 (1)	65 (7)
Southern Cascades	141	67	42 (0)		37 (4)	5 (0)	25 (0)
Upper Klamath	175	35					35 (0)
Northern Great Basin	506	0					
Columbia Plateau	1089	554	392 (97)	6 (0)	289 (89)	97 (8)	160 (22)
Blue Mountains	695	584	378 (14)	13 (0)	326 (14)	39 (0)	206 (2)
Northern Glaciated Mountains	955	193	24 (3)		9 (3)	15 (0)	169 (10)
Lower Clark Fork	415	98					98 (0)
Upper Clark Fork	306	0					
Owyhee Uplands	956	898					887 (0)
Upper Snake	301	0					
Snake Headwaters	387	0					
Central Idaho Mountains	1232	1051	674 (282)	3 (0)	578 (275)	93 (7)	376 (27)
Entire Assessment Area	7498	3754	1711 (478)	23 (0)	1431 (462)	257 (16)	2021 (68)



Map 4.20—Current presence, absence, and estimated probability of occurrence within the historical range for steelhead by subwatershed. Current distributions are based on the current-status database. Probabilities of occurrence are based on the classification tree for presence or absence.



Map 4.21—Current distribution of spawning and rearing areas classified or predicted as strong or depressed for steelhead by subwatershed. Current distributions are based on the current-status database.

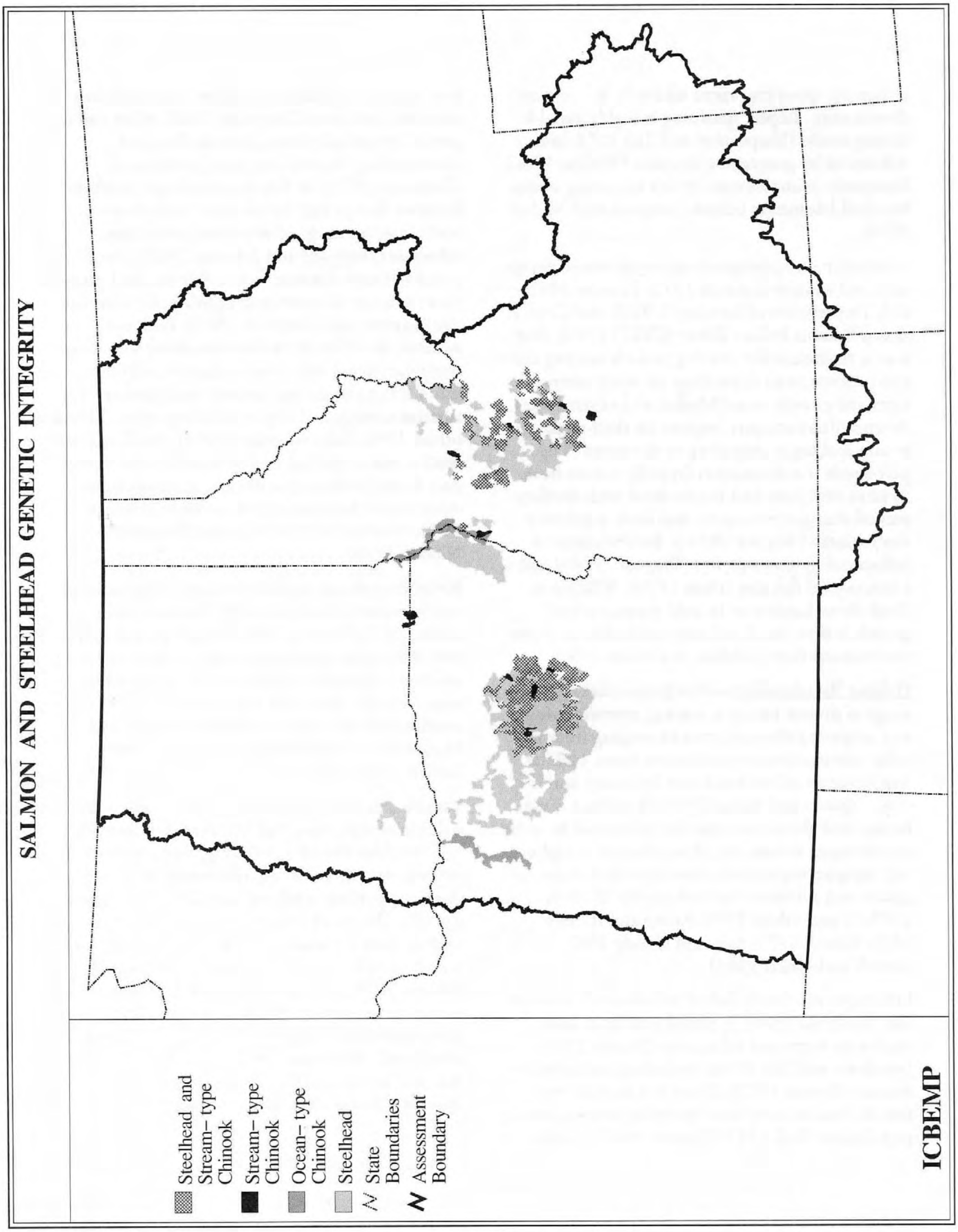
the summer-run steelhead and 33 percent of the winter-run steelhead stocks were judged to be depressed [Washington Department of Fisheries (WDF) and others 1993]. There was not sufficient information to judge the status of the other stocks. Wild/natural steelhead (defined below) in all Oregon basins were less than escapement goals and the 1994 return of wild/natural A-run fish was the lowest on record (Kostow and others 1994). Huntington and others (1994) surveyed stocks of steelhead in the Pacific Northwest and California and reported healthy stocks of summer-run steelhead only in the John Day River basin, Oregon.

Existing steelhead populations consist of four main types: wild, natural (non-indigenous progeny spawning naturally), hatchery, and mixes of natural and hatchery fish. Production of wild anadromous fish in the Columbia River basin has declined about 95 percent from historical levels (Huntington and others 1994). Most existing steelhead production is supported by hatchery and natural fish as a result of large-scale hatchery mitigation production programs. By the late 1960's hatchery production surpassed natural production in the Columbia River basin (NWPPC 1986). Wild, indigenous fish, unaltered by hatchery stocks, are rare (map 4.22) and present in 10 percent of the historical range and 25 percent of the current distribution (table 4.39). Remaining wild stocks are concentrated in reaches of the Salmon Selway rivers in Central Idaho and the John Day River basin in Oregon (map 4.22). Although few wild stocks were classified as strong, the only subwatersheds classified as strong were those sustaining wild stocks (6%) (table 4.40).

Life-history Characteristics—Life histories of steelhead are highly variable, both among and within populations (Shapovalov and Taft 1954; Withler 1966). Divergence of characteristics may be related to genetic and environmental influences (Bulkley 1967; Withler 1966). Their wide genetic repertoire enables steelhead populations to adapt to a wide range of physical circumstances (Thorpe 1994).

Mature adult summer-run steelhead ascend the Columbia River from May through October and winter-run steelhead from November to April (CBFWA 1990; Fulton 1970). Summer-run and winter-run steelhead may differ genetically because of spatial and temporal isolation (Leider and others 1984; Withler 1966). Age at maturity is highly variable depending on both fresh and salt water residence time. Withler (1966) reported 14 different life-history categories and Mullan and others (1992) reported 11 different categories. Length at maturity is positively related to length of ocean residence (Mallet 1974). Most steelhead remain in salt water for one to four years prior to maturation (Shapovalov and Taft 1954; Withler 1966). Fecundity is positively related to fish length (Bulkley 1967; Shapovalov and Taft 1954; Thurow 1987; Withler 1966) and may be genetically and environmentally influenced (Bulkley 1967; Mullan and others 1992). Sex ratios typically approximate 1:1 although many interior stocks have a larger ratio of females to males (Narver 1969). Male residualism may influence sex ratios (Mullan and others 1992).

Summer-run and winter-run steelhead spawn from March to June (CBFWA 1990), typically on a rising hydrograph and prior to peak streamflows (Thurow 1987). Similar to coastal steelhead (Shapovalov and Taft 1954), a dominant male usually pairs with a female, however, several other males, including small (< 200 mm) precocials may fertilize eggs. Spawning of precocial males may be particularly important when adult escapements are low. Parkinson (1984) reported genetic variation among steelhead in adjacent drainages indicating little interchange and a large number of individual populations. Reisenbichler and Phelps (1989) similarly believed that the variation in steelhead allele frequencies and the high degree of homing suggested that the basic breeding unit for steelhead occurred at the drainage or intradrainage scale.



Map 4.22—Watersheds supporting wild, native populations of steelhead and chinook salmon that have had little or no influence from introduced non-indigenous stocks.

Following spawning, spent adults (kelts) migrate downstream. Repeat spawning is highly variable among stocks (Shapovalov and Taft 1954) and is influenced by geographic location (Withler 1966). Iteroparity is uncommon in fish migrating several hundred kilometers inland (Simpson and Wallace 1978).

Incubation and emergence are temperature dependent and variable [Leitritz 1972; Thurow 1987; U.S. Department of Interior (USDI) and Confederated Yakama Indian Tribes (CYIT) 1993]. Parr rear in freshwater for varying periods ranging from two to seven years depending on water temperatures and growth rates (Mullan and others 1992). At smoltification, parr imprint on their natal stream and begin migrating to the ocean. The parr-smolt transformation typically occurs from April to mid-June and is associated with developmental changes in osmotic and ionic regulatory mechanisms (Wagner 1974a). Smoltification is influenced by photoperiod (Wagner 1974b) and is a function of fish size (Hoar 1976). When confined above barriers or in cold systems where growth is slow, steelhead may residualize to a non-anadromous form (Mullan and others 1992).

Habitat Relationships—Steelhead inhabit a wide range of diverse habitats, rearing, overwintering, and migrating through streams ranging from low order tributaries up to mainstem rivers. Habitat requirements of steelhead vary by season and life stage (Bjornn and Reiser 1991). Steelhead distribution and abundance may be influenced by water temperature, stream size, flow, channel morphology, riparian vegetation, cover type and abundance, and substrate size and quality (Everest 1973; Li and others 1994; Reiser and Bjornn 1979; Russell 1975; Salo and Cundy 1987; Stowell and others 1983).

Life stages are closely linked to habitat characteristics. Steelhead spawn in sorted gravels in both mainstem rivers and tributaries (Burner 1951; Needham and Taft 1934), including intermittent streams (Everest 1973). There is a marked variation in time of entry into spawning streams among populations (Ball 1985; Thurow 1987). Incubation success is influenced by fine sediment, temperature, and flow (Chapman 1988). After emergence, fry typically move into shallow and slow-moving channel margins (Everest and Chapman 1972). As fish become larger, preferred habitats change and fry use areas with deeper water, a wider range of velocities, and a larger substrate (Sheppard and Johnson 1985). Focal point velocity, distance from substrate, and maximum velocity all correlated significantly with fish size (Everest and Chapman 1972). In desert streams, densities of rainbow/steelhead were negatively correlated with solar radiation and maximum temperature, and streams with greater riparian canopy had higher standing crops (Li and others 1994; Tait and others 1994). Steelhead parr tend to select specific rearing habitats that segregate them, both temporally and spatially, from other native salmonids (Everest and Chapman 1972; Hanson 1977; Moffit and Bjornn 1984; Nilsson 1963).

Juvenile steelhead typically occupy different habitats in winter than in summer (Bustard and Narver 1975; Everest 1969). Steelhead may adopt two overwintering strategies, migration and concealment. Juveniles typically select areas of low water velocity and enter concealment cover beneath cobble or rubble substrate (Bustard and Narver 1975; Edmundson and others 1968; Everest and Chapman 1972).

Variables including hucorder, ERU, slope, solar radiation, vegetation and management clusters, and variables related to lithology and sensitivity to erosion were useful in discriminating subwatersheds supporting steelhead (see table 4.22; appendix 4E). Our results suggest that spawning and rearing areas for steelhead were likely to be found within specific core ERUs, in small to mid-size streams, in erosive land types and in steeper, higher elevation watersheds. Spawning and rearing also occurred primarily on Forest Service administered lands. Migration corridors represented probable steelhead occurrence in larger mainstem streams at lower elevations. Disturbance-related variables had a relatively minor influence in the model classifications. Management cluster and the number of dams steelhead must pass en route to the ocean were useful but not dominant.

There are several potential reasons for the lack of influence of disturbance variables on steelhead status. First, the current presence of steelhead is influenced by environmental factors within the watersheds in addition to factors outside the Basin. The disturbance variables we have access to may not adequately reflect "outside basin" disturbances including passage mortality, harvest, and ocean survival. Second, the model does not differentiate wild and hatchery origin steelhead. In some heavily disturbed areas, remnant numbers of steelhead may be present only because of annual hatchery supplementation so disturbance is not a good predictor of hatchery steelhead presence. Third, too few strong populations are present to relate their presence to disturbance. As a result, the model does not provide an adequate prediction of risk associated with different management scenarios.

Like the other anadromous fish, the status and distribution of steelhead are confounded by a large number of factors operating at multiple scales in both space and time. Ocean and passage conditions, harvest, and the use of hatchery fish have undoubtedly played a major role in the condition of the remaining populations. Sorting out the role of habitat change and land management effects in the decline of this species will likely result only through specific analysis capable of finer resolution and control of confounding effects.

Key Factors—Predation and competition are biotic factors that may influence the abundance of steelhead. More than 55 introduced species occur within the current range of steelhead and introduced species occupy a majority of the watersheds in the current steelhead range. Because introduced species did not evolve in sympatry with steelhead, there has been no opportunity for adaption to ameliorate competition (Fausch 1988) or predation. In many reaches of the Snake and Columbia rivers, introduced species outnumber indigenous fish (Li and others 1987). Hobbs and Huenneke (1992) suggest that non-natives may pose a greater risk to native species where habitat has been disturbed. Dams have created habitat that is suitable for a variety of non-native predators and potential competitors (Beamesderfer and Rieman 1991). Native predators may also be influenced by anthropogenic habitat alterations. Northern squawfish have become well adapted to habitat created by dams (Beamesderfer and Rieman 1991). Attempts to quantify the effects of predation in the Snake and lower Columbia rivers suggest an annual system-wide loss of 15 to 20 million juvenile salmonids to northern squawfish (Collis and others 1995).

Blocked access to historical habitat is a second factor influencing steelhead status. An estimated 12,452 kilometers of steelhead habitat are no longer accessible in the Columbia River basin in the United States and Canada (NWPPC 1986). Although undocumented, the historical range likely supported diverse and locally adapted populations. Extinctions have resulted in lower diversity and lower total abundance of steelhead. Cumulative habitat changes that eliminate or isolate segments of populations may increase both demographic and environmental stochasticity (thereby increasing the risk of extirpation or extinction of remaining populations) because of lower numbers and lower diversity in population structure or distribution (Rieman and others 1993).

Passage mortality is a third factor influencing steelhead status. Construction and operation of mainstem dams on the Columbia and Snake rivers are considered the major cause of decline of anadromous fish (CBFWA 1990). Hydroelectric development changed Columbia and Snake river migration routes from mostly free-flowing in 1938 to a series of dams and impoundments by 1975. Reservoirs reduce flows in most years by about 50 percent during smolt migration (Raymond 1979). Steelhead must navigate up to nine mainstem dams. At each dam, adult steelhead are delayed during upstream migrations. Smolts may be killed by turbines; become disoriented or injured, making them more susceptible to predation; or become delayed in the large impoundments behind dams (IDFG and others 1990). Smolt-to-adult return rates declined from more than 4 percent in 1968 to less than 1.5 percent from 1970-1974. In 1973 and 1977, low flows resulted in 95 percent of migrating smolts never reaching the ocean (Raymond 1979). The influence of passage mortality on steelhead stocks is illustrated by Huntington and others (1994) who concluded that although much of the Pacific Northwest's best remaining spawning and rearing habitat is in Idaho, no healthy stocks of anadromous fish were found.

Habitat degradation is a fourth factor influencing steelhead status. More than 95 percent of the healthy native stocks of anadromous fish identified by Huntington and others (1994) were judged by biologists to be threatened by some degree of habitat degradation. Similarly, Nehlsen and others (1991) identified habitat loss or degradation as a major problem for 90 percent of the 195 at-risk salmon and steelhead stocks they identified. Fish habitat quality in most watersheds has declined. During the past 50 years, numbers of pools have decreased and fine sediment has increased in selected Northwest watersheds (McIntosh and others 1994a). In addition to hydroelectric development, most alterations of steelhead habitat can be attributed to mining (Nelson and others 1991), timber harvest (Chamberlain and others 1991), agriculture (NWPPC 1986; Platts 1991), industrial development, and urbanization (NWPPC 1986).

Hatcheries are a fifth factor influencing steelhead status. Hatcheries have been widely used in attempts to mitigate losses of steelhead caused by construction and operation of dams (NWPPC 1986). Hatcheries may effect wild steelhead populations through genetic introgression and loss of fitness, creation of mixed-stock fisheries (see harvest below), competition for food and space, and changes in the abundance of disease organisms (Reisenbichler 1977). Studies of the interaction between wild and hatchery fish illustrate that survival of progeny from hatchery or hybrid (wild x hatchery) parentage is less than for progeny of wild fish pairings (Chilcote and others 1986; Reisenbichler and McIntyre 1977). Introduced rainbow trout also have the potential to mature and hybridize with steelhead (Chapman and others 1994b); and the species has been introduced throughout the current steelhead range (see map 4.7). Byrne and others (1992) suggested that supplementation of native stocks with hatchery fish have typically resulted in replacement, not enhancement of native steelhead. Most (107/121) of the healthy anadromous salmonid stocks identified by Huntington and others (1994) have either had no fish culture activities in the home watershed or been exposed to little risk from stock transfers or interaction with hatchery fish. Adult steelhead collection and egg taking operations may also be detrimental to individual populations (Chapman and others 1994b). Introductions of large numbers of hatchery-reared parr may cause localized decreases in the density of juvenile steelhead (Pollard and Bjornn 1973) and induce early migration of wild fish (Hillman and Mullan 1989).

Harvest is a sixth factor influencing steelhead status. Steelhead stocks have historically provided harvest opportunities for tribal, commercial, and sport fisheries. Wild steelhead populations have declined as numbers of hatchery steelhead have increased, creating harvest management problems. Hatchery steelhead that are surplus to egg taking needs can be harvested, in contrast to declining runs of wild steelhead that cannot. In response, sport and commercial harvest of wild adult steelhead has been closed (Chapman and others 1994b; IDFG 1992; NWPPC 1986), harvest of juvenile steelhead has been restricted in some areas (IDFG 1992), and tribal fisheries are regulated (Chapman and others 1994b; NWPPC 1986). Although the harvest of wild stocks has been reduced, declining runs of wild steelhead are still harvested in tribal fisheries, and steelhead are killed during commercial salmon fisheries in the

Columbia River and coastal marine waters, and in high seas driftnet fisheries (Chapman and others 1994b; Cooper and Johnson 1992-cited in Chapman and others 1994b).

<u>**Summary**</u>—The combination of low abundance, decreasing time trends in abundance, widely dispersed spawning populations, fragmented habitats, degraded freshwater and estuarine habitats, high risks of genetic introgression in most stocks, low frequency of wild indigenous stocks, high rates of non-native fish introductions, continued harvest in mixed-stock fisheries, and high-risks of smolt and adult passage mortality result in a high risk for individual steelhead stocks. Long-term persistence of steelhead will likely depend on maintaining and restoring migration corridors, mosaics of high quality habitats, and wild gene pools.

Steelhead are still relatively widely distributed in the Basin, however, they are extinct in nearly 60 percent of the historical range. Although steelhead are widespread throughout the remaining accessible range, most populations are severely depressed and heavily influenced by hatchery supplementation. Wild stocks are rare; core areas for remaining wild populations include the Salmon, Selway and John Day river basins. The only remaining strong populations are found among wild stocks, primarily in the Columbia Plateau and Blue Mountains. Within the Central Idaho Mountains recent steelhead runs have been critically low.

Restoration of habitats and habitat connectivity in both mainstem rivers and subbasins will be necessary for steelhead stocks to persist. The decline of steelhead stocks in the Basin in recent years can be attributed primarily to mainstem dams. However, until passage problems are resolved the resilience and persistence of remaining steelhead stocks will be largely dependent on the quality and diversity of remaining stream habitats. All remaining populations and habitats for steelhead within the Central Idaho Mountains, Blue Mountains, Northern Cascades, Columbia Plateau, Northern Glaciated Mountains, and Southern Cascades are critical to the persistence of this species within the Basin.

Chinook Salmon (*Oncorhynchus tshawytscha*)—Chinook salmon are widely distributed in the Basin. Traditionally they have been described as spring, summer, and fall races, separated primarily by their time of passage over Bonneville Dam (Matthews and Waples 1991). Spring chinook salmon cross Bonneville Dam from March to May, summers from June to July, and falls from August to September (Burner 1951). This nomenclature has led to some confusion across the Basin because stocks of similar run timing may differ considerably between the Snake and Columbia rivers in their spawning areas, life histories, behavior, and genetic characteristics. Gilbert (1912 as cited by Matthews and Waples 1991) first categorized juvenile chinook salmon that migrate seaward as yearlings as stream-type and those that migrate as subyearlings as ocean-type. We adopted these definitions to characterize chinook salmon stocks in the Basin. Within the Snake River basin and tributaries to the Columbia River downstream from the Snake River, stream-type chinook salmon include spring- and summer-run fish and ocean-type chinook salmon include fall-run fish (Fulton 1968; Gebhards 1959; IDFG 1992; Parkhurst 1950). Within the Columbia River upstream from its confluence with the Snake River "stream-type" chinook salmon include spring-run fish and "ocean-type" chinook salmon include summer- and fall-run fish (Matthews and Waples 1991, Mullan and others 1992).

Snake River chinook salmon (stream- and ocean-types) were listed as threatened under the federal Endangered Species Act in 1992. Their status was changed by emergency rule to endangered in 1994 because of continuing declines in abundance. The emergency rule terminated in 1995, and although their status has not improved, the species is again listed as threatened. Snake River chinook salmon populations in Oregon were also listed as threatened under the State endangered species statute in 1993. A petition was filed in 1993 to list mid-Columbia River (Chief Joseph to Priest Rapids dams) summer chinook salmon. The status review by the National Marine Fisheries Service (NMFS) determined that listing was not warranted. In

1994 NMFS began a status review of the remaining chinook salmon stocks in the United States outside Alaska.

Historical Distribution—The historical range of chinook salmon in North America was the eastern Pacific and Arctic oceans and accessible freshwaters (Scott and Crossman 1973). The broad historical range of chinook salmon in the Basin is well documented (Evermann 1896; Fulton 1970; Howell and others 1985a; NWPPC 1986; Parkhurst 1950) (appendix 4D). Chinook salmon were found in all accessible areas of the Snake River downstream from Shoshone Falls (Evermann 1896) and in all accessible areas of the Columbia River downstream from Windermere Lake, British Columbia (Fulton 1970) (maps 4.23 and 4.24). Like steelhead, chinook salmon were found in most accessible watersheds and formerly ascended the Snake River and spawned in reaches of Salmon Falls Creek, Nevada, more than 1,450 kilometers from the ocean. An estimated 16,935 kilometers of stream were accessible to chinook salmon in the Columbia River basin in the United States and Canada (NWPPC 1986).

Stream-type chinook salmon were the most widely distributed, occupying about 46 percent of the subwatersheds in the Basin and occurring in all ERUs except the Northern Great Basin, Upper Clark Fork, Snake Headwaters, and Upper Snake above Shoshone Falls (table 4.41). Ocean-type chinook salmon were much less widely distributed, occupying about 7 percent of the available subwatersheds and occurring in six of 13 ERUs (table 4.42). Ocean-type chinook salmon were historically present in the fewest subwatersheds of any key salmonid. The Snake River was considered the most important production area for ocean-type fall chinook salmon in the Columbia River Basin (Fulton 1968).

Cold water temperatures at high elevations (Mullan and others 1992) and the need for relatively large areas of suitable spawning gravel (Burner 1951) may have restricted the upper limit of chinook salmon but this is not well defined. Chinook salmon parr are also associated with low gradient, meandering, unconstrained stream channels (Scully and others 1990) which may further restrict their distribution.

Historical runs of chinook salmon were immense; estimates of annual run sizes prior to 1850 range from 3.4 to 6.4 million fish (NWPPC 1986). Most native people in the Basin shared a significant dependence on salmon as a subsistence and ceremonial resource (NWPPC 1986). Commercial harvest of chinook salmon in the mainstem Columbia River peaked in 1883 at 2.3 million fish and yield was about 1.3 million fish from 1890-1920 (Mullan and others 1992).

Current Status and Distribution—A total of 1,262 observations was available for stream-type and 224 observations for ocean-type chinook salmon classification-tree analysis, respectively (see table 4.22). We used the classification models to estimate the probability of occurrence of stream-type chinook salmon in 281 subwatersheds (8% of the historical range) that were previously unclassified (table 4.41). We predicted status (strong, depressed, or corridor) in those subwatersheds and an additional 141 subwatersheds (4%) where salmon were classed as present but of unknown status (table 4.41). We also used the classification models to estimate the probability of occurrence of ocean-type chinook salmon in 50 subwatersheds (9% of the historical range) that were previously unclassified (table 4.42). We predicted status (strong, depressed, or corridor) in those subwatersheds and an additional 20 subwatersheds (4%) where salmon were classed as present but of unknown status (table 4.42).

The model was very effective in predicting the distribution of chinook salmon spawning and rearing areas and migration corridors. Subwatersheds predicted to have high probabilities for depressed or strong spawning and rearing areas and subwatersheds predicted to have high probabilities of migration corridors were very similar to known distributions. For stream-type chinook salmon, the overall classification success rate was

Table 4.41—Summary of the current status and distribution classifications (number of subwatersheds) for stream-type chinook salmon throughout the Ecological Reporting Units of the Basin.

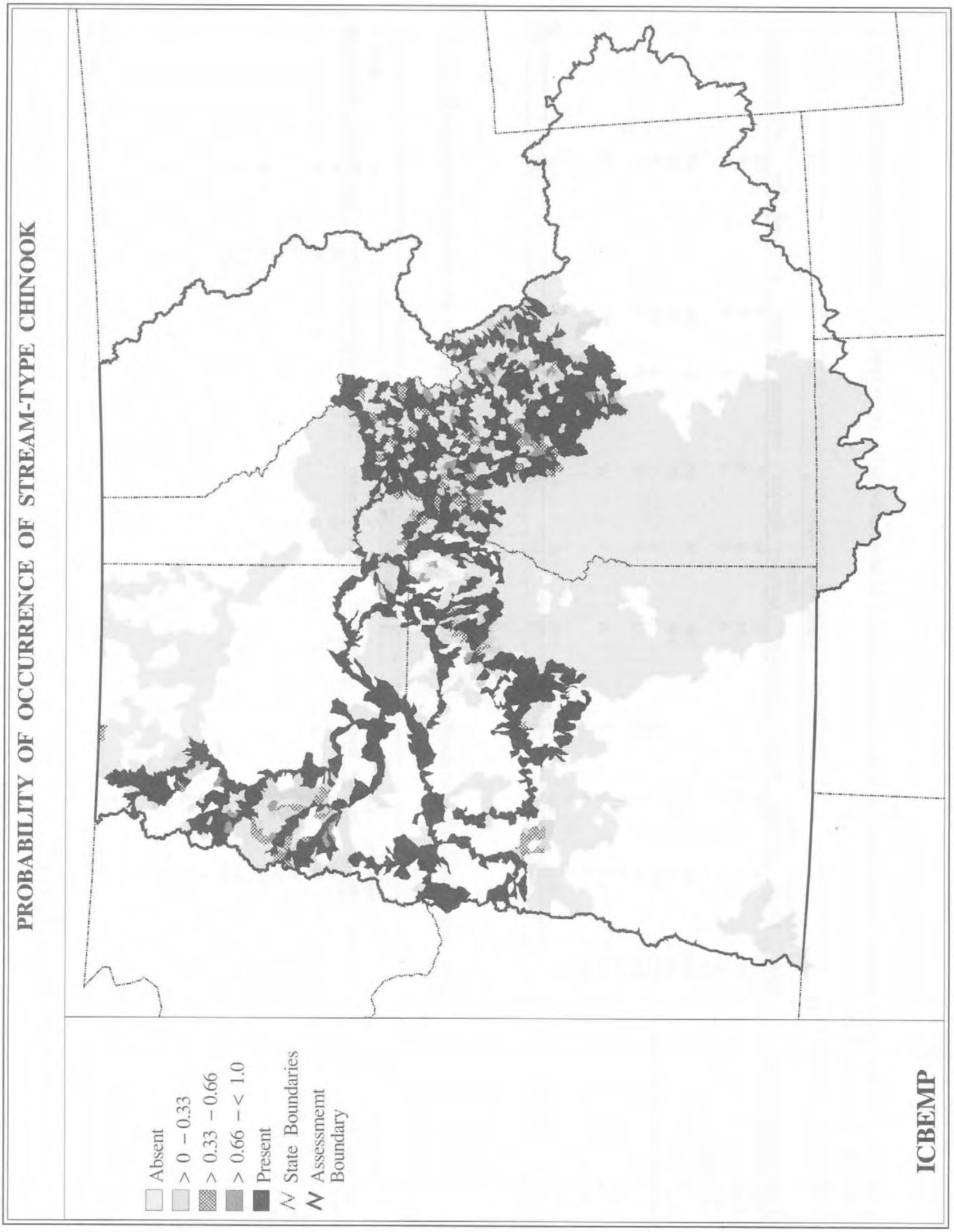
				Status Where Present							
Ecological Reporting Unit	Total	Historical Range	Total Present	Strong*	Depressed*	Status Unknown	Corridor	Introduced	Absent	No Classification	Wild Indigenous
Northern Cascades	340	219	105	0	71	16	18	0	74	40	0
Southern Cascades	141	51	26	0	13	8	5	0	25	0	0
Upper Klamath	175	35	0	0	0	0	0	0	35	0	
Northern Great Basin	506	0	0								
Columbia Plateau	1089	465	133	0	34	19	80	0	285	47	0
Blue Mountains	695	504	199	8	138	7	46	0	273	32	68
Northern Glaciated Mountains	955	172	7	0	1	0	6	0	160	5	0
Lower Clark Fork	415	98	0	0	0	0	0	0	98	0	
Upper Clark Fork	306	0	0								
Owyhee Uplands	956	866	0	0	0	0	0	0	866	0	0
Upper Snake	301	0	0								
Snake Headwaters	387	0	0								
Central Idaho Mountains	1232	1051	405	0	213	91	101	0	489	157	60
Entire Assessment Area	7498	3461	875	8	470	141	256	0	2305	281	128

*Refers to spawning and rearing areas only.

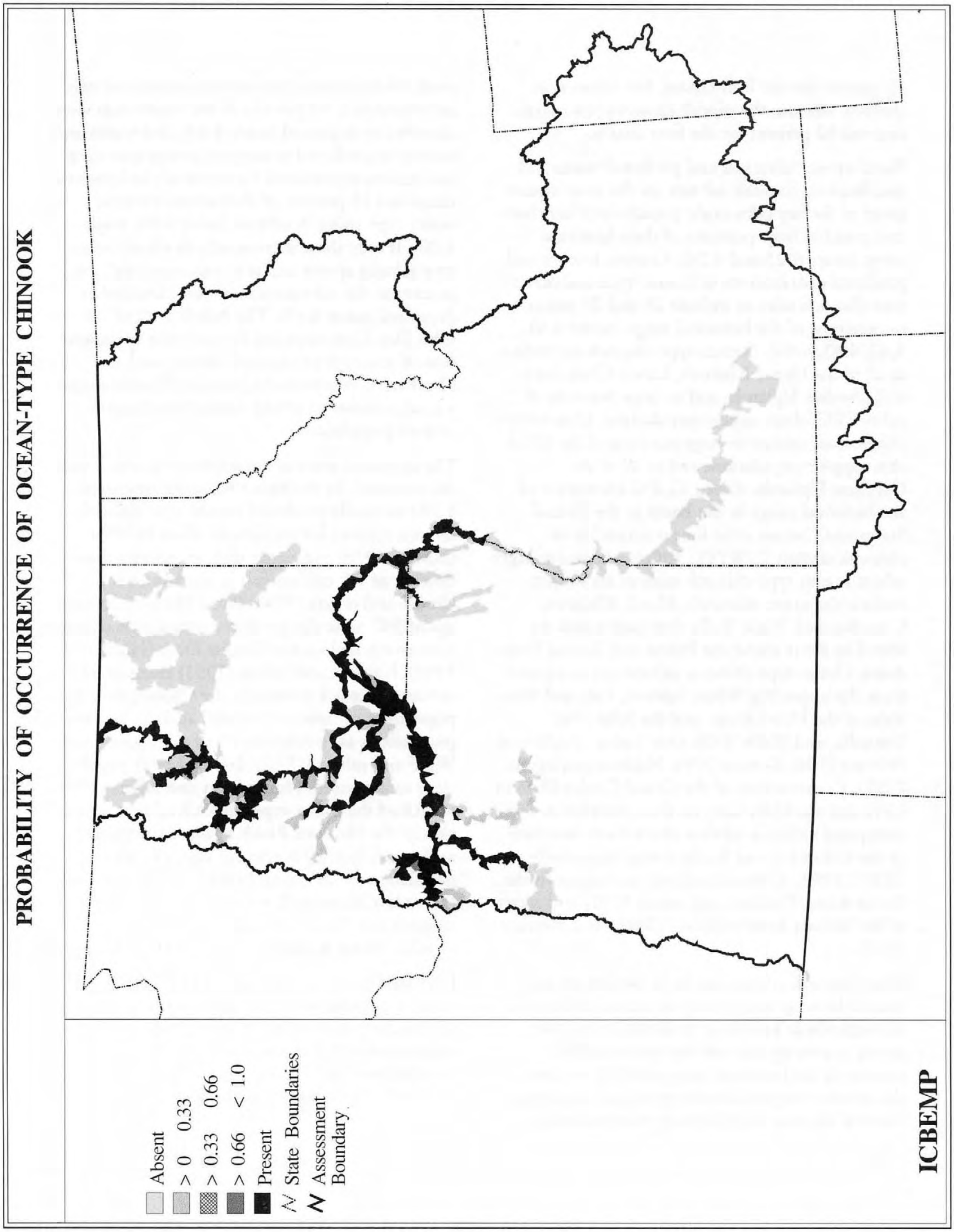
Table 4.42—Summary of the current status and distribution classifications (number of subwatersheds) for ocean-type chinook salmon throughout the Ecological Reporting Units of the Basin.

				Status Where Present							
Ecological Reporting Unit	Total	Historical Range	Total Present	Strong*	Depressed*	Status Unknown	Corridor	Introduced	Absent	No Classification	Wild Indigenous
Northern Cascades	340	79	37	12	12	12	1	0	28	14	0
Southern Cascades	141	31	9	0	0	0	9	0	19	3	0
Upper Klamath	175	0	0								
Northern Great Basin	506	0	0								
Columbia Plateau	1089	147	58	9	11	7	31	0	65	24	11
Blue Mountains	695	99	39	0	25	1	13	0	51	9	16
Northern Glaciated Mountains	955	149	15	0	9	0	6	0	134	0	0
Lower Clark Fork	415	0	0								
Upper Clark Fork	306	0	0								
Owyhee Uplands	956	48	0	0	0	0	0	0	48	0	0
Upper Snake	301	0	0								
Snake Headwaters	387	0	0								
Central Idaho Mountains	1232	0	0								
Entire Assessment Area	7498	553	158	21	57	20	60	0	345	50	27

* = Refers to spawning and rearing areas only



Map 4.23—Current presence, absence, and estimated probability of occurrence within the historical range for stream-type chinook by subwatershed. Current distributions are based on the current-status database. Probabilities of occurrence are based on the classification tree for presence or absence.



Map 4.24—Current presence, absence, and estimated probability of occurrence within the historic range for ocean-type chinook by subwatershed. Current distributions are based on the current-status database. Probabilities of occurrence are based on the classification tree for presence or absence.

83 percent for the four classes. For ocean-type chinook salmon, the overall classification success rate was 92 percent for the four classes.

Based on our observed and predicted status and distribution, chinook salmon are the most endangered of the key salmonids; populations have been extirpated in large portions of their historical range (maps 4.23 and 4.24). Current known and predicted distributions of stream-type and ocean-type chinook salmon include 28 and 29 percent, respectively, of the historical range (tables 4.41, 4.42, 4.43, 4.44). Stream-type chinook are extinct in all of the Upper Klamath, Lower Clark Fork, and Owyhee Uplands; and in large portions of other ERUs that support populations. Ocean-type chinook are extinct in large portions of the ERUs that support populations and in all of the Owyheee Uplands. About 12,452 kilometers of the historical range in the Basin in the United States and Canada is no longer accessible to chinook salmon (NWPPC 1986). Major drainages where stream-type chinook salmon are extinct include the upper Klamath, Hood, Klickitat, Umatilla, and Walla Walla river basins and the Metolius River above the Pelton and Round Butte dams. Ocean-type chinook salmon are extirpated from the upper Big White Salmon, East and West forks of the Hood River, and the John Day, Umatilla, and Walla Walla river basins (Bakke and Felstner 1990; Kostow 1995; Nehlsen and others 1991). Construction of the Grand Coulee Dam in 1941 and the Hells Canyon dam complex in 1967 exterpated chinook salmon above those locations in the Columbia and Snake rivers, respectively (IDFG 1992). Chinook salmon are extinct in the Entiat River (Nehlsen and others 1991) and much of the Yakima River subbasin (Bakke and Felstner 1990).

Most chinook salmon stocks in the remaining accessible range are severely depressed and at risk. Subwatersheds known or predicted to support strong spawning and rearing represented 0.2 percent of the historical range and 0.8 percent of the current range of stream-type chinook salmon (table 4.43; map 4.25). If only those subwatersheds where stream-type salmon spawn and rear are considered, 99 percent of the watersheds were classified as depressed (table 4.43). Subwatersheds known or predicted to support strong spawning and rearing represented 5 percent of the historical range and 15 percent of the current range of ocean-type chinook salmon (table 4.44; map 4.26). If only those subwatersheds where ocean-type salmon spawn and rear are considered, 74 percent of the subwatersheds were classified as depressed (table 4.44). The North Fork of the John Day River contains the only strong population of stream-type chinook salmon and the Northern Cascades and Columbia Plateau support a small number of strong ocean-type chinook salmon populations.

The depressed state of most salmon stocks is well documented. In the Snake River, an estimated 1,882 naturally produced stream-type chinook salmon reached Lower Granite Dam in 1994 (NMFS 1995) compared with an estimated production of 1.5 million fish in the late 1880s (Bevan and others 1994). From 1985-93 an average of 387 naturally produced ocean-type chinook salmon reached Lower Granite Dam (NMFS 1995). Nehlsen and others (1991) categorized summer chinook salmon in the Okanogan as a population of "special concern" and the Methow populations at "moderate risk of extinction" and WDF and others (1993) classified both populations as depressed. Huntington and others (1994) identified the ocean-type chinook salmon population in the Hanford Reach of the Columbia River as the only healthy native fall chinook salmon population in the Basin. NMFS (1994) concluded that mid-Columbia River summer and fall chinook salmon, as a whole, were healthy relative to other chinook salmon stocks in the Basin.

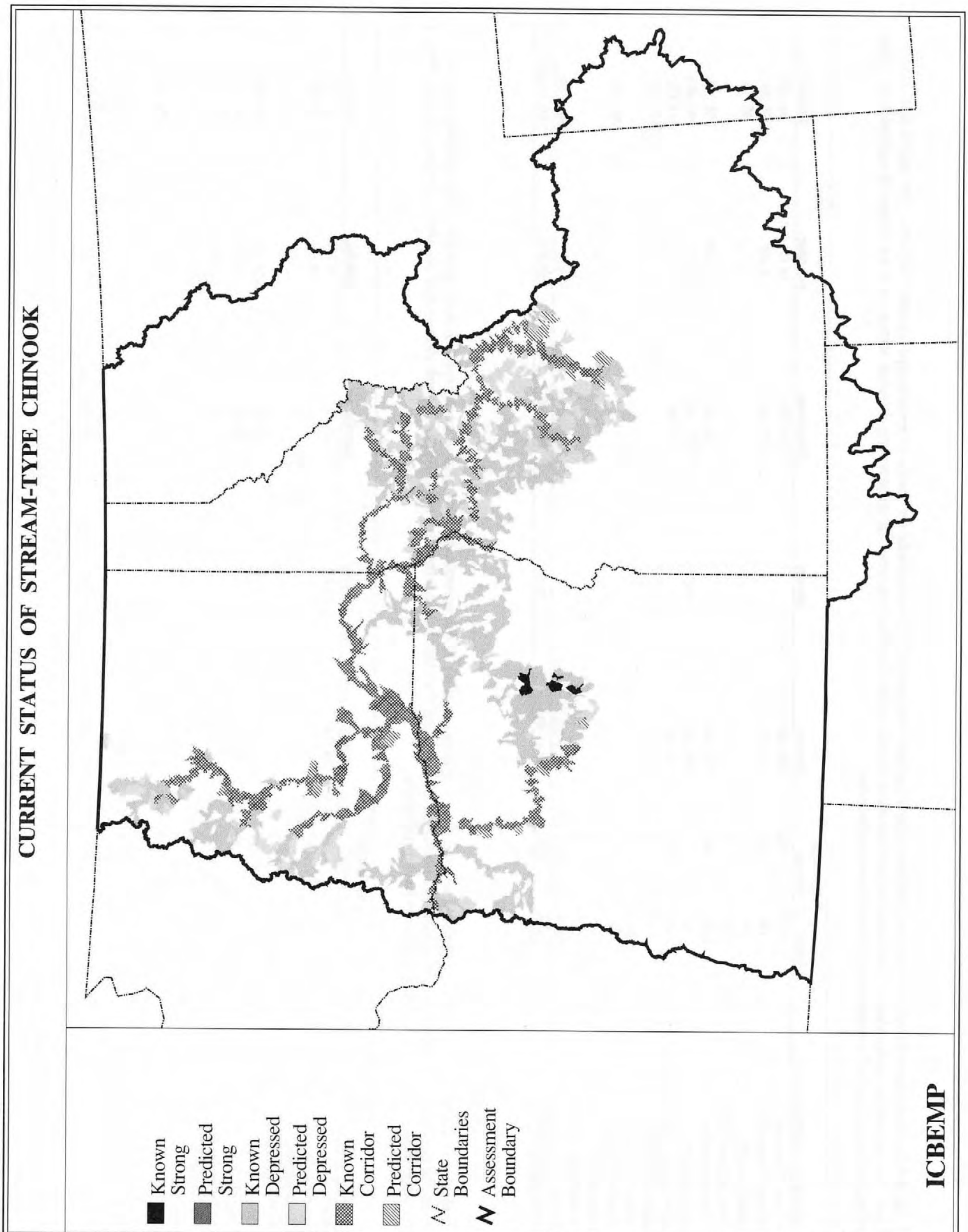
Like steelhead, many remaining chinook salmon populations have been influenced by hatchery-reared fish. Production of wild anadromous fish in the Columbia River basin has declined by about 95 percent from historical levels (Huntington and others 1994).

Table 4.43—Summary of total known and predicted classifications (number of subwatersheds) for status of stream-type chinook salmon. The classifications are from the current-status database. The numbers predicted are based on the classification trees for stream-type chinook salmon and are shown in parentheses. Eight subwatersheds classified as unknown did not have predictions.

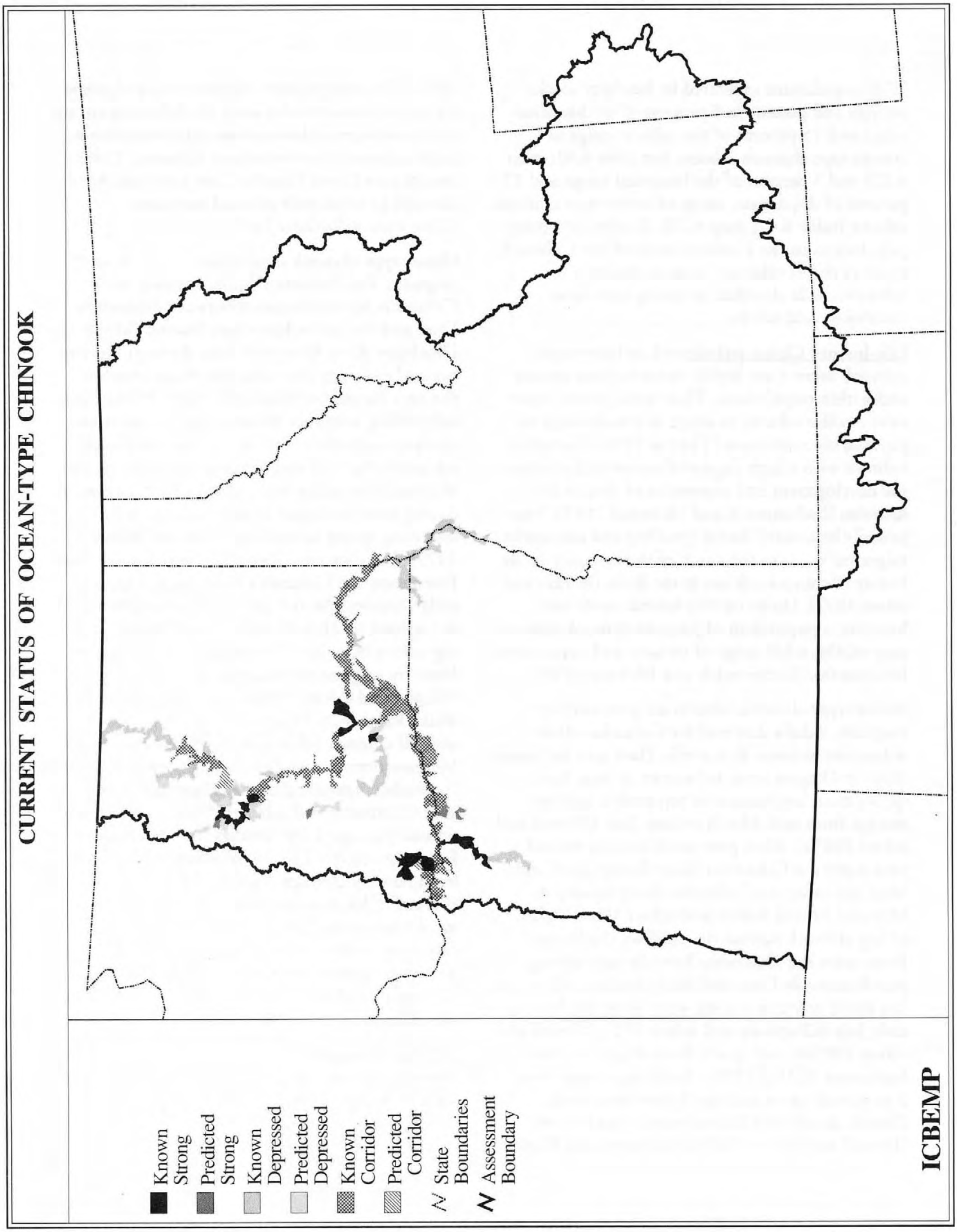
Ecological Reporting Unit	Total	Historical Range	Present	Strong	Depressed	Corridor	Absent
Northern Cascades	340	219	123 (34)		104 (33)	19 (1)	94 (20)
Southern Cascades	141	51	26 (0)		21 (8)	5 (0)	25 (0)
Upper Klamath	175	35					35 (0)
Northern Great Basin	506	0					
Columbia Plateau	1089	465	129 (15)		39 (5)	90 (10)	331 (46)
Blue Mountains	695	504	208 (16)	8 (0)	152 (14)	48 (2)	295 (22)
Northern Glaciated Mountains	955	172	7 (0)		1 (0)	6 (0)	165 (5)
Lower Clark Fork	415	98					98 (0)
Upper Clark Fork	306	0					
Owyhee Uplands	956	866					866 (0)
Upper Snake	301	0					
Snake Headwaters	387	0					
Central Idaho Mountains	1232	1051	465 (151)		351 (138)	114 (13)	586 (97)
Entire Assessment Area	7498	3461	958 (216)	8 (0)	668 (198)	282 (26)	2495 (190)

Table 4.44—Summary of total known and predicted classifications (number of subwatersheds) for status of ocean-type chinook salmon. The classifications are from the current-status database. The numbers predicted are based on the classification trees for ocean-type chinook salmon and are shown in parentheses. Five subwatersheds classified as unknown did not have predictions.

Ecological Reporting Unit	Total	Historical Range	Present	Strong	Depressed	Corridor	Absent
Northern Cascades	340	79	39 (14)	13 (1)	21 (9)	5 (4)	39 (11)
Southern Cascades	141	31	9 (0)			9 (0)	22 (3)
Upper Klamath	175	0					
Northern Great Basin	506	0					
Columbia Plateau	1089	147	57 (6)	12 (3)	14 (3)	31 (0)	86 (21)
Blue Mountains	695	99	41 (3)		27 (2)	14 (1)	58 (7)
Northern Glaciated Mountains	955	149	15 (0)		9 (0)	6 (0)	134 (0)
Lower Clark Fork	415	0					
Upper Clark Fork	306	0					
Owyhee Uplands	956	48					48 (0)
Upper Snake	301	0					
Snake Headwaters	387	0					
Central Idaho Mountains	1232	0					
Entire Assessment Area	7498	553	161 (23)	25 (4)	71 (14)	65 (5)	387 (42)



Map 4.25—Current distribution of spawning and rearing areas classified or predicted as strong or depressed for stream-type chinook by subwatershed. Current distributions are based on the current-status database. Predictions are based on the full classification tree for spawning and rearing areas.



Map 4.26—Current distribution of spawning and rearing areas classified or predicted as strong or depressed for ocean-type chinook by subwatershed. Current distributions are based on the current-status database. Predictions are based on the full classification tree for spawning and rearing areas.

Wild populations unaltered by hatchery stocks are rare and present in 5 percent of the historical range and 15 percent of the current range of stream-type chinook salmon (see table 4.41; map 4.22) and 5 percent of the historical range and 17 percent of the current range of ocean-type chinook salmon (table 4.42; map 4.22). Except for strong populations in the Hanford reach of the Columbia River in the Northern Cascades, the only subwatersheds classified as strong were those sustaining wild stocks.

Life-history Characteristics—Life histories of chinook salmon are highly variable, both among and within populations. Their wide genetic repertoire enables salmon to adapt to a wide range of physical circumstances (Thorpe 1994). Complex habitats with a high degree of connectivity permit the development and expression of diverse life histories (Lichatowich and Mobrand 1995). Two juvenile behavioral forms (yearling and subyearling migrants) account for much of the diversity in life history of chinook salmon in the Basin (Mullan and others 1992). Under healthy habitat conditions, however, a population of juvenile chinook salmon may exhibit a full range of stream- and ocean-type life histories (Lichatowich and Mobrand 1995).

Stream-type chinook salmon are primarily age-1 migrants. Adults destined for Columbia River tributaries between Bonneville Dam and the Snake River in Oregon enter tributaries in mid-April, spawn from late August to September, and fry emerge from mid-March to mid-June (Howell and others 1985a). Most parr smolt in their second year, enter the Columbia River during April and May, and enter the Columbia River estuary in May and June (Lindsay and others 1986). Adult spring chinook salmon destined for the Snake River enter the Columbia River in early spring, pass Bonneville Dam and reach the Snake River by late April, arrive at staging areas from late May to early July (Chapman and others 1991; Howell and others 1985a), and spawn from August to mid-September (IDFG 1992). Adult ages range from 3 to 6, with age-4 and age-5 dominant in the Grande Ronde and Salmon rivers, respectively (Howell and others 1985a; Matthews and Waples 1991). Fry emerge from February to April; some fry rear in natal streams until the following spring, but most migrate downstream into mainstem or larger tributaries to overwinter (Gaumer 1968). Smolts pass Lower Granite Dam from late April through June on their seaward migration (Chapman and others 1991).

Ocean-type chinook salmon are primarily age-0 migrants. Fall chinook salmon destined for Columbia River tributaries between Bonneville Dam and the Snake River pass Sherars Falls on the Deschutes River from mid-June through November and spawn in the mainstem from October through November (Aho and others 1979). Most subyearling migrants return at age-4, and most yearling migrants return at age-5 (Howell and others 1985a). Fry emerge from February to mid-May and rear in the main stem before emigrating during June to August as subyearlings or the following spring as yearlings (Aho and others 1979). Fall chinook salmon destined for the Snake River enter the Columbia River from August to early October; the run past Ice Harbor Dam peaks in September (Howell and others 1985a). Spawning occurs from late October to mid-November. Most fry migrate to the ocean as subyearlings (Waples and others 1991b) and move through the Snake River from March through June. Summer and fall chinook salmon in the Wenatchee and Methow rivers spawn from late September to early November; spawning in the Okanogan River is later (Chapman and others 1994b). Adults mature primarily at age-5. Fry emerge between mid-February and April in the Wenatchee River and from January through April in the Wells spawning channel (Chapman and others 1994b). Fry leave natal areas within weeks after emergence, and continue to rear in the Columbia River. Parr primarily smolt as subyearlings (NMFS 1994), although numbers of yearling migrants vary annually (Chapman and others 1994b).

Habitat Relationships—Habitat requirements of chinook salmon vary by season and life stage, and the fish occupy a diverse range of habitats. Distribution and abundance of chinook salmon may be influenced by cover type and abundance, water temperature, substrate size and quality, channel morphology, and stream size.

Cover is essential for adult chinook salmon prior to spawning, especially early migrants that may remain in tributaries for several months (Reiser and Bjornn 1979). In the John Day River, for example, adult spring chinook salmon stage primarily in deep pools (> 1.5 m.) with cover (Lindsay and others 1986). Fish also use deep water near cover or undercut banks.

Temperature may influence suitability of spawning habitat. The primary evolutionary factor determining time of spawning may be the number of temperature units required for successful incubation of embryos (Heggberget 1988). In the Snake River, suitable spawning areas for fall chinook salmon may have been restricted to mainstem reaches that accumulated at least 960 temperature units from November 15 (spawning) to late April to early May.[19] Fry that emerge later than mid-May may not be large enough to begin their downstream migration as age-0 fish. Spawning in the Hanford Reach may now occur about a month later than it did in the early 1800s, possibly as a result of changes in water temperatures (Chapman and others 1994b). Extremely cold temperature can also influence egg and fry mortality when anchor ice reduces water interchange in gravel (Reiser and Bjornn 1979). Survival and emergence success of chinook salmon embryos is also influenced by fine sediment and flow (Chapman 1988). Other factors that reduce egg-to-fry survival include: redd disturbance and excavation, bottom scour, and microbial infestation (Beauchamp and others 1983; Healey 1991).

After emergence, fry concentrate in shallow, slow water near stream margins with cover (Hillman and others 1989a; 1989b). As fry grow they occupy deeper pools with submerged cover during the day (Reiser and Bjornn 1979) and shallower inshore habitat at night. Areas of shade near stream margins are the preferred habitat of juvenile salmonids including chinook salmon (Chapman 1966; Everest and Chapman 1972). Suspended sediment may effect juvenile fish by damaging gills, reduced feeding, avoidance of sedimented areas, reduced reactive distance, suppressed production, and increased mortality (Castillo and others 1994; Hicks and others 1991; Reiser and Bjornn 1979). Sediment deposition can also reduce habitat capacity (Reiser and Bjornn 1979).

Key habitat factors for juvenile rearing include streamflow, pool morphology, cover and water temperature (Steward and Bjornn 1990). Chinook salmon parr tend to select specific rearing habitats that segregate them, both temporally and spatially, from other native salmonids (Everest and Chapman 1972; Nilsson 1963). They also tend to be most abundant in low gradient, meandering stream channels (Scully and others 1990).[20] Parr are sensitive to alteration of water temperatures, particularly increases during the summer. Bell (1973) reported a preferred temperature range for juvenile chinook salmon of 7.3° to 14.6° C and Brett (1952) an upper lethal temperature of 25.1° C. Water temperatures in reaches of the John Day, upper Grande Ronde, and other basins in eastern Oregon commonly exceed the preferred ranges and often exceed lethal temperatures (McIntosh and others 1994a; ODFW and others 1990).

Juvenile chinook salmon often occupy different habitats in winter than in summer with two overwintering strategies, migration or concealment. Juveniles select areas of low water velocity and enter concealment cover beneath cobble or rubble substrate or beneath undercut banks (Edmundson and others 1968; Hillman and others 1987).

[19]Personal communication. 1995. B. Connor, U.S. Fish and Wildlife Service, Orofino, ID.

[20]Personal communication. 1995. K. Overton and R. Thurow, U.S. Forest Service, Intermountain Research Station, Boise, Idaho. Personal communication of unpublished data.

In our classification analysis, variables including number of dams, *hucorder*, precipitation, management clusters, and mean temperature were useful in discriminating watersheds supporting stream-type chinook salmon (see table 4.22, appendix 4E). Spawning and rearing areas were more likely to be found in mid-size streams, above fewer mainstem dams, in wetter, cooler landscapes, and on Forest Service-administered lands. Larger mainstem streams at lower elevations were more likely to represent migratory corridors. A notable exception was the distribution of ocean-type chinook where remaining spawning sites were associated with mainstem segments of the Columbia and Snake rivers.

Key Factors—A number of factors including habitat degradation, disease, predation, harvest, artificial propagation (Bevan and others 1994; Chapman and others 1991), and fluctuations in ecosystem productivity (Lichatowich and Mobrand 1995) have influenced the status of chinook salmon in the Basin.

Habitat degradation has influenced the status of chinook salmon. The overall pattern of decline of chinook salmon suggests the species is sensitive to habitat degradation throughout their entire range (for example, Nehlsen and others 1991). Livestock grazing, timber harvest, and irrigation diversions effect habitat (Beschta and others 1991; Henjum and others 1994). Reduced stream habitat complexity has been one of the most pervasive cumulative effects of forest management practices and may have substantially altered fish communities (Bisson and others 1992). Forest management practices, including timber harvest activities, have reduced salmon habitat quantity, reduced habitat complexity, increased sedimentation and eliminated sources of woody debris needed for healthy salmon habitat (Henjum and others 1994; Lichatowich and Mobrand 1995).

The integrity of salmon ecosystems is linked to the condition of riparian and upland areas and their influence on water temperature (Marcot and others 1994; Theurer and others 1985), sediment (Castillo and others 1994), the aquatic food base (Li and others 1994, Tait and others 1994), and pools (McIntosh and others 1994b). Land-use histories in eastern Oregon and Washington illustrate habitat degradation in the Columbia River basin (McIntosh and others 1994b; Wissmar and others 1994b). A comparison of managed (roaded) and unmanaged (wilderness or roadless) basins in eastern Oregon and Washington illustrated that managed basins had a significantly lower frequency of coarse woody debris and a significant decrease in large pools (> 20 m^2 surface area and > 0.9 m deep) compared with unmanaged basins (McIntosh and others 1994b). In the Snake River basin, more than 80 percent of the salmon production occurs on Forest Service and BLM lands, and most salmon spawning occurs in grazed areas (Bevan and others 1994). In Idaho, more than 80 percent of the riparian areas managed by the BLM are in degraded condition (Chapman and others 1991). In portions of the Snake River basin still accessible to salmon, management practices on Forest Service lands have been largely responsible for reducing the suitability of about 3,100 kilometers of stream (Haugen 1991 as cited in Bevan and others 1994). Chinook salmon habitat in upper Columbia River tributaries, especially the upper Methow and Wenatchee rivers, remains minimally altered by development compared with the main stem of the Columbia River and its other tributaries (Chapman and others 1994b; Mullan and others 1992). Restoring connectivity in reaches of lower subbasins should be a management priority (Lichatowich and Mobrand 1995).

Hydropower development is a second factor influencing chinook salmon status. Construction and operation of mainstem dams on the Columbia and Snake rivers are considered the major cause of decline of anadromous fish (CBFWA 1990). Similar to steelhead, adult chinook salmon are delayed during upstream migrations and smolts may be killed by turbines; become disoriented or injured, making them more susceptible to predation; or become delayed in the large impoundments behind dams (Bevan and others 1994; Chapman and others 1994b; IDFG and others 1990). Development and operation of hydropower facilities in the Basin have reduced salmon and steelhead production by about eight million fish: four million from blocked access to habitat above Chief Joseph and Hells Canyon dams, and four million from ongoing passage losses at other facilities (NWPPC 1986). Passage losses are cumulative depending on the number of dams; chinook salmon in the Basin must pass one to nine dams. Losses of mid- and upper-Columbia River ocean-type chinook salmon were estimated to be about 5 percent per dam for adults and 18 to 23 percent per dam for juveniles (Chapman and others 1994b).

Hatcheries are a third factor influencing chinook salmon status. Hatcheries have been used extensively in attempts to compensate for losses, primarily from hydroelectric projects, of natural production (Howell and others 1985a; Matthews and Waples 1991). Salmon of hatchery origin comprise about 80 percent of the Columbia river salmon run (Lichatowich and Mobrand 1995). Problems associated with hatchery production include genetic introgression from non-native stocks and loss of fitness, reduced wild spawning escapement from the collection of broodstock, ecological interactions between hatchery and wild fish, mixed hatchery and wild stock fisheries, and transmission of diseases (Bevan and others 1994). Most (107/121) of the healthy anadromous salmonid stocks identified by Huntington and others (1994) have either had no fish culture activities in the home watershed or have been exposed to little risk from stock transfers or interaction with hatchery fish. Adult chinook salmon collection and egg-taking operations may also be detrimental to individual populations (Chapman and others 1994b). Introductions of large numbers of hatchery-reared parr may induce early migration of wild fish (Hillman and Mullan 1989).

Harvest is a fourth factor influencing chinook salmon status. Harvest has contributed to the decline of spring and summer chinook salmon in the Basin since the late 1800s (Fulton 1970) and to the decline of fall chinook salmon after 1920 (Lichatowich and Mobrand 1995). Historical ocean and river harvest rates exceeded 80 percent (Ricker 1959 as cited in Bevan and others 1994). Thompson (1951) reported that as a result of excessive harvest, by 1919 the characteristics of the Columbia River chinook salmon run had changed. Formerly large portions of the run were reduced, thus making smaller portions of the run more important to the fishery. The once nearly continuous run of salmon became segregated into more discrete groups. Lichatowich and Mobrand (1995) divided the fishery into four phases: initial development (1866 to 1888), sustained production (1889 to 1922), resource decline (1923 to 1958), and maintenance at a depressed level (post 1958). Declining runs of wild chinook salmon are still harvested in mixed-stock commercial and tribal fisheries. Sport harvest of wild chinook salmon has been curtailed in most states. In Idaho, for example, sport harvest of chinook salmon has been closed since 1975, except for periodic terminal fisheries on hatchery stocks.

Predation and competition are biotic factors influencing chinook salmon status. Predation is one of the major causes of mortality to fry and fingerling chinook salmon (Healey 1991). Introduced species may prey upon and compete with native fishes. Many middle and lower reaches of the Columbia River are dominated by introduced species (Li and others 1987). Northern squawfish, a native predator, have become well adapted to the habitat created by dams (Beamesderfer and Rieman 1991). It has been estimated that 15 to 20 million juvenile salmonids in the Snake and lower Columbia rivers annually succumb to northern squawfish predation (Collis and others 1995).

<u>Summary</u>—Prior to overfishing and habitat alterations, migrating chinook salmon in the Columbia River formed a continuum from March to October with the largest part of the run likely consist-

ing of summer chinook salmon (Thompson 1951). At present, the combination of low abundance, decreasing time trends in abundance, widely dispersed spawning populations, fragmented habitats, degraded freshwater and estuarine habitats, high risks of genetic introgression in most stocks, low frequency of wild indigenous stocks, high rates of non-native fish introductions, continued harvest in mixed-stock fisheries, and high-risks of smolt and adult passage mortality result in a high risk of extinction for individual chinook salmon stocks.

Chinook salmon represent the most imperiled of the key salmonids we considered. Both forms of chinook salmon are extinct in more than 70 percent of the historical range. The distribution of stream-type chinook appears to be widespread throughout the remaining accessible range, but most populations are severely depressed and heavily influenced by hatchery supplementation. The only remaining strong populations are restricted to relatively small areas of the John Day River basin within the Blue Mountains. Within the central Idaho Mountains recent runs of stream-type chinook salmon have been critically low and most populations are believed to be on the brink of extinction. Ocean-type chinook salmon are found in a more restricted range linked principally to mainstem rivers and larger tributary systems. Populations associated with the Snake River Basin in Idaho are also considered on the verge of extinction. The remaining distribution of spawning and rearing includes very few subwatersheds in each occupied ERU and many areas of contiguous occupied habitat are small and disjunct.

Restoration of habitats and habitat connectivity in both mainstem and subbasins will be necessary for salmon stocks to persist. The declines of chinook salmon stocks in the Basin in recent years can be attributed primarily to mainstem dams, however, until passage problems are resolved, the resiliency and persistence of remaining chinook salmon stocks will be largely dependent on the quality and diversity of remaining stream habitats. All remaining populations and habitats for both forms of chinook salmon within the Central Idaho Mountains, Blue Mountains, Northern Cascades, Columbia Plateau, Northern Glaciated Mountains, and Southern Cascades are critical to the persistence of this species within the Basin.

Introduced Salmonids

Eight species of non-native salmonids have been introduced and maintain self-sustaining populations in the Basin. They include lake whitefish; Atlantic salmon; Arctic grayling; and brook, brown, golden, and lake trout; and Sunapee char (see table 4.16). Several species native to the Basin have been introduced outside their natural range. These include Lahontan, Yellowstone, and westslope cutthroat trout; redband trout and several other forms of rainbow trout; chinook and coho salmon; kokanee; and steelhead. All were introduced to create or expand fishing opportunities and are managed and protected as game fishes by state agencies. Established populations of these fishes now support many important fisheries throughout the region.

Current Distribution and Status

Arctic grayling, golden trout and Sunapee char, have relatively narrow distributions in the Basin. Grayling were reported in 1.3 percent of the subwatersheds, golden trout in 0.3 percent, and Sunapee char in only one subwatershed (see table 4.16). These species were primarily introduced in high elevation and formerly fishless alpine lakes to provide unique sport fishing opportunities.

In contrast to the many examples of introduced species contributing to declines in native fishes and other aquatic organisms, introduced Sunapee char may represent one of the last remaining genetic refuges for the species. The Sunapee char, a landlocked Arctic char endemic to Sunapee Lake, New Hampshire, became extinct in New Hampshire in the mid-1900s as a result of competition with introduced species (Kircheis and others 1995). In 1925, prior to extinction in Sunapee Lake, Sunapee char were introduced into two high elevation and fishless lakes in Idaho's Sawtooth Mountain Range (Kircheis and others 1995) and established a reproducing population. The American Fisheries Society listed the Sunapee char and blueback trout (a phenotypically distinct form endemic to Maine) as a Threatened species (Williams and others 1989). Because the species is non-native, Idaho does not list it as a species of special concern. However, protection of the Sunapee char and its habitat is suggested as a priority (IDFG 1991). Sunapee char differed from other North American lacustrine char in their unique coloration, large size, and use of a mid-lake spawning shoal (Kircheis and others 1995). The fish stocked in Idaho were not "rediscovered" until 1978 (IDFG 1991). Kircheis and others (1995) recently examined the genetic identity of 17 Sunapee char collected from one Idaho lake. The samples displayed four haplotypes, three of them uniquely divergent from other landlocked arctic char and the other similar to a blueback char haplotype. The authors reported that the divergent portion of the Idaho char population represented a distinct, native population formerly in Sunapee Lake. Despite the unique haplotype, Kircheis and others (1995) recommended against using Idaho Arctic char to restock New Hampshire waters because brook trout were stocked in the Idaho lakes in the 1940s. The authors, however, presented no data assessing the degree of introgression with brook trout.

Lake whitefish, kokanee, Gerrard and Kamloops rainbow trout, Atlantic salmon, and lake trout were introduced primarily in lakes and reservoirs that also held native salmonids. Lake whitefish supported an early commercial fishery around 1900 on Pend Oreille Lake (Simpson and Wallace 1978), but are a less important sport fish today. Introduced lacustrine rainbow and lake trout currently support trophy fisheries in larger lakes and have been particularly important where kokanee provide a forage base (Wydoski and Bennett 1981). Kokanee have supported some of the most heavily used fisheries in the region (Rieman and Maiolie 1995) and have been introduced into virtually every lake and reservoir deep enough to sustain a population.

Brook, introduced rainbow, and brown trout, are among the most widely distributed species in the region. All are found in both lowland and alpine lakes but have become much more widely established in streams. Introduced rainbow trout were found in 78 percent of the watersheds in our database, while brook trout were recorded in about 50 percent (see map 4.7). They are the most widely distributed species in the Basin. Brown trout were found in 23 percent of the watersheds (see map 4.7). All three provide fishing opportunities, but rainbow and brown trout have become particularly popular, supporting nationally and internationally recognized sport fisheries. They are well adapted to habitats ranging from small streams and to large rivers, often preying on native and introduced fishes. They are prized by anglers because of their ability to achieve relatively large size in these more productive environments.

Expansion of ranges through introduction of the non-native and native salmonids has been common in alpine lakes, most of which were historically fishless (Bahls 1992). Many of these systems do not support self-sustaining populations, but the introductions have resulted in the establishment of new populations downstream. Cutthroat, rainbow and brook trout appear to have been particularly successful through this means of expansion.

Like the native salmonids, the life histories of introduced forms are highly variable and include adfluvial and fluvial migratory forms and nonmigratory resident forms. In some cases (such as kokanee in Pend Oreille Lake, Idaho) the apparent variation in life history is the result of stocking several evolutionarily distinct forms that have both adapted to the new system (Rieman and Bowler 1980). We have little knowledge of whether the expression of varied life histories among the introduced species represents adaptation to new environments through recent selection or reflects the expression of a plastic or broad set of environmental tolerances.

The broad distribution of many of the introduced species suggests that they are more tolerant of habitat disturbance than many of the native forms. Brown trout for example, occupy a wide range of habitats ranging from alpine lakes to small, low-elevation reservoirs and large low-elevation rivers, to small springs. Brown trout tolerate some degradation of habitat that would be lethal to native salmonids. For short periods, brown trout tolerate water temperatures exceeding 27° C, oxygen levels near 5 milligrams/liter, and high levels of suspended sediment (Brynildson and others 1963). To sustain wild, self-sustaining populations, however, brown trout still require abundant and high-quality habitats similar in many aspects to those required by native salmonids. Brown trout spawn in sorted gravels, often in areas used by native salmonids (Scott and Crossman 1973). Newly emerged fry seek shallow, low-velocitiy areas along stream margins. Cover is one of the most important habitat features controlling the number of rearing brown trout (Lorz 1974).

Key Factors

Habitat degradation has influenced the status of introduced salmonids. Although the introduced salmonids may be more tolerant, like native salmonids they have declined as a result of habitat degradation. Activities such as dam construction, water diversion, grazing, mineral extraction, road construction, and residential development have degraded habitats that affect these species. In the Deschutes River, for example, brown trout were effected by loss of migrating fish in irrigation canals, reductions of riparian areas by grazing and housing developments, and siltation of stream substrate (Lorz 1974).

Angler harvest is a second factor that has an important influence on the status of most of the introduced salmonids. Special angling restrictions designed to limit mortality are common throughout the region. Numerous studies have demonstrated the influence of angling on population size structures and abundance and clearly show a negative response of populations to increasing or unrestricted harvest.

Summary

The introduced salmonids are an important and permanent part of many aquatic ecosystems, often desired by anglers. The introduction and expansion of non-native salmonids have played a definite role in the decline of native species. Displacement likely occurs through competition, predation, and hybridization (Fausch 1988; Leary and others 1993; Rieman and McIntyre 1993). The expansion of non-native fishes represents a loss of biological diversity and integrity for many systems, but ironically, a desirable social and economic component in some cases. In most cases, elimination of the non-native forms is infeasible and may be socially unacceptable. Conservation management of these species will be an important consideration of aquatic ecosystem management. The widespread establishment of some introduced salmonids suggests that they may be less sensitive to habitat disruption. Any attempts to conserve or strengthen habitats for the native species will likely either benefit or have no influence on most introduced salmonids.

Community Integrity and Conservation Emphasis Areas

The species assemblage and key salmonid information collectively highlight areas that retain their historical species diversity and ecological structure and point to other areas that are severely altered. In this section, we attempt to further delineate areas of high community integrity and diversity. Such areas could be useful in defining conservation emphasis areas necessary for rebuilding and maintaining healthy and productive aquatic systems. We also consider areas that support important fisheries sustained by wild salmonids of introduced origin. Although the latter areas may be of less ecological significance than native salmonid assemblages, they represent high social and economic values.

Four criteria were used in our analysis, as follows: 1) population strongholds for key salmonids, 2) native species richness, 3) diversity and evenness, and 4) uniqueness. The first three were combined to create a numerical measure of integrity that is applicable at the watershed level. We also listed important wild trout waters as a final criterion available for further analysis. However, we did not map the wild trout waters or use them to rank integrity.

Population Strongholds

Subwatersheds that support strong populations of any of the key salmonids likely represent a fortuitous balance of habitat quality, climatic and geologic constraint, and geographic location which effectively minimize cumulative threats to these species. The classification of strong included only subwatersheds that supported spawning and rearing. Because full life-history expression was part of the criteria for defining strong populations, however, the occurrence of strong populations may also indicate the relative integrity of the larger system of watersheds. The most productive, abundant, and diverse populations are likely to be most resistant and resilient to environmental disturbance and most likely to survive catastrophic disturbance. Thus, they are more likely to serve as sources for supporting weak or at risk populations, refounding locally extinct populations, or refounding habitats made available through restoration. Strong populations are potentially critical for short-term persistence and long-term recovery. To identify important strongholds, we summarized the number of strong species known or predicted to occur within subwatersheds throughout the Basin.

We estimate that less than 0.01 percent of the subwatersheds support three strong salmonids, 3 percent support two, and about 20 percent support one. The largest patches of contiguous or clustered habitats supporting strong populations are associated with the major river subbasins found in the Central Idaho Mountains, Blue Mountains, Northern Cascades, and the Snake Headwaters (map 4.27). Smaller patches are found in the Upper Clark Fork and extreme eastern fringe of the Northern Glaciated Mountains. Strong populations of chinook salmon and steelhead were rare or absent, even in relatively undisturbed habitats in the Central Idaho Mountains. This suggests that factors outside the Basin strongly influence the status of anadromous species. Most of the subwatersheds supporting strong populations were found on Forest Service administered lands (75%) and a substantial number (29%) are located within designated Wilderness areas or National Parks. Strong populations occurred more frequently in areas with lower road densities (fig. 4.20).

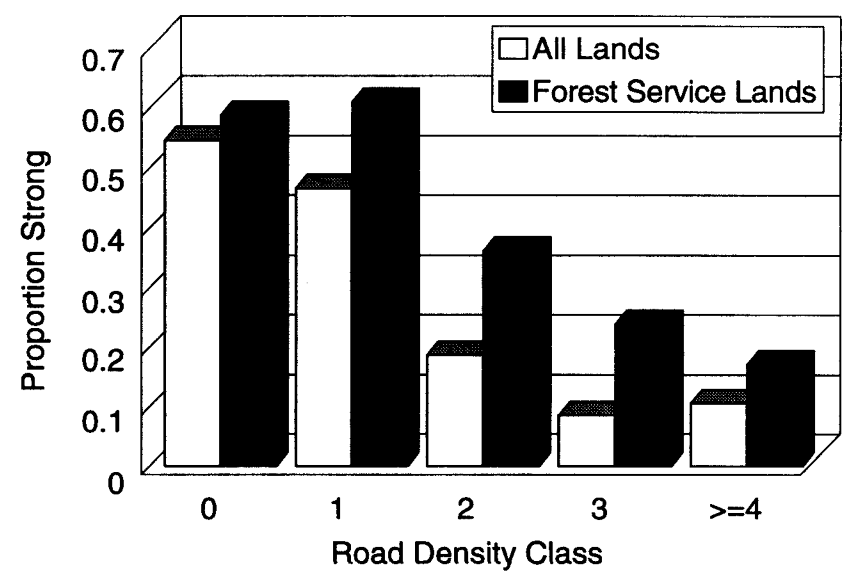
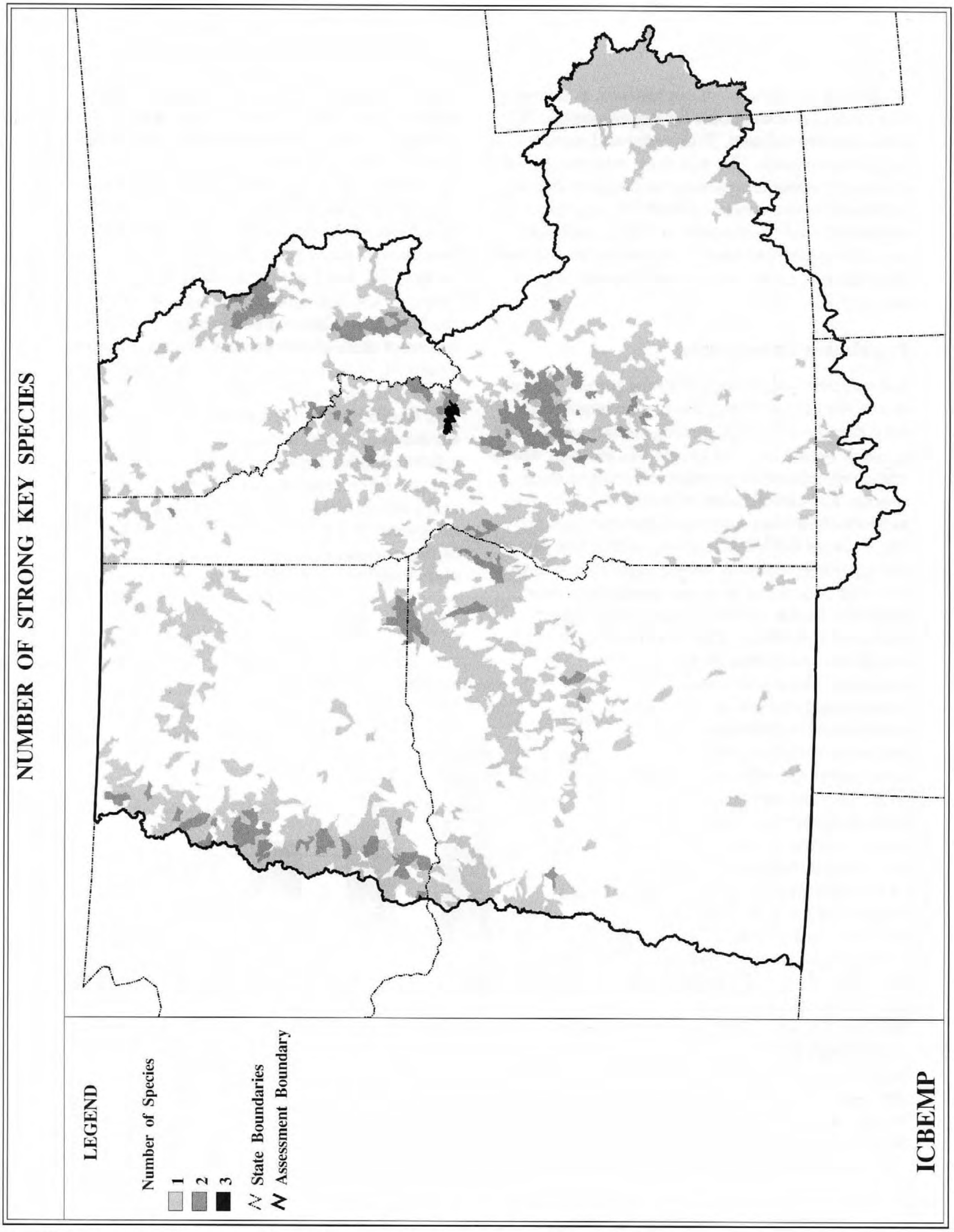


Figure 4.20. Proportion of subwatersheds supporting strong populations of key salmonids by road density class and land ownership.



Map 4.27—Number of key salmonids classified or predicted as strong by subwatershed.

Native Species Richness

The number of native species present in a watershed is an important element of diversity, and reflects heterogeneity in the physical environment. A high degree of species overlap might reflect strong habitat diversity. Even within a fairly narrow group like salmonids, each species relies on different habitats and environments, with variable and wide-ranging life-history patterns. The co-occurrence of several salmonids suggests suitable habitats exist over relatively large landscapes, not just those directly tied to the local subwatershed. High richness may also indicate critical common areas that serve as corridors, wintering areas, or seasonal refuges for the varied life histories in the assemblage. The loss of such areas could portend a loss of richness on both local and regional scales.

To characterize species richness, we used two measures, the number of key salmonids known or predicted to occur within each subwatershed, and the total number of native species known to occur in each watershed. We recorded from 0 to 28 native species within watersheds throughout the basin. The largest number of native species was found in the large river corridors, particularly the lower and middle Columbia and the lower Snake rivers. Moderate native-species numbers were observed in lower-order watersheds associated with the Blue Mountains, along the transition between the North Cascades and Columbia Plateau, and throughout much of the Basin within Montana (map 4.28).

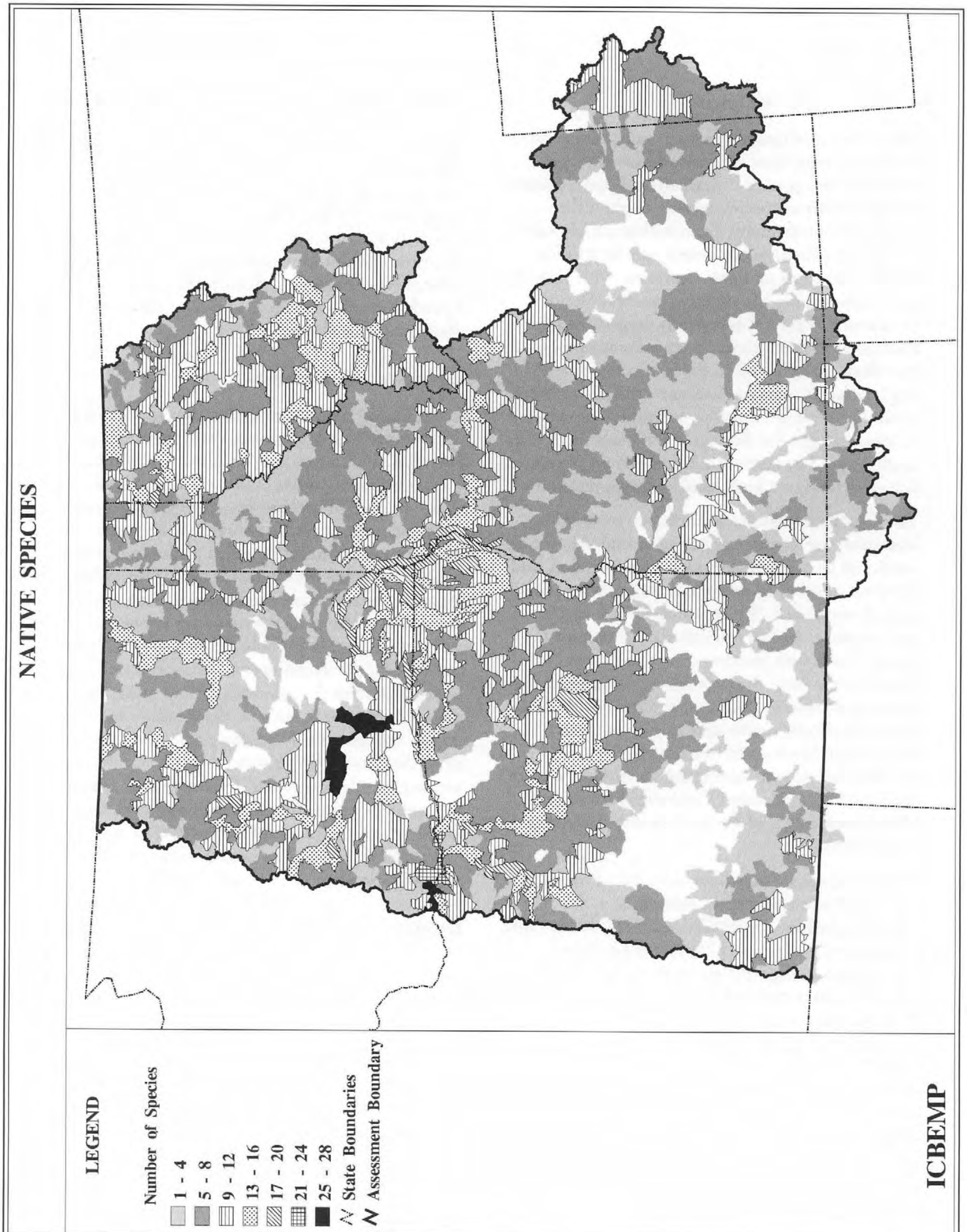
Historically there may have been broad overlap in the distribution of the key salmonids at the subwatershed level. The Central Idaho Mountains, Northern Cascades, and Northern Glaciated Mountains, along with the river corridors connecting all of the watersheds accessible to anadromous fish, probably supported multiple species. We estimate that 97 percent of the subwatersheds could have supported at least one key salmonid historically and 74 percent could have supported two or more, about 2 percent could have supported six (map 4.29). We estimate that about 74 percent of the subwatersheds currently support at least one, and less than 38 percent support two or more, about 0.5 percent support 6 (map 4.30). The largest remaining regions of high species overlap are associated with the Central Idaho Mountains, the Blue Mountains, the Northern Cascades and, notably, their connecting river corridors. Two or more species are still found in subwatersheds scattered throughout Montana and a patchwork of watersheds and river corridors throughout the Basin.

Diversity and Evenness

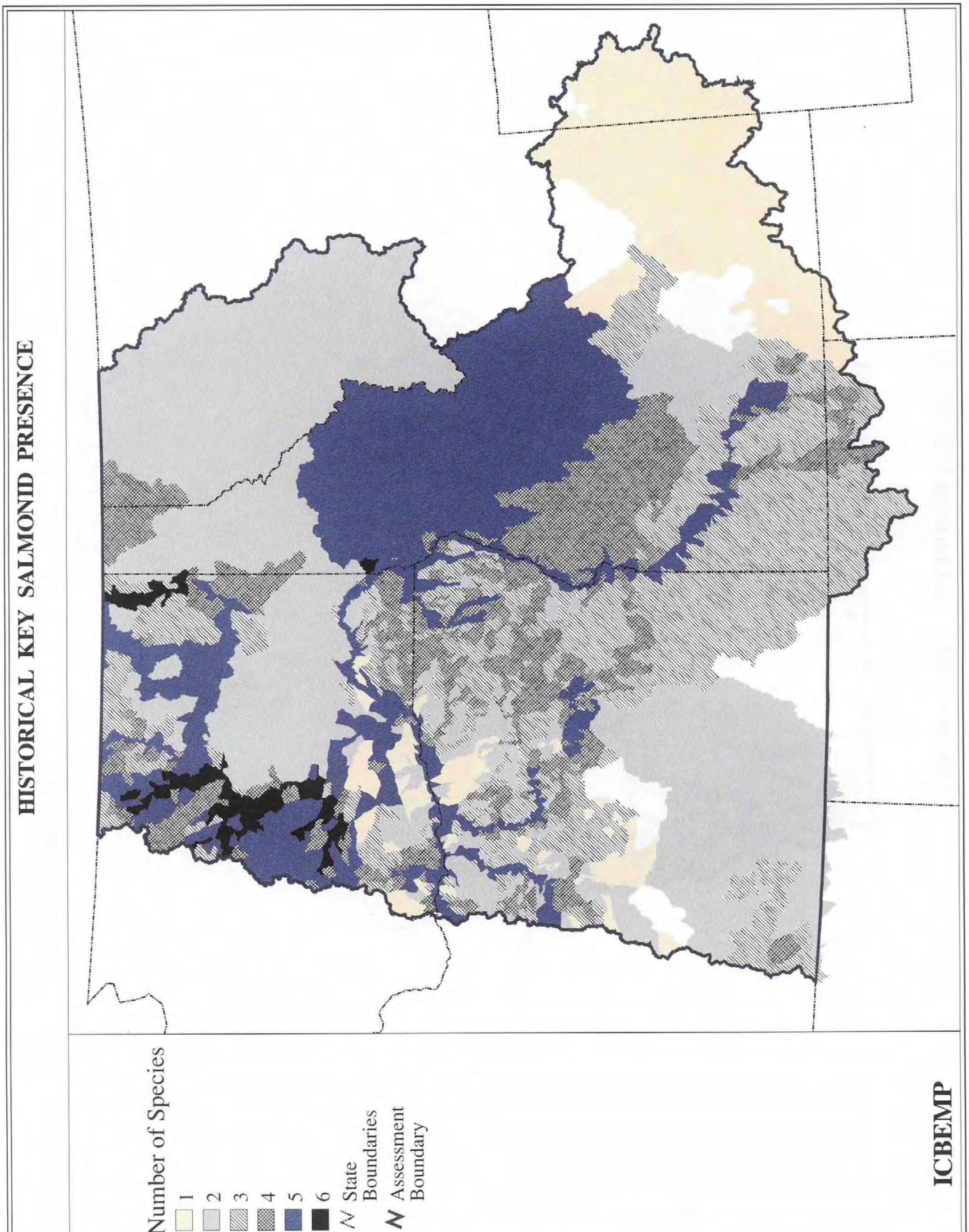
In the ecological literature, diversity refers to both the number of species present and their relative abundance. Thus, an area with many abundant species is more "diverse" than an area with an equal number of species, few of which are abundant and most of which are rare. As its name suggests, evenness simply refers to the degree to which species are evenly distributed. Our indices of native diversity and evenness were described earlier. Since the diversity measures apply to an assemblage class as a whole rather than individual watersheds, the diversity and evenness maps would be equivalent to the assemblage map presented in map 4.8. Note that because the diversity measures do not rely exclusively on the presence or absence of species from a single watershed, they more accurately reflect *potential* diversity rather than realized diversity within a given watershed. Both diversity and evenness are instructive, in that they reflect the capacity of an area to accommodate multiple species.

Uniqueness

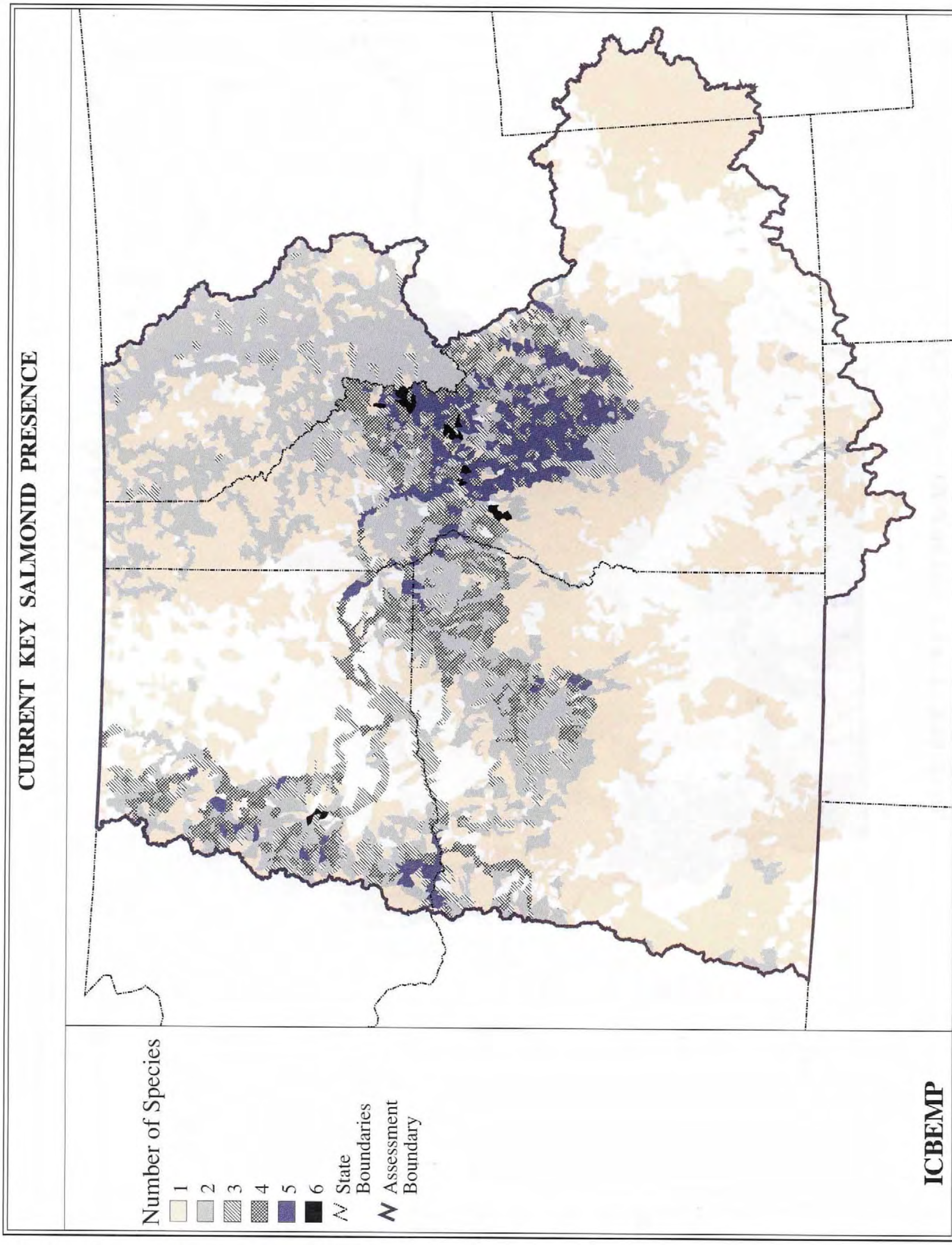
We defined uniqueness by four measures: 1) watersheds representing fringes of a species range; 2) watersheds supporting unusual genetic integrity; 3) watersheds supporting narrowly distributed endemic species; and 4) watersheds with species formally listed under the federal Endangered Species Act.



Map 4.28—Number of native species reported by watershed.



Map 4.29—Historical number of key salmonids potentially present by subwatershed.



Map 4.30—Current number of key salmonids (classified or predicted as present) by subwatershed.

Fringe Distributions—Because extreme or fringe environments may support a disproportionately large part of the genetic diversity within a species, populations persisting in these areas may represent distinct adaptations and an important source of genetic variability for that species (Lesica and Allendorf 1995; Scudder 1989). Populations that maintain native gene complexes and the widest possible diversity likely offer the best resources for refounding extinct populations in similar environments. Populations that historically were distributed over broad geographic areas have likely evolved under relatively distinct environments with little gene flow across the species range (Lesica and Allendorf 1995). Conservation of the genetic diversity in these species then implies sustaining populations over the broad geographic area (Allendorf and Leary 1988; Leary and others 1993).

Because the ERUs represent distinct biophysical environments, the potential for adaptive divergence for any species distributed across these units may be strong. As a first step to identify fringe areas, we summarized key salmonid presence in spawning and rearing areas by ERU (fig. 4.21) and noted regions where a particular species was weakly distributed, but still within its historical range. Each species was generally widely distributed, that is, found in more than 100 spawning and rearing subwatersheds within an ERU (often with several strong populations) near the core of their respective ranges. We assumed that a species with known or predicted presence in fewer than 30 subwatersheds within an ERU indicated a weak or fringe distribution. We then considered whether the limited distributions identified within an ERU were isolated from other parts of the range.

Bull trout are found in a limited number of subwatersheds within the Southern Cascades, the Upper Klamath, the Owyhee Uplands, and the Walla Walla and Umatilla subbasins within the Columbia Plateau. Although the distribution of bull trout in several ERUs is weak, we recognized only those subwatersheds within the Owyhee Uplands and Upper Klamath as distinctly isolated fringe distributions. Bull trout within the Southern Cascades probably have not been strongly isolated from similar environments in the lower Columbia River tributaries of the Northern Cascades. Bull trout in the Columbia Plateau are limited to streams of the Blue Mountains and were therefore considered an extension of the populations found in the Blue Mountains.

We placed the fringe of the range for westslope cutthroat trout within the Blue Mountains (map 4.31). The Columbia Plateau also met our criteria for fringe distribution (that is, less than 30 subwatersheds). The subwatersheds included in that ERU, however, are in the lower Clearwater River and are really part of a much larger distribution of westslope cutthroat trout in the Lower Clark Fork. For that reason, we did not include the Columbia Plateau as part of the fringe for westslope cutthroat trout.

We modified our criteria to consider the fringe distribution for redband trout. Although the taxonomic status of redband trout is uncertain, studies suggest long-term isolation of the allopatric form has occurred in a number of isolated subbasins (Behnke 1992; Berg 1987) that fall within individual ERUs. As a result, our criteria based on distribution within an ERU may not represent the degree of isolation and genetic differentiation that has occurred with this species (Williams and others 1989). Important isolated groups are associated with the Wood, Yaak, and Kootenai rivers, several subbasins in the Klamath River basin and a complex of isolated subbasins in south central Oregon (Goose Lake, Warner, Catlow Valley, and Harney). Because of the clear isolation, we considered any of these subbasins supporting fewer than 30 subwatersheds to be part of the fringe distribution for allopatric redband trout. Our formally defined fringe distribution included the Big and Little Wood, Warner Lake, Guano, and Goose Lake subbasins. However, because of potentially unique characteristics in the other isolated subbasins and because the distribution and status are so poorly known in these waters (map 4.19), managers should consider all these

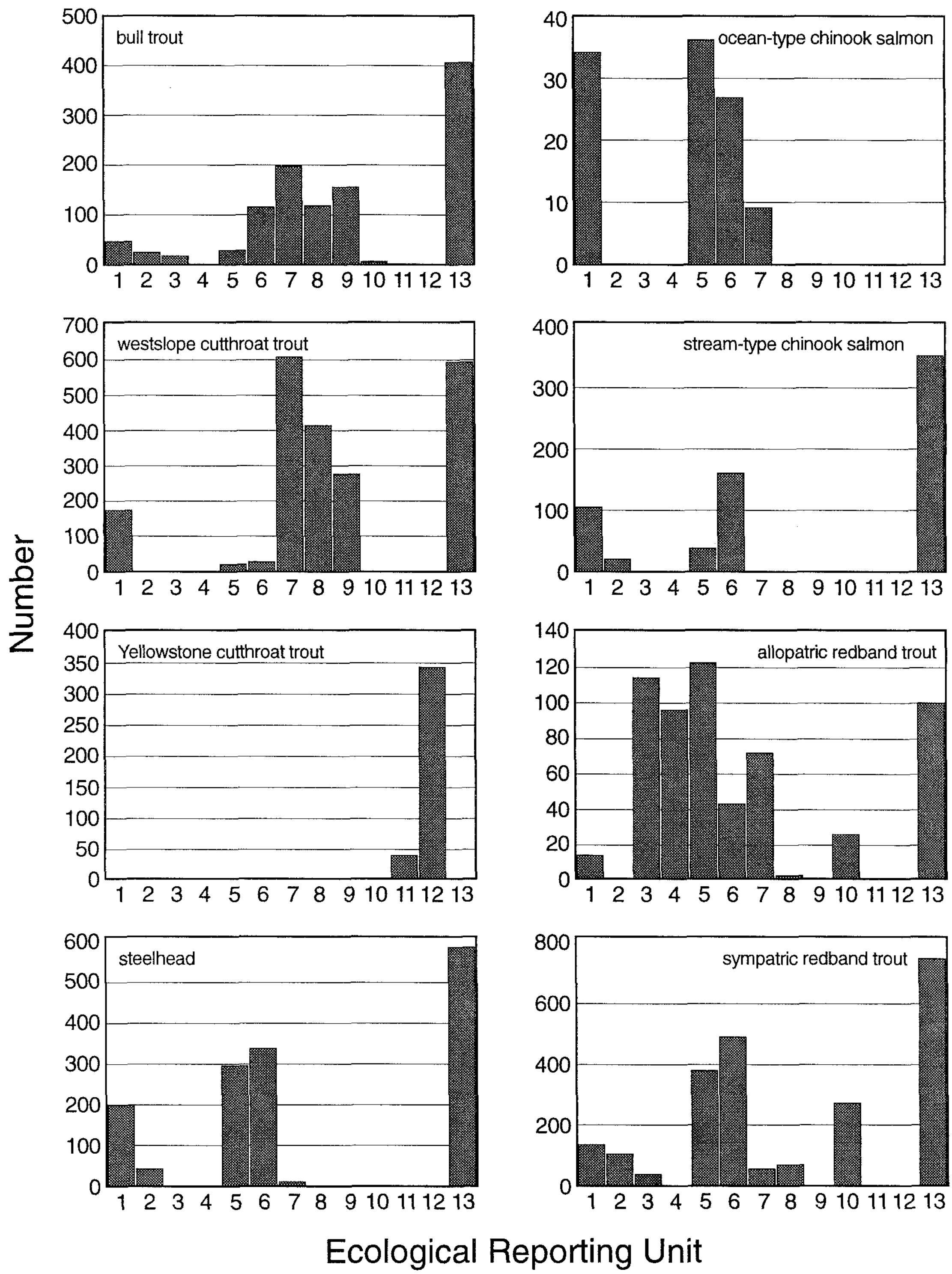
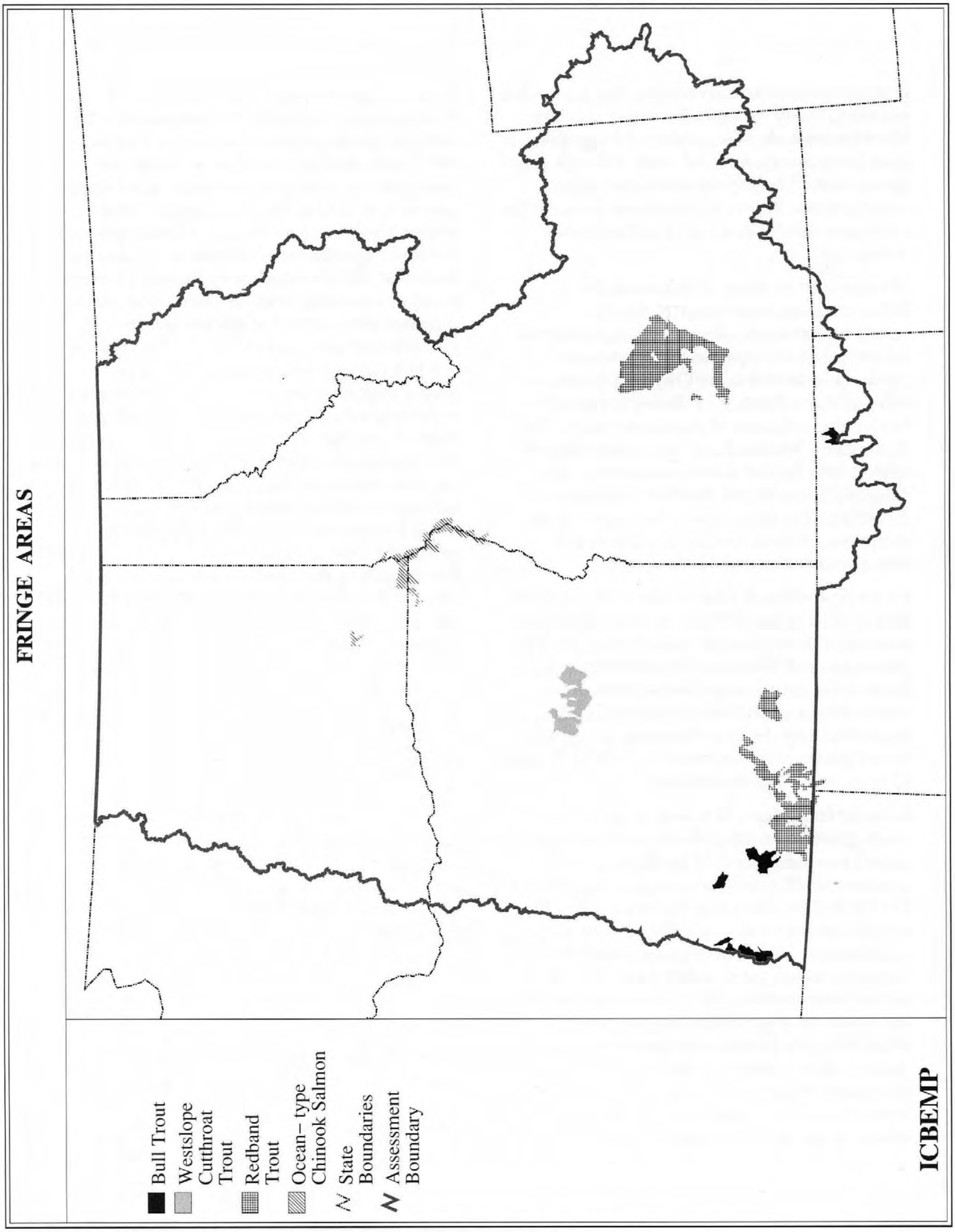


Figure 4.21—Number of subwatersheds classified as known or predicted spawning and rearing areas for each of the seven key salmonids, including both allopatric and sympatric redband trout, within each of the Ecological Reporting Units. ERUs: (1) Northern Cascades, (2) Southern Cascades, (3) Upper Klamath, (4) Northern Great Basin, (5) Columbia Plateau, (6) Blue Mountains, (7) Northern Glaciated Mountains, (18) Lower Clark Fork, (9) Upper Clark Fork, (10) Owyhee Uplands, (11) Upper Snake, (12) Snake Headwaters, (13) Central Idaho Mountains.



Map 4.31—Subwatersheds classified as fringe areas in the distribution of key salmonids.

populations as potentially unique. Further work is needed to clarify the current status of this form. No subwatersheds were considered fringe distributions for sympatric redband trout, although populations in the Klamath are candidates. Again, confusion over the extent of isolation between the two forms clouds the status of redband trout within this ERU.

We identified no fringe distributions for Yellowstone cutthroat trout, steelhead, or stream-type chinook salmon. Although only one subwatershed was reported for Yellowstone cutthroat trout within the Owyhee Uplands, it is found above Shoshone Falls and is part of the historical distribution of that species within the Snake River. Steelhead and stream-type chinook salmon have limited distributions within the Southern Cascades and Northern Glaciated Mountains, but these populations appear to be extensions of much broader distributions in adjacent subbasins and ERUs.

Ocean-type chinook salmon have a limited distribution in all of the ERUs across the range. They occurred in fewer than 30 watersheds in the Blue Mountains and Northern Glaciated Mountains. Because the latter group of subwatersheds is closely associated with the Northern Cascades, we considered only the Blue Mountain group, represented primarily by the lower Snake River in Hells Canyon, as a fringe distribution.

Genetic Integrity—Hatchery programs may erode genetic diversity and alter co-adapted gene complexes characteristic of locally adapted stocks of salmonids (Reisenbichler, in press; Waples and Do 1994). The effects may include a loss of fitness or performance (such as growth, survival, and reproduction) and a loss of genetic variability important to long-term stability and adaptation in varying environments. To map areas of potential importance as wild, indigenous gene pools, we asked biologists familiar with specific subwatersheds to identify chinook salmon and steelhead spawning and rearing areas that are unaltered by hatchery stocking, regardless of whether the populations are strong or depressed. Because data describing genetic purity of populations was not available across the Basin, we chose not to rely solely on genetic analysis. Instead, we defined wild, indigenous areas as subwatersheds with a low probability of strays from non-indigenous sources spawning with indigenous fish and as: 1) subwatersheds that had no history of hatchery-reared or non-indigenous introductions; or 2) subwatersheds that had been stocked rarely with hatchery-reared or non-indigenous fish, but where evidence suggested poor survival of stocked fish and a low probability of introgression; or 3) subwatersheds that had been stocked regularly in the past, but genetic analyses found existing wild fish identical to the original wild gene pool. Several biologists believed our criteria were too conservative because some potentially intact gene pools were not included. Biologists cited examples of stocks that had been heavily supplemented with hatchery-reared fish but where genetic analysis revealed substantial differences between wild and hatchery fish, suggesting that hatchery fish survived poorly. None of these cases, however, met the third criterion. The areas important to the genetic integrity of the anadromous salmonids are found principally within the Blue Mountains and Central Idaho Mountains (see map 4.22).

We could not identify similar pure populations for non-anadromous salmonids because (excepting Montana) there has been little genetic inventory (Rieman and Apperson 1989; Young 1995) or documentation of stocking that could be used in this scale of analysis. We also did not use the presence of introduced trout as an indicator of genetic integrity because those data were available only at the watershed level, potentially overestimating the influence of stocking in subwatersheds considered here. As a result, our analysis of genetic integrity is incomplete and would require a finer level of analysis for a consistent application to non-anadromous salmonids.

Narrowly Distributed Endemic and Federally Listed Species—We recognized species that are very narrowly distributed and endemic either to the Basin or to a relatively small region including the Basin as potentially sensitive and evolutionarily distinct. Fishes listed under the federal Endangered Species Act (see table 4.16) have a similar though more formal recognition. We summarized the occurrence of both groups across watersheds. Narrowly distributed endemics were found principally in Oregon and southern Idaho. The Upper Klamath was a particularly important area for endemism, with up to six species reported in a single watershed (map 4.32). Most of the endemic species are associated with closed basins and many are isolated in relatively small portions of watersheds. The Central Idaho Mountains and the major river corridors emerged as important areas of convergence for federally listed species (map 4.33).

Integrity of the Fish Assemblage

Biotic integrity provides a conceptual metric for identifying aquatic systems that more closely approximate historical levels of natural diversity. Karr and Dudley (1981) define biotic integrity as "the ability to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitat of the region". Unlike species diversity, integrity specifically refers to *native* biota that reflect natural evolutionary and biogeographic processes (Angermeir and Karr 1994). Several measures of biotic integrity have been proposed, which often reflect different attributes for communities of invertebrates and amphibians, and fish (Fisher 1989; Lyons and others 1995).

We developed a simple measure of integrity that reflects the status, diversity and structure of the native fish assemblage. Our measure includes three metrics described above: key salmonid strongholds, native species richness, and our composite index (Z) of native diversity and evenness. We built our index at the watershed level, using information from subbasins, watersheds, and subwatersheds. We totaled strong populations in subwatersheds within each watershed, used species richness at the subbasin level to assign a value to every watershed contained therein, and used our Z values from each watershed. The first two metrics were normalized such that they could assume a maximum value of 1, the same as Z. Our index of integrity was defined simply as the arithmetic average of the three normalized indices.

Values for our index of integrity range from 0.23 to 0.87, units with missing data were assigned a value of 0. The spatial pattern of the integrity index reflects the overlap of population strongholds, high species richness, and a predominance of native species (map 4.34). The highest concentrations of high-integrity values are found in the Northern and Southern Cascades, the Blue Mountains, the Central Idaho Mountains, and the southern edge of the Columbia Plateau. Smaller areas of high integrity are found in the Lower Clark Fork. These ERUs had the highest median values, and with the exception of the Lower Clark Fork, also exhibited the widest variability in integrity (fig. 4.22). There are varied reasons for each score. Many of the high-value areas are forested areas within the range of anadromous fish. Rangeland and agricultural areas tend to have lower integrity values.

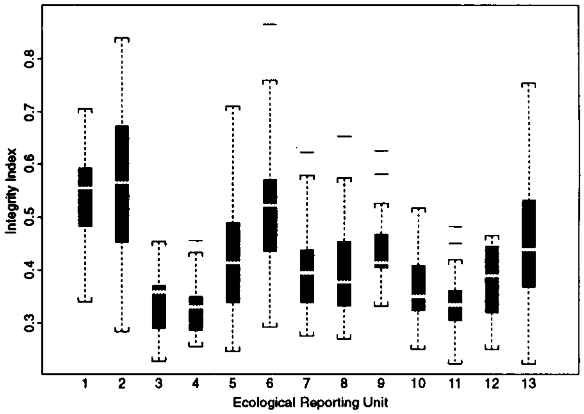
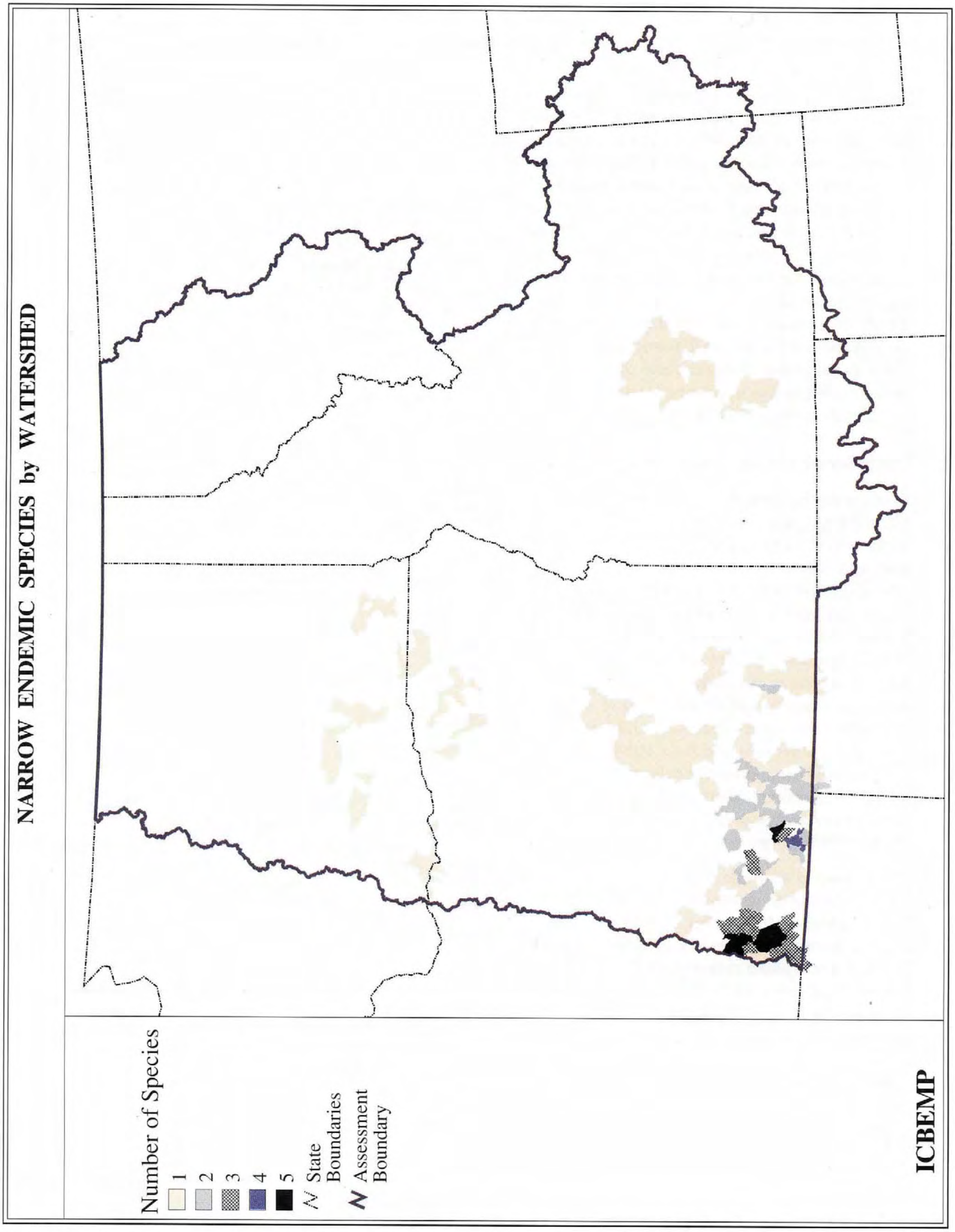
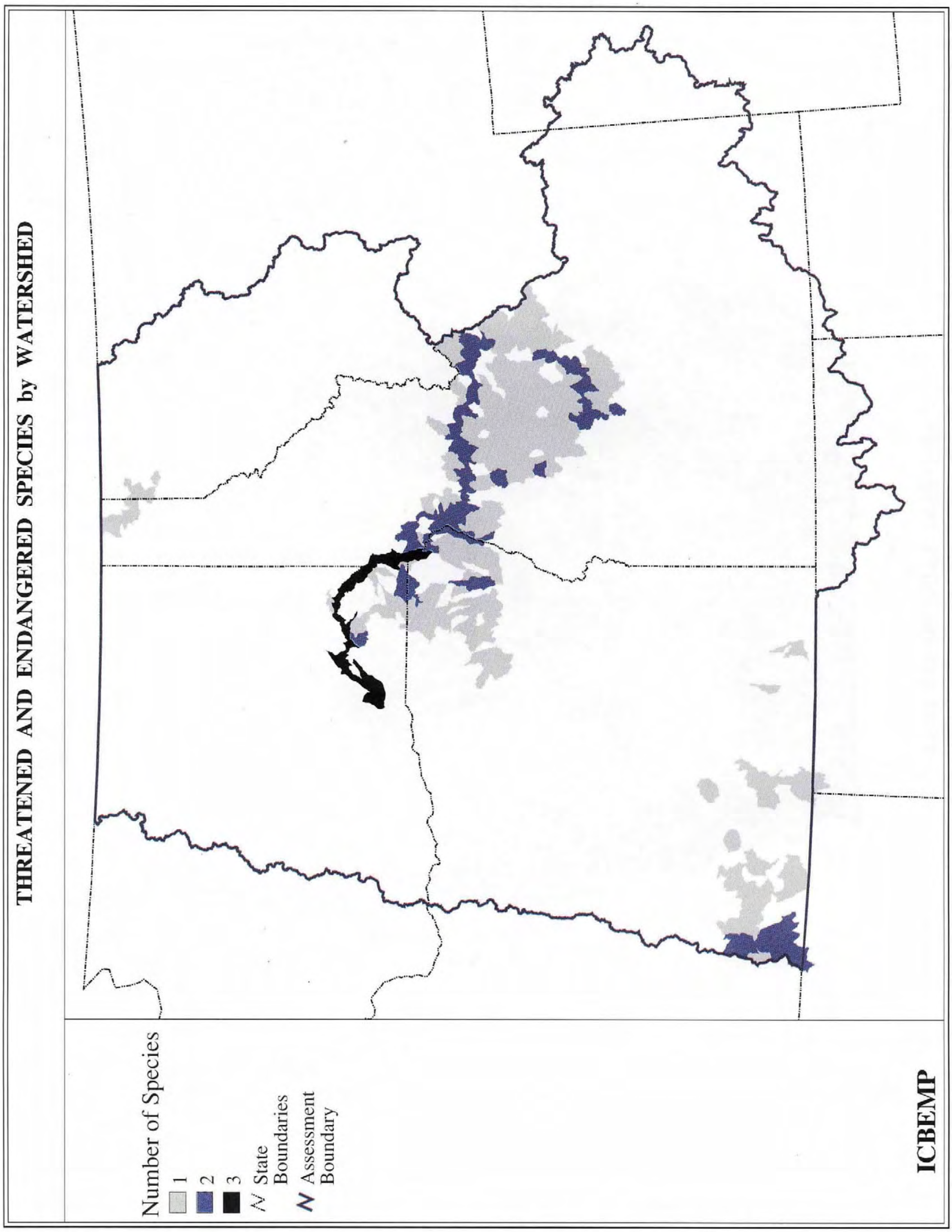


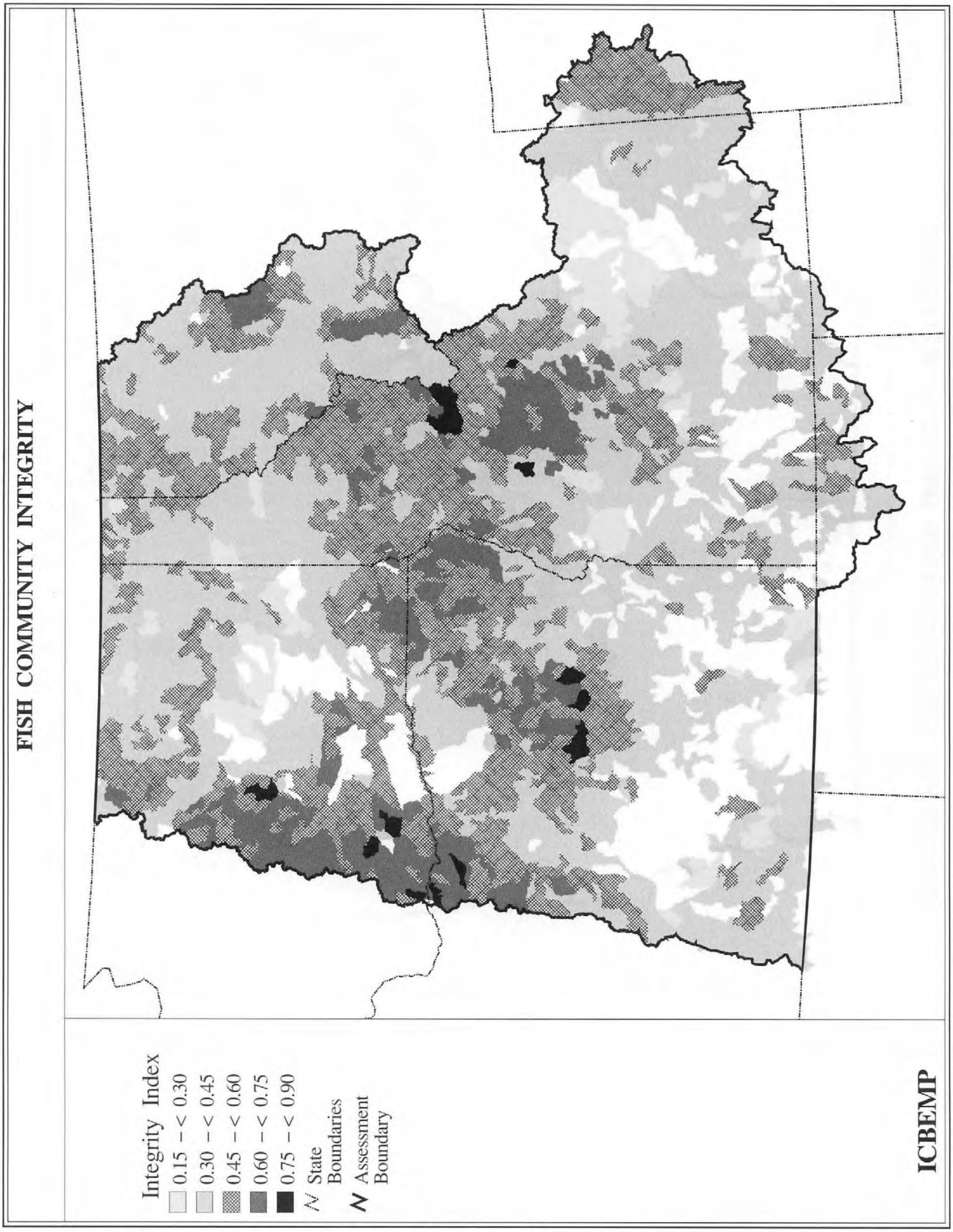
Figure 4.22. Box and whisker plots illustrating the range of fish assemblage integrity values by Ecological Reporting Unit. The horizontal line in the interior of the box represents the median, the box height equals the interquartile distance, and the dotted lines extend to the extreme values of the data. Data points outside the dotted lines may be outliers and are indicated by horizontal lines.



Map 4.32—Number of narrowly distributed endemic species reported by watershed.



Map 4.33—Number of species formally listed under the Endangered Species Act reported by watershed.



Map 4.34—Distribution of fish assemblage integrity scores by watershed.

Wild Trout Waters

While our analysis focused primarily on native fish, we recognize that many subwatersheds support high-value, wild-trout fisheries, but are supported by introduced salmonids including brook, brown, rainbow, and lake trout. Habitat in these subwatersheds remains suitable for natural reproduction of salmonids, although native salmonids may be depressed or extinct. In the Henry's Fork Snake River, Idaho, for example, native Yellowstone cutthroat trout are virtually extinct in large portions of their historical range but wild, self-sustaining populations of introduced rainbow trout thrive and support an internationally recognized trout fishery. In the Blackfoot River, Montana, native westslope cutthroat trout similarly have been replaced by introduced brown and rainbow trout that support a renowned wild-trout fishery.

We contacted state and tribal biologists in Idaho, Montana, Nevada, Oregon, Washington, and Wyoming and compiled a list of high-value recreational fisheries supported by wild, self-sustaining salmonid populations, regardless of original origin (appendix 4F). We encountered many definitions of wild, high-value salmonid fisheries, including waters described as Trophy, Quality, Blue Ribbon, and Wild Trout. All these waters shared three characteristics, as follows: 1) they were at least partially supported by wild, naturally spawning salmonids; 2) fisheries were regulated by special regulations consisting of reduced bag limits, size limits, gear restrictions, season closures, or mandatory catch-and-release; and 3) the waters typically received increased emphasis on habitat protection. The list includes a wide variety of salmonid assemblages and regulations. In some cases, special regulations (such as size limits) were applied to large numbers of streams to sustain wild native trout (such as cutthroat trout) while differential general regulations were applied to other species (such as rainbow trout) in the same streams. In other cases, special regulations, such as catch-and-release, were applied to both native and introduced salmonids.

Supplemental Analysis of Road Impacts

Much of our previous discussion and results points to negative consequences that often accompany roads. The effects associated with roads reach beyond their direct contribution to disruption of hydrologic function and increased sediment delivery to streams. Roads provide access, and the activities which accompany access magnify the negative effects on aquatic systems beyond those due solely to roads themselves. Activities associated with roads include, but are not limited to, fishing, recreation, timber harvest, livestock grazing, and agriculture. Roads also provide avenues for stocking non-native fishes. Unfortunately, we do not have adequate broad-scale information on many of these attendant effects to identify their component contributions accurately. Thus, we are forced to use roads as a catch-all indicator of human disturbance.

In preceding sections, we presented results based on analyses which used the variable, *roaddn*, to indicate road density. Each of these analyses supports the general conclusion that increasing road density is correlated with declining aquatic habitat conditions and aquatic integrity. Yet to some reviewers, our arguments are not convincing given the weak statistical relationships found within the data. Here, we reinforce and expand our conclusions based on a supplemental analysis of road density and the distributional data of four key salmonids. This analysis is based in part on information that was compiled after the earlier analyses were completed, and seeks to rectify some of the shortcomings of the earlier analyses. Time constraints did not allow us to redo the earlier analyses.

Focus on Non-anadromous Salmonids

Land-use disturbances within a given watershed affect anadromous and non-anadromous fishes in similar ways. The complex life history of anadromous fishes, however, confounds the effect of any single activity. The effects of habitat degradation within a spawning area on population status, for example, may be masked by migrational and ocean-related losses. Analytical resolution is further compounded by the fact that anadromous species often inhabit larger rivers than their non-anadromous relatives. These larger rivers both integrate and diffuse the effects of land-use activities over large areas, often well beyond the local subwatershed. It is not surprising to find little apparent relationship between subwatershed features and the population status of anadromous species found there. We simply have a poor experimental design for single-variable analyses of fine-scale factors on anadromous fishes. In contrast, non-anadromous fishes more readily display the effects of local disturbance because they spend a much larger portion of their life within the confines of a single watershed.

In our analyses reported earlier, the seven key salmonids followed this pattern. Status of anadromous fishes was principally tied to stream size, the number of dams between the subwatershed and the ocean, and broad geographical descriptors such as ERU and elevation. The non-anadromous fishes showed increased sensitivity to more local factors such as management and ownership, vegetation indices, and road density class. Therefore, we focused on analysis of the non-anadromous species in our supplemental analysis of road density. Similar exploratory analyses of anadromous species proved inconclusive in terms of identifying consistent relationships between population status and road density, as expected.

Methods

Part of the reason for the weak statistical relationships found in earlier analyses is due to the lack of resolution provided by our measure of road density. The predicted road density class, *roaddn*, is a categorical variable that is limited to six levels. As such, it is useful for making broad characterizations but lacks the resolution of continuous variables such as the arithmetic mean from which *roaddn* is derived. *Roaddn* was created by calculating the area-weighted, arithmetic mean road density (*rdmean*) for each subwatershed using the 1 square kilometer road-density map (see Landscape Dynamics chapter of this report), and assigning one of six levels to each subwatershed based on the value of *rdmean* (fig. 4.23). Most subbasins are grouped within the moderate and high categories, which cover a disproportionate share of the observed range. The resulting frequency histogram for *roaddn* masks much of the variability inherent in *rdmean.*

To recover information lost in the derivation of *roaddn*, we choose to use *rdmean* instead of *roaddn* in our supplemental analyses. We also calculated the area-weighted, geometric mean road density (*rdgeo*), which is defined as the n-th root of the product of n values, where each value is the midpoint of the road density class assigned to each of n 1 square kilometer pixels within each subwatershed. The geometric mean minimizes the effect of outliers with large positive values, thus increasing the weight given to less-roaded areas of a subwatershed. Geometric means are consistently less than the arithmetic means and show a more skewed frequency histogram (fig. 4.23). Since streams may be affected by upstream processes beyond the local subwatershed, we also calculated the arithmetic mean road density for the entire area that drains into each subwatershed (*ups_rdms*).

We used classification trees to identify areas within the range of each species where roads appear influential and then tested the strength of these relationships using logistic regression. Classification trees were constructed using a reduced set of the available landscape variables and the newer measures of road density (table 4.45). The classification trees were used to help identify portions of each species's range that might be sensitive to road density. For example, the classification tree for redband trout suggested roads were important in forested areas outside the range of anadromous fish. For each of the four species, we then performed a two-step logistic regression on selected subgroups of the data. The bull trout and Yellowstone cutthroat trout subgroups included all subwatersheds that were predominantly managed by the Forest Service or BLM. Westslope cutthroat trout were split into two subgroups; one included

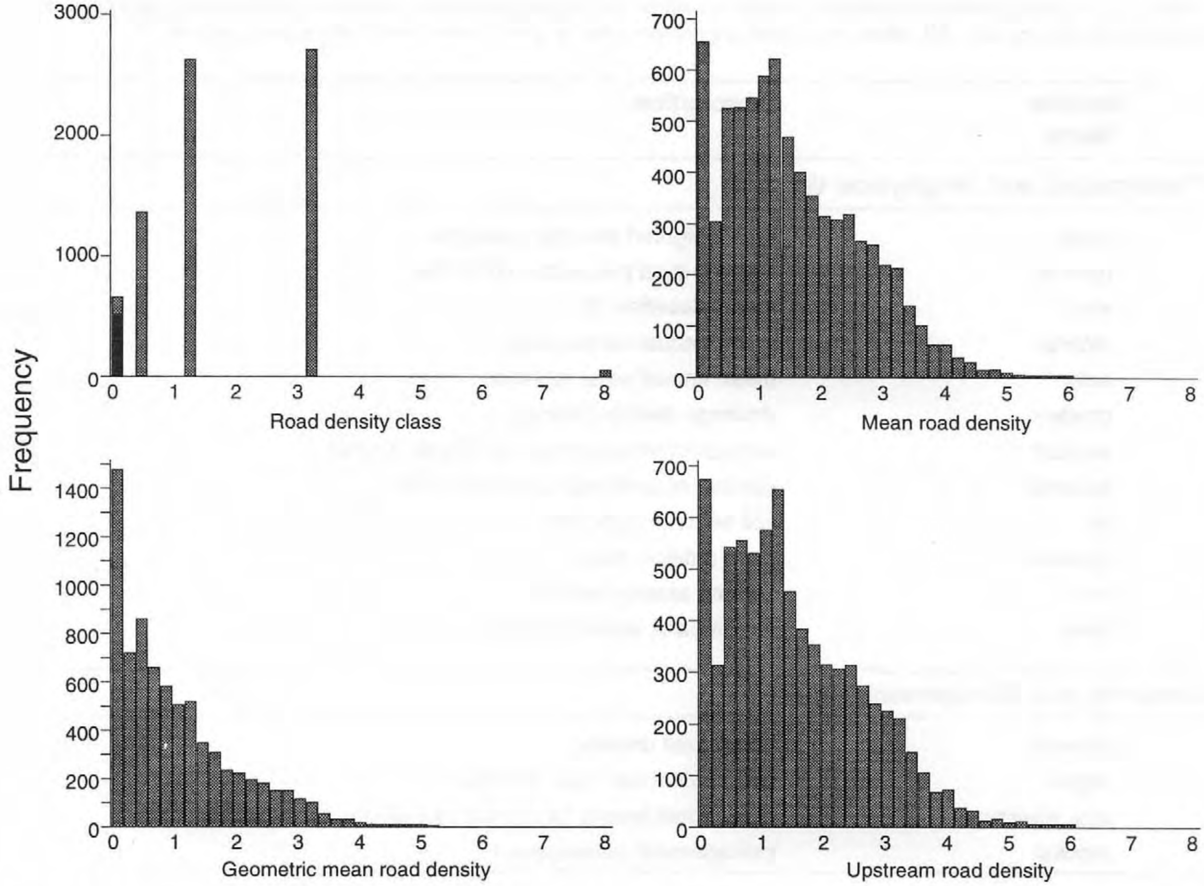


Figure 4.23—Histograms showing the frequency of road density class, mean road density, geometric mean road density, and upstream road density. Plotted values are midpoints of road density classes: none (solid black bar under 0) = 0 - 0.02; very low (hatched bar under 0) = 0.02 - 0.1; low = 0.1 - 0.7; moderate = 0.7 - 1.7; high = 1.7 - 4.7; extreme = > 4.7.

FS/BLM subwatersheds outside the range of anadromous fish, and the second included all subbasins within the range of anadromous fish where the mean annual air temperature is less than 5.8° C. The redband trout subgroup was limited to subwatersheds with a large forest component outside the range of anadromous fish.

In the first step of the logistic regression, the response variable indicated whether or not the species used the subwatershed for spawning and rearing. A positive response (1) was recorded if the species's status was known strong or depressed. Negative responses (0) were recorded for known absence or migration corridors. Possible explanatory variables included *rdgeo* and *ups_rdms* to reflect road density, and the additional landscape variables described in table 4.46. We used an iterative-selection modeling process in which variables were iteratively added and removed from the model according to minimum significance criteria (SAS Institute Inc. 1989). In the second step, we used the same set of potential explanatory variables to distinguish between strong and depressed responses.

Table 4.45—Descriptions of landscape variables used in the logistic regression analysis of subsampled groups of non-anadromous salmonids. All values expressed as percents refer to the percent area of the subwatershed.

Variable Name	Description
Physiographic and Geophysical Variables	
slope	area weighted average midslope
pprecip	mean annual precipiation (PRISM)
elev	mean elevation (ft)
mtemp	mean annual temperature
solar	mean annual solar radiation
drnden	drainage density (mi/mi^2)
anadac	access for anadromous fish (0=no, 1=yes)
hucorder	number of upstream subwatersheds
hk	soil texture coefficient
baseero	base erosion index
ero	surface erosion hazard
bank	streambank erosion hazard
Ownership and Management Variables	
rdmean	mean road density
rdgeo	geometric mean road density
ups_rdmsss	mean road density for current and all upstream subwatersheds
mgclus	management classification

Results

All subgroups except one showed significant road effects—either when distinguishing occupied spawning and rearing areas from unoccupied areas (step 1), or when distinguishing strong from depressed status (step 2). The geometric mean road density had a significant effect in step 1 for bull trout in FS/BLM lands (p=0.0001), in step 2 for westslope cutthroat trout in FS/BLM lands outside the range of anadromous fish (p=0.0007), and in step 2 for redband trout in forested areas outside the range of anadromous fish (p=0.0001). The arithmetic mean road density for all upstream areas had a significant effect in step 2 for bull trout in FS/BLM lands (p=0.0001), and in step 1 for westslope cutthroat trout within the cooler portions of the range of anadromous fish (p=0.002). In all cases, the sign of the coefficient associated with the road density measurements was negative (table 4.46), which suggests a decreasing likelihood of occupancy, or a decreasing likelihood of strong status if occupied, with increasing road density. Other variables also were found to be significant in different combinations for each species. As indicated by the sign of the coefficients associated with each variable (table 4.46), no other variables except *slope* showed the consistent patterns across all species shown by the road density measures.

The influence of roads on population status indicated in the logistic regression analysis is reinforced by graphical displays of the cumulative

Table 4.46—Results of logistics regression analysis for subsampled groups of non-anadromous salmonids. The variables listed refer to only those variables found to be significant. Direction indicates the sign of the estimated coefficient associated with each variable.

	Bull trout		Westslope cutthroat subgroup 1		Westslope cutthroat subgroup 2		Yellowstone cutthroat		Redband trout	
Response[1]	variables	direction	variables	direction	variables	direction	variables	direction	variables	direction
*spawning/rearing	rdgeo	-	slope	+	ups_rdms	-	elev	+	solar	+
vs.	hucorder	-			hucorder	-	mtemp	-	hk	-
absent/migration	mtemp	-			drnden	+	pprecip	-		
	pprecip	+								
*strong	ups_rdms	-	rdgeo	-	mtemp	-	drnden	-	rdgeo	-
vs.	solar	+	hucorder	+	solar	-	hk	+		
depressed	slope	+	elev	+	slope	+				
	drnden	+	mtemp	+						
	hk	+	bank	-						
			hk	+						

[1] Modeled responses are indicated by asterisks.

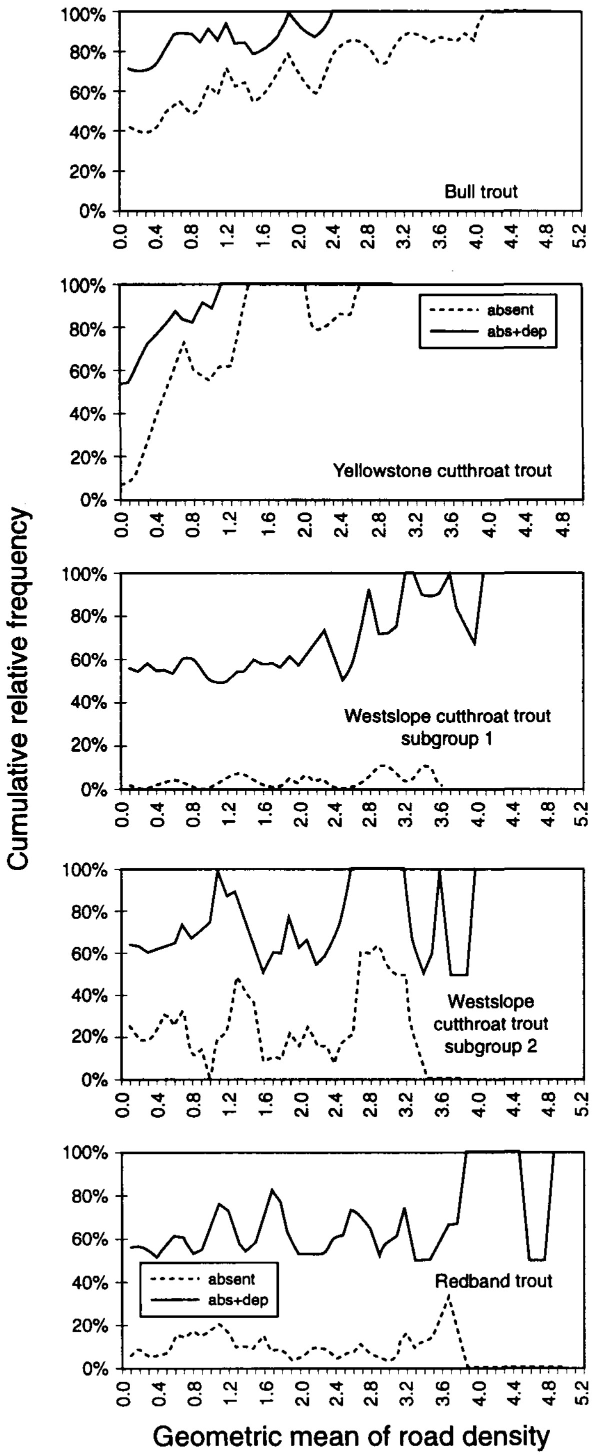


Figure 4.24—Cumulative relative frequencies for absent, and absent or depressed versus geometric mean density (*rdgeo*, table 4.46), using a moving window on *rdgeo*.

relative frequencies for absent, and absent or depressed versus *rdgeo*, using a moving window on *rdgeo* (fig. 4.24). These graphs clearly show increasing absence and decreasing proportion of strong populations with increasing road density for several subgroups. Additional evidence is suggested by the mean values for *rdgeo* and *ups_rdms* by status level for each subgroup (table 4.47). In all cases, the lowest mean road density values are associated with strong population status.

The graphical trend and the progression in mean road density values are quite apparent for Yellowstone cutthroat trout, even though it was the only subgroup that did not show a significant road effect in the logistic regression exercise. The lack of statistical significance in the face of apparent trends points to complex interactions among the explanatory variables that are not adequately addressed in the relatively simple logistic model that looks only at additive main effects irrespective of interactions. The fact that we detected consistent, significant effects for other species using a limited model is further testament to the strength and pervasiveness of the effects.

Discussion

Our results clearly show that increasing road densities and their attendant effects are associated with declines in the status of four non-anadromous salmonid species. They are less likely to use highly roaded areas for spawning and rearing, and if found are less likely to be at strong population levels. This is a consistent and unmistakable pattern based on empirical analysis of 3,327 combinations of known species' status and subwatershed conditions, limited primarily to forested lands administered by the Forest Service and BLM. We would not expect the relationship to be as strong on the non-forested, lower-gradient lands administered by the BLM. Of the four species examined, these lands tend to support only redband trout. Our results suggested that redband trout status could be clearly associated with road density only within forested, higher-elevation areas.

Table 4.47—Average road densities by population status for subsampled groups of non-anadromous salmonids.

Species		Status	Frequency	rdgeo[1]	ups_rdms[2]
Bull trout					
		Absent	932	1.13	1.71
		Depressed	477	0.67	1.36
		Strong	155	0.18	0.45
Westslope cutthroat trout					
	subgroup 1	Absent	23	1.67	1.87
		Depressed	443	1.45	2.14
		Strong	165	0.59	1.16
	subgroup 2	Absent	113	0.6	1.04
		Depressed	174	0.66	1.07
		Strong	144	0.31	0.55
Yellowstone cutthroat trout					
		Absent	42	1.16	1.81
		Depressed	54	0.27	1.01
		Strong	126	0.1	0.41
Redband trout					
		Absent	46	1.46	2.32
		Depressed	274	1.5	2.19
		Strong	159	1.03	1.73

[1]rdgeo = geometric mean of road density (mi/mi^2)

[2]ups_rdms = arithmetic mean of road density of all upstream subwatersheds (mi/mi^2)

The non-anadromous salmonid species are important elements in aquatic communities and deserve increased stature in policy deliberations. In some ways, their importance has been overshadowed in recent years due to the plight of the anadromous species in the Columbia River Basin. This is unfortunate given their distribution and contribution to aquatic integrity. The historical range of the four non-anadromous salmonids examined above includes 96 percent of the subwatersheds under substantive FS and BLM management. In contrast, only 27 percent of the subwatersheds so classified currently are accessible to anadromous fishes and fall within the range of the three anadromous key salmonids. Thus, the non-anadromous salmonids provide a much broader picture of watershed conditions. One of the elements that we used to identify areas of high aquatic integrity was whether or not a watershed supported strong populations of key salmonids. There were 52 known strong populations of anadromous fish at the subwatershed scale, and 927 known strong populations of non-anadromous fish were recorded.

The declines in population status of the non-anadromous key salmonids should be viewed as an indication of the types of responses that may be experienced by other native species in similar habitats. Those most like the non-anadromous key salmonids in distribution or habitat requirements would be expected to show the most similar responses. This would include the anadromous species such as steelhead, stream-type chinook salmon, and Pacific lamprey that broadly overlap in range with the non-anadromous salmonids and use many of the same habitats for significant portions of their life. There are no logical reasons to expect anadromous fishes to be immune to the effects of habitat change evident in the non-anadromous species. Other species, including sculpins, dace, and some suckers also have considerable overlap in range and may follow similar trends in population abundance and distribution.

Summary and Conclusions

We have assessed the current status of select fishes, identified areas with high biotic integrity and diversity, and identified linkages between fish, the biophysical environment, and humans. In this final section, we synthesize our results and consider the implications for management. We organize our conclusions along four major themes.

1. The composition, distribution, and status of fishes within the Basin are very different than they were historically. The overall changes are dramatic and extensive, and in many cases irreversible. Some forms, such as the Alvord cutthroat trout (*Oncorhynchus clarki* spp.) and the Miller Lake lamprey (*Lampetra minima*), are extinct (table 4.16). Many others, especially anadromous fish, are extirpated from large portions of their historical range. While we may be able to reestablish some depressed taxa in portions of the range, reopening large tracts of the former range to anadromous salmonids is unlikely in the short term.

Our clearest understanding of fish status comes from the analysis of the seven key salmonids.

Our analysis of available data suggests the historical condition of native fishes has changed dramatically. Several of the key or sensitive species are distributed throughout most of their historical range (notably cutthroat and redband trout), however, declines in abundance, the loss of important life histories, genetic introgression, local extirpations, and fragmentation and isolation of high-quality habitats into smaller patches are apparent for all of the key salmonids and many other species. Yellowstone and westslope cutthroat trout appear to be the most successful of the key salmonids. They were found or predicted in 71 and 85 percent of the historical range, respectively. With the exception of Yellowstone cutthroat trout, however, none of the key salmonids have known or predicted strong populations in more than 22 percent of their historical ranges (table 4.48). The non-anadromous trouts may be in even worse condition than suggested because of introgressive hybridization with introduced salmonids. Chinook salmon are the most restricted of the key salmonids. Both forms of chinook salmon are extinct in more than 70 percent of their historical ranges, steelhead in more than 50 percent, and all are approaching extinction in important portions of their remaining ranges. With few exceptions, most subwatersheds supporting chinook salmon and steelhead are also likely to be influenced by hatchery stocks.

If current distributions of the key salmonids are good indicators of aquatic ecosystem health, many systems remain only as remnants of what were larger, more complex, diverse and connected systems. Even with no further habitat loss the fragmentation and isolation may place remaining populations at risk. With the exception of the Central Idaho Mountains, Snake Headwaters, and perhaps the Northern Cascades, most of the important areas for the key salmonids exist as patches of scattered subwatersheds. Many are not well connected or are restricted to much smaller areas than historically. Many of the important subwatersheds are associated with high-elevation, steep, and more erosive landscapes. These may be more extreme or variable environments contributing to higher variability in the associated populations, and higher sensitivity to watershed disturbances. Risks could be aggravated with further development.

Table 4.48—Summary comparison of the known and predicted present and strong distributions for each of the seven key salmonids. The proportion of subwatershed areas in each class is shown within the Basin and within the historical range. The unknown subwatersheds included both unclassified and present but status unknown that required predictions for a complete distribution.

	Assessment Area		Historical Range		
Species	Historical	Present	Present	Strong	Unknown
Bull trout	0.59	0.26	0.45	0.06	0.38
Yellowstone cutthroat	0.09	0.07	0.66	0.32	0.39
Westslope cutthroat	0.35	0.27	0.85	0.22	0.38
Redband trout					
Sympatric	0.56	0.38	0.69	0.17	0.67
Allopatric	0.17	0.08	0.49	0.09	0.62
Steelhead	0.50	0.23	0.46	0.01	0.15
Stream-type chinook	0.46	0.13	0.28	<0.01	0.12
Ocean-type chiniook	0.07	0.02	0.30	0.05	0.13

The patchwork of important watersheds also suggests that remaining populations of salmonids are not well distributed within the subbasins. Watersheds that were once likely to support a complex of life-history patterns and subpopulations within larger regional or metapopulations are now often fragmented. The loss of spatial diversity in population structure and of the full expression of life-history pattern may lead to a loss of productivity and stability important to long-term persistence. If connectivity among populations is limited by a matrix of poor quality habitats interspersed among remaining high quality areas, gene flow and the potential for refounding or demographic support among populations will also be limited. Local extirpations may occur through random events even in high-quality environments with no further habitat change, but in many cases the spatial and life-history diversity necessary to mitigate the losses is no longer present.

All populations are likely variable in time and many have patchy distributions even in high-quality environments. Local extirpation may be a natural and perhaps common element in the dynamics of many species (Rieman and McIntyre 1995). Natural climate change and associated events such as glaciation and the Bonneville and Missoula glacial floods, have undoubtedly influenced the current distribution of these fishes (Behnke 1992). Climatic variation associated with fluctuating ocean conditions, drought and flood patterns, and local storm frequency and intensity may influence abundance and distributions in shorter time scales. The recent declines in Columbia River salmon, for example, have been influenced by recent ocean conditions (for example, Lichatowich and Mobrand 1995) and a period of extended drought. Although environmental variability is a factor and may be the proximate cause of decline or extinction in some cases, the effects of human-caused disturbance appear to be far more important. Population declines for many of the fishes, including Columbia River salmon, can be associated with a variety of factors that include habitat disruption linked to land management; watershed development for hydropower and irrigation; competition, hybridization and predation linked to the introduction of non-native species, races or stocks of fish; harvest; and even the intentional eradication of some populations. The native-fish assemblages have generally persisted best in the areas least influenced by humans.

Population fragmentation and lost resilience may place many remaining populations at increased risk to natural and human-caused disturbance. Recognition and conservation of important or sensitive populations and habitats will likely be critical to broad-scale persistence of many of these fishes.

Our analysis of species assemblages led to similar conclusions. Although we were unable to map the distribution of all species accurately, the trends are clear. We found large numbers of introduced species throughout all major river systems. The changes are most severe in lowland rivers, but higher-elevation tributaries are also affected. The current distributions of the native fishes likely reflect specific habitat requirements, the availability of suitable habitats constrained by the biophysical environment and dispersal mechanisms, and the persistence of local stocks influenced by population dynamics and habitat quality. The distributions of native fishes probably reflect natural disturbance patterns, but human influences have profoundly altered those distributions and the condition of populations throughout the Basin.

2. Though much of the native ecosystem has been altered, core areas remain for rebuilding and maintaining functional native aquatic systems. Even though they are reduced in numbers and distributions, native trouts remain one of the most widely distributed taxa within the basin. These indicators of environmental quality suggest that we have serious problems, particularly in the larger rivers and in the low-elevation agricultural and range lands. Many of these areas likely were fringe areas for several species. The situation is somewhat better in the forested lands, especially in those that have experienced less disturbance. We see a higher proportion of strong populations in higher-elevation forested lands than others, and the proportion declines with road density. Most of the areas exhibiting high integrity fall within forested areas, with the exception of areas inherently high in native species richness near the southern edge of the Basin.

The largest areas of contiguous or clustered watersheds supporting strong populations of key salmonids are associated with the major river subbasins found in the Central Idaho Mountains, the Snake Headwaters and the Northern Cascades. Important areas are also found in the Blue Mountains, Upper Clark Fork and the Northern Glaciated Mountains, but are scattered or generally restricted to only portions of interior river subbasins. Each of the key salmonids supported some strong populations, ranging from less than 1 percent of the historical range for stream-type chinook salmon to 32 percent for Yellowstone cutthroat trout (map 4.27). There was, however, little overlap in the strong distributions at our scale of analysis. Differences in habitat requirements and life-history patterns likely lead to the lack of overlap. With the exception of the Central Idaho Mountains and Northern Cascades there are few clusters of subwatersheds likely to provide highly productive habitat for multiple species, but collections of subwatersheds still exist within larger subbasins.

Native species still dominate most of the Basin. When we examined species assemblages, we could not detect trends indicating that introduced species were replacing native species. We know that introduced species can and do replace natives. For example, introduced rainbow trout have replaced Yellowstone cutthroat trout in large portions of the Henrys Fork Snake River (IDFG 1991). Our inability to detect such replacement may be an artifact of the scale at which our data were collected, and be influenced by our lack of information on introgression. Introduced and native trouts are sometimes spatially segregated within streams (Rieman and Apperson 1989), but appear to overlap within a watershed because of the resolution of our data distributions. The general pattern, however, based on over 2,000 sample points, indicates native species richness persisting even as non-native richness increases.

Our contention is that although native fish species have declined and fish communities have been altered, important elements remain. A core of habitats and populations for maintaining and perhaps restoring much of the biological diversity associated with historical aquatic communities still exists. Where introgressive hybridization has seriously compromised native gene pools, hybrid and introduced trouts probably cannot be eliminated. Hybrids may adapt well to existing conditions, however, and provide at least some of the function and value of the historical forms and species complexes.

3. Protection and maintenance of system integrity and functioning will require innovative approaches. Simple solutions such as setting aside small, scattered subwatersheds probably will not be adequate for the persistence of even current distributions and diversity. The problems are too complex and too pervasive. However, there are several actions which could be taken to maintain or restore the integrity of aquatic ecosystems.

First, conservation of watersheds and habitats that support remaining areas of high intrinsic value or condition for aquatic species is critical. These include areas supporting strongholds for one or multiple species, areas of high genetic integrity, fringe distributions, and areas that support narrowly distributed endemic or listed species. Though it may be unpopular to suggest land-use restrictions to protect fishes that are poorly known or support little recreational and economic opportunity, maintenance of biological diversity and integrity implies recognition of values other than those defined solely by charismatic game species. As Norton (1991) points out, it is not so important why one chooses to protect biodiversity—there are any number of reasons—but that you actively do it.

Narrowly distributed endemics are generally associated with closed basins; obvious exceptions are margined sculpin and Shoshone sculpin. All would benefit from special emphasis. Most endemic species have been isolated in basins over geologic time, contributing to their speciation and narrow distribution. Many of these species are associated with the interior Oregon Lakes and Klamath basins, and their presence emphasizes the special significance of these areas. Management and analysis of these aquatic systems probably can proceed independently of the Columbia River Basin. Conservative approaches to land management are more important here because virtually all these species and most of the waters that support them exist as small, isolated systems. Problems associated with habitat disruption and non-native species magnify themselves in such restricted spring and stream systems. Climate change alone may place many of these fishes at extreme risk.

Second, reconnection and expansion of the mosaic of strongholds for widely distributed species such as the key salmonids will enhance the integrity of larger systems. For wide-ranging fishes such as salmon, steelhead, and other migratory trouts, this includes protection of water quality and passage in migratory corridors as well as protection of spawning and rearing areas. Conservation and restoration of important habitats for key salmonids should provide habitat for associated species and will sustain important processes that influence structure and function within these systems. We have used existing information to identify important subwatersheds in the current distribution of the key salmonids. These subwatersheds should be incorporated into future conservation strategies, but they cannot be the only element of those strategies.

The diverse characteristics of the key salmonids and associated assemblages involve more than single subwatersheds. The full expression of life history in any species often involves whole river basins and a mosaic of habitats throughout those basins (Gresswell and others 1994; Healey 1994; Lichatowich and Mobrand 1995). It is also apparent from our analysis that although many subwatersheds support several species, few support spawning and rearing areas for more than one key salmonid. Any conservation strategy focused on protecting or restoring habitat for a single species or life-history form at the subwatershed scale will not provide for multiple species or complete communities. Frissell and others (1995) and Gresswell and others (1994) reach similar conclusions. Consideration of larger spatial and temporal scales would be an important aspect of any effective conservation strategy for aquatic ecosystems. Conservation and restoration of subwatersheds that support the remaining populations are critical for short-term persistence. Conservation and restoration of habitat networks throughout larger subbasins, however, will be necessary for long-term stability, persistence and the full expression of biological diversity.

Restoration and management of watersheds only on Federal lands will not be sufficient. River corridors surrounded largely by private lands are a particularly important part of the habitat networks. Much of the overlap in species distributions occurs in the larger river corridors because many of the species range widely. For example, anadromous species use the entire system of rivers as migratory corridors and also for overwintering and short-term rearing. The connections and habitat provided by larger river systems are thus critical to the maintenance of anadromous populations. The construction of dams and reservoirs and their complex effects on migration and survival is viewed as the single greatest threat to the persistence of salmon and steelhead in the upper basins. Although much of the highest-quality habitat for anadromous fish probably remains in the Central Idaho Mountains, no strong populations persist there, due largely to passage mortality in migration corridors. These corridors provide a critical link maintaining the complex life histories of other species as well. Non-anadromous species that retain migratory life-history patterns such as bull, redband, Yellowstone cutthroat, and westslope cutthroat trout may migrate hundreds of kilometers annually. These species may move repeatedly between small rivers and headwater streams used for spawning and initial rearing and large rivers or lakes used for subadult rearing, overwintering or seasonal foraging. Bull, cutthroat, and redband trout persist in basins isolated from the larger rivers and lakes. Where connections are disrupted and the full expression of life histories is restricted, however, the persistence, resilience, and diversity of aquatic communities are limited.

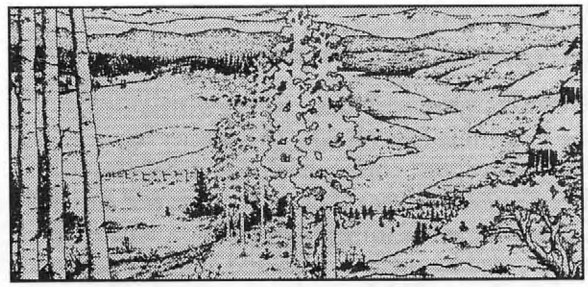
Restoring or maintaining the integrity of migration corridors will be challenging. Lowland rivers and lakes integrate the effects of disturbance over huge areas under many ownerships. No single agency oversees the management of entire basins or has the ability to balance competing demands and uses (for example, irrigation, hydropower, recreation, housing, waste disposal, timber harvest). Habitats associated with lower-elevation and lower-gradient valley bottoms are more desirable for human development. Many of these areas are in private ownership and attempts to protect private properties have severed the ties between rivers and their floodplains. Channelization, riprapping, construction of levees, and elimination of riparian vegetation have dramatically simplified stream and river habitats and disrupted the natural processes that form and maintain those habitats (Johnson and others 1995). Areas that may have been critical seasonal habitats for some species have been eliminated or degraded.

Third, containment of non-native species in certain areas will benefit native species. We are not advocating wholesale eradication of introduced fishes. We do suggest, however, that land management agencies could work with state fishery management agencies to reduce or eliminate stocking of non-native and hatchery-reared fish in areas capable of supporting self-sustaining native species. Containment of introduced fishes will help with the first two approaches as well. Interestingly, the average Z values, our measure of native diversity and evenness, were significantly higher in areas with two or more strong populations of key salmonids. This suggests a meaningful relationship between introduced fishes and abundance of native species.

4. Although the integrity of aquatic systems is most easily maintained in watersheds having little or no human influence, evidence suggests that many species persist in some intensively managed areas. We found stronger fish populations in areas with minimal disturbance. But we also found that intensive land use did not necessarily eliminate all strong populations or areas of higher fish community integrity. The scope of this assessment precludes us from examining individual watersheds in detail. Thus, we cannot discern whether intensively managed areas that retain relatively healthy populations and communities are anomalies, regions where the effects on streams lag behind the change on lands, or are areas where intensive management and fish simply coexist. We therefore recommend a more careful examination of these areas.

There are many factors that contribute to the productivity of individual watersheds and systems of watersheds. Those watersheds that are intensively managed, yet support strong populations or areas of high integrity, should be among those of greatest concern. Watershed responses often lag behind management changes. The cumulative effects of land disturbance, for example, may not be evident in streams until triggered by low-frequency storm events. We suggest prioritization of these areas for detailed review of watershed conditions and trends. Such review could recognize populations at risk and highlight practices that could be used to benefit aquatic resources in other intensively managed areas.

RARE AND SENSITIVE SPECIES: STATUS SUMMARIES



Narratives summarizing the status for 39 rare and sensitive species were prepared. These narratives are arranged in phylogenetic order and are divided as follows: Distribution and Status, Habitat Relationships, and Key Factors Influencing Status. Accompanying each narrative is a map that typically illustrates the probable historic and current distribution plus any major introduction sites mapped at the watershed level. The actual distributions of these species may include small portions of the watersheds and therefore, are overestimated on the figures. General categories of research and information needs are listed for each species in table 4.53 (see following section "Information and Research Needs."

These status narratives include a variety of narrowly distributed endemics, largely unknown species, and other native species that may be important and wide ranging but for which the assessment area represents a small portion of their range. The Sunapee char, an introduced species of special note, also is included. All these species are worthy of special management attention and several already receive protection under the Endangered Species Act. Resident freshwater threatened or endangered species are listed under the Endangered Species Act by the U.S. Fish and Wildlife Service, whereas the National Marine Fisheries Service has responsibility for listing anadromous fishes. Some rare and sensitive species are recognized as requiring special protection by the States of Oregon, Washington, Idaho, or Montana. Many are managed as sensitive species by the USDA Forest Service and/or USDI Bureau of Land Management. Several were considered by the U.S. Fish and Wildlife Service to be Category 2 Candidates for listing until a February 27, 1996 nationwide decision to delete all Category 2 taxa from Candidate species status.

Although we know less about the rare and sensitive species than the seven key salmonids, analyses of existing distribution and reviews of available literature provide important insights about common threats and appropriate management needs. Many of these taxa occur in isolated areas of the Columbia River Basin, in isolated subbasins of the Great Basin, or are restricted to the upper Klamath Basin. They typically occur in relatively depauperate subbasins, perhaps with only one or two native fish species present and therefore, may not be recognized in management strategies that focus on areas of high native species diversity. Many of these taxa occur in very restricted areas, often occupying one or two small habitat patches within subwatersheds. Consequently, broad or mid-scale assessments may not adequately describe their distributions.

White Sturgeon (*Acipenser transmontanus*)

White sturgeon were once widely distributed in the Columbia River basin. The species has been an important sport, commercial, and tribal resource. The Kootenai River (Idaho, Montana and British Columbia) white sturgeon population is listed as endangered by the U.S. Fish and Wildlife Service and the State of Idaho and as sensitive by the BLM. The Snake River white sturgeon is listed as a species of concern by the State of Idaho and as a sensitive species by Region 1 of the Forest Service.

Distribution and Status

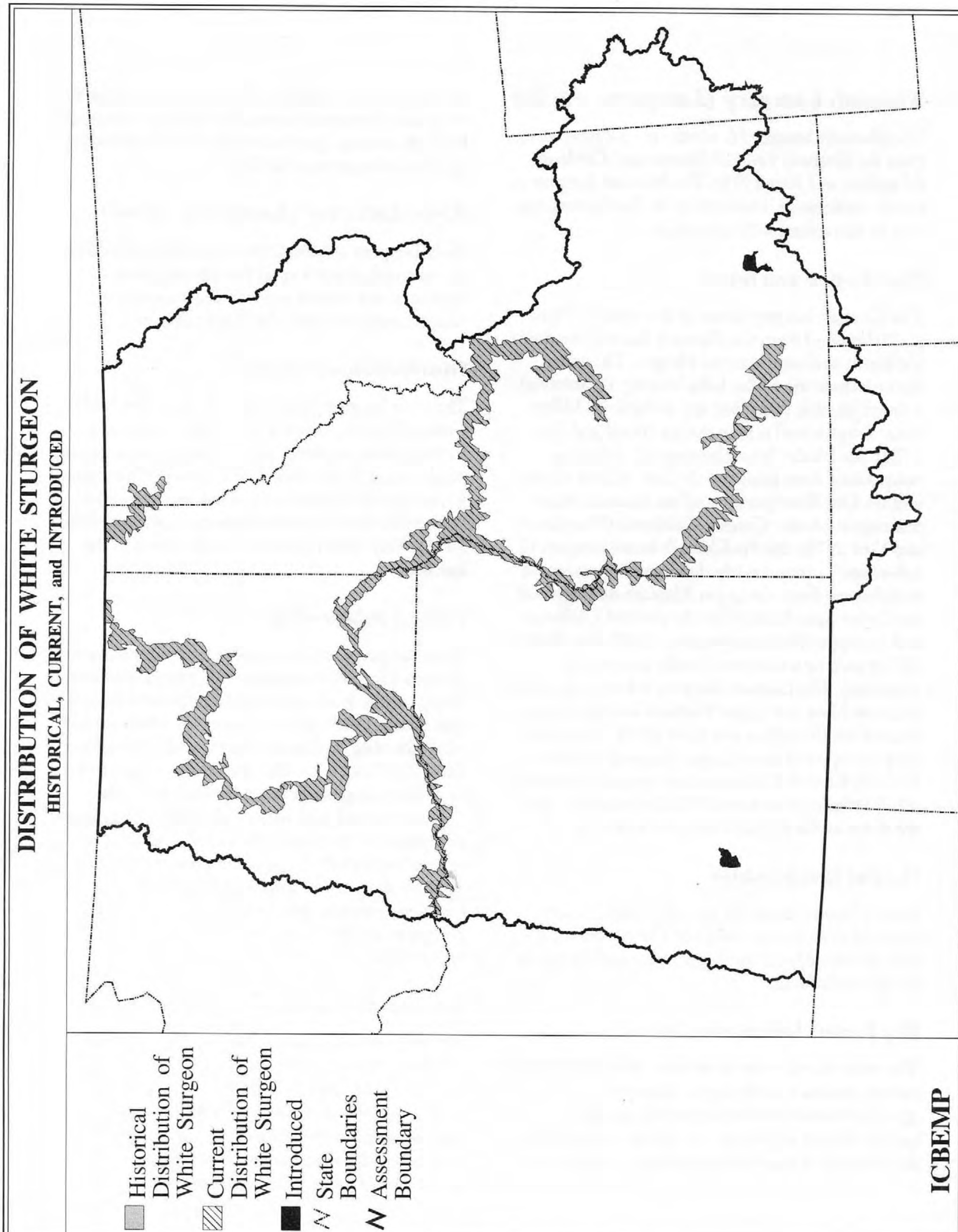
Along the Pacific Coast, white sturgeon were found in accessible freshwater from the Aleutian Islands south to central California. The Columbia River represents one of three large river basins in the Pacific Northwest where white sturgeon reproduce. Historically, prior to dam construction, white sturgeon were anadromous and migrated within the Columbia River Basin up to impassable falls (map 4.35). The Kootenai River white sturgeon has been isolated from other white sturgeon populations since the last glacial age (Apperson and Anders 1991). The white sturgeon is restricted to 695 river kilometers in the Kootenai River Basin between Kootenai Falls, Montana downstream to Cora Linn Dam at Kootenay Lake, British Columbia, Canada. The Kootenai River white sturgeon have not successfully spawned in recent years. The current population, which has a no harvest restriction, has decreased to about 880 individuals (Apperson and Anders 1991). Snake River white sturgeon are found in the Snake River in Idaho up to Shoshone Falls, and in the Salmon River in Idaho likely upstream to the confluence of the East Fork Salmon River. The present distribution of Snake River white sturgeon has been fragmented into discrete subpopulations confined between mainstem dams (Hanson and others 1992). The Hells Canyon reach along the Oregon-Idaho border contains the highest densities of Snake River white sturgeon. In the Columbia River downstream from McNary Dam, on the Washington-Oregon border, the annual abundance of white sturgeon greater than 53 centimeters was 893,800 fish from 1986 to 1992. In contrast to the Snake and Kootenai rivers where all captured sturgeon must be released, a consumptive sturgeon fishery continues in the lower Columbia River.

Habitat Relationships

Substrate size and water velocity influence selection of spawning areas by white sturgeon. Spawning generally occurs in water over three meters deep and over cobble substrate. In the Columbia River system, reproduction has been greater during years of high flows compared with years of low flow (Hanson and others 1992). Spawning also occurs earlier and at lower temperatures during high flow years (Hanson and others 1992). Adults and juveniles prefer deep-pool habitat with a fine bottom substrate. Adults tend to move downstream in the summer and fall months and upstream in the winter and spring months. Fish tend to stay in shallower water during the spring and summer and move to deeper waters during the winter.

Key Factors Influencing Status

Hydropower dams on the Columbia and Snake rivers have prevented migration, fragmented riverine populations, and reduced the effectiveness of natural propagation (Hanson and others 1992). Dams have also reduced spawning success, by decreasing the amount of suitable spawning areas or creating poor incubation environments. In general, the length of time required to reach sexual maturity, typically 10 to 15 years, results in low rates of natural recruitment. Most reproductively capable fish are protected by harvest regulations based on fish length. The decrease in spring river flows below Libby Dam, Montana appears to have contributed to spawning failures of the Kootenai River population in recent years (Apperson and Anders 1991). Land management activities are considered a secondary impact to white sturgeon.



Map 4.35—Historical and current distribution plus introduced sites of white sturgeon.

Klamath Lamprey (*Lampetra similis*)

The Klamath lamprey (*L. similis*) is a parasitic form from the Klamath River of Oregon and California (Vladykov and Kott 1979). The Klamath lamprey is poorly understood, restricted in its distribution, and may be threatened with extinction.

Distribution and Status

The Klamath lamprey is one of five species of lampreys described from the Klamath Basin of northern California and south-central Oregon. The other forms include the Miller Lake lamprey (*L. minima*), a dwarf parasitic form that was endemic to Miller Lake, Oregon, and is now extinct (Bond and Kan 1973); the Modoc brook lamprey (*L. folletti*), a nonparasitic form known only from Willow Creek and the Lost River portions of the Klamath River drainage in Modoc County, California (Vladykov and Kott 1979); the Pit-Klamath brook lamprey (*L. lethophaga*), a more widely distributed nonparasitic form known from the upper Klamath drainage and the Goose Lake drainage in Oregon and California and the upper Pit River system in California (Hubbs 1971); and the widespread Pacific lamprey (*L. tridentata*). The Klamath lamprey is known from the Klamath River and upper Klamath Lake in Oregon (map 4.36) (Vladykov and Kott 1979). The Pacific lamprey reported from Copco Reservoir on the Klamath River in California may instead represent *L. similis* (Moyle and others 1989). Information about the status of the Klamath lamprey is lacking.

Habitat Relationships

Little is known about the specific habitat requirements of the Klamath lamprey. The species is parasitic, presumably on the larger trouts and suckers in the Klamath Basin.

Key Factors Influencing Status

The many dams, water diversions, and other modifications common to the upper Klamath River drainage are presumed to have disrupted the distribution and life history of this species. Many native fishes in the Klamath Basin, which would serve as hosts for the lamprey, have declined. Based on the restricted range and threatened status of many host species, the Klamath lamprey may be at high risk of extinction. Specific information is lacking.

River Lamprey (*Lampetra ayresi*)

River lamprey are anadromous and parasitic. They are rare and, according to the limited available literature, apparently migrate short distances inland compared with the Pacific lamprey.

Distribution and Status

The river lamprey is distributed along the Pacific Coast of North America from the Sacramento River, California north to Tee Harbor near Juneau, Alaska (map 4.37) (Kan 1975). In the Columbia River, the river lamprey has been reported from Bonneville Dam and locations downstream (Kan 1975). Very little is known on the status of this species.

Habitat Relationships

River lampreys require small, clear water streams for spawning, with ammocoetes (the larval form) living in silty backwaters of such streams (Moyle and others 1989). Known hosts for adults include smelt, herring, kokanee salmon, and coho salmon (Kan 1975). In Canada, the adults begin their spawning migration in freshwater between September and late winter after they have spent two years in the ocean (Beamish 1980). They spawn the following April to June, and subsequently die. At the appropriate age, the young begin to metamorphose in July but do not emigrate to the ocean until the following May to July (Beamish 1980).

Key Factors Influencing Status

Relative to Pacific lamprey, the river lamprey's tendency to use areas close to the coast may protect it to some degree from mortality associated with mainstem dam passage. The limited distribution of river lamprey relative to Pacific lamprey may increase their vulnerability. Information to assess that or other risks is not available.

Pacific Lamprey (*Lampetra tridentata*)

The Pacific lamprey is an anadromous and parasitic lamprey widely distributed along the Pacific Coast of North America and Asia. Traditionally, Pacific lamprey were an important ceremonial and subsistence resource for native peoples. In the assessment area, they occur in all waters that remain accessible to salmon and steelhead (Simpson and Wallace 1978). The Idaho Department of Fish and Game lists Pacific lamprey as a state endangered species.

Distribution and Status

Within the Columbia River Basin, Pacific lamprey are believed to have migrated to all waters accessible to anadromous salmonids (Simpson and Wallace 1978). Kan (1975) suggested that the primary consideration for presence of lamprey was access rather than distance from the ocean. Historic runs were large; in some years up to 400,000 lampreys were counted as they migrated past Bonneville Dam (Harrison 1995). Counts of lamprey passing Ice Harbor Dam totaled 40 in 1993 and 399 in 1994, compared with the 1960s when nearly 50,000 were counted annually (Harrison 1995).

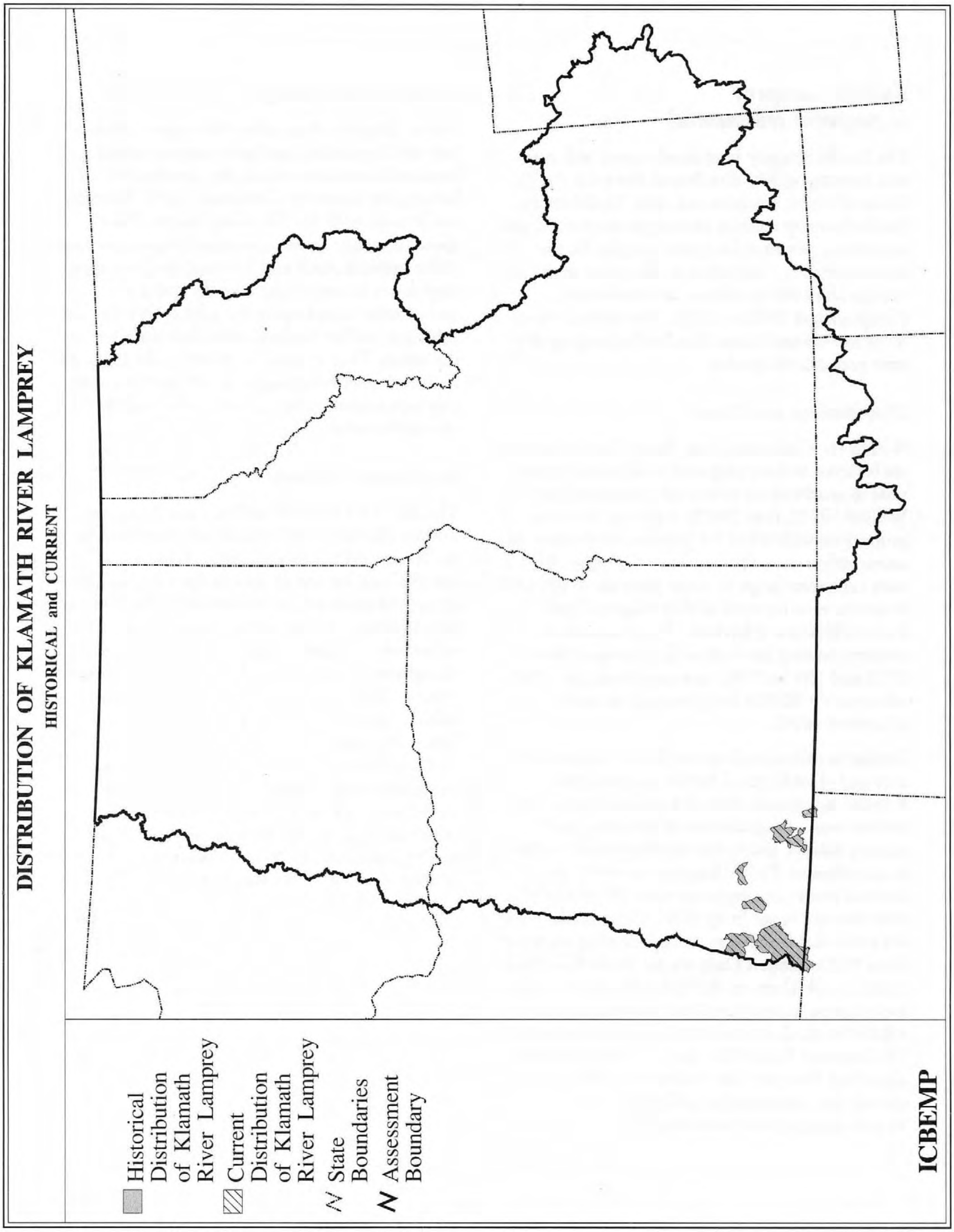
Similar to other anadromous fishes, the distribution and abundance of Pacific lamprey has been reduced by construction of dams and water diversions as well as degradation of spawning and rearing habitat. Using the distribution of steelhead as an indicator, Pacific lamprey currently are blocked from entering more than 50 percent of their historic range (map 4.38). Consequently they are precluded from large areas, including upstream from Hell's Canyon Dam on the Snake River and Chief Joseph Dam on the Columbia River. Landlocked populations have been found in areas from which the anadromous form has been precluded (Wallace and Ball 1978), but they have not persisted and Beamish and Northcote (1988) concluded that metamorphosed lamprey were unable, in such areas, to survive to maturity.

Habitat Relationships

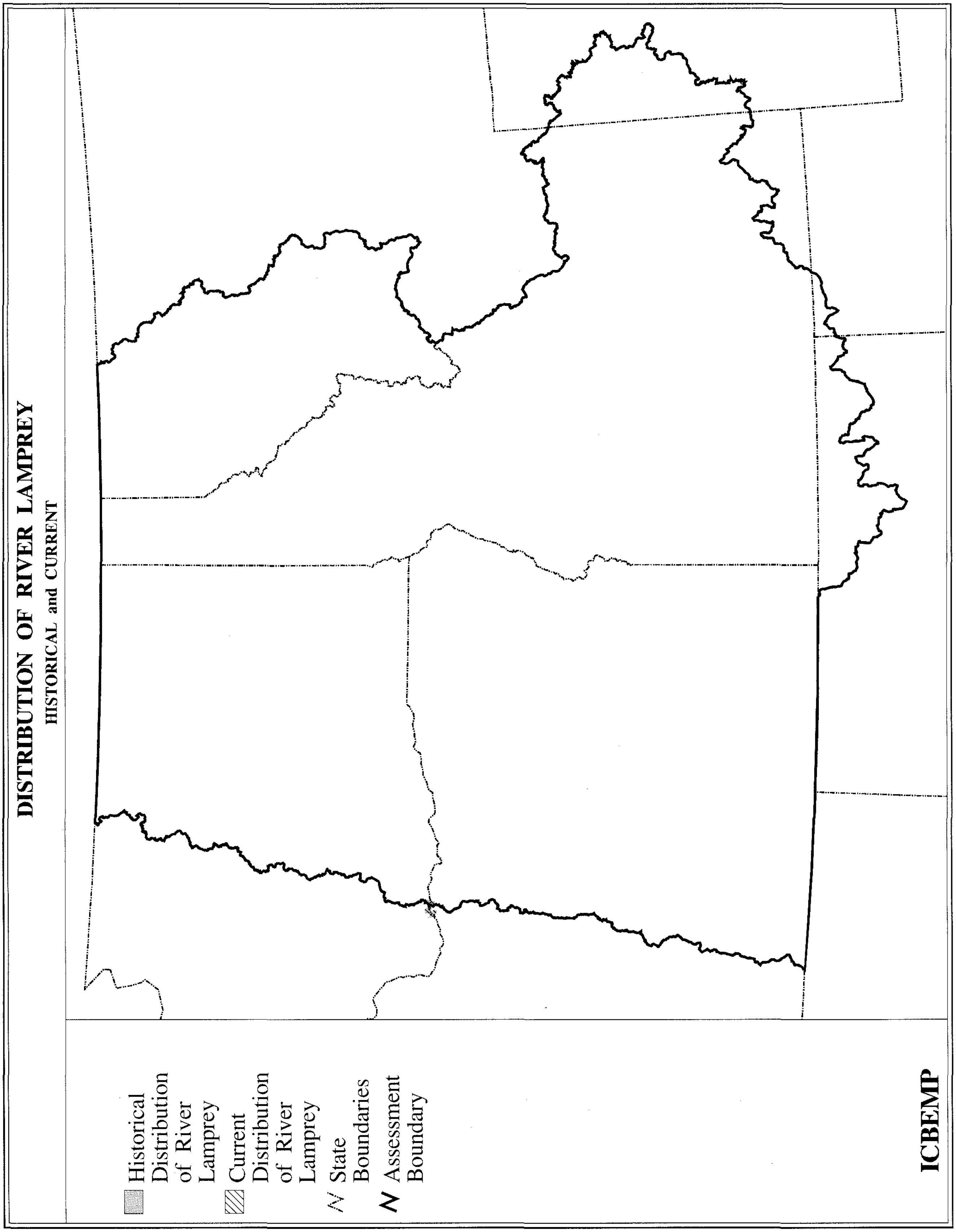
Pacific lamprey adults enter freshwater between July and September, and may migrate several hundred kilometers inland (for details of life history, see Scott and Crossman 1973). They do not mature until the following March. They spawn in sandy gravel immediately upstream from riffles between April and July and die soon after. Eggs hatch in two to three weeks and the ammocoetes spend up to the next six years in soft substrate as filter-feeders before they emigrate to the ocean. They remain in the ocean for 12 to 20 months before returning to freshwater to spawn. Diatoms appear to be a primary food supply for the ammocoetes.

Key Factors Influencing Status

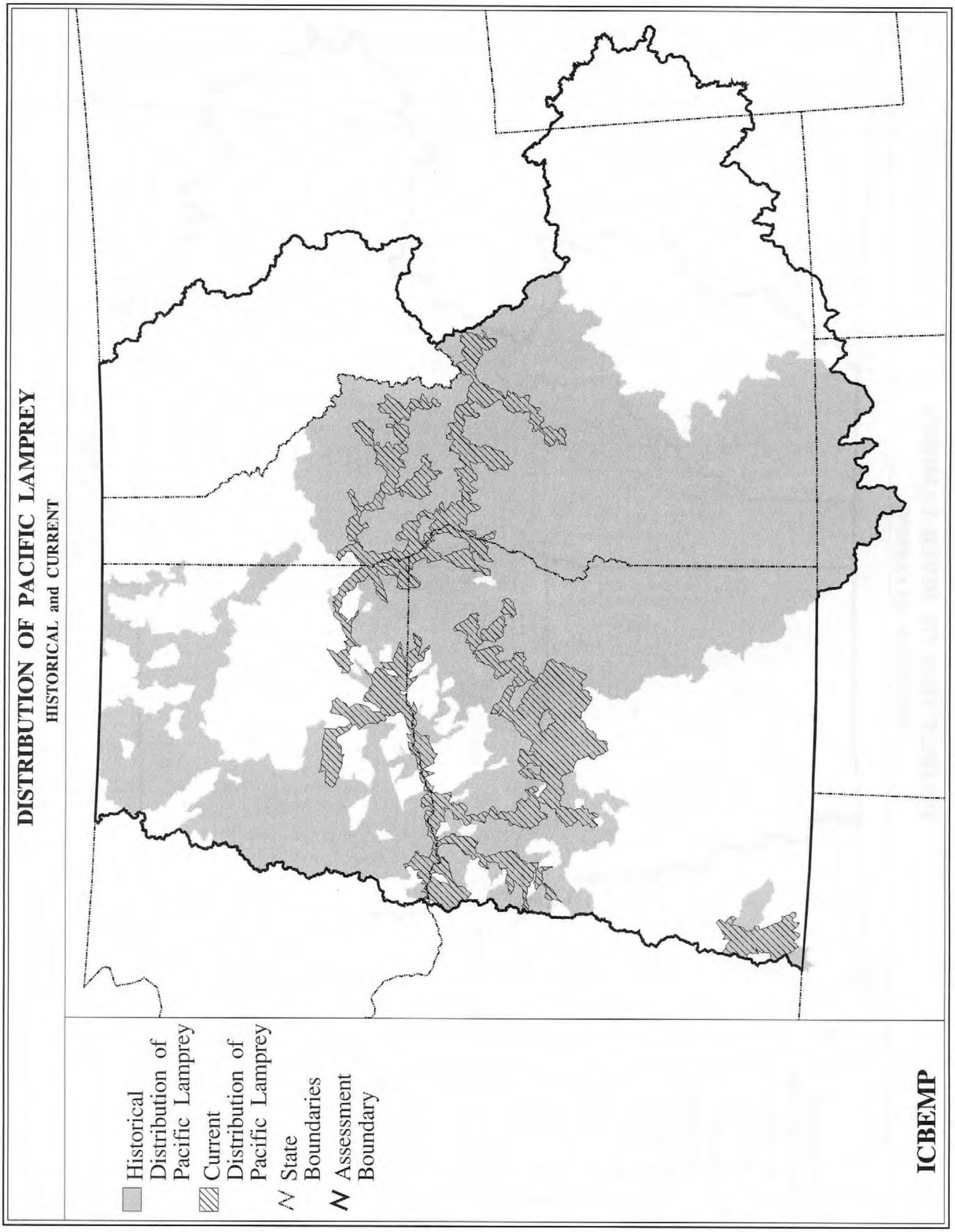
The Idaho Chapter of the American Fisheries Society (IDAFS 1995) concluded that dams on the Snake and Columbia rivers, alteration of streams, and harvest of ammocoetes by bait fisheries are the most serious threats to the Pacific lamprey in Idaho. Pacific lamprey, similar to salmonids, are likely to be vulnerable to land disturbances that cause sedimentation in nursery streams. The ammocoetes depend on quality habitat in freshwater for up to six or seven years before they emigrate to the ocean. Such an extended period in freshwater makes them especially vulnerable to degraded stream conditions. Their anadromous life-history necessitates maintenance of access to spawning and rearing areas. Water quality consistent with robust diatom production may be a key factor for their continued success.



Map 4.36—Historical and current distribution of Klamath lamprey.



Map 4.37—Historical and current distribution of river lamprey.



Map 4.38—Historical and current distribution of Pacific lamprey.

Goose Lake Lamprey (*Lampetra tridentata* ssp.)

The Goose Lake lamprey is considered a subspecies of parasitic Pacific lamprey that were isolated in the Goose Lake basin during the early Pleistocene (Moyle and others 1989). Although it is a State of Oregon sensitive species, and a State of California species of special concern, it currently receives no special management attention.

Distribution and Status

The species is unique to the Goose Lake Basin of Oregon and Washington (map 4.39). In California, they are reported from Lassen and Willow creeks. In Oregon, they were found in Drews Creek, Drews Reservoir, Thomas Creek, Cottonwood Creek, and Cottonwood Reservoir (ODFW 1992). They also occur in Goose Lake. Beyond this, the extent of their distribution and abundance is unknown.

Habitat Relationships

Specific habitat requirements of the Goose Lake lamprey are unknown. The parasitic adults live for a time in Goose Lake, a shallow (less than 7 meters), turbid lake. Lake temperatures range from 1° to 24° C (Johnson and others 1985). They migrate to lake tributaries to spawn. The young remain in the tributaries for an unknown amount of time before they metamorphose and move to the lake. Requirements in nursery streams are thought to include gravel spawning areas and soft substrate for the filter-feeding ammocoetes.

Key Factors Influencing Status

Goose Lake dried in 1992 because of drought. Extended drought could lead to extinction of the species. Water diversions during the period that ammocoetes emigrate to the lake would likewise contribute to the demise of the species. The subspecies does not have a wide distribution and is therefore vulnerable to the effects of small population size and fragmentation. Declines of native hosts, particularly chub, sucker, and redband trout, would likely be detrimental to the lamprey.

Pit-Klamath Brook Lamprey (*Lampetra lethophaga*)

This brook lamprey completes its life-cycle in freshwater and is nonparasitic. It occurs in the Klamath and upper Pit river systems of northern California and south-central Oregon.

Distribution and Status

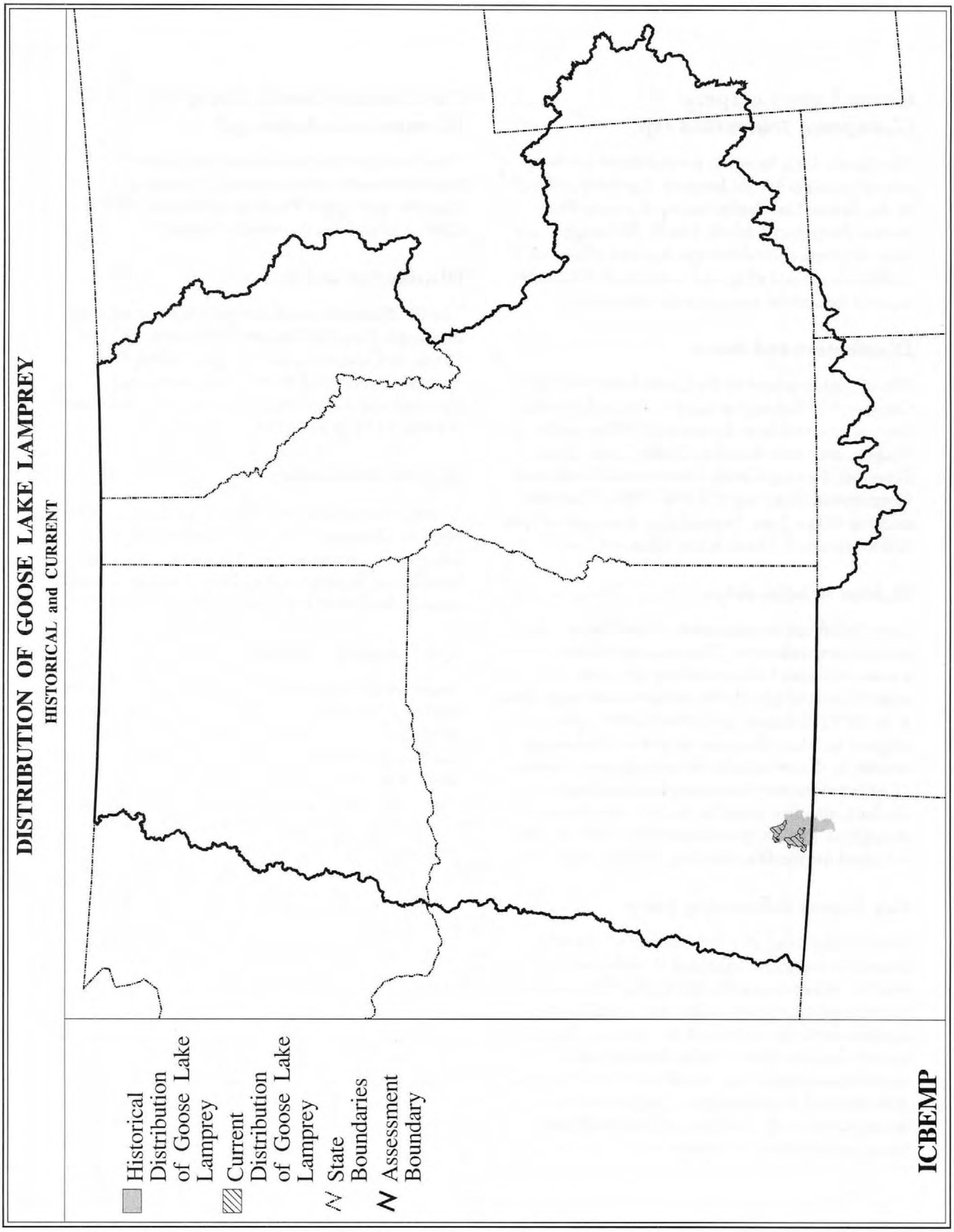
The Pit-Klamath brook lamprey is reported from the North Fork Pit, Pit, and Fall rivers, and Hat Creek, in California; the Sprague, North Fork Sprague, Sycan, and Williamson rivers, and Crooked and Meryl creeks in Oregon (map 4.40) (Hubbs 1971; Kan 1975).

Habitat Relationships

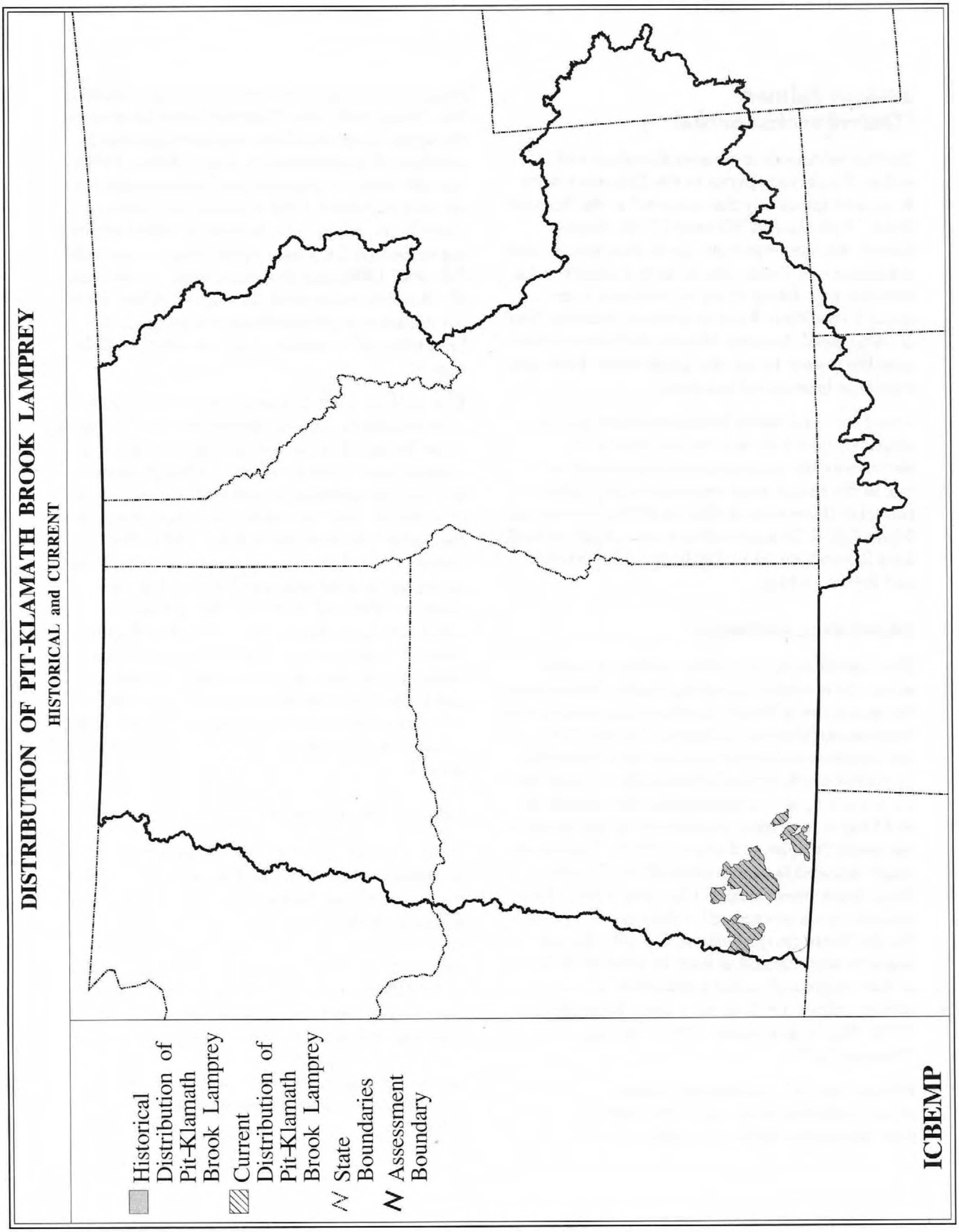
Metamorphosis may occur anytime between May and late October (Kan 1975) and neotenic ammocoetes are common. Spawners have only been found from March to May. The larval period extends for five or six years.

Key Factors Influencing Status

Factors influencing the habitat of the species have not been described. Stream alterations that reduce productivity for diatoms and other food items, and that increase water temperature would be likely to be deleterious for the species. The species is narrowly distributed and is therefore vulnerable to the effects of small population size and fragmentation.



Map 4.39—Historical and current distribution of Goose Lake lamprey.



Map 4.40—Historical and current distribution of Pit-Klamath brook lamprey.

Sockeye Salmon (*Oncorhynchus nerka*)

Sockeye salmon were once an abundant and widely distributed species in the Columbia River Basin and apparently also occurred in the Klamath Basin (Wydoski and Whitney 1979). Sockeye salmon were an important sport, commercial, and tribal resource. Native stocks have declined and a remnant population of sockeye salmon in the upper Snake River Basin in Idaho is federally listed as endangered. Sockeye salmon exhibit two dominant life history forms, the anadromous form and a resident form called kokanee.

Introduced and native kokanee salmon populations that now exist in lakes and reservoirs throughout the assessment area are viewed as one of the Basin's most important sport fishery resources (Rieman and Maiolie 1995; Rieman and Myers 1992). Kokanee salmon are a highly valued food fish and are also a key forage fish (Wydoski and Bennett 1981).

Distribution and Status

The historic range of sockeye salmon extended across the northern rim of the Pacific Ocean down the west coast of North America as far south as the Sacramento River in California (Burgner 1991). Anadromous sockeye salmon are now believed to be extinct south of the Columbia River Basin and the remaining runs represent the most southerly and longest migrating populations of the species in the world (Waples and others 1991a). The historic range included large segments of the Columbia River Basin where natural lakes and surrounding watersheds are connected by river systems to the Pacific Ocean (maps 4.41a and 4.41b). Eleven major watersheds and at least 24 lakes are believed to have supported native populations of sockeye salmon within the Columbia River Basin (Fulton 1970; Waples and others 1991a; Wydoski and Whitney 1979).

Historic runs of anadromous adults at the mouth of the Columbia River may have numbered more than two million before the turn of the century. Numbers are now consistently less than 100,000 fish. Today, only lakes Wenatchee and Osoyoos in the upper Columbia River produce significant numbers of anadromous sockeye (Mullan 1986). A single remnant population of anadromous fish remains in Redfish Lake in the upper Salmon River Basin, Idaho. The number of adults returning to Redfish Lake has ranged from zero to eight fish since 1990, and that population is now federally listed as endangered (Bevan and others 1994). An intensive, captive brood-stock program has been initiated to conserve the remaining population.

The resident form, kokanee salmon, still appears to be widely distributed throughout the Columbia River Basin. All of the historic rearing lakes still support some kokanee salmon although eradication programs and subsequent reintroduction of other stocks may have eliminated most if not all of the native fish from several lakes within the Stanley Basin (Chapman and others 1990). Nonnative kokanee salmon have been widely introduced to lakes and reservoirs throughout the Columbia River Basin. These introduced populations are now far more widely distributed than native populations were historically (map 4.41 a and b). Introgressive hybridization may have compromised the genetic integrity of many populations but no study of such effects has been conducted.

Habitat Relationships

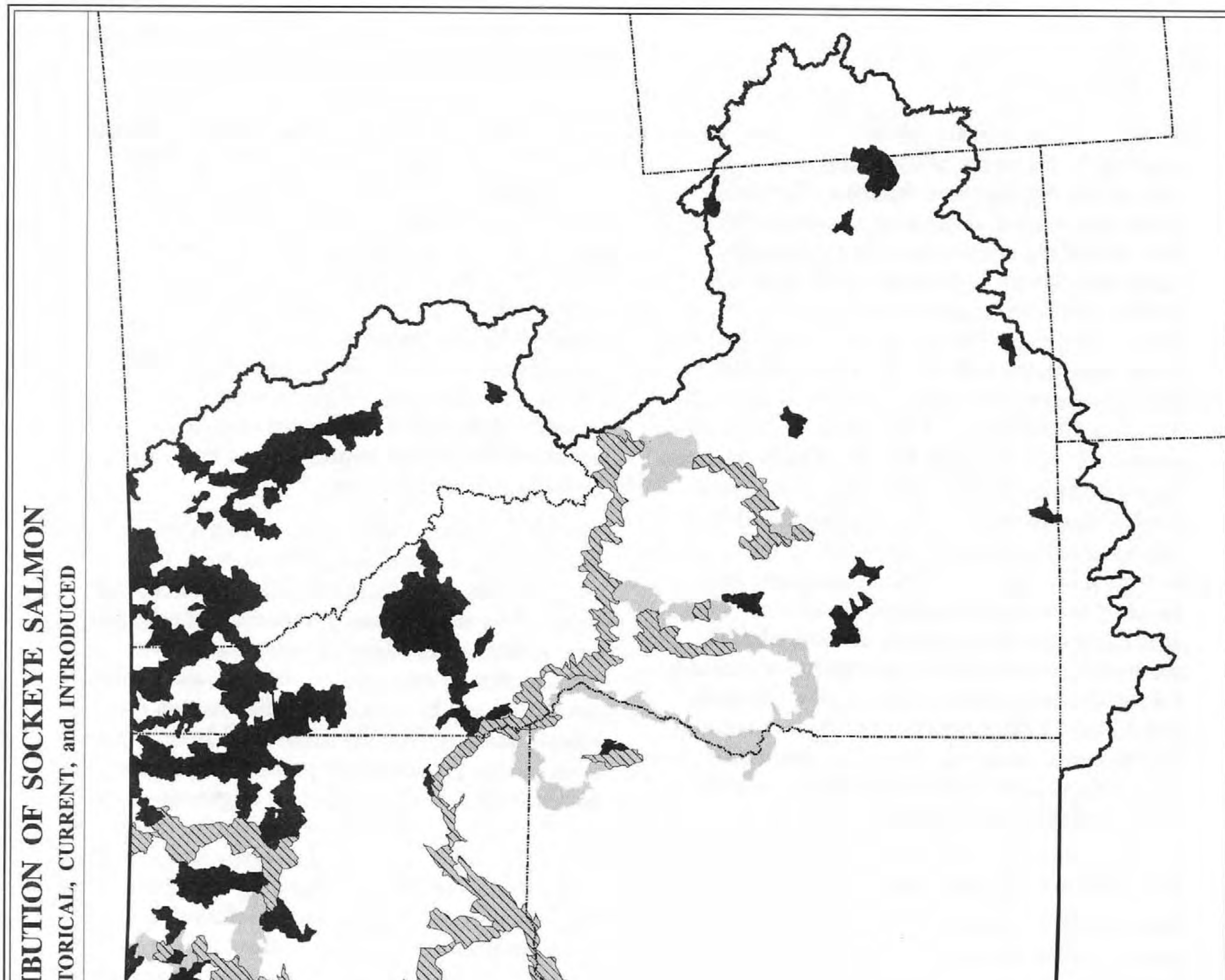
Sockeye salmon exhibit some of the more complex life history patterns among Pacific salmon in that they often rely on both stream and lake environments for early rearing (Burgner 1991). Adults often spawn in lake inlet streams but lake shoal and outlet spawning also occurs. Sockeye salmon in the Columbia River Basin are typically late summer/fall spawners. Juveniles emerge from the gravel the following spring and move into the lake for one to two years where they feed on zooplankton before migration. Kokanee salmon will continue to rear for a total of three to five years before maturing.

Within the two primary life histories, there is wide variation in spawning, rearing, and maturation timing, site selection and duration. The indigenous distribution of kokanee salmon coincides with that of the anadromous form; natural kokanee populations have probably developed repeatedly from anadromous populations and not through dispersal of kokanee from other systems (Foote and others 1989). Non-migratory progenies of anadromous fish are known as residuals (Burgner 1991) and are believed to represent the transitional link between the two dominant forms. Native kokanee salmon still persist throughout much if not most of the historic range and thus represent an important component of the original biological diversity within the assessment area. Residual sockeye and kokanee salmon may also have the potential to support or even refound anadromous forms should change in the available habitat and migratory corridors allow (Rieman and others 1994). Conservation of all remaining native stocks, therefore, should be considered important regardless of whether they currently support anadromous returns.

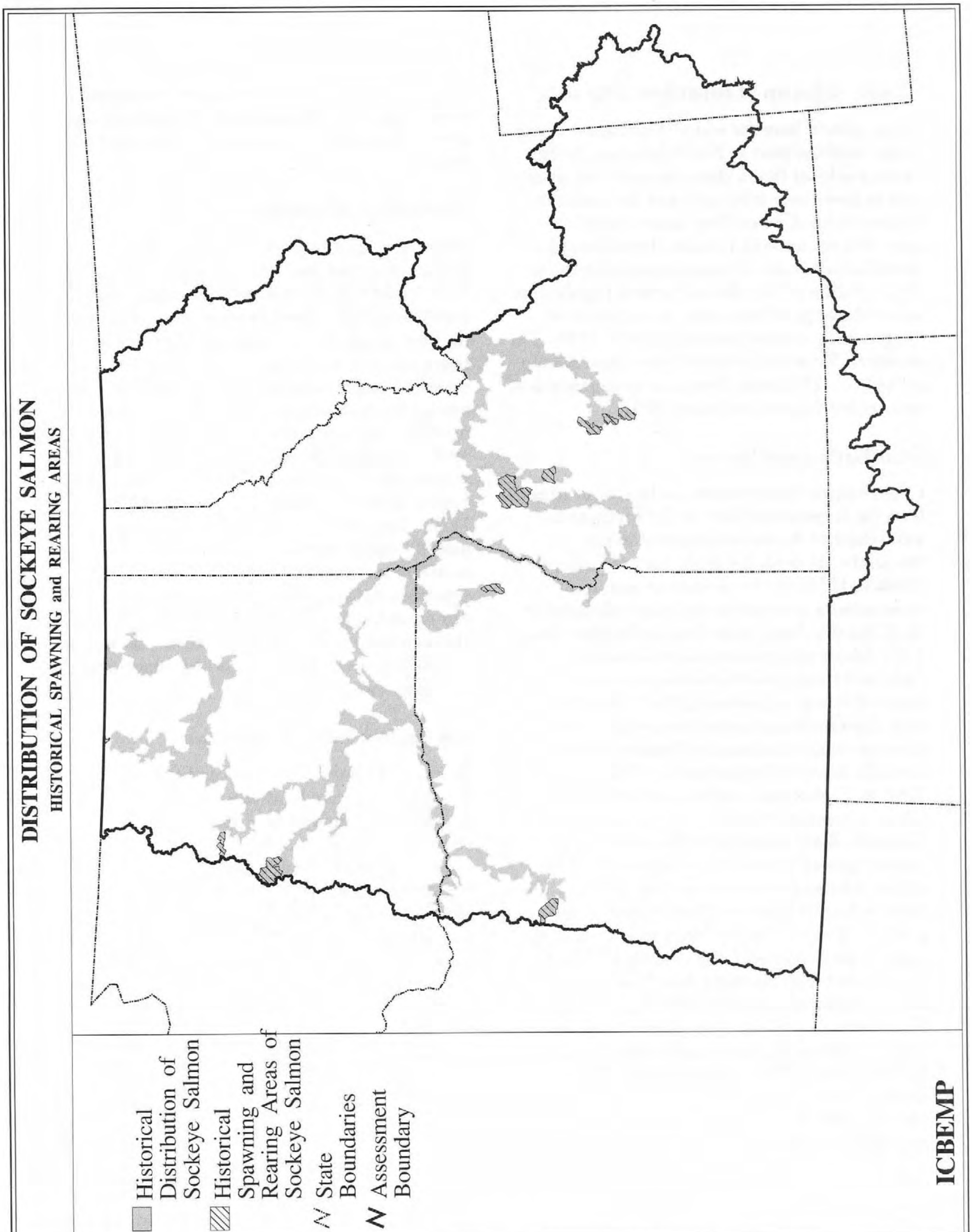
Key Factors Influencing Status

Much of the decline in anadromous sockeye salmon can be attributed to dams blocking access to spawning and rearing streams in the early 1900s and to increased mortality of migrants caused by dams in the migratory corridors of the Snake and Columbia rivers constructed in later years (Fulton 1970; Mullan 1986; Nehlsen 1995). Available lake rearing habitat has been reduced from approximately 86,880 surface hectares to 4,400 surface hectares (Mullan 1986). Numbers of spawning sockeye were also seriously reduced by commercial fisheries around the turn of the century. Although fishing was important historically, there is no longer any legal harvest in the Columbia Basin for anadromous sockeye except a small American Indian fishery in the Priest Rapids, Washington pool and a recreational harvest in Lake Wenatchee, Washington when escapement goals are met. Ocean conditions probably account for some of the variation in annual returns and may influence productivity of stocks over long periods. Forest management may influence the quality of spawning habitats and the productivity of lake environments. Sockeye salmon are likely susceptible to factors that may increase sediment in spawning gravel and scour of redds. Factors influencing rearing lake environments, such as accelerated eutrophication, could also be important. Ultimately, however, it is unlikely that any substantial recovery of historic anadromous populations will occur without major improvements in conditions in the migratory corridors.

Overall, kokanee and sockeye salmon appear to be in little danger of extinction throughout their range. However, individual stocks are susceptible to loss. The anadromous component of sockeye is near extinction in Idaho's Stanley Basin. The introduction of exotic fishes, fishing, and habitat condition may be important influences on the relative productivity and survival of those populations but, in general, most populations appear to be relatively abundant. The loss of genetic integrity in native stocks, however, may represent an important loss of biological integrity and could compromise the potential for long-term persistence of wild populations or the refounding of anadromous runs. A basic inventory of the integrity and status of native stocks would be useful.



Map 4.41a—Historical and current distribution plus introduced sites of sockeye salmon.



Map 4.41b—Historical spawning and rearing areas of sockeye salmon.

Chum Salmon (*Oncorhynchus keta*)

Chum salmon have the widest distribution of any of the Pacific salmon in North America. In the Columbia River Basin, chum salmon were abundant in lower river tributaries and the mainstem Columbia River, where they supported tribal, sport, and commercial fisheries. Populations had dwindled to a point of minor importance by the 1950s (Fulton 1970). Because several populations are on the verge of extinction, the species is an Oregon state sensitive species (ODFW 1990). The American Fisheries Society lists the chum salmon in the lower Columbia River as at moderate risk of extinction (Nehlsen and others 1991).

Distribution and Status

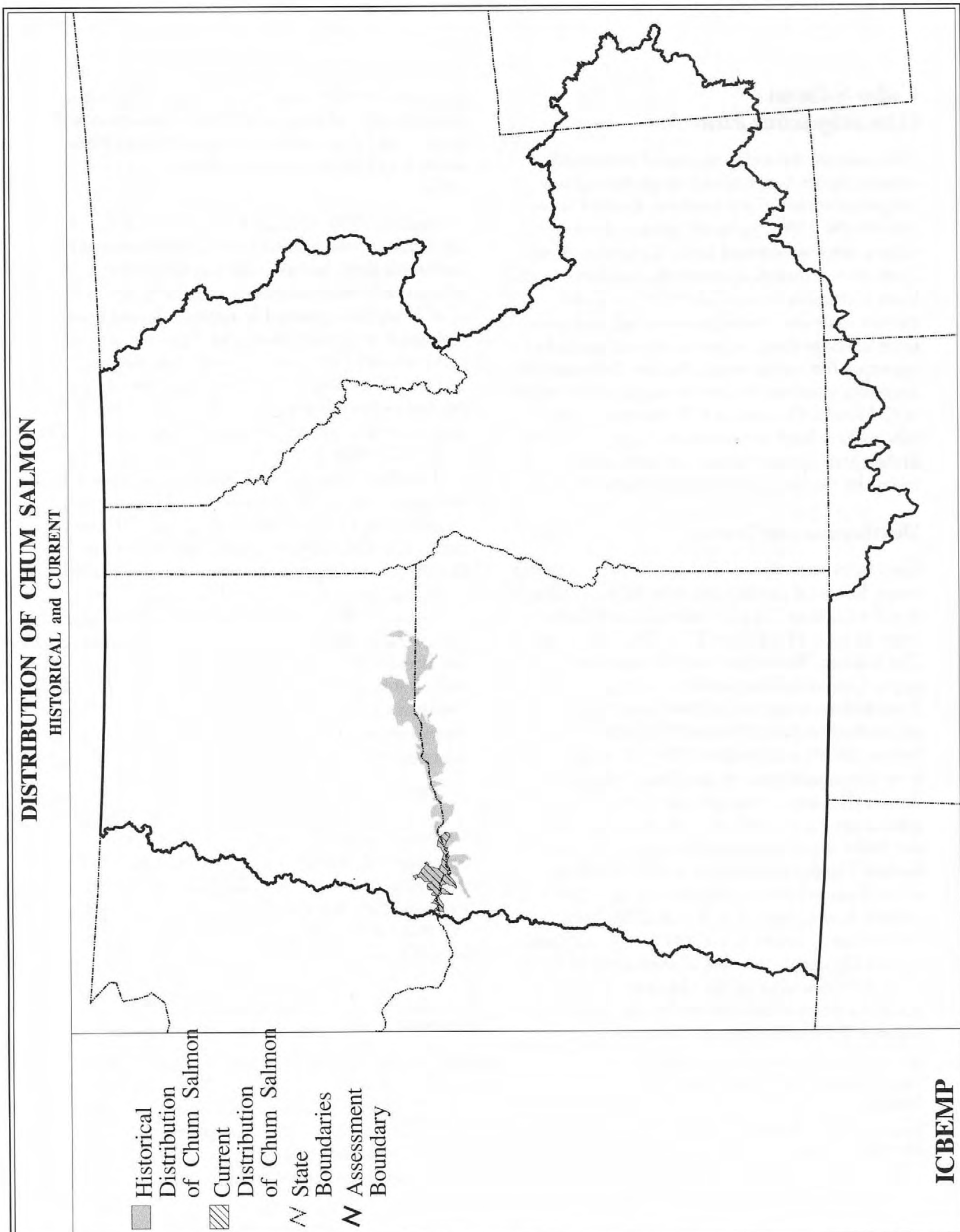
Chum salmon have spawned and reared in streams from the Sacramento River in California to the arctic shore of Alaska and eastward to the Mackenzie River on the arctic coast of Canada (Bakkala 1970). In the assessment area, most chum salmon spawned in the lower tributaries of the Columbia River below Bonneville Dam (map 4.42). Major wild production areas include Grays Basin and Hardy and Hamilton creeks in Washington (Howell and others 1985a). Chum salmon were also distributed above Bonneville Dam in the lower portions of tributaries upstream to the Umatilla River in Oregon and the Walla Walla River in Washington (Nelhsen and others 1991). Chum salmon are essentially extinct in the Columbia River Basin above Bonneville Dam. Former spawning areas above Bonneville Dam included lower portions of the Little White Salmon River; Hamilton, Rock, and Herman creeks; and areas along the margins of river banks in the main Columbia River (Fulton 1970). Chum salmon are currently found only in the lower sections of tributaries entering the Columbia River below Bonneville Dam (Howell and others 1985a). Historical commercial harvest ranged from 450,000 to 3.9 million kilograms, peaking in 1928 at about 700,000 fish (Chaney and Perry 1976). Since 1978, commercial harvest has not exceeded 2,000 fish (ODFW and WDF 1981). Chum salmon have also experienced similar declines coast-wide. No significant hatchery production of chum salmon occurs in the Columbia River.

Habitat Relationships

Most chum salmon spawn within the lower reaches of streams and frequently within the tidal zone. In the Columbia River Basin, spawning usually occurred a short distance upstream from the head of tidewater, in margins of mainstem rivers, and at the mouths of lower tributary streams. Clean, abundant gravel in these areas is needed for successful spawning. Water temperatures for spawning range from 4° to 16° C (Neave 1966). Eggs hatch between about 1.5 and 4.5 months after fertilization. Survival from egg to fry stage is usually less than 10 percent (Bakkala 1970). Survival is related to flows and temperatures during incubation. The greatest cause of egg mortality is fluctuating streamflows, dislodged eggs from shifting gravels, and sediments deposited on and in gravels impeding intragravel flows. Unlike most other anadromous salmonids, chum salmon fry enter saltwater soon after emergence and form schools in estuaries.

Key Factors Influencing Status

Dramatic declines in run size can be attributed to siltation and stream blockages from logging activities and inundation of spawning areas following construction of Bonneville Dam (Fulton 1970). Poor condition of chum populations is attributed to their sensitivity to poor water quality primarily caused by habitat degradation and loss from forest and agricultural practices, urbanization, and pollution; incidental over-harvest in mainstem fisheries directed at coho and chinook salmon; and competition with hatchery fish in streams (Nehlsen and others 1991).



Map 4.42—Historical and current distribution of chum salmon.

Coho Salmon
(*Oncorhynchus kisutch*)

Coho salmon are native to coastal streams of western North America and range throughout temperate waters of the northern Pacific Ocean. Prior to the 1900s, naturally produced coho salmon were widespread in the Columbia River basin, with a historical center of abundance in the lower Columbia River (Fulton 1970). Coho salmon supported tribal, commercial, and sport fisheries. Now there are essentially no naturally spawning fish in the basin. Because their naturally spawning numbers are low throughout their range in California, Oregon, and Washington, coho salmon have been petitioned for listing under the Endangered Species Act and are now under status review by the National Marine Fisheries Service.

Distribution and Status

Early literature, reports, and accounts reveal some major historical production areas for coho salmon in the middle and upper Columbia and Snake rivers (map 4.43) (Fulton 1970; Mullan 1984). The Yakima, Wenatchee, and Methow rivers in the upper Columbia River and the Grande Ronde River system in the Snake River were major natural production areas (Fulton 1970; Mullan 1984). Fulton (1970) and Mullan (1984) also report some substantial runs in the Hood, John Day, Entiat, Okanogan, and Spokane rivers. Other reports appear to verify that historic coho runs in the Snake River also occurred in the Clearwater, Lochsa, Touchet, Tucannon, and Walla Walla rivers (Fulton 1970; Schoning 1947b).[1,2] In the Grande Ronde River, the Wenaha, Wallowa, Minam, and Lostine rivers, and Catherine Creek historically supported runs of coho salmon. Other accessible tributaries in the Columbia and Snake River systems with low gradients and sufficient water in September through November probably supported small runs of coho salmon. The longest distance coho salmon are known to have migrated in the Columbia River was to the Spokane River, about 1,125 kilometers from the ocean (Fulton 1970).

Chapman (1986) estimated the peak 5-year average of adult coho salmon that annually entered the Columbia River before 1900 was 618,000 fish, of which somewhere around 30 percent or upwards of 200,000 fish spawned in the middle and upper Columbia and Snake rivers (Mullan 1984). Wahle and Pearson (1987) indicated that less than 25,000 coho salmon were spawning naturally in the Columbia River Basin by the early 1980s. This number remained the same into the late 1980s (CBFWA 1990). Naturally spawning fish included feral hatchery fish that spawned in streams near hatcheries, returns from hatchery outplants to streams away from hatcheries, and naturally produced fish. Currently, natural coho salmon production (including hatchery spawning fish) in the Columbia River Basin above Bonneville Dam occurs in the Wind, Big White Salmon, Klickitat, Yakima, and Wenatchee rivers in Washington and the Hood, Deschutes and Grande Ronde rivers and Lindsey, Viento, Mosier, Chenowith and Mill creeks in Oregon (Johnson and others 1991). The current abundance is less than 6 percent of historic abundance (Johnson and others 1991).

The in-river harvest of Columbia River coho salmon reached a peak of about 880,000 fish in 1925 (Beiningen 1976). Thereafter, an almost-continuous decline in the fishery occurred until the 1960s, when a widespread hatchery enhancement program was initiated. Since the 1960s, the vast majority of coho salmon returning to the Columbia River have been of hatchery origin (CBFWA 1990).

Habitat Relationships

Habitat requirements for coho salmon were reviewed by Bjornn and Reiser (1991). Coho salmon migrate upstream at water temperatures ranging from 7.2° to 15.6° C. Spawning and egg incubation occur at water temperatures from 4° to 13° C. Adults spawn in as sorted gravels. After

[1]Also, personal communication. 1995. Stacy Gebhards, Idaho Department of Fish and Game, Boise, Idaho.

[2]Also, personal communication. 1995. Chuck Huntington, Clearwater BioStudies, Inc., Portland, Oregon.

emergence, coho fry move into available rearing space, especially areas with lower velocities, such as stream margins and off-channel habitat. As fry grow, they rear in deeper pool habitats. Preferred summer water temperatures for rearing are 12° to 14° C, with upper lethal temperatures near 26.0° C. During winter rearing, young coho salmon seek areas with more cover, such as woody debris, root wads, and overhanging vegetation and lower velocity areas such as side channels, sloughs, and beaver ponds.

Key Factors Influencing Status

Human activities have profoundly influenced the past and present status of coho salmon in the mid and upper Columbia and Snake River systems. These activities have resulted in lost spawning and rearing habitat, migration delays, passage mortality, predation, diseases, water pollution, genetic introgression, and overfishing (Horner and Bjornn 1981). Habitat degradation has resulted from water withdrawals for irrigation, increased water temperatures caused by removal of riparian vegetation, and excessive sediment and simplification of stream channels from accelerated land use activities. The National Marine Fisheries Service has essentially determined that mid and upper Columbia and Snake River wild stocks of coho salmon are below levels necessary for long-term survival (Johnson and others 1991). Commercial exploitation, dams, and degradation of spawning and rearing habitat, beginning around the 1890s, dramatically reduced natural spawning populations and resulted in extirpations in much of the historical range. All middle and upper Columbia River and Snake River runs were drastically reduced or destroyed by construction of impassable mill dams, unscreened irrigation diversions, habitat loss, and over-harvest, prior to completion of the Grand Coulee Dam in 1941 (Mullan 1984). However, the advent of extensive artificial propagation in hatcheries "rebuilt" some runs to or above historic levels. In the process, coho salmon runs once dominated by naturally spawning fish have changed to predominately hatchery-maintained runs.

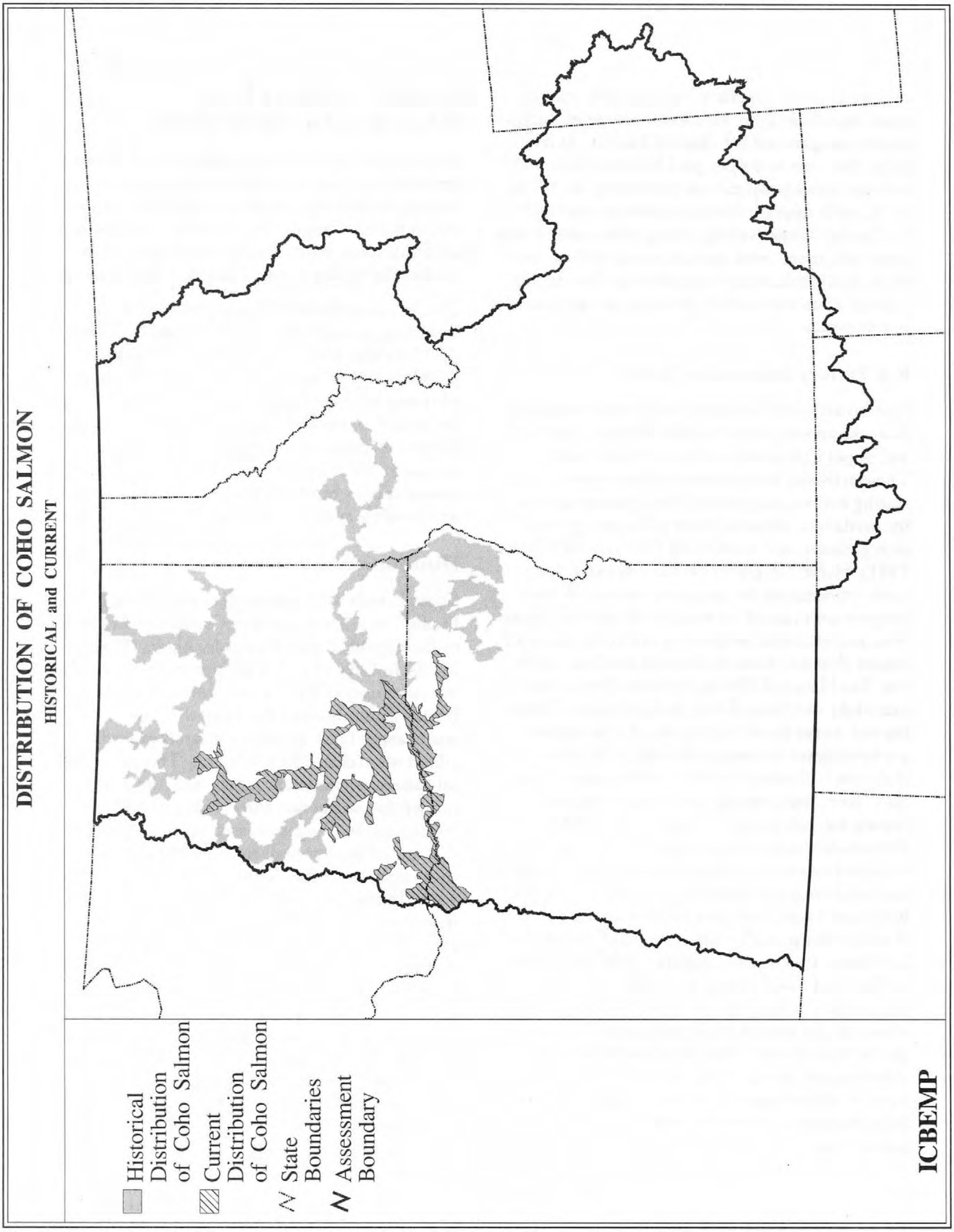
Coastal Cutthroat Trout (*Oncorhynchus clarki clarki*)

Historically abundant and widespread, the coastal cutthroat trout is particularly sensitive to environmental disturbance and is now in decline throughout its historic range (Trotter 1989). The coastal cutthroat trout includes both anadromous and resident forms that support popular sport fisheries.

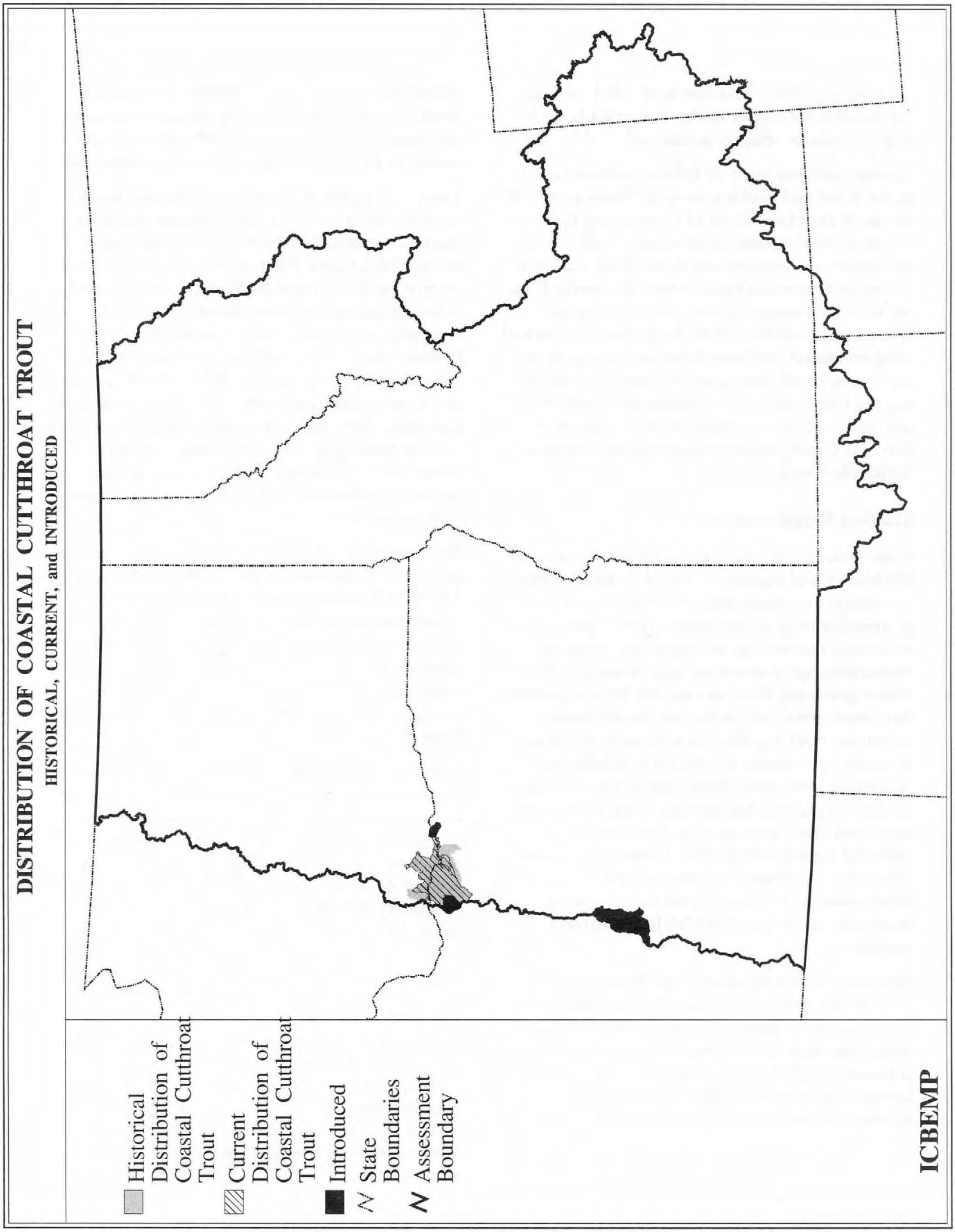
The American Fisheries Society identified all remaining populations of coastal cutthroat trout in the Columbia River as at high risk of extirpation (Nehlsen and others 1991). In July 1994, the National Marine Fisheries Service proposed to list the coastal cutthroat trout in the Umpqua Basin, Oregon as threatened. Listing was finalized in August 1996. Also in Oregon, it is listed as a state critical species by the Oregon Department of Fish and Wildlife and a sensitive species by the BLM.

Distribution and Status

Coastal cutthroat trout are distributed along the Pacific Coast from northern California's Eel River to Gore Point, Kenai Peninsula, Alaska (Behnke 1979). In Oregon and Washington, they extend to the east slopes of the Cascade Mountains and in British Columbia and Alaska to the crest of the coast range. Their distribution rarely extends inland more than 160 kilometers. This geographical pattern corresponds closely with the distribution of the coastal rain forest belt in the Pacific Northwest. In Oregon, coastal cutthroat trout are distributed in almost all rivers from the Winchuck River north into the Columbia River system (map 4.44). In the Columbia River Basin, coastal cutthroat trout resided in tributaries east to Fifteenmile Creek in Oregon and Rock Creek in Washington, including the Willamette Basin to its headwaters. The abundance of sea-run cutthroat trout in the lower Columbia Basin appears to have significantly declined in recent years. Although these populations are not routinely monitored, angler surveys have shown decreases from catches of around 5,000 fish in the 1970s to catches as low



Map 4.43—Historical and current distribution of coho salmon.



Map 4.44—Historical and current distribution plus introduced sites of coastal cutthroat trout.

as 50 in the 1980s. Effective as of 1994, all wild cutthroat trout caught in the Columbia River by anglers must be released unharmed.

Sea-run cutthroat trout are believed to be extirpated in the Wind and Klickitat rivers of Washington. The stocks in the Hood River of Oregon and Rock Creek in Washington are listed as at high risk of extinction. All populations in small tributaries in the lower Columbia Basin below Bonneville Dam are identified as at moderate risk of extirpation (Nehlsen and others 1991). Both anadromous and resident coastal cutthroat trout are present in the mainstem Hood River and its tributaries, including the East Fork. In the Fifteenmile Creek drainage, cutthroat trout are known to be present in Fivemile Creek and are thought to be present in Eightmile Creek.

Habitat Relationships

Coastal cutthroat trout exhibit diverse patterns in life history and migration. Populations of coastal cutthroat trout show marked differences in their preferred rearing environments (river, lake, estuary, or ocean), size and age at migration, timing of migrations, age at maturity, and frequency of repeat spawning. Four distinct life history patterns have been described for the subspecies: anadromous (sea-run) populations migrate to the ocean or estuary for usually less than a year before returning to freshwater; fluvial populations undergo in-river migrations between small spawning tributaries and main river sections downstream; adfluvial populations migrate between spawning tributaries and lakes or reservoirs; and nonmigratory (resident) forms occur in small headwater streams and exhibit little instream movement.

Spawning occurs in riffles 15 to 45 centimeters deep in pea-sized, clean gravel in low gradient stream reaches. Spawning sites are located near or in shallow areas of deep pools close to escape cover (Hunter 1973). Newly emerged fry move into low-velocity stream margins, backwaters, and side-channels adjacent to main channel pools and riffles (Moore and Gregory 1988). Selection of small tributaries for spawning and first-year rearing serves to isolate sea-run cutthroat trout and minimize their interactions with other salmonids.

There is evidence of negative interactions among juvenile cutthroat trout, coho salmon, and steelhead during this life history stage (Bisson and others 1988; Glova 1984, 1986) which may limit sea-run cutthroat trout population size. Generally in spring, young fish move downstream into mainstem rivers and, with the onset of winter freshets, often move back upstream into smaller tributaries. In the Columbia River, smolts remain in a large estuary during their first migration. The Columbia River plume provides a large, near-shore area for feeding in the ocean. Some sea-run cutthroat trout overwinter in salt water, but most return to freshwater in the same year they migrate (Johnston 1981).

Resident cutthroat trout are primarily drift feeders and reside at the head of pools (Wilzbach and Hall 1985). In forested streams, large woody debris creates productive habitat by forming pools, meanders, secondary channels, and undercut banks (Bisson and others 1987). Pools formed by large wood support more and older resident cutthroat trout, and also provide winter refugia during high flows (House 1995).

Campton (1981) and Campton and Utter (1987) found genetic differences among coastal cutthroat trout stocks at four locations in Washington. If this genetic variation holds true throughout the subspecies range, then the total subspecies gene pool could be composed of literally hundreds of genetically distinct breeding units (Trotter and others 1993).

Key Factors Influencing Status

There are three main factors that contribute to coastal cutthroat trout declines (Trotter and others 1993): present or potential destruction, modification, or blockage of habitat by logging, urban and rural development, or mainstem passage; overharvest from recreational fishing; and negative interactions with hatchery stocks and/or introduced species.

Lahontan Cutthroat Trout (*Oncorhynchus clarki henshawi*)

The Lahontan cutthroat trout is native to the Pleistocene Lake Lahontan Basin of northwestern Nevada, northeastern California, and a small adjacent portion of southeastern Oregon. This subspecies is federally listed as threatened pursuant to the Endangered Species Act throughout its native range. It has been introduced elsewhere in southeastern Oregon and eastern Washington.

Distribution and Status

Lahontan cutthroat trout are native to the McDermitt Creek drainage, a Quinn River tributary of the larger Lahontan Basin, in southeastern Oregon (map 4.45). In 1991, genetic analyses of cutthroat trout in Willow and Whitehorse creeks, also in southeastern Oregon, found the inhabitants to be genetically indistinguishable from *O. c. henshawi* (Williams 1991). During the 1970s, trout from Willow and Whitehorse creeks were introduced into Denio, Van Horn, Pike, Mosquito, Little McCoy, Big Alvord, Little Alvord, Cottonwood, and Willow creeks in the adjacent Alvord Basin. Surveys conducted in 1991 confirmed that Lahontan cutthroat trout still persist in many Alvord Basin streams.[3] Trout from Willow and Whitehorse creeks were also introduced into Guano Creek in 1957 (Hanson and others 1993). Lahontan cutthroat trout also have been introduced into Oregon's Mann Lake although recent information questions their genetic purity.[4] In addition, the Washington Department of Fisheries has widely introduced Lahontan cutthroat trout into eastern Washington.

[3]Personal communication. 1995. Wayne Bowers, Oregon Department of Fish and Wildlife, Hines, Oregon. Personal communication of unpublished data.

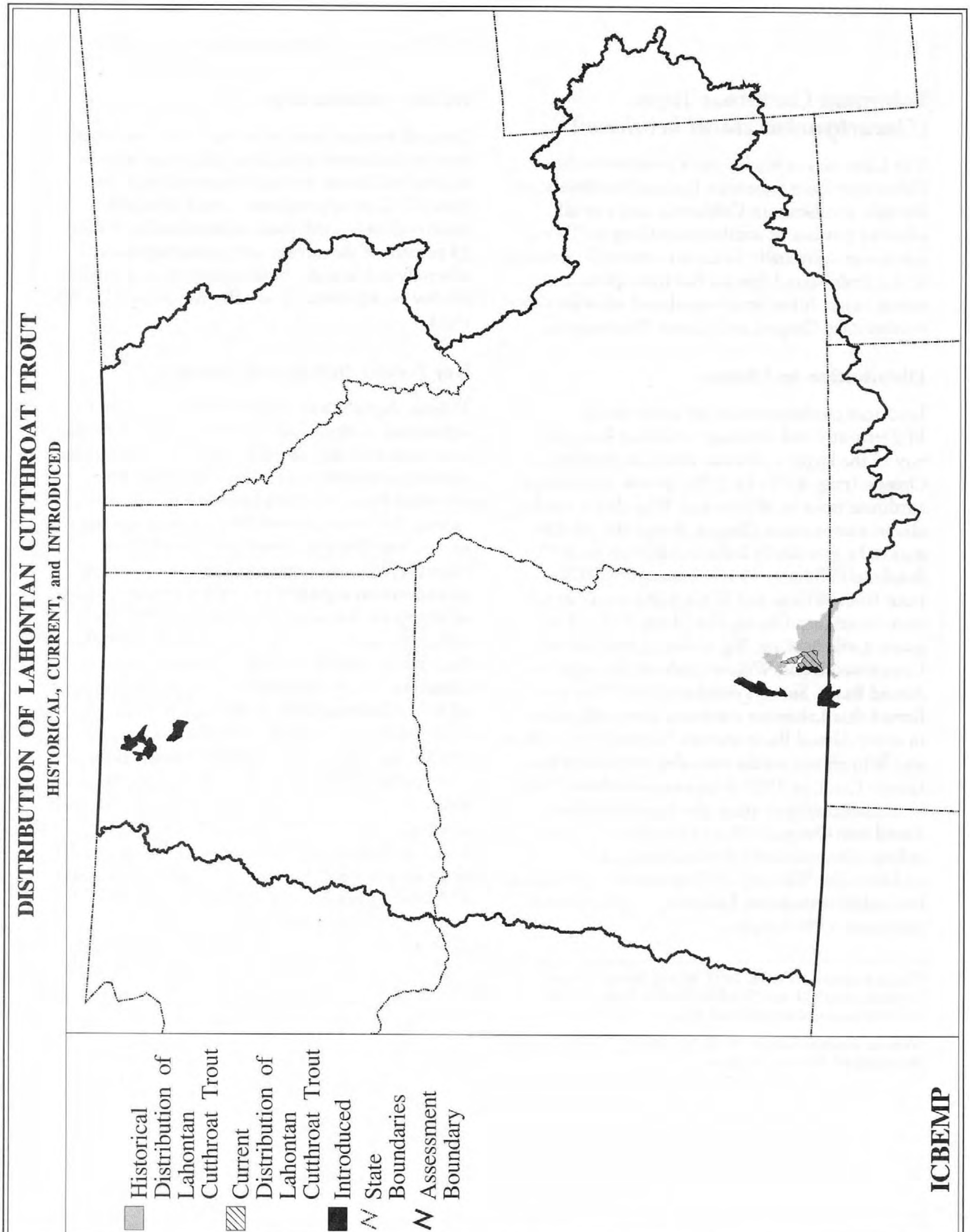
[4]Personal communication. 1995. Ron Wiley, Bureau of Land Management, Portland, Oregon.

Habitat Relationships

Optimal riverine habitat for Lahontan cutthroat trout is characterized as clear, cold water with an average maximum summer temperature of less than 22° C; an approximate 1-to-1 pool:riffle ratio; well-vegetated, stable stream banks; at least 25 percent of the stream area providing cover; a relatively stable water flow regime; and a relatively silt-free rocky substrate in riffle-run areas (USFWS 1993a).

Key Factors Influencing Status

Habitat degradation, especially loss of riparian vegetation, is identified as a key factor in declining Oregon stream populations. Loss of vegetation has resulted in stream temperatures that have far exceeded those considered optimal for the subspecies. Dissolved oxygen levels in such reaches are too low. Drought conditions coupled with extremely low temperatures during winter have caused stream segments to freeze completely. Loss of vegetation has resulted in the loss of forage organisms and cover (Hanson and others 1993). Excessive turbidity and sedimentation also contribute to habitat degradation problems because of their effects on food production, spawning areas, and feeding ability (Hanson and others 1993). Water diversions and the introductions of non-native salmonids are also key factors. Because native populations of Lahontan cutthroat trout in southeastern Oregon are naturally small and isolated, they are at risk. Many introduced populations are especially vulnerable because the number of founding fish was less than 30.



Map 4.45—Historical and current distribution plus introduced sites of Lahontan cutthroat trout.

Sunapee Char (*Salvelinus alpinus oquassa*)

The Sunapee char, a landlocked Arctic char endemic to Sunapee Lake, New Hampshire, was extirpated from New Hampshire in the mid-1900s as a result of competition with introduced species (Kircheis and others 1995). In 1925, prior to extirpation from Sunapee Lake, Sunapee char were introduced into two high elevation and fishless lakes in Idaho's Sawtooth Mountain Range (Kircheis and others 1995) where they established reproducing populations. The fish stocked in Idaho were not "rediscovered" until 1978 (IDFG 1991). To help protect the taxon, no further information on the location of introduction sites is provided.

The American Fisheries Society listed the Sunapee char and blueback trout (a phenotypically distinct form endemic to Maine) as threatened (Williams and others 1989). Because the subspecies is introduced, Idaho does not list it as a state species of special concern. However, protection of the fish and its habitat is suggested as a priority because of its unique genotype (IDFG 1991).

Sunapee char differ from other North American lacustrine char in their unique coloration, large size, and use of a mid-lake spawning shoal (Kircheis and others 1995). Kircheis and others (1995) recently examined the genetic identity of 17 Sunapee char collected from one Idaho lake. The samples displayed four haplotypes, three of them uniquely divergent from other landlocked arctic char and the other similar to a blueback char haplotype. The authors reported that the divergent portion of the Idaho char population represented a distinct, native population formerly in Sunapee Lake. Despite the unique haplotype, Kircheis and others (1995) recommended against using Idaho arctic char to restock New Hampshire waters because of the potential for dilution of the gene pool by brook trout, which were stocked in the Idaho lakes in the 1940s. The authors, however, presented no data assessing the degree of introgression with brook trout.

Pygmy Whitefish (*Prosopium coulteri*)

The pygmy whitefish may have been widely distributed across North America during the last ice age, however, once the glaciers receded only isolated populations remained. Little is known about the species in the southern range of its distribution, including Washington, Idaho, and Montana. The pygmy whitefish is listed as a Washington state monitor species.

Distribution and Status

The pygmy whitefish has a discontinuous distribution in the Columbia River system. No reports have found this species to be present below Grande Coulee Dam on the Columbia River (map 4.46). In Montana, the pygmy whitefish is found in Bull Lake and Lake MacDonald and their tributaries, Flathead, Little Bitterroot, Ashley, Swan and Seeley lakes and Hungry Horse Reservoir. In Washington, the fish has been reported in Diamond Lake near Spokane and Lake Roosevelt. In Idaho, the pygmy whitefish occurs in Lake Pend Oreille and Priest Lake. No estimates of abundance are available.

Habitat Relationships

The pygmy whitefish inhabits lakes and cold streams. All reported data are from lakes. In Canada, fish were found in depths ranging from 4.6 to 36.6 meters (McCart 1963). The upper limit may coincide with the depth of the warmer epilimnial water. Shallower distributions in some lakes may be the result of low dissolved oxygen levels on the bottom (McCart 1963). There is no evidence of horizontal or vertical movements or change in depth distribution in pygmy whitefish with seasonal changes (Scott and Crossman 1973).

Key Factors Influencing Status

Little is known about the distribution and abundance of the species or its ecology. One factor that may influence species abundance is chemical treatment of lakes. Increased eutrophication of lake environments, through nutrient additions from increased human development and sediment input from watershed practices, could negatively influence the species persistence. Lack of information on species distribution, abundance, and ecology at the southern edge of its range leads to a cautionary approach. If pygmy whitefish are isolated to a few lakes in the Columbia River Basin, species persistence is at risk.

Burbot (*Lota lota*)

The burbot, a freshwater member of the cod family and a popular sport fish, has a holarctic distribution that includes Alaska, much of Canada, and the northern portion of the United States from the Columbia River Basin to Maine (Scott and Crossman 1973). Within the assessment area, they are native to the Kootenai and Columbia rivers, and scattered deep lakes of eastern Washington, which represent the southern extent of their distribution in the Pacific Northwest. Burbot in the Kootenai River, Idaho, are listed as threatened by the State and as sensitive by Region 1 of the Forest Service.

Distribution and Status

In Washington, Wydoski and Whitney (1979) reported burbot from the Columbia River and deep lakes in the Columbia River Basin, including Kachess, Keechelus, Banks, and Cle Elum (map 4.47). In Idaho and the Columbia River drainage of Montana, burbot are native only to the Kootenai River system, where they provided a popular winter game fishery until numbers began to decline in 1965 (Simpson and Wallace 1978). According to the Idaho Chapter of the American Fisheries Society (IDAFS 1995), Idaho Fish and Game personnel could fill trap nets set for burbot and individual anglers could catch up to 50 per night during the 1950s. Traditional Idaho spawning tributaries included Deep, Snow, Caribou, Parker, Smith, Mission, Boundary, and Myrtle creeks. Stocking of burbot in Clark Fork River, Montana, has resulted in introduced populations in the Clark Fork drainage of Idaho and Montana as well as in Lake Pend Oreille.

Habitat Relationships

Burbot prefer clear, cold water (15.6° to 18.3° C) lakes or large rivers. In summer, they move into deeper pools or hypolimnion areas of lakes. Spawning occurs in midwinter, typically in water between 0.6° and 1.7° C. Young burbot may occur in shallow areas of lakes, large rivers, or smaller cold water streams. As burbot increase in size, they prefer deeper water and become nocturnal.

Key Factors Influencing Status

Populations of burbot in the Kootenai River dropped precipitously in numbers following completion of Libby Dam in Montana and associated changes in the river's hydrograph (IDAFS 1995). In the Pacific Northwest, dam construction typically leads to collapse of burbot populations below the dams, although populations may persist in the reservoirs. Impoundments and changes in river hydrographs appear to disrupt historic spawning patterns and result in reduced recruitment (IDAFS 1995). Most burbot populations in the Columbia River drainage suffer from reduced recruitment, low population densities, and fragmented populations.

Sand Roller (*Percopsis transmontana*)

The sand roller is an uncommon, poorly understood species that is endemic to larger rivers of the Columbia River system. It is a species of special concern in Idaho, and a monitor species in Washington.

Distribution and Status

Historic distribution has not been fully documented but sand roller were known to occur from the Columbia River at Horseshoe Island Slough (about 40 kilometers from the mouth of the Columbia River) upstream to West Bar south of Wenatchee on the Columbia River and in the Snake River to the lower Clearwater River in Idaho (map 4.48) (Gray and Dauble 1979). The sand roller is currently uncommon throughout its range. Within its historic range in the assessment area, it occurs in the Yakima River upstream to Ellensburg and the Umatilla and Walla Walla rivers (Pratt and Whitt 1952). It is unlikely that the fish presently occurs in Idaho. Because of its seclusive behavior during the day, the sand roller is a difficult species to sample and may be more abundant than it appears. Little information is available to assess its current range and population size.

Habitat Relationships

Sand rollers seem to prefer large rivers in areas with low water velocities. Because of their seclusive nature, large numbers of sand rollers have only been observed when they enter shallow open waters to feed at night. During the day, they inhabit quiet areas near exposed roots and undercut banks with depths greater than 15 meters.

Key Factors Influencing Status

Several non-native predators now occupy waters in the historic and current ranges. Walleye, smallmouth bass, channel catfish, and native squawfish are known to prey on sand rollers. For instance, sand rollers were found to be the primary food of walleyes between 350 and 550 millimeters in length (Poe and others 1991). Alterations of larger rivers may degrade habitat preferred by the native species.

Northern Roach (*Hesperoleucus symmetricus mitrulus*)

The northern roach is native to Oregon and California and was classified as a subspecies of the California roach in 1948 (Murphy 1948). The California roach occurs in the Sacramento River Basin (Moyle 1976). The northern roach is currently listed as sensitive by the BLM and as sensitive (peripheral/rare) in Oregon.

Distribution and Status

Historically, the northern roach was found in the upper Pit River system and tributaries to Goose Lake (map 4.49) (Moyle 1976). In the Goose Lake Basin, the fish was known only from tributaries in the Oregon side, although the actual extent of their historic distribution is poorly known. They were known to occur in Cottonwood, Drews, and Muddy creeks and were thought to be introduced in Deep and Mud creeks in the Warner Valley (ODFW 1992). Outside the Goose Lake Basin, the distribution of the northern roach has been reduced in the Pit River system (Moyle and Daniels 1982). Surveys conducted in 1983 and 1988 revealed a limited distribution of the northern roach, with populations still occurring in Thomas and Dent creeks and possibly Cottonwood Creek.[5] Northern roach have also been introduced as bait fish into Summer and Harney lakes and tributaries of the Chewaucan River (Bills 1977). The northern roach was abundant in Spring Creek, tributary to Cottonwood Creek, but is rare in other areas. Presently, populations are considered stable (ODFW 1992).

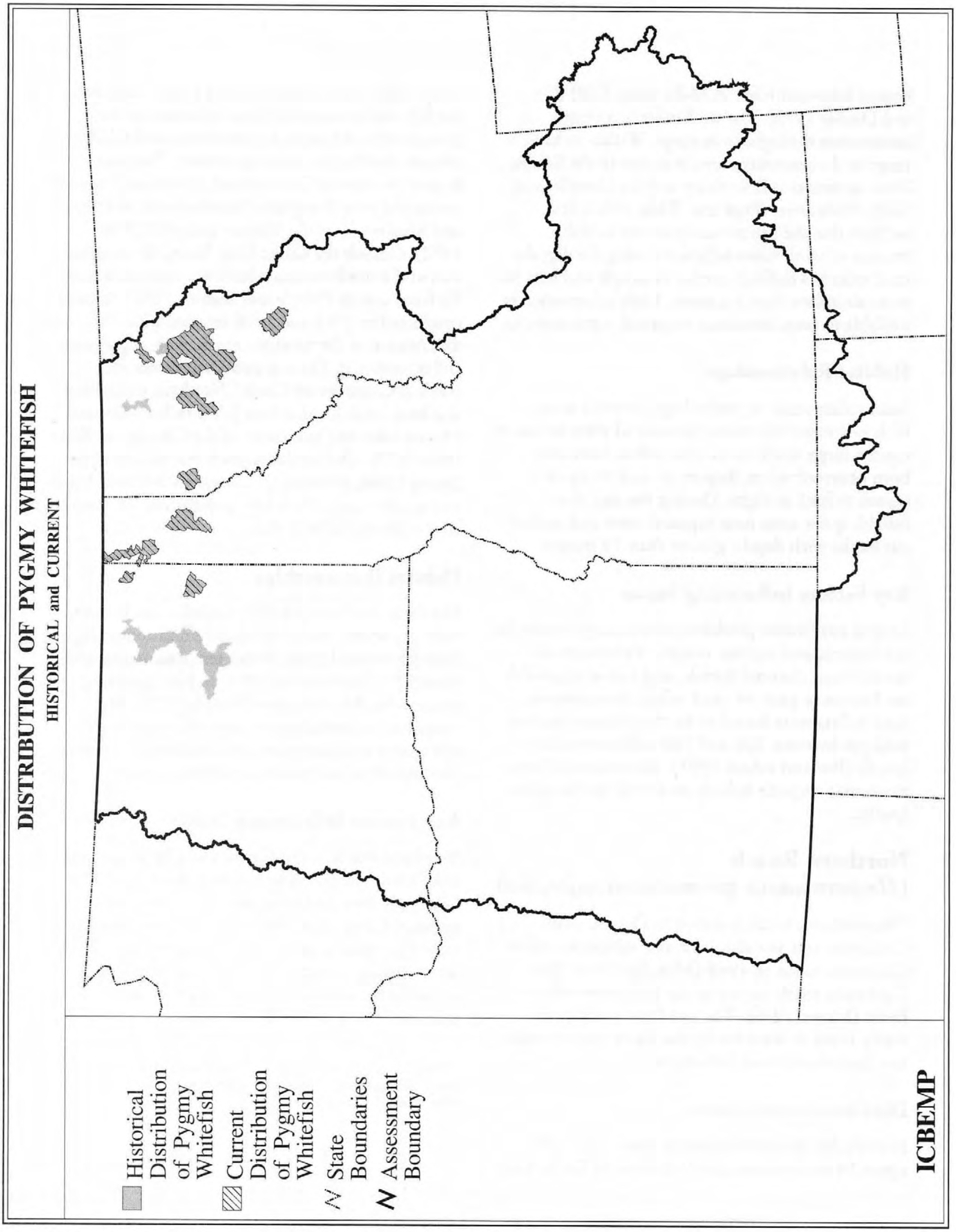
Habitat Relationships

Northern roach are generally found in small, intermittent streams with silty bottoms and containing frequent isolated pools. Water temperatures ranged from 5.5° C in March to 29° C in July. Spawning occurs from March to June (Moyle 1976). Fry remain in interstitial spaces until they are free-swimming and then move into shallow areas with moderate flow and gravel or rubble substrate.

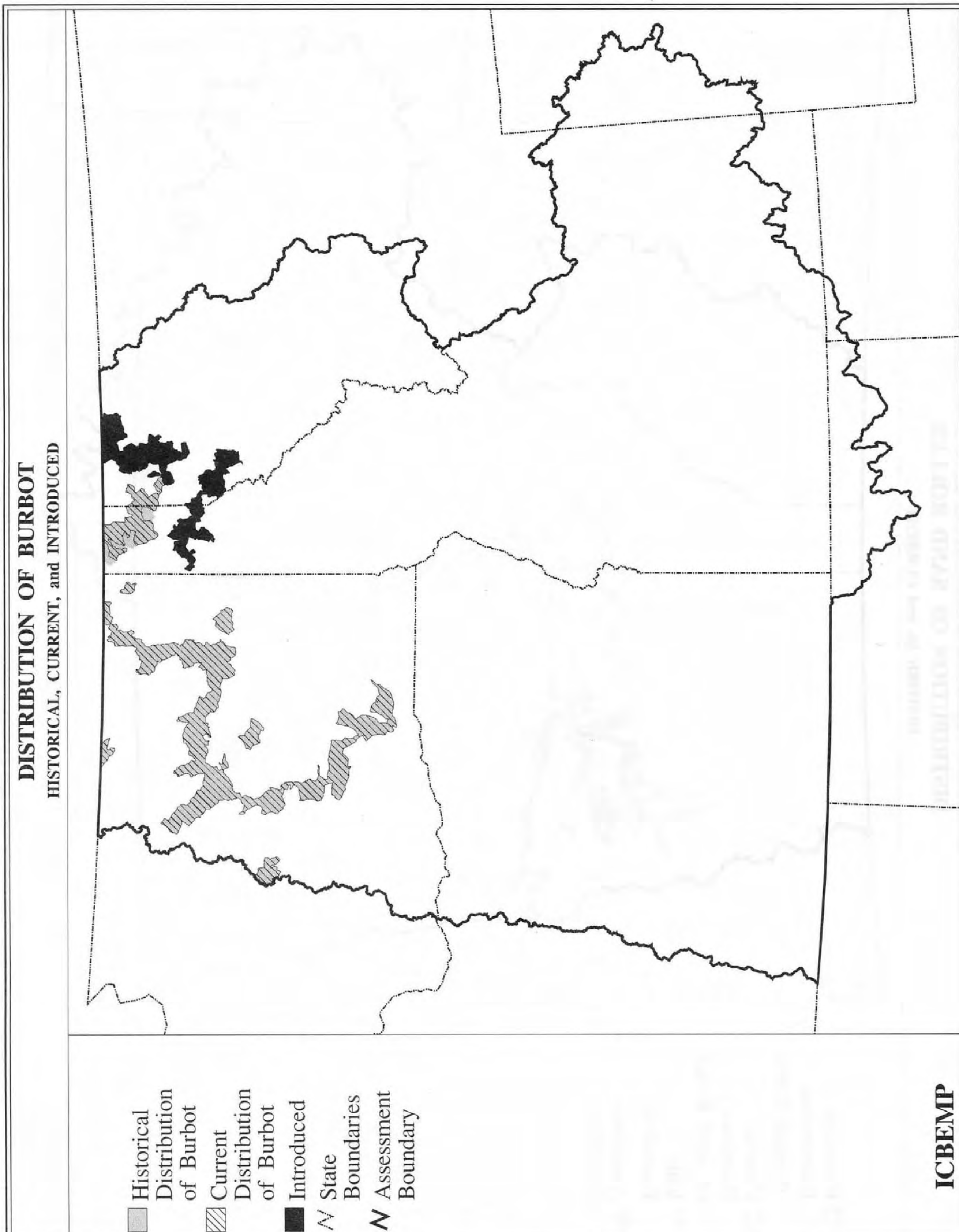
Key Factors Influencing Status

Northern roach in the Goose Lake Basin are very rare. Their rarity and apparent lack of use of the lake limit their ability to recolonize extirpated streams. Conditions that would negatively affect existing stream populations include loss of surface water during extreme drought, timing of water diversions, construction of reservoirs, and introduction of non-native fishes.

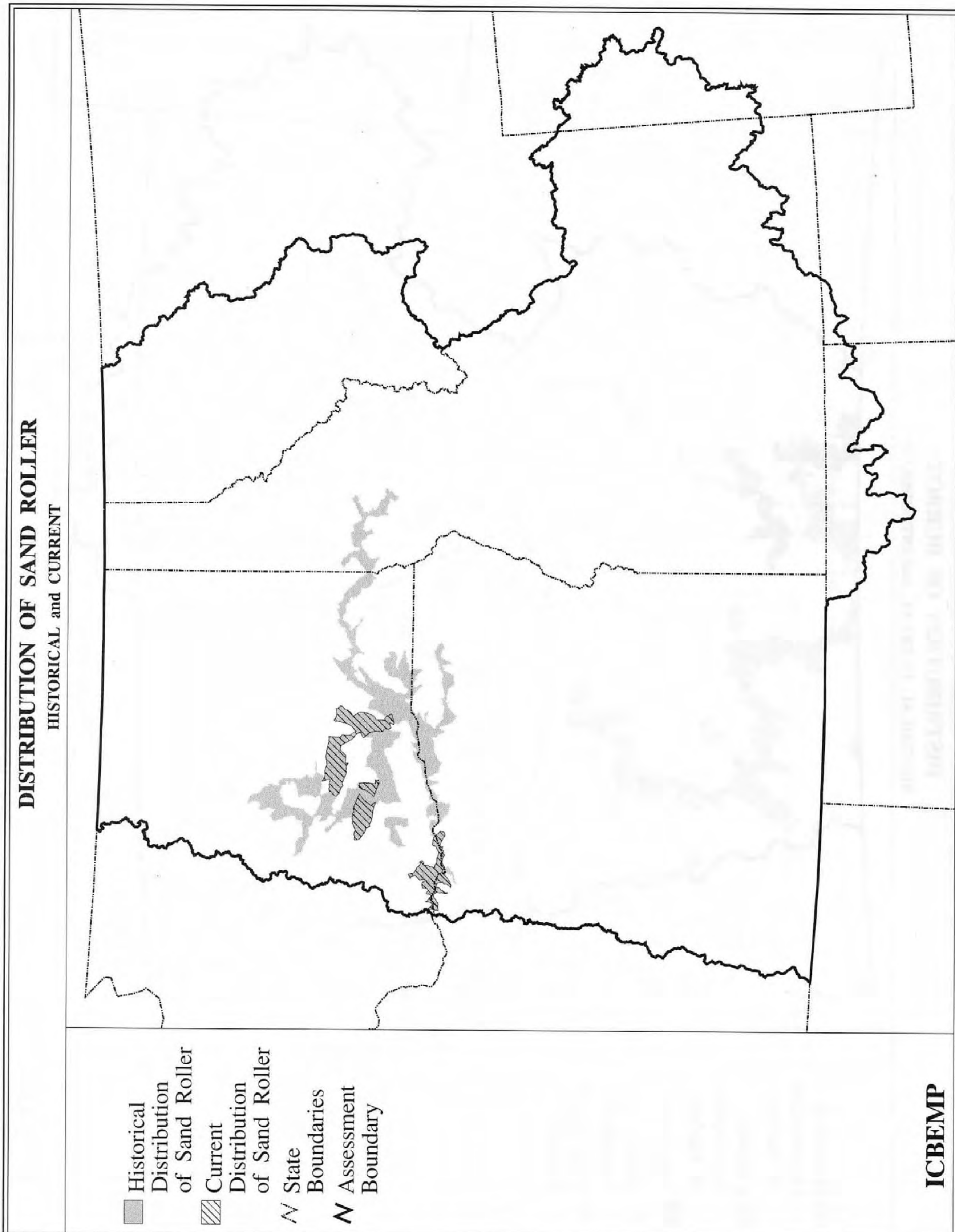
[5]Personal communication. 1995. Jack Williams, Bureau of Land Management, Boise, Idaho. Personal communication of unpublished data.



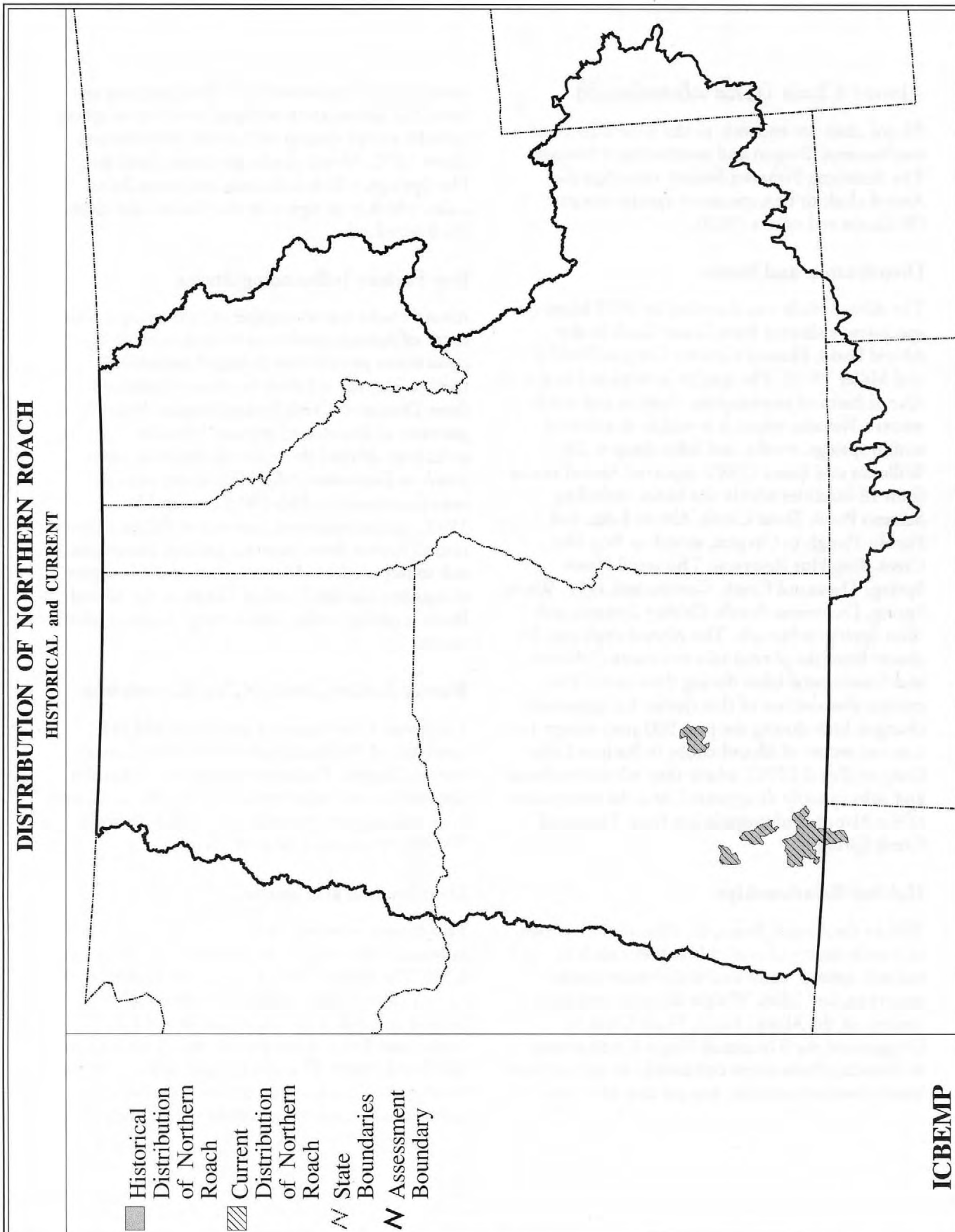
Map 4.46—Historical and current distribution of pygmy whitefish.



Map 4.47—Historical and current distribution plus introduced sites of burbot.



Map 4.48—Historical and current distribution of sand roller.



Map 4.49—Historical and current distribution of northern roach.

Alvord Chub (*Gila alvordensis*)

Alvord chub are endemic to the Alvord Basin of southeastern Oregon and northwestern Nevada. The American Fisheries Society considers the Alvord chub to be a species of special concern (Williams and others 1989).

Distribution and Status

The Alvord chub was described in 1972 based on specimens collected from Trout Creek in the Alvord Basin, Harney County, Oregon (Hubbs and Miller 1972). The species is restricted to the Alvord Basin of southeastern Oregon and northwestern Nevada, where it is widely distributed within springs, creeks, and lakes (map 4.50). Williams and Bond (1983) reported Alvord chubs from 16 localities within the basin, including Serrano Pond, Trout Creek, Alvord Lake, and Pueblo Slough in Oregon, as well as Bog Hot Creek, Bog Hot Reservoir, Thousand Creek Spring, Thousand Creek, Continental Lake, Warm Spring, Dufurrena Ponds, Gridley Springs, and West Spring in Nevada. The Alvord chub may be absent from the pluvial lake remnants of Alvord and Continental lakes during drier years. The current distribution of this species has apparently changed little during the past 100 years except for a recent report of Alvord chubs in Juniper Lake, Oregon (Bond 1974), where they were introduced and subsequently disappeared, and the extirpation of the Alvord chub population from Thousand Creek Spring.

Habitat Relationships

Within the Alvord Basin, the Alvord chub occurs in a wide variety of available habitats such as isolated springs, cool- and warm-water creeks, reservoirs, and lakes. Within the principal creek systems in the Alvord Basin, Trout Creek in Oregon and the Thousand-Virgin Creek system in Nevada, chubs occur commonly in the mid and lower elevation sections, but are rare or absent entirely from high elevations. Within spring systems, the Alvord chub occupies a variety of spring habitats except springs with water temperatures above 31°C. Alvord chubs are absent from Bog Hot Springs, which is fishless, and from Borax Lake, which is occupied by the Borax Lake chub (*G. boraxobius*).

Key Factors Influencing Status

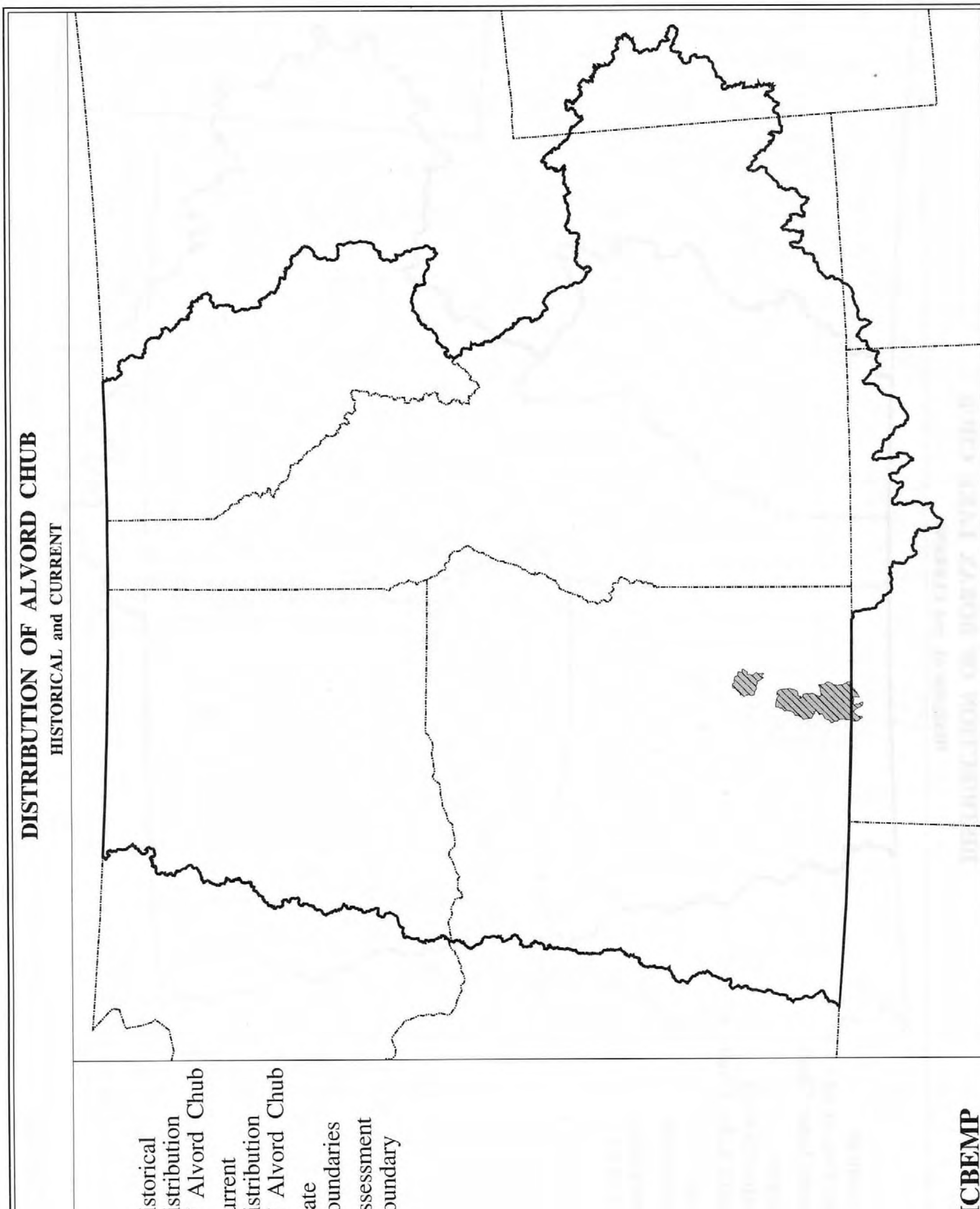
Alvord chubs appear capable of occupying a wide range of habitat conditions as long as relatively clean water persists that is free of introduced species. The Alvord chub has been eliminated from Thousand Creek Spring because of the presence of introduced guppies (*Poecilia reticulata*). Alvord chubs are absent from some ponds at Dufurrena, which are dominated by introduced centrarchids (Williams and Bond 1983). Introductions of non-native fish and diversion of stream flows pose the greatest immediate risk to populations. Maintenance of the integrity of aquifers that feed surface waters in the Alvord Basin is critical to the long-term persistence of this species.

Borax Lake Chub (*Gila boraxobius*)

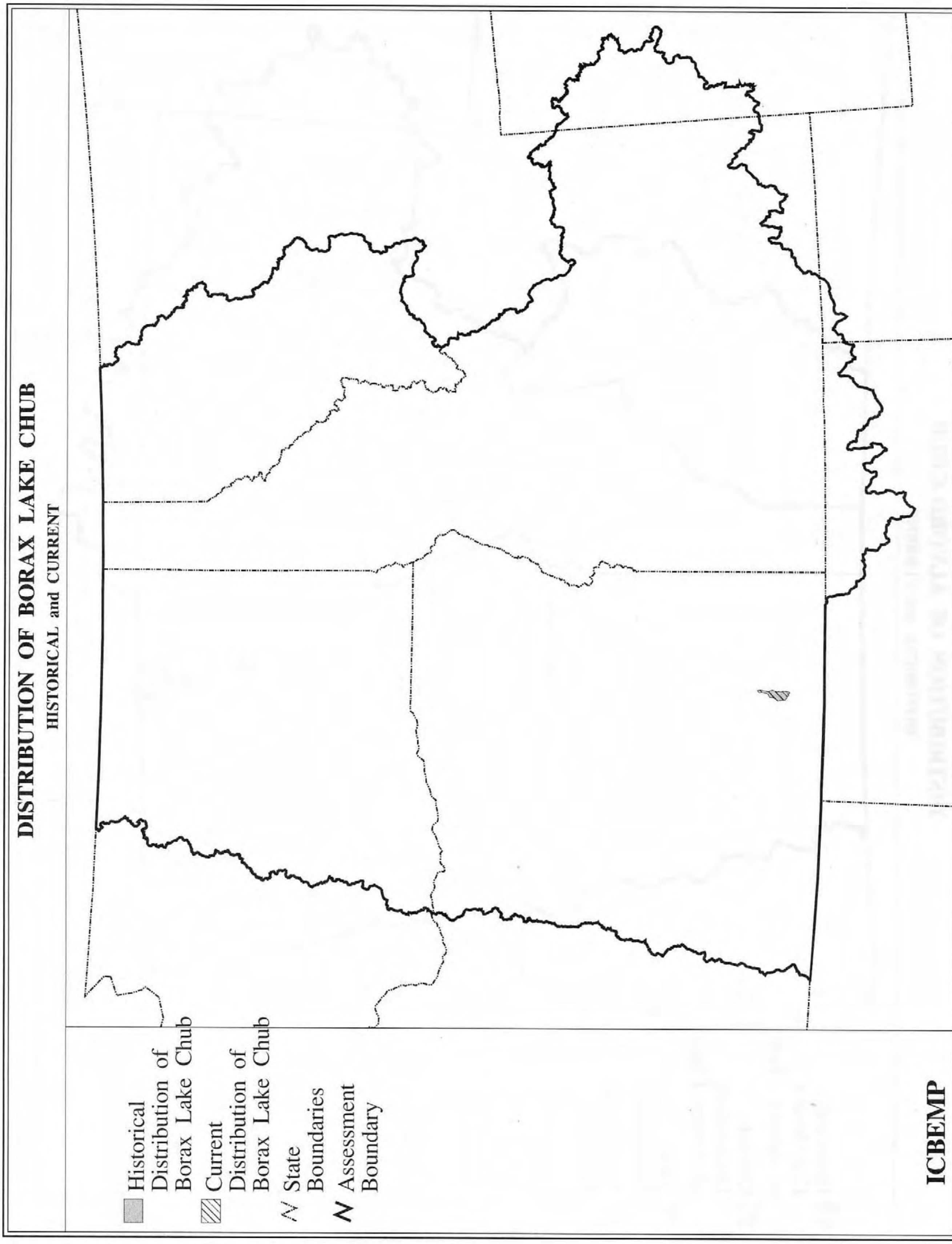
The Borax Lake chub is a small cyprinid fish restricted to the Borax Lake ecosystem of southeastern Oregon. Because of its restricted distribution and threats to its remaining habitat, it is listed as an endangered species by the U.S. Fish and Wildlife Service and State of Oregon.

Distribution and Status

This species is known only from Borax Lake and associated waters in Harney County, Oregon (map 4.51). The Borax Lake chub is a sister taxon of the Alvord chub (*Gila alvordensis*) from which it became isolated as the waters of pluvial Lake Alvord receded (Hubbs and Miller 1972; Williams and Bond 1980). The Borax Lake chub occurs in Borax Lake, its associated outflows including Lower Borax Lake, surrounding marsh, and pools.



Map 4.50—Historical and current distribution of Alvord chub.



Map 4.51—Historical and current distribution of Borax Lake chub.

From 1986 to 1988, population estimates for the Borax Lake chub ranged from 3,934 to 13,319 depending on the year and season (Williams 1995).[6,7] Based on water conditions, hundreds of chubs also may occur in outflow creeks and, during wet years, up to a few thousand also may occur in Lower Borax Lake.[6,7]

Habitat Relationships

The Borax Lake chub is restricted to the thermal waters of Borax Lake and its outflows. Waters flow out from the elevated rim of Borax Lake in many directions, but more typically to the southwest, where they enter a marsh and then flow into Lower Borax Lake (a reservoir). Reproduction is limited to Borax Lake; Borax Lake chubs in other habitats gain access through interconnected outflows and marshes. In Borax Lake, the species occurs throughout the lake except in hot spring inflows, where temperatures exceed approximately 34°C.

Key Factors Influencing Status

Threats of geothermal energy exploration and manipulation of surface flows from Borax Lake were the primary factors that resulted in the 1980 listing of the species by emergency provision under the Endangered Species Act. Changes in thermal flows that enter the lake could cause slight temperature increases or decreases that would be detrimental to the species. Alterations in surface flows from Borax Lake could isolate subpopulations adjacent to the lake causing their desiccation. Because of the restricted size of the lake, threats also exist from introductions of chemicals or nonnative species. Protection of the fragile salt crusts that maintain water level at Borax Lake is also critical (USFWS 1987). Livestock grazing and physical damage from off road vehicles and humans are the primary risks to shoreline salt crusts. The species is also at risk because of its highly restricted range and specialized habitats.

[6]Also, personal communication. 1995. Jack Williams, Bureau of Land Management, Boise, Idaho. Personal communication of unpublished data.

[7]Also, The Nature Conservancy. 1995. Unpublished data. Portland, OR: The Nature Conservancy.

Catlow Tui Chub (*Gila bicolor* ssp.)

Catlow tui chub are endemic to the Catlow Valley of southeastern Oregon. Because of their restricted distributions and threats to remaining habitat, the subspecies is considered of special concern by the American Fisheries Society (Williams and others 1989).

Distribution and Status

Historically, Catlow tui chubs occurred in three streams (Threemile, Skull, and Home creeks) that drain the west flank of the Catlow Rim and in Rock Creek along the western edge of Catlow Valley (map 4.52) (Bills 1977; Kunkel 1976). The Catlow tui chub has a restricted range, but appears to be locally abundant in streams and in Threemile Reservoir. An exception is Rock Creek, where only a few were found in 1994.[8]

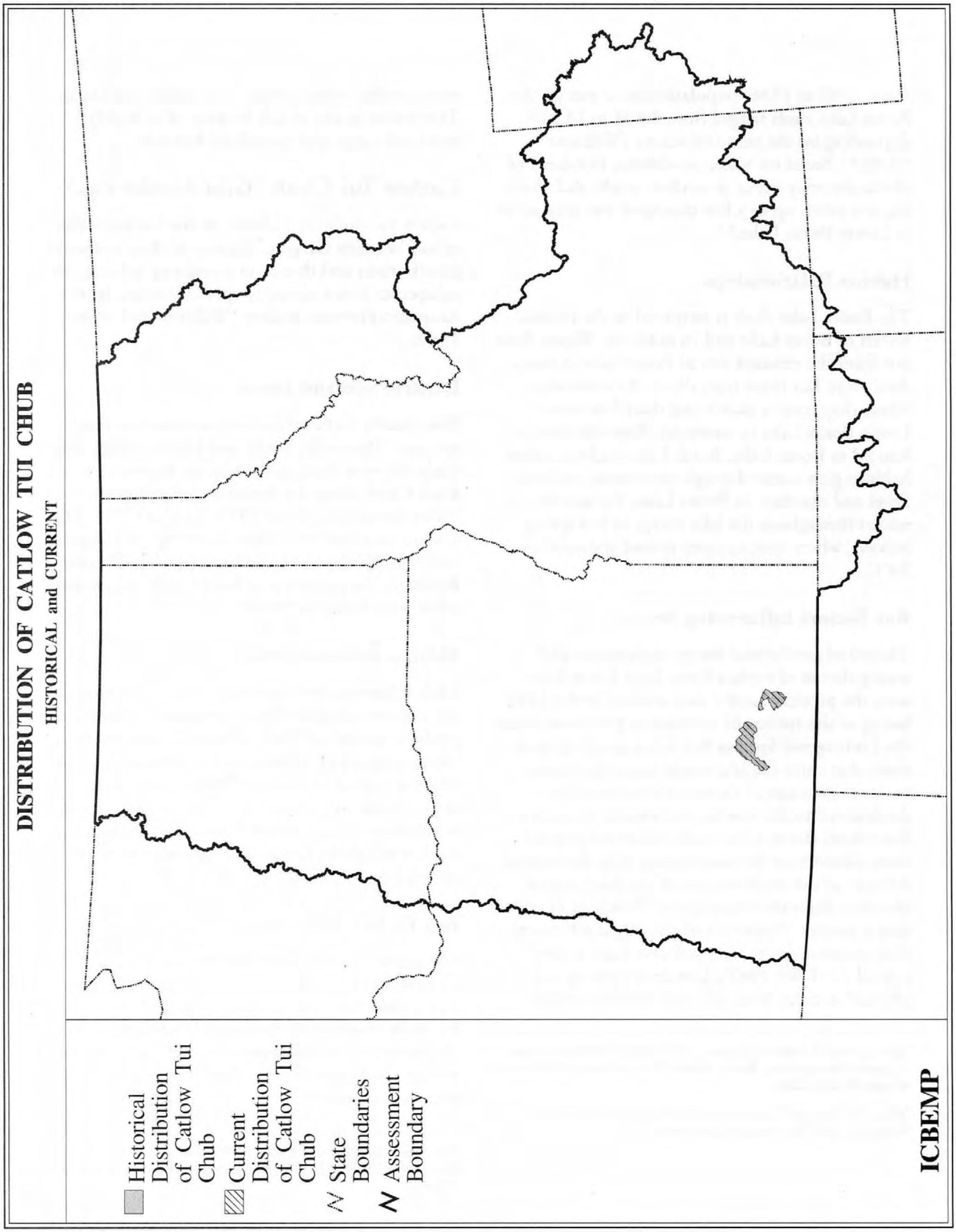
Habitat Relationships

Little is known about the habitat relationships of the Catlow tui chub. Their preference for low gradient reaches of Skull, Threemile, and Home creeks suggests an affinity for low velocity habitats, which is typical of most tui chubs. They also appear to be well-adapted to Threemile Reservoir, at the downstream end of Threemile Creek. Catlow tui chubs occur in streams occupied by redband trout (Kunkel 1976).

Key Factors Influencing Status

Diversions of creek flows for irrigation reduce Catlow tui chub habitat. The low gradient reaches that it prefers are also subject to degradation from livestock overgrazing. Because of the Catlow tui chub's restricted distribution, disturbances such as drought or fire and human land use practices place populations at risk.

[8]Oregon Department of Fish and Wildlife. 1995. Unpublished data. Hines, OR: Oregon Department of Fish and Wildlife.



Map 4.52—Historical and current distribution of Catlow tui chub.

Oregon Lakes Tui Chub (*Gila bicolor oregonensis*)

The Oregon Lakes tui chub, as defined here, is endemic to the Abert Lake Basin of south-central Oregon (Bills 1977). Remaining populations are classified by the State of Oregon as vulnerable. The American Fisheries Society lists the Oregon Lakes tui chub as a species of special concern (Williams and others 1989), although they use the common name XL Spring tui chub for this form.

Distribution and Status

The Oregon Lakes tui chub complex, as originally described by Snyder (1908), consisted of tui chub populations in five isolated basins of south-central Oregon: Silver, Summer, Abert, Alkali, and Warner. The pioneering work of Bills (1977) demonstrated that morphological divergence had occurred among these long-isolated populations and he recognized that the complex of tui chubs actually consists of four subspecies. Only populations in the Abert Lake Basin are retained in *Gila bicolor oregonensis*. They occur in XL Spring to the north of Abert Lake and in the Chewaucan River (map 4.53). Abert Lake is, in general, fishless but records of chubs exist, presumably in areas of spring or river inflow (Snyder 1908). No changes between historic and current distributions are known although the Abert Lake Basin has not been adequately sampled.

Habitat Relationships

In general, tui chubs occupy a wide variety of habitats (Moyle 1976). In the Abert Lake Basin, the Oregon Lakes tui chub inhabits springs, rivers, and ditches. The Oregon Lakes tui chub is absent from higher gradient portions of the Chewaucan River system.

Key Factors Influencing Status

Agricultural practices, including ditching and diverting stream flows as well as livestock grazing, are the principal factors influencing the distribution and abundance of this subspecies. No recent surveys of habitats occupied by the Oregon Lakes tui chub are known. Thus, any additional, recent factors influencing its status are unknown. The introduction of non-native fishes also threatens the continued existence of this subspecies. The type locality population, at XL Spring, is particularly vulnerable to loss because of its restricted habitat.

Summer Basin Tui Chub (*Gila bicolor* spp.)

The Summer Basin tui chub is endemic to springs and outflows in the Summer Basin of south-central Oregon. The form was considered of uncertain taxonomic status and possibly extinct by Bills (1977) during a thorough review of the Oregon tui chub complex in southern Oregon. Summer Basin tui chubs were rediscovered in 1985. This subspecies is listed as a C1 Candidate by the U.S. Fish and Wildlife Service and as endangered by the American Fisheries Society (Williams and others 1989).

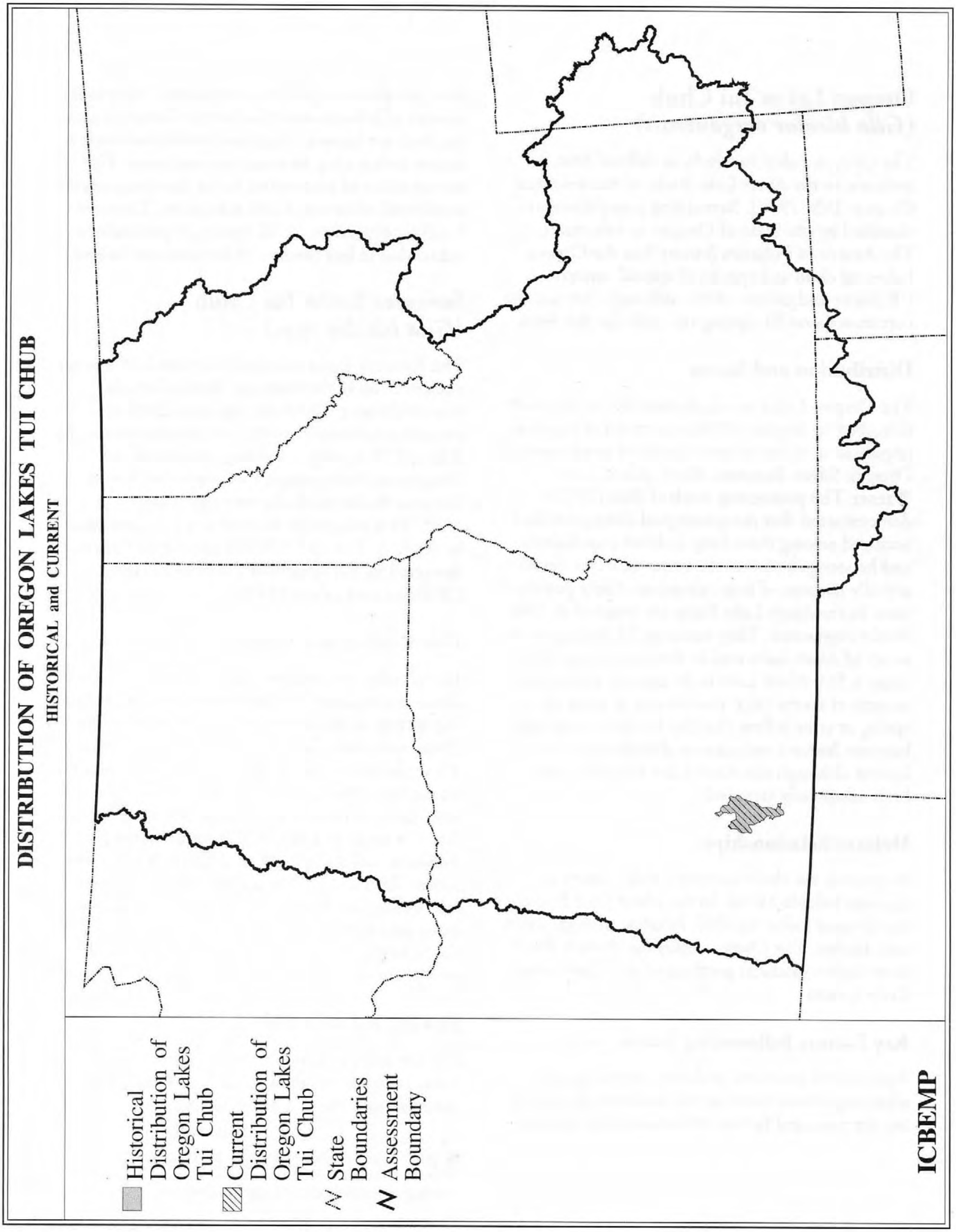
Distribution and Status

Historically, the Summer Basin tui chub occurred at various localities within Summer Basin, including springs at the Summer Lake Post Office, Ana River, and source springs of the Ana River (map 4.54) (Snyder 1908). Collections in these localities during the 1960s and 1970s indicated divergence from the form that was native to the Summer Basin, a result of numerous applications of fish toxicants and transplants of chubs from adjacent basins (Bills 1977). The native form was considered extinct until rediscovered from a small, previously unsampled spring system on the west side of the Summer Lake bed in 1985.[9] The subspecies now occupies a small portion of its historic range.

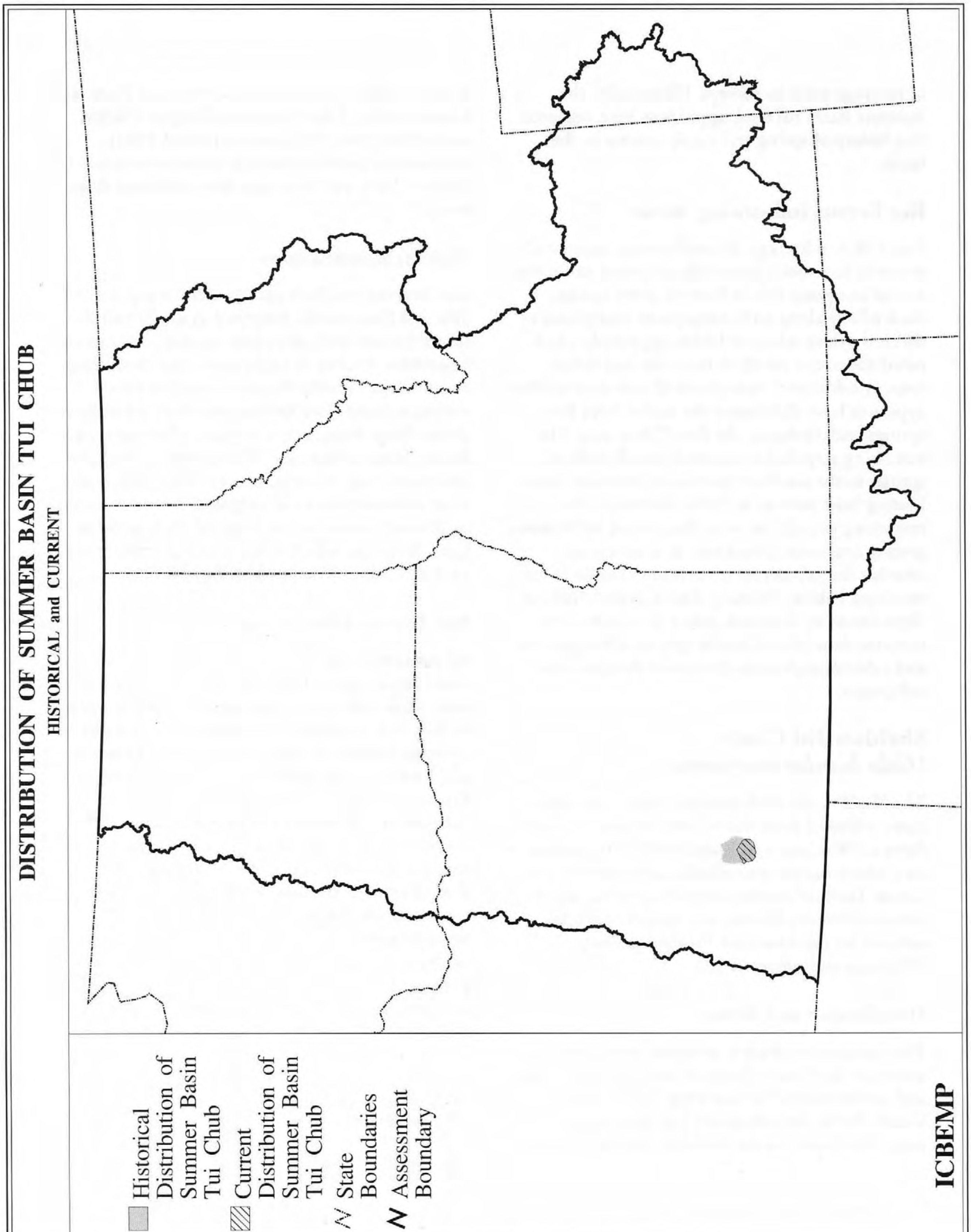
Habitat Relationships

Habitat relationships are poorly known because habitats in the Summer Basin have been greatly modified and the fish were eliminated from much

[9]Personal communication. 1995. C.E. Bond and J.E. Williams, Oregon State University, Corvallis, Oregon. Personal communication of unpublished data.



Map 4.53—Historical and current distribution of Oregon Lakes tui chub.



Map 4.54—Historical and current distribution of Summer Basin tui chub.

of its range prior to surveys. Historically, the Summer Basin tui chub appears to have occurred in a variety of spring and creek systems in the basin.

Key Factors Influencing Status

Fishes in Ana Springs, its outflow and associated reservoir have been repeatedly poisoned to rid the area of nongame fish in favor of game species. Such efforts along with subsequent transplants of tui chubs from adjacent basins apparently eliminated the native tui chub from the Ana River system. Additional transplants of non-native fishes appear to have eliminated the native form from springs and ditches in the Post Office area. The remaining population occurs in small, isolated springs in the southern portion of Summer Basin. During brief surveys in 1985, habitats of the remaining population were threatened by livestock grazing and water diversions. It is uncertain whether the subspecies continues to persist in its restricted habitat. Primary threats include habitat degradation by livestock, water diversions, and introductions of non-native species. Drought, fire, and other disturbances also could threaten this subspecies.

Sheldon Tui Chub (*Gila bicolor eurysoma*)

The Sheldon tui chub was described from specimens collected from Fish Creek, Washoe County, Nevada (Williams and Bond 1981). This subspecies, which occurs sporadically in the mostly arid Guano Basin of southeastern Oregon and northwestern Nevada, is listed as a species of special concern by the American Fisheries Society (Williams and others 1989).

Distribution and Status

The Sheldon tui chub is restricted to isolated waters of the Guano Basin of southeastern Oregon and northwestern Nevada (map 4.55). Within Guano Basin, the subspecies has been reported from Fish Creek on the Sheldon National Wildlife Refuge, Washoe County in Nevada, and Piute and Guano creeks, Lake County in Oregon (Hubbs and Miller 1948, Williams and Bond 1981). Sheldon tui chubs apparently are extremely rare in Guano Creek and have only been collected there twice.[10]

Habitat Relationships

The Sheldon tui chub occurs in those portions of Fish and Piute creeks described as small, turbid desert streams with abundant aquatic and riparian vegetation. During drought years, the chubs may be restricted to isolated pools in intermittent stream sections. The Sheldon tui chub typically is absent from downstream reaches, which often dry during summer months. When water is abundant, chubs also may occur in terminal lakes and reservoirs of these streams as suggested by the discovery of skeletal remains of Sheldon tui chub in Swan Lake Reservoir, which is the terminal water body on Fish Creek (Williams and Bond 1981).

Key Factors Influencing Status

All streams where the Sheldon tui chub occur could be described as "marginal" in reference to their small and often intermittent nature. Grazing by livestock apparently has limited the amount of available habitat by reducing riparian vegetation and limiting soil water retention, which leads to drying of additional stream segments during summer and autumn (Williams and Bond 1981). Reductions in stream flow due to overgrazing and water diversion appear to be the primary threat to this subspecies. Because of the species restricted range and small habitat size, such factors will exacerbate risks associated with environmental stochasticity, such as drought. Although introduced fish have not been documented from water inhabited by the Sheldon tui chub, any introductions could be detrimental to this subspecies.

[10]Personal communication. 1995. J. Williams, Bureau of Land Management, Boise, Idaho. Personal communication of unpublished data.

Hutton Tui Chub (*Gila bicolor* spp.)

The Hutton tui chub, collected as early as 1908, has been found in only two surface flow areas of Hutton Spring, Oregon. The Hutton tui chub was listed as threatened in 1985 by the U.S. Fish and Wildlife Service. It is also considered threatened by the American Fisheries Society (Williams and others 1989).

Distribution and Status

Hutton tui chubs still inhabit their historical locations (map 4.56). The subspecies is restricted to Hutton Spring and a small nearby spring in the Alkali Lake Basin of south-central Oregon (Bills 1977). In 1977, population estimates were 300 at Hutton Spring and 150 at the unnamed spring.

Habitat Relationships

The Hutton tui chub lives its entire life in spring habitats. Little is known about their habitat requirements, with the exception that dense aquatic vegetation is needed for spawning and rearing of young.

Key Factors Influencing Status

The habitat of the Hutton tui chub, along with that of the Foskett speckled dace, are the most restricted of any fish in the assessment area. Because of their limited distribution and small population size, Hutton tui chubs are at risk. A hazardous waste dump at Alkali Lake threatens to contaminate surface floodwater, groundwater, and air at Hutton Spring. Persistence of the Hutton tui chub is threatened by catastrophic events to spring sources, pollution from toxic chemicals, vandalism, introduction of non-native species, and no natural sources of recolonization.

Leatherside Chub (*Gila copei*)

The native range of the leatherside chub in Utah and Wyoming consists of the eastern and southern parts of the Bonneville Basin in rivers draining into the Great Salt Lake. In Idaho, the species is presumed to occur naturally as a result of pluvial discharges from the Bonneville Basin into the Snake Basin. Its current distribution in Idaho may, in part, also be due to its release as a bait fish. The leatherside chub is presently listed by the BLM as a sensitive species, and by the Idaho Department of Fish and Game as a species of special concern, Category C undetermined status.

Distribution and Status

In 1934, Carl Hubbs made the first reported collection of leatherside chub in Idaho.[11] The leatherside chub was collected in the 1970s in the Raft River and Goose Creek tributaries of the Snake River and the Little Wood River, all in Idaho (map 4.57).[12] In 1995, leatherside chubs were collected from Trapper, Goose, and Beaver Dam creeks, all part of the Goose Creek drainage near the Nevada-Utah border.[13] Otherwise, because of a lack of surveys, their current status is unknown.

Habitat Relationships

Little is known about the habitat requirements of leatherside chub. They typically occur in cool to cold creeks and rivers, with adults residing in pools and riffles and young inhabiting brushy, quiet pockets near the shoreline. Water quality ranges from clear to occasionally turbid. Leatherside chub are typically found associated with gravel substrate, but they also use all substrate sizes.

Key Factors Influencing Status

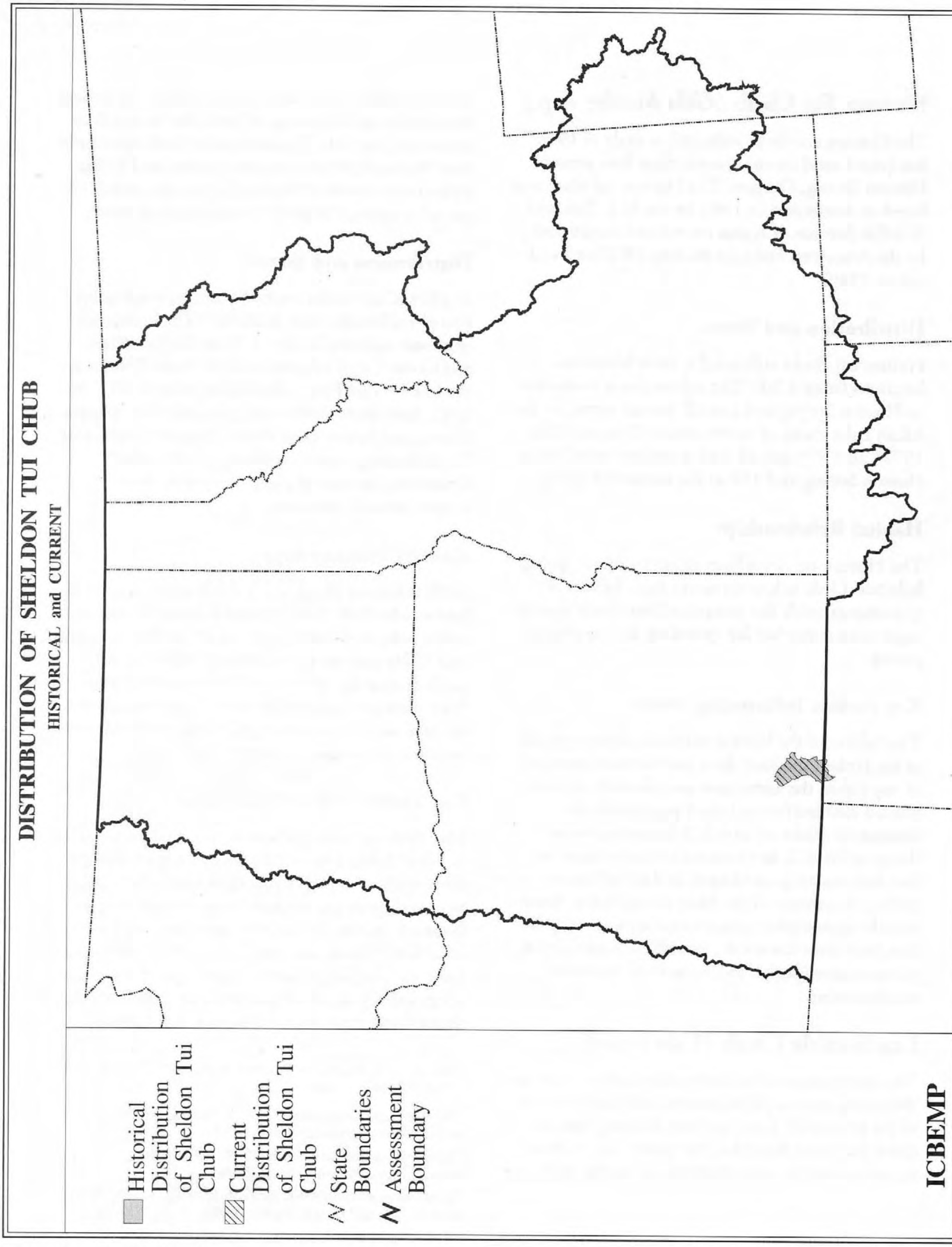
The most significant threat to the leatherside chub is loss of habitat caused by development of irrigation projects in the 1930s. However, habitat losses have and continue to occur from overgrazing of livestock, mining, timber harvest, and road construction.[14] Predation and competitive interactions from the stocking of exotic game species may also affect leatherside chub populations. However, the threat from these introductions is not known.

[11]University of Michigan Museum of Zoology. 1995. Unpublished collection records.

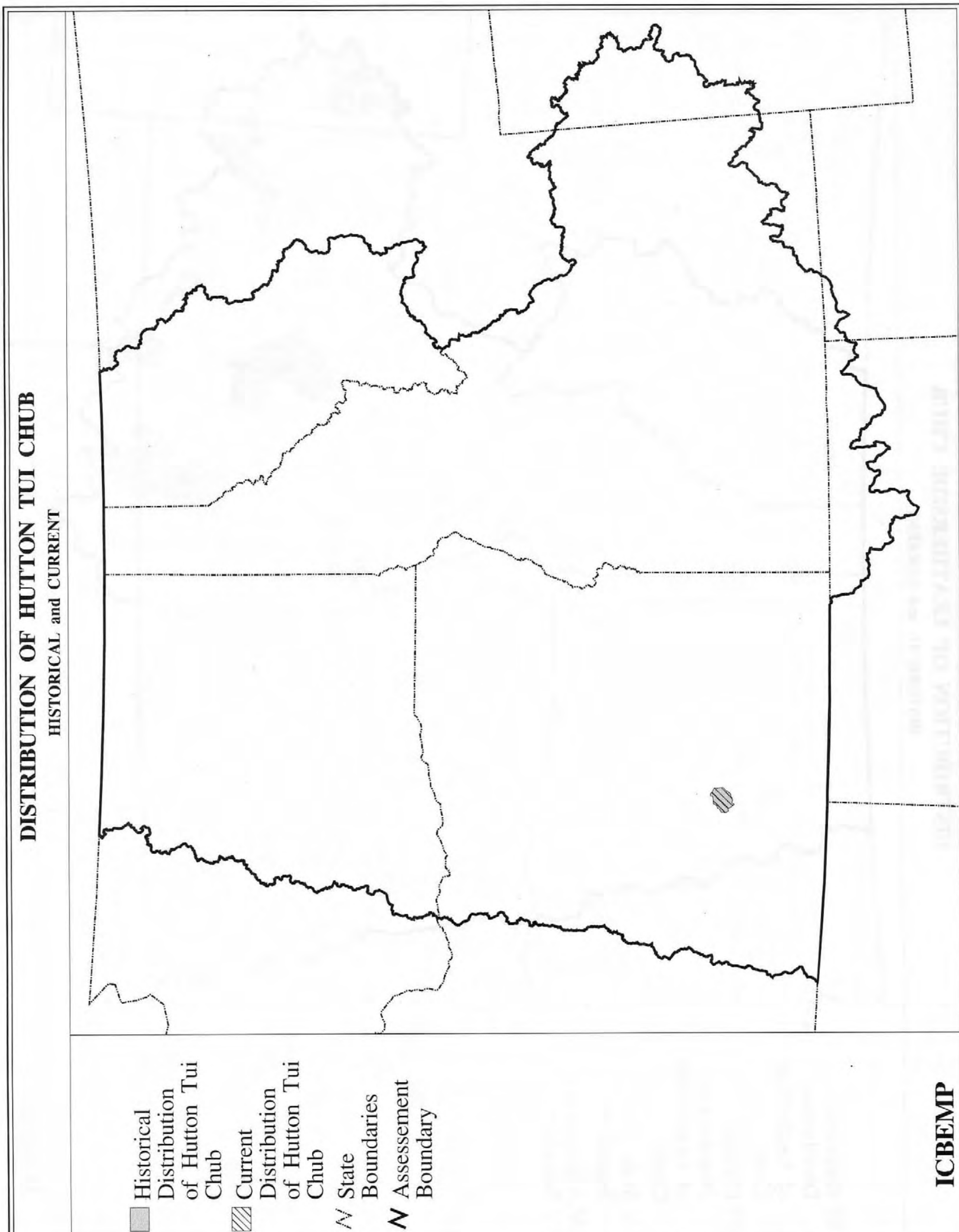
[12]Personal communication. 1995. F. Partridge, Idaho Department of Fish and Game, Jerome, Idaho.

[13]Personal communication. 1995. B. Horton, Idaho Department of Fish and Game, Boise, Idaho.

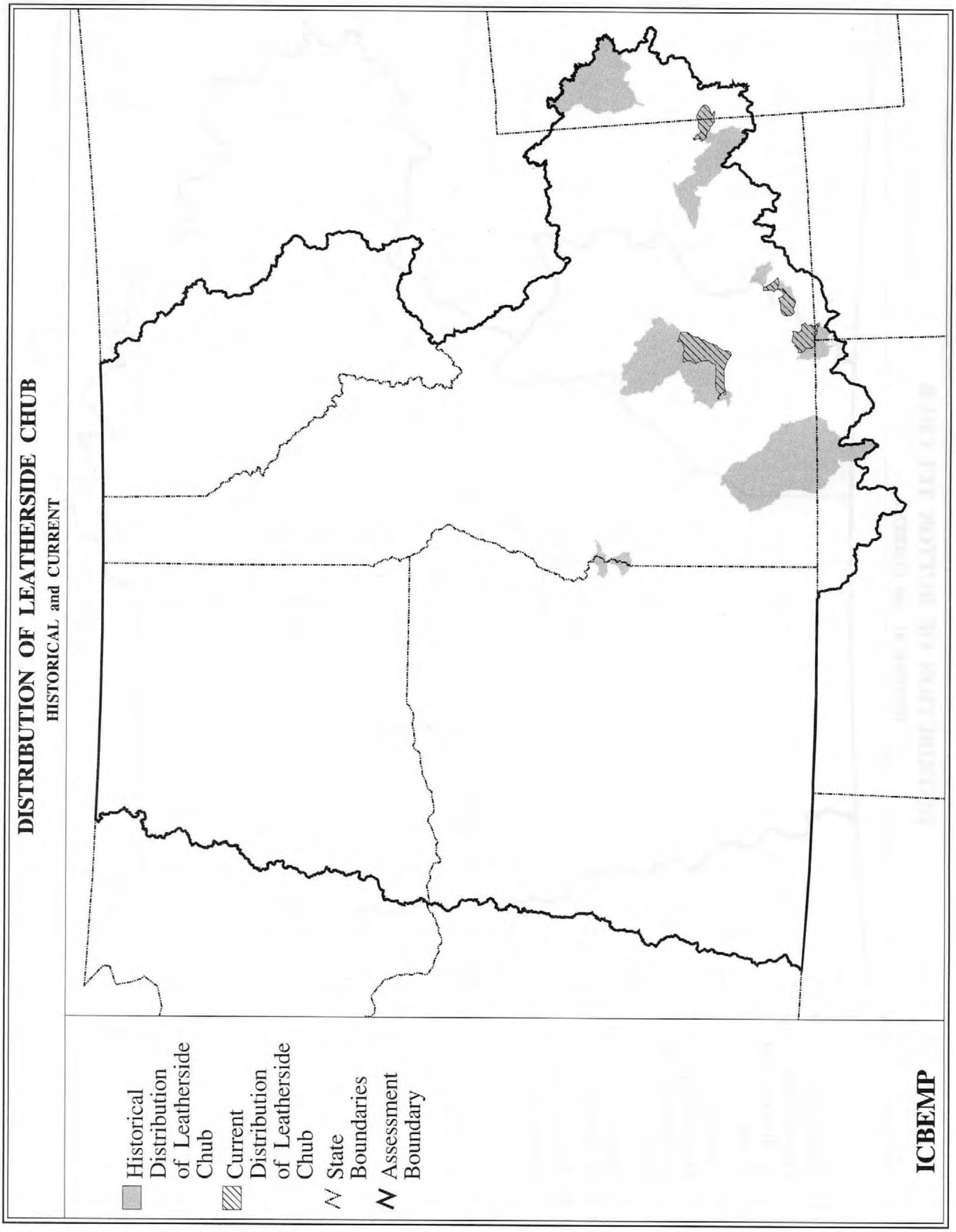
[14]Personal communication. 1995. F. Partridge, Idaho Department of Fish and Game, Jerome, Idaho.



Map 4.55—Historical and current distribution of Sheldon tui chub.



Map 4.56—Historical and current distribution of Hutton tui chub.



Map 4.57—Historical and current distribution of leatherside chub.

Foskett Speckled Dace (*Rhinichthys osculus* spp.)

The Foskett speckled dace is an undescribed subspecies restricted to springs in the Coleman Valley of southeastern Oregon. The dace is federally listed as threatened and as vulnerable by the State of Oregon. The American Fisheries Society also lists the Foskett speckled dace as threatened (Williams and others 1989).

Distribution and Status

The Foskett speckled dace historically was known only from Foskett Spring, located along the west side of Coleman Lake bed in Lake County, Oregon (map 4.58) (Bond 1974). Coleman Lake is dry except during years of exceptional rainfall. During 1979, dace from Foskett Spring were transplanted into Dace Spring, located 1.5 kilometers south of Foskett Spring to create a second population. The population in Dace Spring had reproduced and appeared to be established (Williams and others 1990), but has recently become extirpated.[15] Although the watershed that contains Coleman Lake extends into Nevada, the dace does not occur in that state. Total population size is approximately 2,000 in Foskett Spring (Bond 1974).

Habitat Relationships

Foskett Spring is a small, cool-water (approximately 16° to 17° C) spring. The Foskett speckled dace are abundant in the small spring pool and associated outflow until surface flow disperses and disappears near the edge of Coleman Lake. The species were also abundant at Dace Spring, including an adjacent cattle trough located just outside the fenced area. No other fish has been reported from Coleman Valley.

[15]Personal communication. 1995. Alan Munhall, Bureau of Land Management, Lakeview, Oregon.

Key Factors Influencing Status

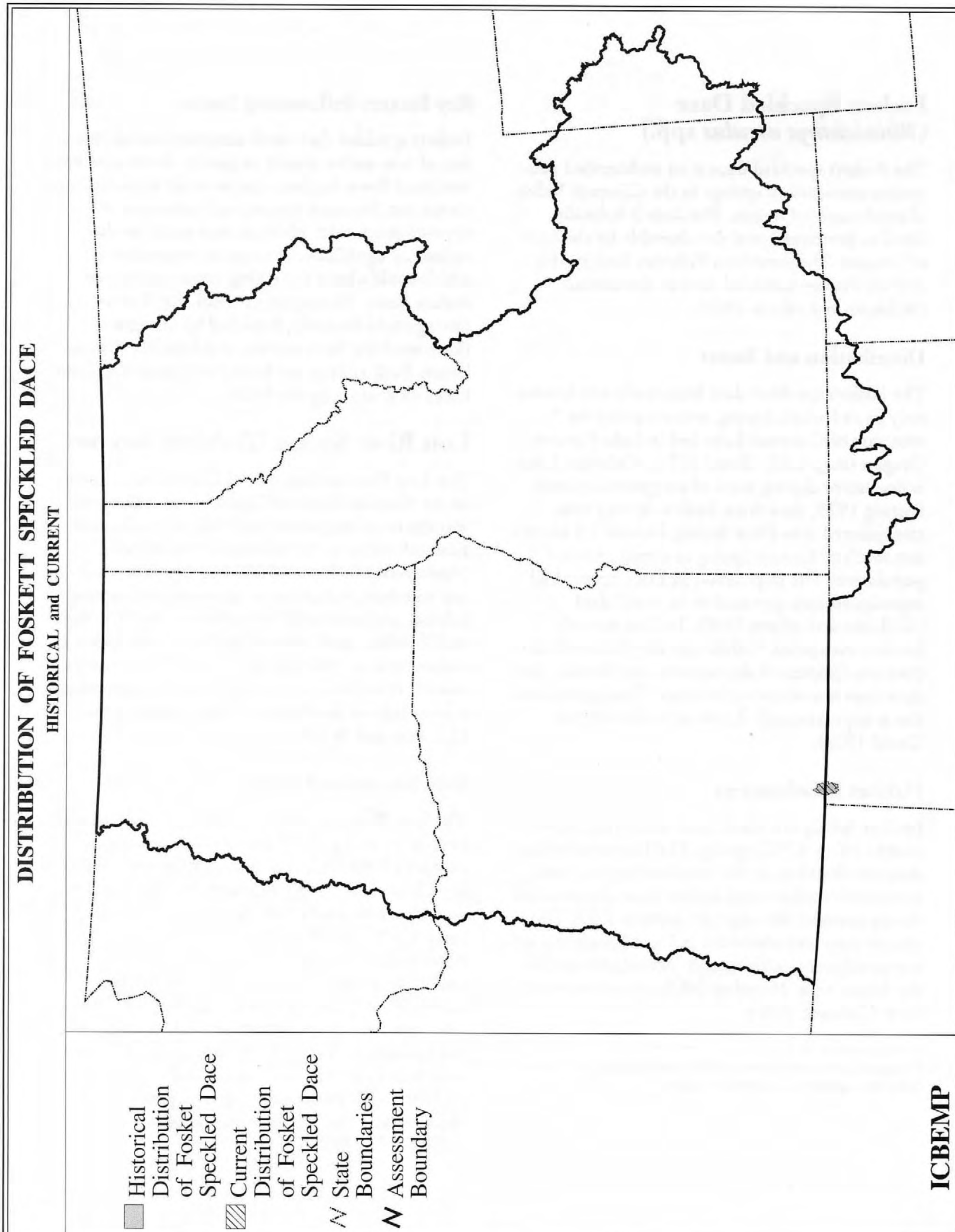
Foskett speckled dace need adequate surface flows free of non-native species to persist. Because of the restricted flows, habitats can be easily disturbed by visitor use, livestock grazing and reduction of riparian vegetation. Habitats also could be disturbed by significant increases in vegetation, which could choke the spring system or reduce surface flows. Because of its small size, Foskett Spring could be easily disturbed by changes in flows resulting from surface or subsurface disturbance. Both springs are fenced and protected from livestock grazing by the BLM.

Lost River Sucker (*Deltistes luxatus*)

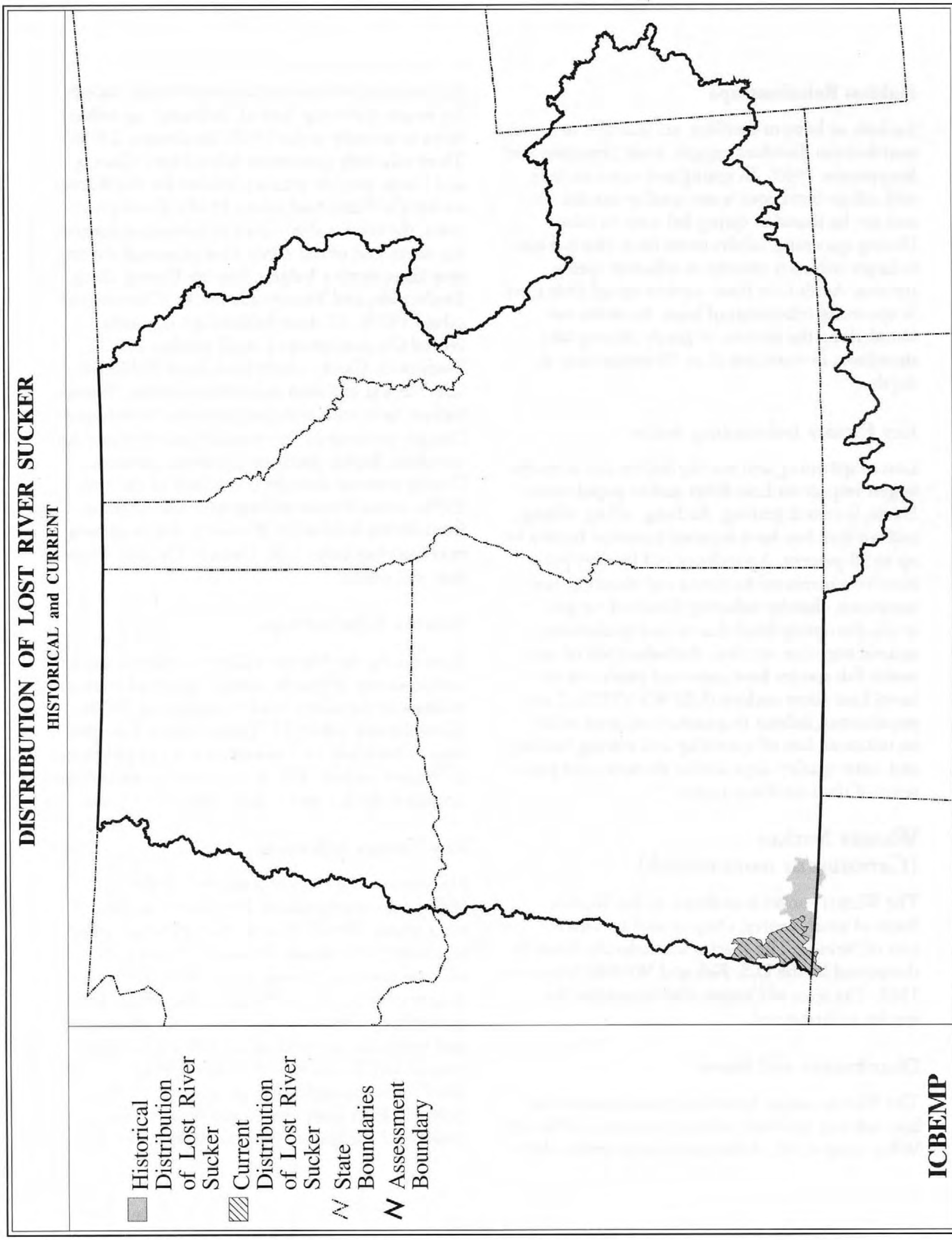
The Lost River sucker, one of four native suckers in the Klamath Basin of California and Oregon, was the most important food fish of Modoc and Klamath tribes in the Klamath Lake Region (Andreasen 1975b; USFWS 1993b). As a result of low numbers, reduction in spawning and rearing habitat, and poor adult recruitment, the U.S. Fish and Wildlife Service listed the Lost River sucker as endangered in 1988. In 1994, 185,000 hectares of stream, river, lake, and shoreline were proposed as critical habitat for the Lost River sucker by the U.S. Fish and Wildlife Service.

Distribution and Status

The Lost River sucker historically occurred in the Lost River and upper Klamath River systems of northern California and southern Oregon, including Clear, Tule, Lower Klamath, Sheepy, Agency, and Upper Klamath lakes and their tributaries (map 4.59) (USFWS 1993b). Currently, Lost River sucker populations and available habitat have been greatly reduced. Lost River suckers currently are found in Upper Klamath and Tule lakes and Clear Lake Reservoir, with possible populations in Sheepy Lake and Iron Gate Reservoir. Recruitment of juvenile Lost River suckers to adult classes is poor, resulting in populations of older suckers and few new spawning individuals (USFWS 1993b).



Map 4.58—Historical and current distribution of Foskett speckled dace.



Map 4.59—Historical and current distribution of Lost River sucker.

Habitat Relationships

Suckers, as bottom dwellers, are sensitive to low near-bottom dissolved oxygen levels (Buettner and Scoppetone 1990). In spring and summer, they seek refuge from poor water quality conditions and can be found at spring fed areas in lakes. During spawning, adults move from lake habitats to larger tributary streams or adjacent spring systems. Adult Lost River suckers spend little time in spawning tributaries of lakes. Juveniles are found along the bottom of gently sloping lake shorelines, in water less than 50 centimeters in depth.

Key Factors Influencing Status

Loss of spawning and rearing habitat has been the largest impact on Lost River sucker populations. Dams, livestock grazing, ditching, tilling, diking, and wetland loss have reduced historical habitat by up to 90 percent. Agriculture and forestry practices have increased nutrients and chemical contaminants, thereby reducing dissolved oxygen levels, disrupting food chains, and minimizing aquatic organism survival. Introductions of nonnative fish species have increased predation on larval Lost River suckers (USFWS 1993b). Low populations, habitat fragmentation, poor adult recruitment, loss of spawning and rearing habitat, and water quality degradation threatens the persistence of the Lost River sucker.

Warner Sucker (*Catostomus warnerensis*)

The Warner sucker is endemic to the Warner Basin of south-central Oregon and an adjacent area of Nevada. The species was federally listed as threatened by the U.S. Fish and Wildlife Service in 1985. The state of Oregon also recognizes the species as threatened.

Distribution and Status

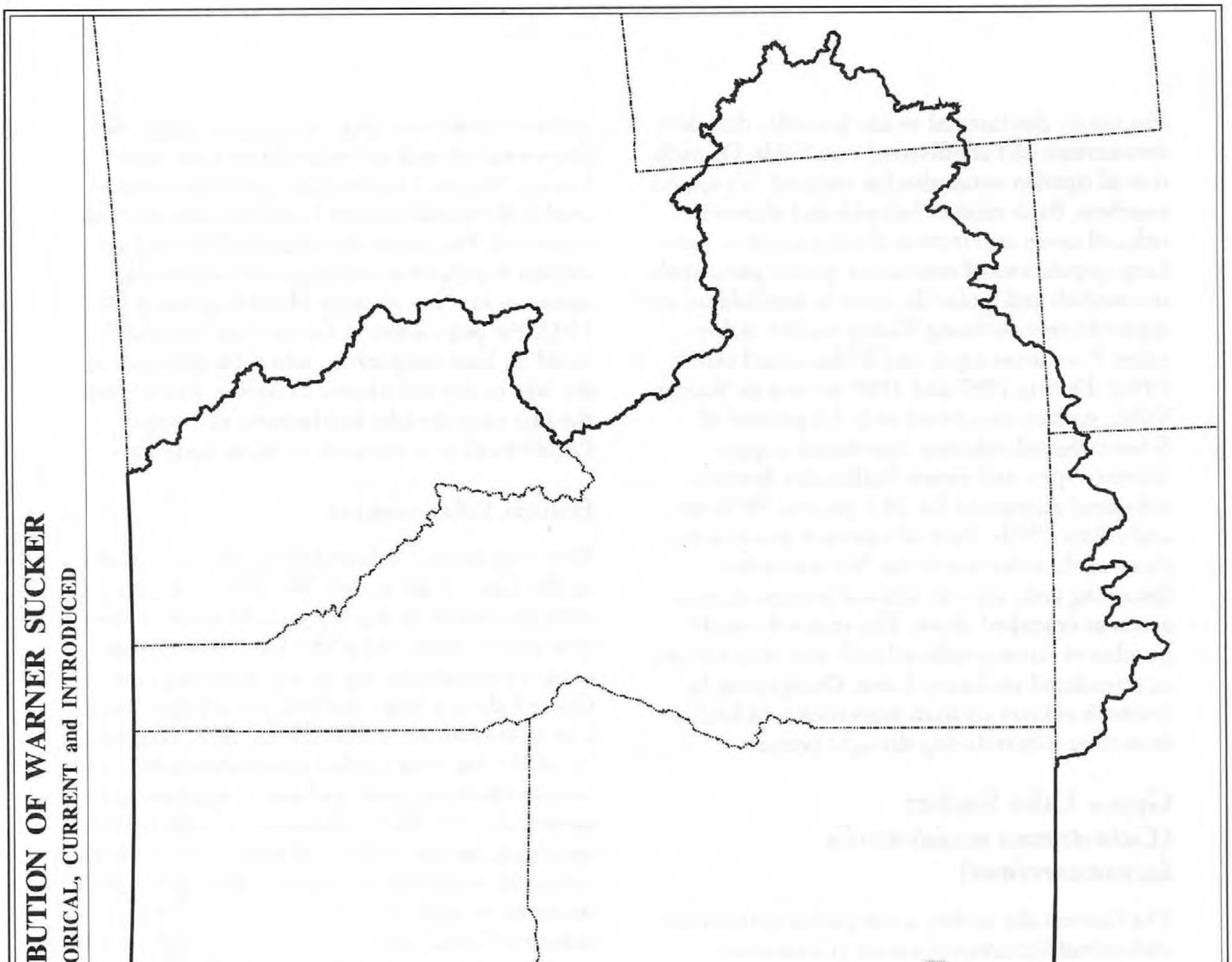
The Warner sucker historically was restricted to lake habitats and their tributary streams in Warner Valley (map 4.60). Accounts of early settlers documented large Warner sucker populations, including major spawning runs of "redhorse" up tributaries as recently as the 1930s (Andreasen 1975b). Three relatively permanent lakes, Hart, Crump, and Deep, provide primary habitat for the Warner sucker (Williams and others 1990). During wet years, the species also occurs in ephemeral lakes in the north end of the valley. Five perennial streams provide spawning habitat: Snyder, Honey, Deep, Twelvemile, and Twentymile creeks (Coombs and others 1979). All these habitats are in south-central Oregon, except a small portion of Twelvemile Creek, which flows from California into Nevada and then north into Oregon. Warner suckers have been collected from the Nevada and Oregon portions of Twelvemile Creek but not the upstream, higher-gradient California sections. During extreme drought conditions of the early 1990s, some Warner suckers were transplanted from drying habitats in Warner Valley to an area near Summer Lake, Lake County, Oregon, where they still persist.

Habitat Relationships

Historically, the Warner sucker probably inhabited a wide variety of marsh, slough, lake, and stream habitats in the valley. Studies during the 1970s (Coombs and others 1979) documented the presence of both lake and resident stream populations of Warner suckers. The species also occurs in larger irrigation ditches and shallow, ephemeral lakes.

Key Factors Influencing Status

Historically, the lakes and streams of Warner Valley were interconnected by slough and marsh areas which allowed Warner suckers broad access to a variety of habitats. Draining of wetlands coupled with installation of numerous irrigation diversion structures on lower reaches of tributaries greatly fragmented the distribution of the species and precluded access to many historic spawning areas as well as access to inflowing streams as lakes dried during drought periods. During 1992, for example, Hart Lake completely dried and nearly eliminated the species. Water diversion practices



Map 4.60—Historical and current distribution plus introduced sites of Warner sucker.

also can be detrimental to any juveniles that drift downstream and are diverted into fields. Degradation of riparian zones also has reduced this species' numbers. Bank erosion has widened channels, reduced cover, and increased sedimentation rates. Large populations of non-native species, particularly centrarchids and ictalurids, occur in lake habitats and appear to prey on young Warner suckers as they return from spawning areas (Williams and others 1990). During 1987 and 1989 surveys in Warner Valley, suckers comprised only 2.5 percent of fishes collected, whereas introduced crappie (*Pomoxis* spp.) and brown bullheads (*Ameiurus nebulosus*) accounted for 38.0 percent (Williams and others 1990). Poor recruitment continues to threaten the existence of the Warner sucker. Spawning areas are very limited because of poor access as described above. The relatively small number of larvae produced each year may fall prey to introduced predatory fishes. Overgrazing by livestock reduces riparian vegetation and further intensifies effects during drought periods.

Goose Lake Sucker (*Catostomus occidentalis lacusanserinus*)

The Goose Lake sucker, a subspecies of the more widespread Sacramento sucker (*Catostomus occidentalis*), is endemic to the Goose Lake Basin of south-central Oregon and northeastern California. The Goose Lake sucker is a Forest Service Region 6 sensitive species, and a State of Oregon sensitive species.

Distribution and Status

The subspecies is found in Goose Lake and many of its larger tributaries (map 4.61) (Moyle and others 1989). Goose Lake suckers were collected during the 1980s from Corral, Long Branch, Badger-Cloud, Davis, Lassen, and Willow creeks in Modoc County, California, and Augur, Bauers, Thomas, Cox, Cottonwood, Fall, Dry, Drews, Dog, and Hay creeks in Lake County, Oregon.[16] The subspecies also has been collected from Cottonwood Reservoir, Dog Lake, and Goose Lake. Spawning fish migrate from Goose Lake into Lassen, Willow, Thomas, Dry, and Cottonwood creeks (King and Hansen 1966; also, see previous footnote). The actual abundance of Goose Lake suckers is unknown, although they are locally common in many streams where they occur. In 1992, the population in Goose Lake was eliminated, at least temporarily, when drought caused the lake to dry completely. However, this was not the first time the lake had become desiccated. Recolonization is expected to occur naturally.

[16]Personal communication. 1995. J. Williams, Bureau of Land Management, Boise, Idaho.

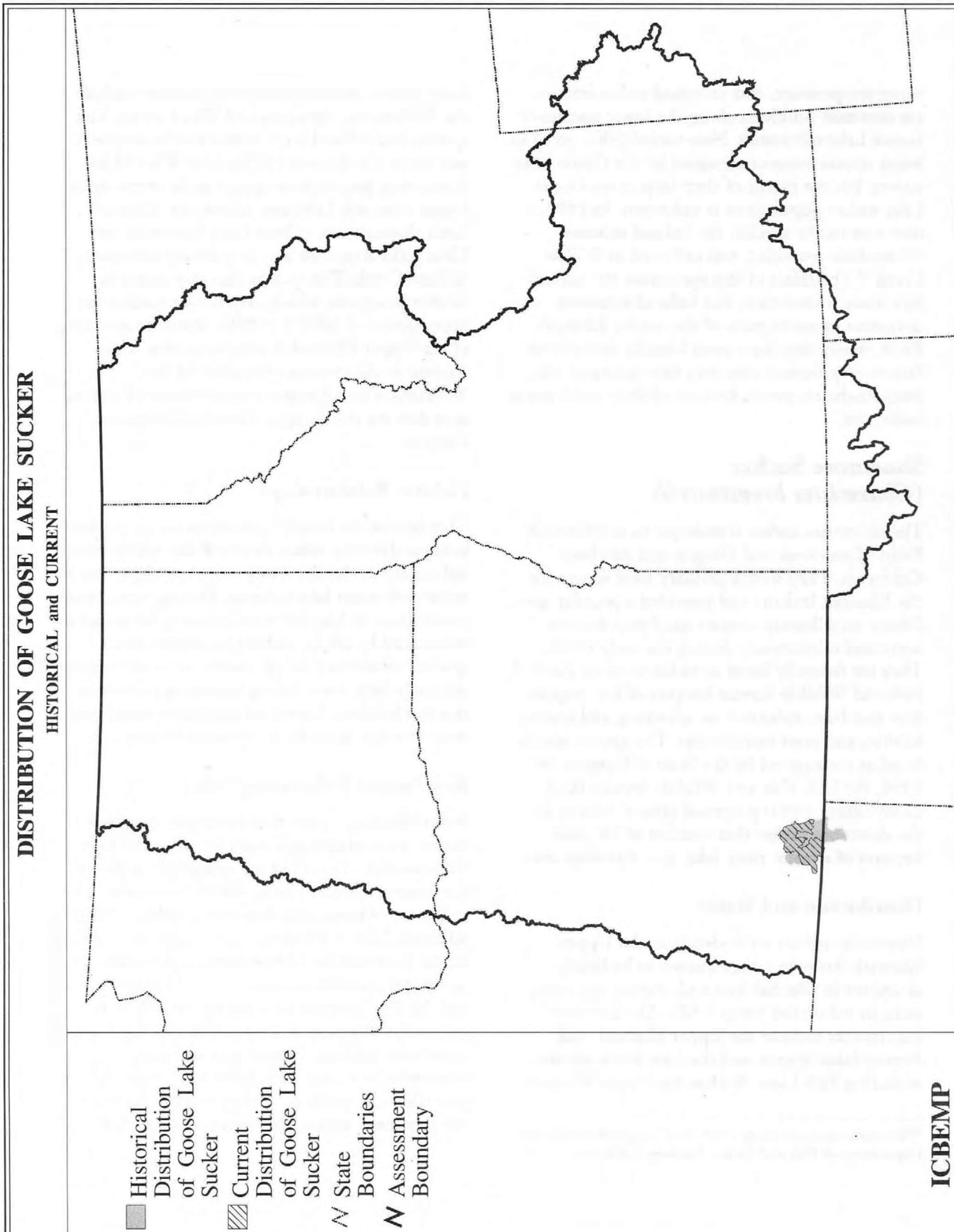
Habitat Relationships

Very little specific habitat information is available on the Goose Lake sucker. Two life history patterns are present in the Goose Lake Basin: a resident stream form, and a lake form that ascends tributary streams during spring spawning runs. Goose Lake is a large, shallow, and alkaline natural lake where bottom sediments are easily suspended by winds that create turbid conditions. Lake temperatures vary with ambient conditions and range from 1° to 24° C (Johnson and others 1985). Spawning habitat in lower Willow Creek is characterized by a complex of cobble-pebble-gravel with moderate to swift current.[17] In streams during summer, Goose Lake suckers seem to prefer pool habitats, with young of year rearing in shallows.

Key Factors Influencing Status

Historically, Goose Lake provided refuge to a relatively diverse native fish fauna free of non-native species that occur in tributary streams and elsewhere. Lower reaches of major tributary streams served as refuges to lake-dwelling Goose Lake suckers during drought periods. Irrigation diversions and physical barriers disrupted this pattern and compounded risk to this subspecies during periods of naturally reduced water availability. Many stream habitats have been degraded by agricultural practices. Overgrazing by livestock and resulting loss of riparian vegetation, increased

[17]Personal communication. 1995. J. Williams, Bureau of Land Management, Boise, Idaho. Personal communication of unpublished data.



Map 4.61— Historical and current distribution of Goose Lake sucker.

water temperature, and increased sedimentation are common problems along the lower reaches of Goose Lake tributaries. Non-native fishes occur in many stream systems occupied by the Goose Lake sucker, but the extent of their impact on Goose Lake sucker populations is unknown. In 1994, a new non-native species, the fathead minnow (*Pimephales promelas*), was collected in Willow Creek.[18] The effect of this species on the native fish fauna is uncertain, but fathead minnows dominate in many parts of the nearby Klamath Basin, where they have been broadly introduced. Stream populations also may face increased risk from stochastic events because of their small population size.

Shortnose Sucker (*Chasmistes brevirostris*)

The shortnose sucker is endemic to the Klamath Basin of south-central Oregon and northern California. They were a primary food source for the Klamath Indians and provided a popular sport fishery on tributary streams until populations decreased substantially during the early 1980s. They are federally listed as endangered by the U.S. Fish and Wildlife Service because of low population numbers, reduction in spawning and rearing habitat, and poor recruitment. The species also is listed as endangered by the State of Oregon. In 1994, the U.S. Fish and Wildlife Service (U.S. Government 1994) proposed critical habitat for the shortnose sucker that consists of 185,000 hectares of stream, river, lake, and shoreline areas.

Distribution and Status

Shortnose suckers are endemic to the Upper Klamath Basin and were known to be locally abundant in lake habitats and, during spawning runs, in tributaries (map 4.62). Documented occurrences include the Upper Klamath and Agency lakes system and the Lost River system, including Tule Lake. Within the Upper Klamath Lake system, primary spawning habitats include the Williamson, Sprague, and Wood rivers. The species has suffered large reductions in numbers and range (Andreasen 1975b; USFWS 1993b). Remaining populations appear to be restricted to Upper Klamath Lake and tributaries, Klamath River downstream to Iron Gate Reservoir, and Clear Lake Reservoir and its primary tributary, Willow Creek. The species also may occur in Gerber Reservoir, which may be the result of an introduction (USFWS 1993b). Riverine spawning of the Upper Klamath Lake population is restricted to the lowermost reaches of the Williamson and Sprague rivers because of a diversion dam on the Sprague River at Chiloquin, Oregon.

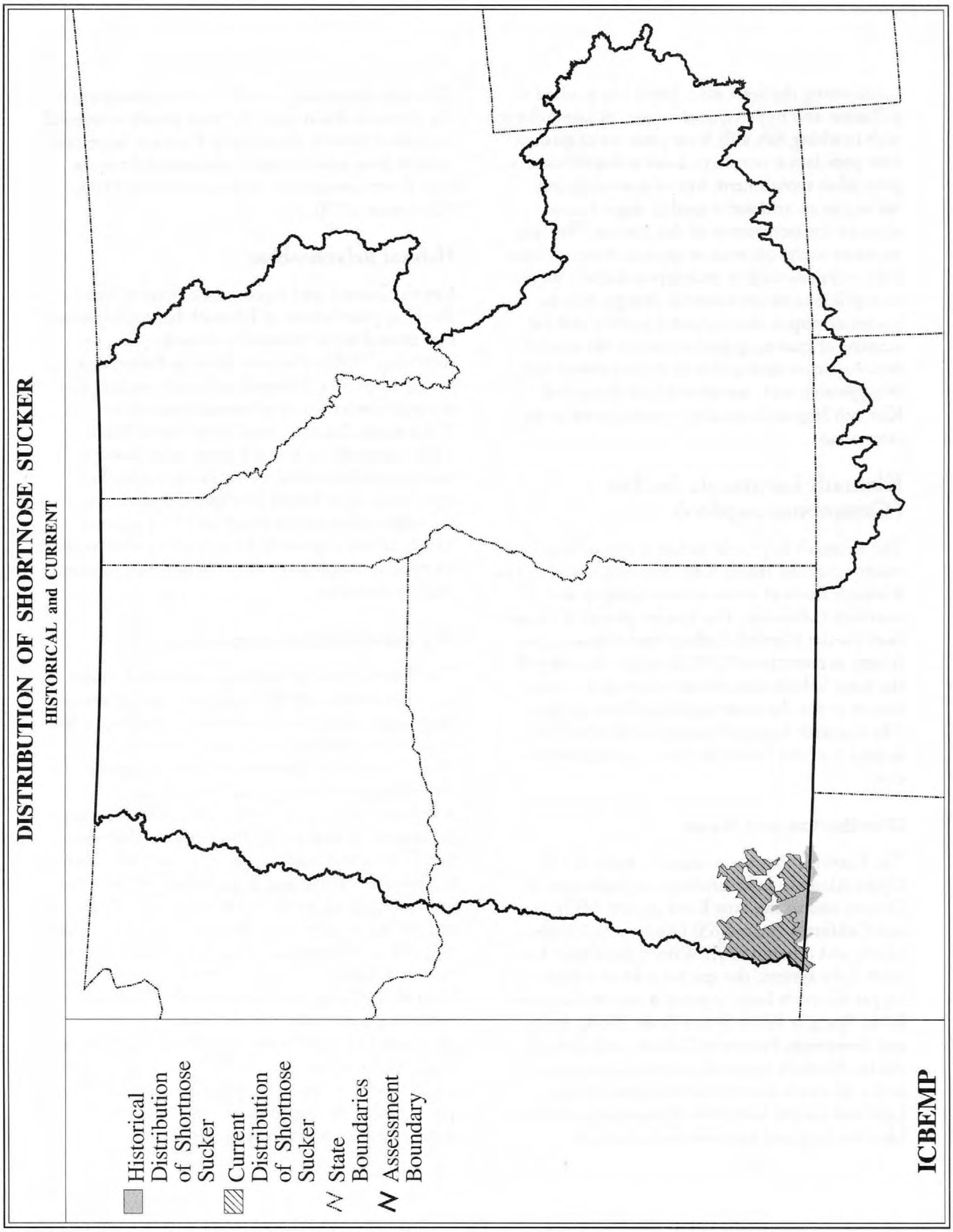
Habitat Relationships

Members of the family Catostomidae are primarily bottom-dwelling fishes although the nearly terminal mouth of the shortnose suggests adaptation for more mid-water lake habitats. During recent years, occurrences in lake habitats appear to be primarily influenced by adults seeking improved water quality conditions. Adult shortnose suckers spend relatively little time during spawning periods in riverine habitats. Larvae are surface oriented and tend to occur in shallow, shoreline habitats.

Key Factors Influencing Status

Factors leading to the decline of the shortnose sucker are multiple and complex. Like the Lost River sucker, the lack of recruitment is probably the greatest threat to its continued existence. The influence of numerous non-native fishes in Upper Klamath Lake is presumed to be negative because of the potential for competition or predation on larval and juvenile suckers. Non-native species may be a primary factor causing the lack of recruitment observed in Upper Klamath Lake and other lake habitats. Draining of marshes, channelization, and dams have been major factors contributing to loss of habitat and the decline of the shortnose sucker. Replacement of wetlands

[18]Personal communication. 1995. Paul Chappell, California Department of Fish and Game, Redding, California.



Map 4.62— Historical and current distribution of shortnose sucker.

surrounding the lakes with farms has resulted in pollution and hypereutrophication of lake habitats with resulting fish kills from poor water quality. Low population numbers, habitat fragmentation, poor adult recruitment, loss of spawning and rearing areas, and water quality degradation threaten the persistence of this species. The risk increases as the decrease in genetic diversity from habitat fragmentation makes populations more susceptible to environmental change. Also as spawning populations become smaller and the amount of spawning area decreases, the species becomes more susceptible to hybridization and introgression with remaining Lost River and Klamath largescale suckers, which spawn in the same areas.

Klamath Largescale Sucker (*Catostomus snyderi*)

The Klamath largescale sucker is one of four members of the family Catostomidae native to the Klamath Basin of south-central Oregon and northern California. The species provided a staple food for the Klamath Indians and a major sport fishery as recently as 1980. It occurs in many of the same habitats and shares many of the same threats as the shortnose and Lost River suckers. The Klamath largescale sucker is classified by Region 6 of the Forest Service as a sensitive species.

Distribution and Status

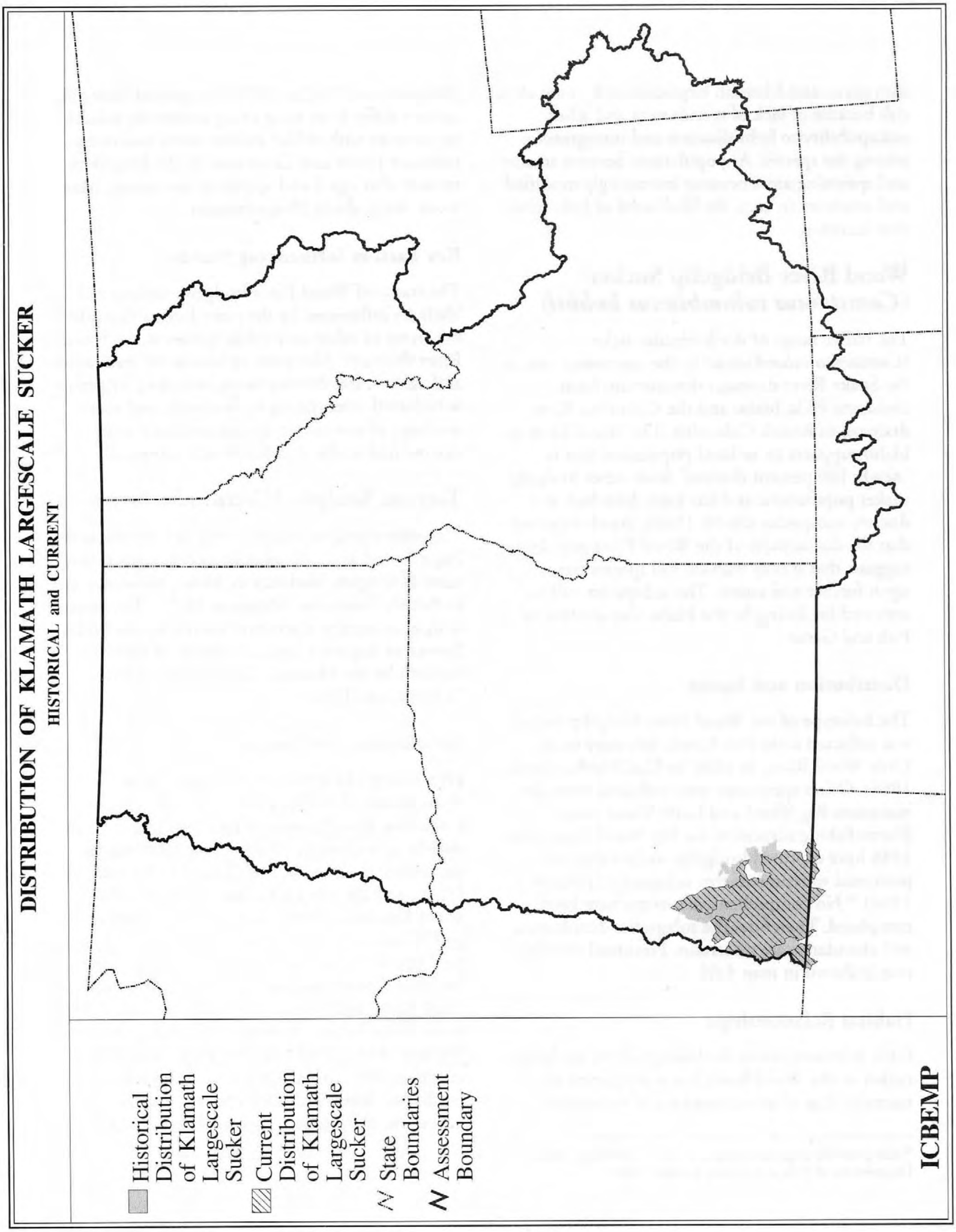
The Klamath largescale sucker is native to the Upper Klamath Lake drainage in south-central Oregon and to the Lost River system of Oregon and California (map 4.63) (Andreasen 1975b; Moyle and others 1989). Within the Upper Klamath Lake system, the species is known from Upper Klamath Lake, Agency Lake, Williamson River, Sprague River, Sycan River, Wood River, and Sevenmile, Fourmile, Odessa, and Crystal creeks. Klamath largescale suckers also may occur in the Klamath River between Upper Klamath Lake and Copco Reservoir. Populations of all the lake-dwelling and big river sucker species (Klamath largescale, Lost River, and shortnose) in the Klamath Basin have declined greatly compared with their historic abundance. Klamath largescale suckers have been virtually eliminated from the Lost River system since at least the early 1970s (Contreras 1973).

Habitat Relationships

Upper Klamath and Agency lakes harbor lake-dwelling populations of Klamath largescale suckers that ascend major tributaries during spring for spawning. Unlike the Lost River and shortnose suckers, however, Klamath largescale suckers also occur as resident riverine populations in the Williamson, Sprague, and Sycan rivers. While adult largescale suckers are oriented to bottom substrates of lakes and rivers, larvae are surface oriented and are found over gravel and cobble substrates of rivers (Buettner and Scoppettone 1990). In lakes, juvenile Klamath largescale suckers tend to congregate along the bottoms of gently sloping shorelines.

Key Factors Influencing Status

The historic marshes and interconnected waterways common to the Klamath and Lost rivers have been nearly completely modified and replaced by major irrigation projects. These projects have drained wetlands, channelized streams, and diverted large amounts of water for irrigation. Nearly one-third of the wetlands in the Klamath Basin were eliminated by the Klamath Reclamation Project and many more areas were inundated by reservoirs. Including degradation and loss from intensive grazing by livestock, ditching, tilling, and diking, total wetland loss approximates 75 to 90 percent of historical extent. As a result of these factors, habitats have been fragmented, access to historic spawning areas has been blocked, and remaining lakes have suffered from hypereutrophication and massive blooms of toxic blue-green algae. Modified habitats in the Klamath Basin now support large numbers of non-native fishes, which cause further decline in native sucker numbers through predation on young. The Lost River,



Map 4.63— Historical and current distribution of Klamath largescale sucker.

shortnose, and Klamath largescale suckers are all at risk because of lack of recruitment and adult susceptibility to hybridization and introgression among the species. As populations become smaller and spawning areas become increasingly modified and restricted in area, the likelihood of hybridization increases.

Wood River Bridgelip Sucker (*Catostomus columbianus hubbsi*)

The native range of the bridgelip sucker (*Catostomus columbianus*) in the assessment area is the Snake River drainage downstream from Shoshone Falls, Idaho and the Columbia River drainage to British Columbia. The Wood River in Idaho supports an isolated population that is "nearly 100 percent distinct" from other bridgelip sucker populations and has been described as a discrete subspecies (Smith 1966). Smith reported that the distinctness of the Wood River population suggests that it may warrant full species status upon further evaluation. The subspecies will be reviewed for listing by the Idaho Department of Fish and Game.

Distribution and Status

The holotype of the Wood River bridgelip sucker was collected from Fish Creek, tributary to the Little Wood River, in 1934 by Carl Hubbs (Smith 1966). Other specimens were collected from the mainstem Big Wood and Little Wood rivers. Electrofishing surveys of the Big Wood River since 1986 have captured bridgelip suckers that are presumed to represent this subspecies (Thurow 1990).[19] No comprehensive surveys have been completed. Therefore, the subspecies distribution and abundance are uncertain. Presumed distribution is shown in map 4.64.

Habitat Relationships

Little is known about the biology of the bridgelip sucker in the Wood River but it is believed to resemble that of other members of the species (Simpson and Wallace 1978). In general, bridgelip suckers differ from most other suckers by inhabiting streams with colder, swifter water and rocky substrate (Scott and Crossman 1973). Fish likely mature after age-2 and spawn in late spring. Maximum size is about 25 centimeters.

Key Factors Influencing Status

The status of Wood River bridgelip suckers will likely be influenced by the same factors that affect the status of other native fish species in the Wood River drainage. Alteration of habitat by residential and agricultural development, including irrigation withdrawal, overgrazing by livestock, and introductions of non-native species are likely to be detrimental to the viability of this subspecies.

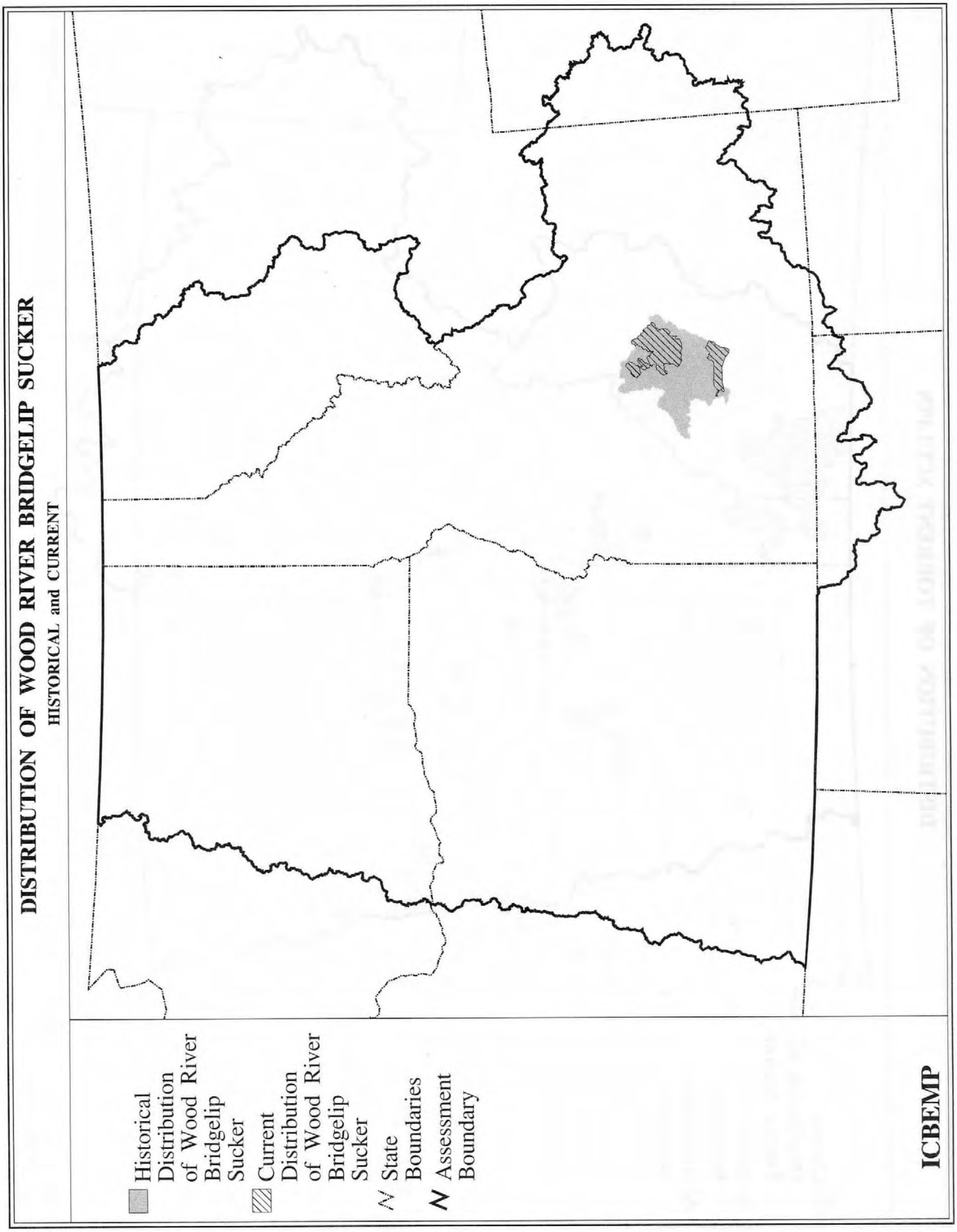
Torrent Sculpin (*Cottus rhotheus*)

The torrent sculpin inhabits rivers and streams in the Puget Sound and Columbia River drainages in the states of Oregon, Washington, Idaho, Montana, and in British Columbia (Maughan 1976). The torrent sculpin is listed as a sensitive species by the Forest Service in Region 1 and as a species of special concern by the Montana Department of Fish, Wildlife, and Parks.

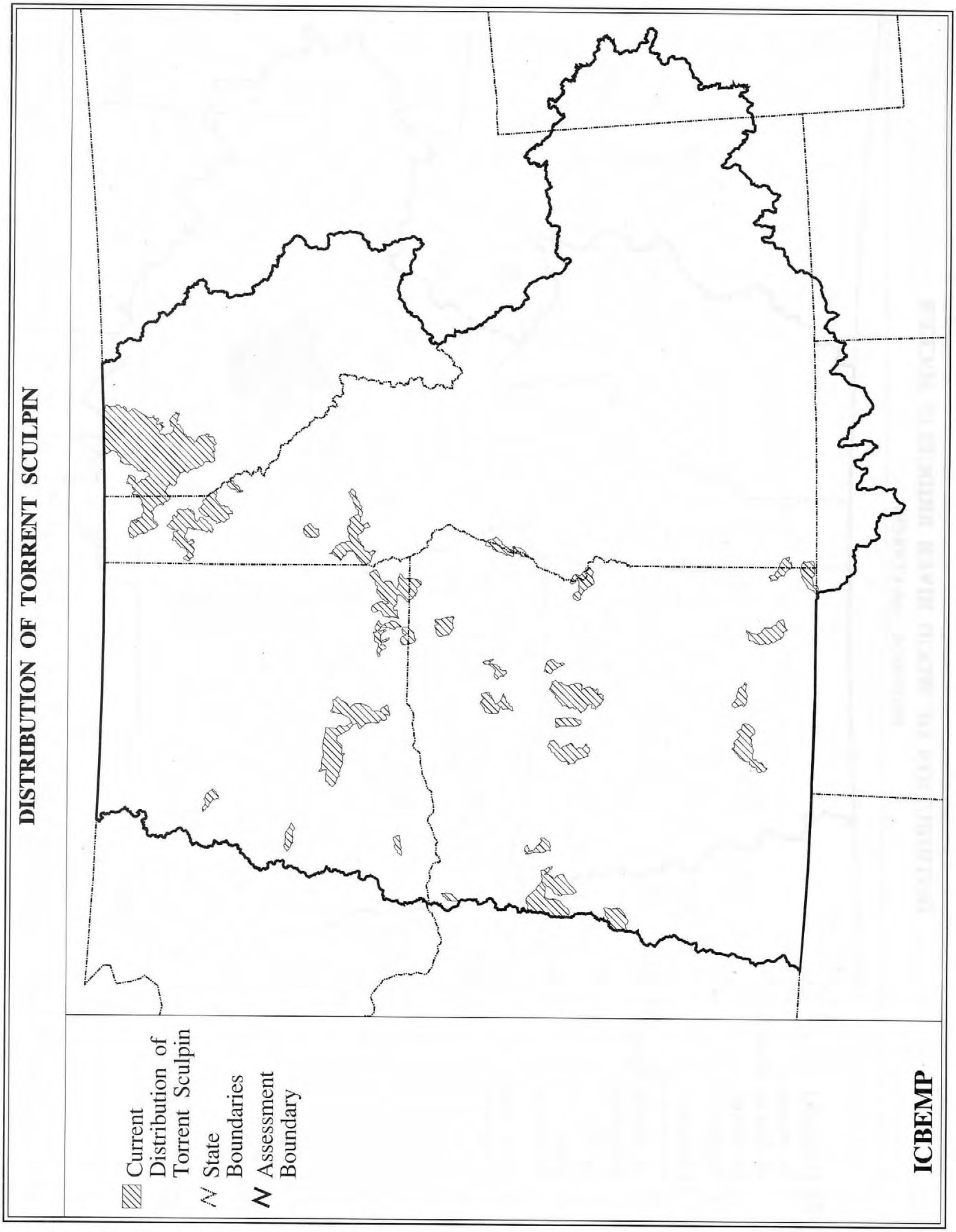
Distribution and Status

Historically, the torrent sculpin was found throughout tributaries of the mid- and upper Columbia River Basin, overlapping range with the shorthead sculpin. In Oregon and Washington, the torrent sculpin has been found in the Yakima, Entiat, and Palouse rivers and tributaries of the upper Deschutes River (map 4.65) (Wydoski and Whitney 1979). In Idaho, the torrent sculpin has been found in the Kootenai, Pend Oreille, Spokane, Clearwater, Salmon, St. Maries, upper Clark Fork, and Palouse river drainages and in the Snake River below Shoshone Falls (Simpson and Wallace 1978). MacPhee (1966) also found the torrent sculpin in Rochat Creek, a tributary to the St. Joe River. Although the current status is unknown, the present range probably remains

[19]Also personal communication. 1995. F. Partridge, Idaho Department of Fish and Game, Jerome, Idaho.



Map 4.64— Current distribution of Wood River bridgelip sucker.



Map 4.65— Current distribution of torrent sculpin.

similar to the historic range. When found, the torrent sculpin was in low numbers compared with the shorthead sculpin (MacPhee 1966).

Habitat Relationships

The torrent sculpin is primarily a benthic stream species but also occurs in lakes (Wydoski and Whitney 1979). In streams, they inhabit swift, cool, clear riffles with a stable bottom of scattered rubble, gravel, and boulder substrate (Simpson and Wallace 1978). Torrent sculpin were found only where rock substrate was present. Spawning, which occurs in riffle areas with rubble or boulder substrate, starts in early spring and lasts to late April.

Key Factors Influencing Status

The torrent sculpin is probably susceptible to land use practices (for example, livestock grazing, road construction, and mining) and climatic events that degrade water quality and diminish its natural habitat. Pollution, increased water temperatures, and sedimentation are the major potential negative impacts to sculpin populations.

Shorthead Sculpin (*Cottus confusus*)

The shorthead sculpin inhabits rivers and streams in the Puget Sound and Columbia River drainages in the states of Oregon, Washington, Idaho, Montana, and in British Columbia (Bailey and Bond 1963). The shorthead sculpin is listed by the Forest Service as sensitive and by the Montana Department of Fish, Wildlife, and Parks as a species of special concern.

Distribution and Status

In the assessment area, the shorthead sculpin is found in the upper reaches and tributaries of the Deschutes, Yakima, Wenatchee, and John Day river systems in the lower Columbia River Basin and the Snake, Spokane, Pend Oreille, and Kootenai River drainages in the mid and upper Columbia River Basin (map 4.66) (Bailey and Bond 1963). In the mid-Columbia River Basin, the species has been collected in the Coeur d'Alene and St. Joe river systems in Idaho (Maughan 1976), and in the upper Columbia in the Flathead, Little Blackfoot, and Blackfoot river systems in Montana and the Flathead River in British Columbia (Bailey and Bond 1963). In the Snake River drainage, the species occurs in the mainstem and tributaries of the Grande Ronde and Imnaha river systems in Oregon and the Clearwater, Salmon, Boise, Owyhee, Big Lost, and Little Lost rivers and Birch, Medicine, and Beaver creeks in Idaho (Maughan 1976). In 1994, surveys found shorthead sculpin in the upper Deschutes drainage in Oregon, the upper and lower Kootenai, Yaak, and Fisher rivers in Montana, the upper Yakima in Washington, and the upper and South Fork Coeur d'Alene, and Little Lost rivers, and Birch Creek in Idaho.[20]

Habitat Relationships

The shorthead sculpin prefers riffle areas and small coldwater rivers and streams (Bowler 1974). It has been collected in emergent grasses and slow-moving waters along shorelines and in backwaters. Spawning, which occurs in riffle areas with a rubble/boulder substrate, starts in early spring and lasts to late April.

Key Factors Influencing Status

The shorthead sculpin is probably susceptible to land use practices (for example, timber harvest activities, road construction, and mining) and/or climatic events that degrade water quality. Pollution, increased water temperatures, and sedimentation are the major potential negative impacts to sculpin populations. Once stream populations are eliminated, sculpin are poor recolonizers from adjacent areas as compared with trout.

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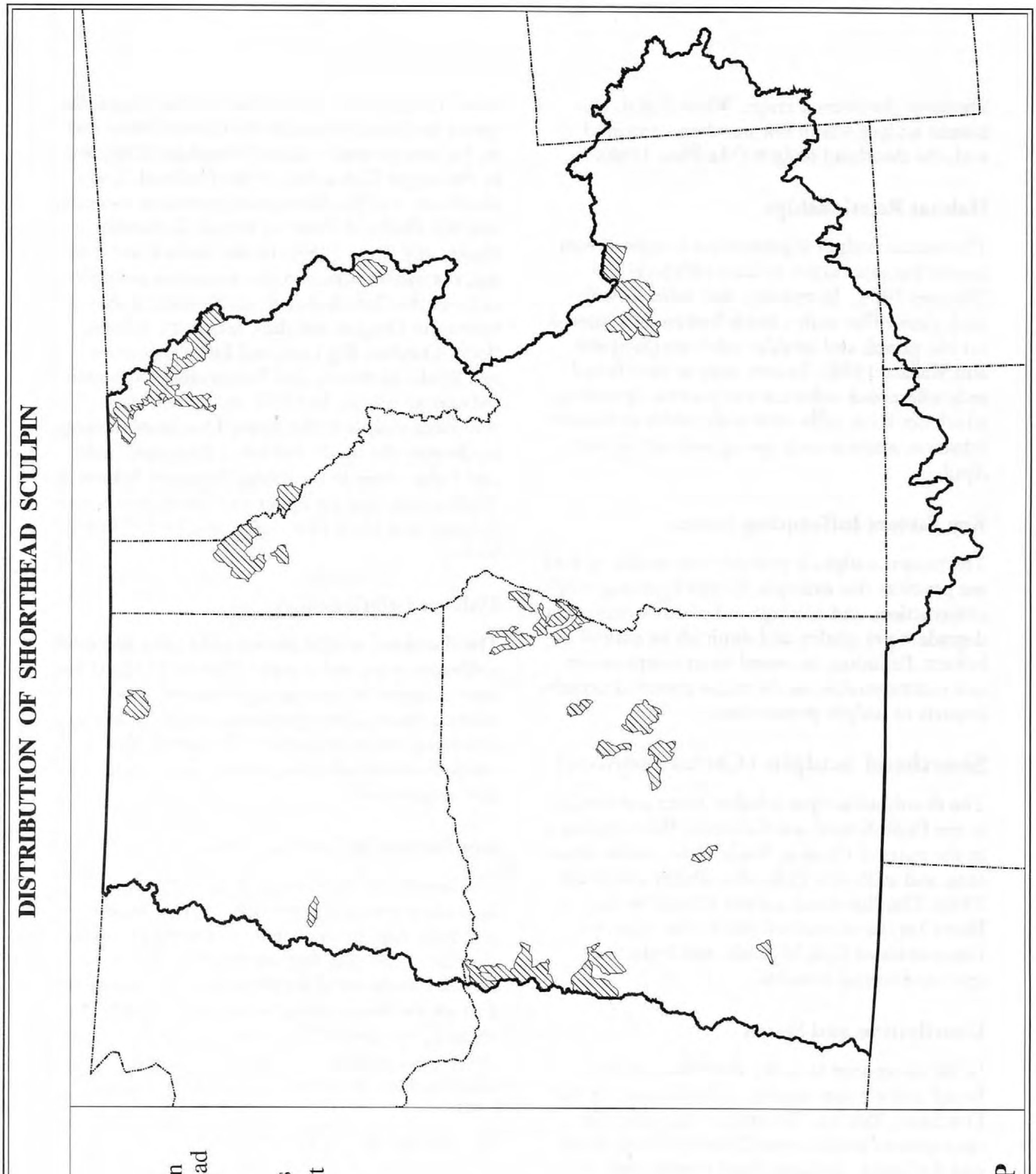
[20]Personal communication. 1995. Lori Leatherbury, Forest Service, Boise, Idaho.



Map 4.66— Current distribution of shorthead sculpin.

Pit Sculpin (*Cottus pitensis*)

Pit sculpin were historically widespread in the Pit River system, from the Goose Lake Basin to the Sacramento River (Moyle 1976). In Oregon, the Pit sculpin is currently listed as sensitive by the Forest Service and BLM, and as a peripheral/rare species by the Oregon Department of Fish and Wildlife.

Distribution and Status

In Oregon, the extent of the historical distribution of Pit sculpin in the Goose Lake Basin is unknown. However, the sculpin was recorded both historically and recently (1979), and can still be found, in a few tributaries of Goose Lake, specifically Cottonwood and Drews creeks (map 4.67) (Long and Bond 1979; ODFW 1992). Pit sculpin also were recorded historically in Thomas Creek, Oregon, but surveys in 1983 and 1988 yielded no sculpin.[21] The Pit sculpin has never been found in Goose Lake. In California, the Pit sculpin is still widespread throughout the Pit River system and has recently been found in Lassen, Gold, and Willow creeks, all tributaries of Goose Lake. The Pit sculpin, where found, tends to be rare.

Habitat Relationships

Pit sculpin are not known to migrate. They are a benthic species found mainly in small, cold water tributaries (Moyle 1976). They show a strong preference for riffle habitat with small gravel substrate (Moyle 1976).[22] In the winter, they use the interstices among large gravels and cobbles during near-freezing and ice formation periods. They occupy water that ranges in temperature from 6.5° C in March to 18.8° C in June.[23]

[21]Personal communication. 1995. J. Williams, Bureau of Land Management, Boise, Idaho. Personal communication of unpublished data.

[22]Personal communication. 1995. J. Williams, Bureau of Land Management, Boise, Idaho. Personal sommunication of unpublished data.

[23]Personal communication. 1995. J. Williams, Bureau of Land Management, Boise, Idaho. Personal communication of unpublished data.

Factors Influencing Status

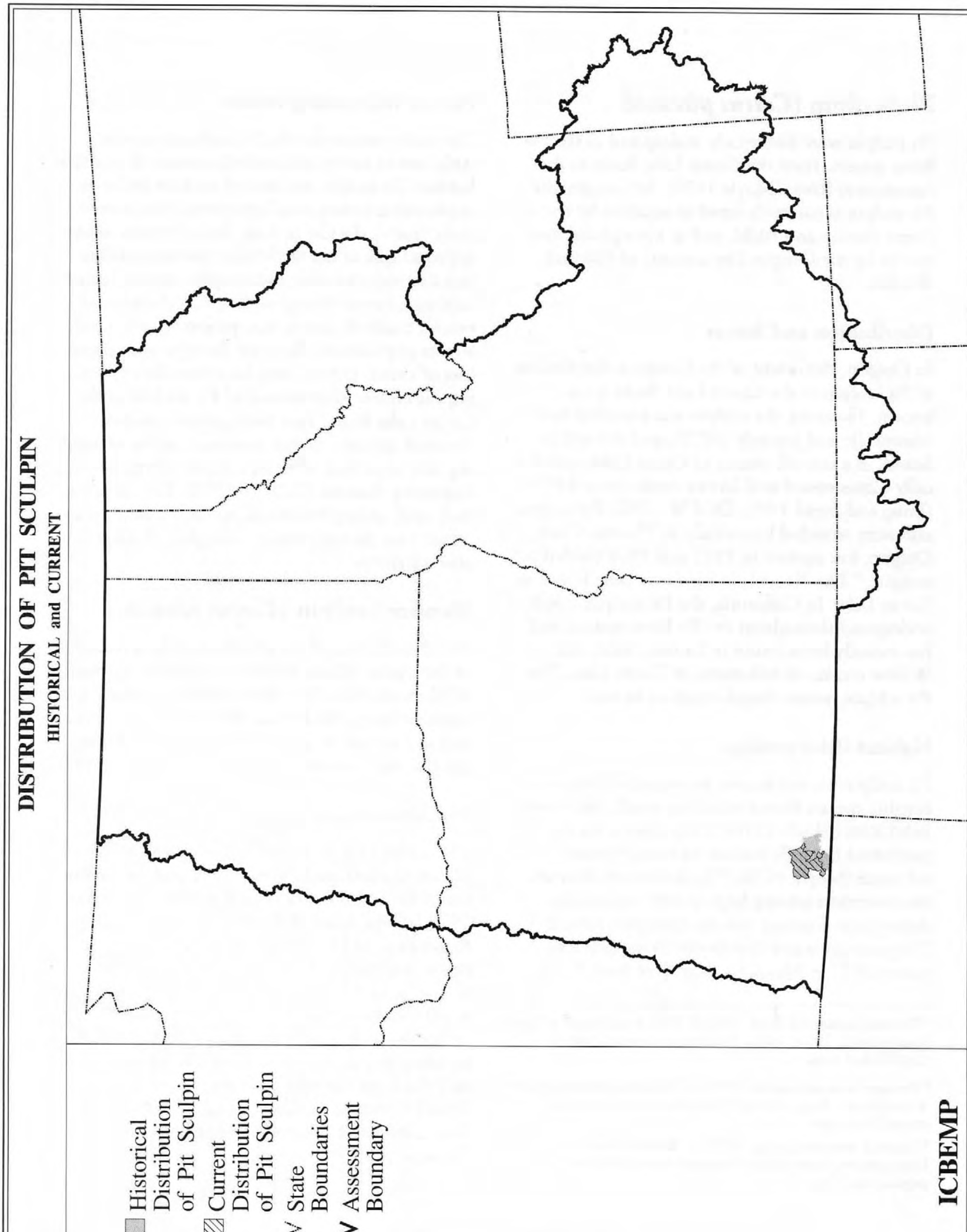
The main reasons for the Pit sculpin's current status are its rarity and limited amount of available habitat. Pit sculpin are limited in their ability to recolonize a stream once extirpated, this is especially true in the Goose Lake Basin because of the apparent lack of use of the lake. Because of their inability to recolonize, catastrophic events, loss of habitat, extreme droughts, and introductions of exotics could all lead to extirpation of individual stream populations. Extreme drought events and loss of stream habitat may have been factors limiting the current distribution of Pit sculpin in the Goose Lake Basin. Past management, such as livestock grazing, timber activities, and road building, has negatively affected sculpin habitat by increasing siltation (ODFW 1992). Loss of perennial, cool, spring-fed stream reaches, which act as refuge sites during extreme droughts, also has affected status.

Slender Sculpin (*Cottus tenuis*)

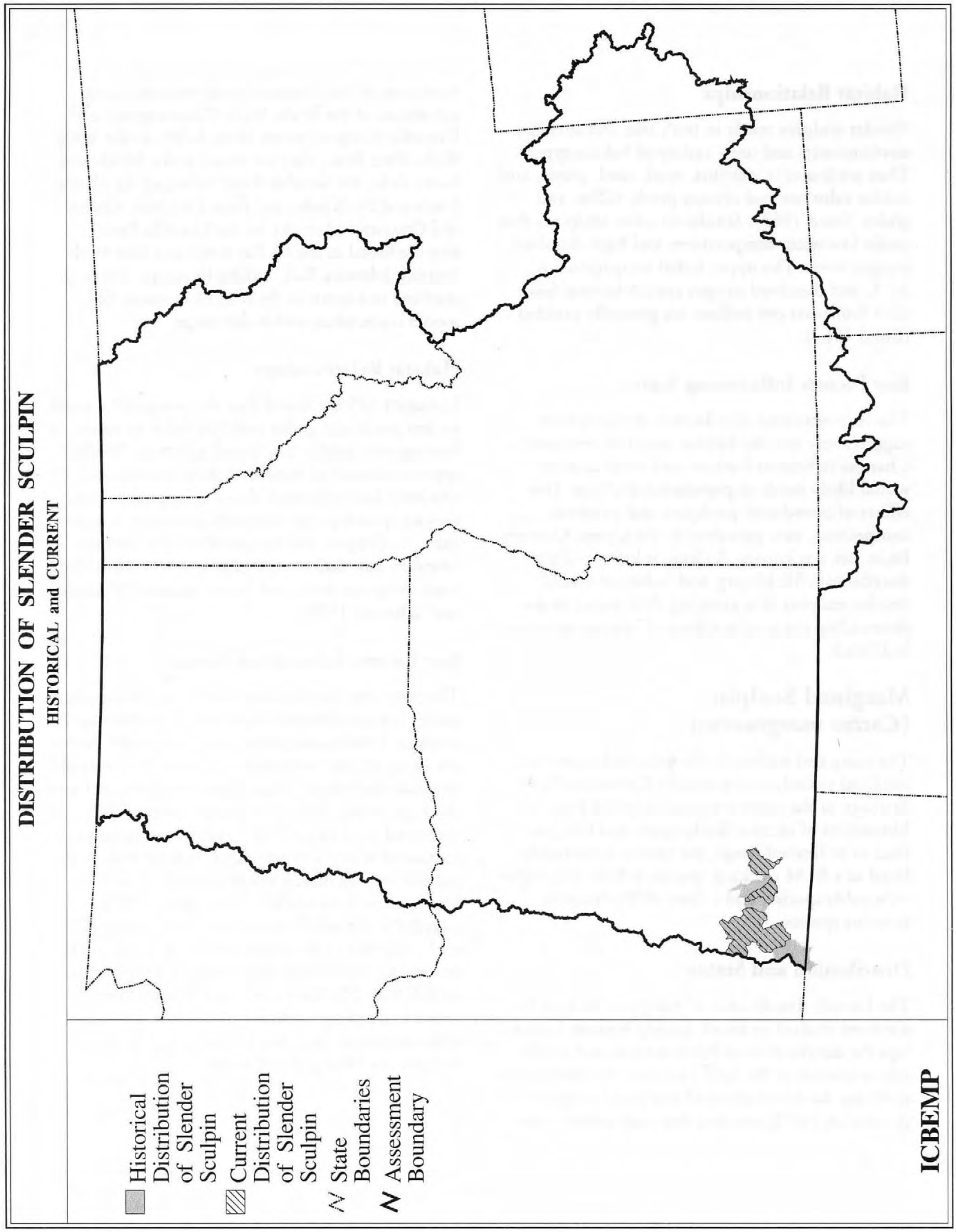
The distribution of the slender sculpin is restricted to the Upper Klamath Basin in Oregon, upstream of Klamath Falls. The slender sculpin is listed as a sensitive species by Region 6 of the Forest Service and as a species of special concern by the American Fisheries Society (Williams and others 1989).

Distribution and Status

The sculpin was historically distributed in the Upper Klamath and Agency lakes and their tributaries. Recorded distributions, between 1934 and 1976, are the lower Williamson and Sycan rivers, South Fork and mainstem Sprague and Wood rivers and Odessa, Denny, Crystal Camporee Spring, Crooked, Fort, and Sevenmile creeks (map 4.68) (Ford and Thomas 1993). The current status of slender sculpin is not well known, but sampling by Oregon State University and the Forest Service in 1992 found slender sculpins in sections of the Wood River and Larkin, Crystal, Crooked, and Fort creeks, and Fourmile Springs (Ford and Thomas 1993).



Map 4.67— Historical and current distribution of pit sculpin.



Map 4.68— Historical and current distribution of slender sculpin.

Habitat Relationships

Slender sculpins reside in both lake and stream environments and use a variety of habitat types. They are found in detritus, mud, sand, gravel, and rubble substrate and occupy pools, riffles, and glides (Bond 1963). Similar to other sculpins, they prefer low water temperatures and high dissolved oxygen levels. The upper lethal temperature is 31° C and dissolved oxygen concentrations less than four parts per million are generally avoided (Bond 1963).

Key Factors Influencing Status

The very restricted distribution of this species suggests very specific habitat needs or preferences. Changes in habitat features and water quality would likely result in population declines. The effects of introduced predators and potential competitors, now prevalent in the Upper Klamath Basin, are not known. So little is known of the distribution, life history, and habitat needs of slender sculpins that assessing their status or understanding the precise effects of human activities is difficult.

Margined Sculpin (*Cottus marginatus*)

The margined sculpin is the only freshwater fish confined entirely to the middle Columbia River drainage in the northern portion of the Blue Mountains of eastern Washington and Oregon. Due to its limited range, the species is currently listed as a BLM tracking species, a State of Oregon vulnerable species; and a State of Washington monitor species.

Distribution and Status

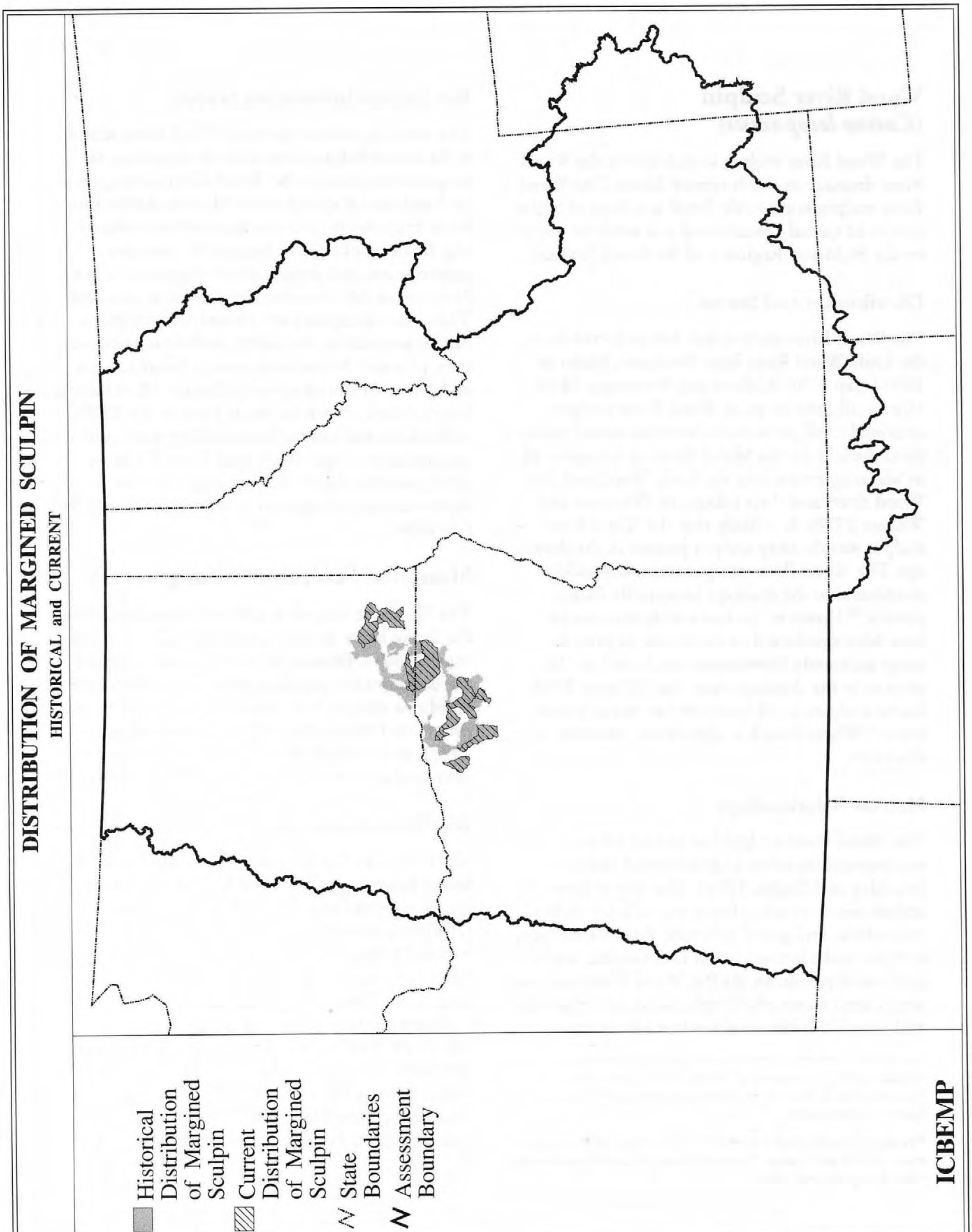
The historic distribution of margined sculpin has not been studied in detail, mainly because it overlaps the distribution of Paiute sculpin and is difficult to identify in the field. The most comprehensive study on the distribution of margined sculpins (Lonzarich 1993) revealed that they occur in the mainstem of the Tucannon and mainstem and tributaries of the Walla Walla (Washington) and Umatilla (Oregon) rivers (map 4.69). In the Walla Walla River Basin, they are found in the North and South forks, the Touchet River including the North, South and Wolf forks, and Pine, Dry, Mill, Couse, and Cottonwood creeks. In the Umatilla Basin, they are found in the McKay River and East Birch, Pearson, Johnson, Rail, and Butler creeks. Although confined to streams in the Blue Mountains, the species is abundant within this range.

Habitat Relationships

Lonzarich (1993) found that the margined sculpin prefers pools and glides with low velocity water flowing over cobble and gravel substrate. While it appears tolerant of variable habitat conditions, relatively little is known about this species. Most sculpin species prefer relatively low water temperatures. In Oregon, the margined sculpin is often found in association with rainbow trout, speckled dace, longnose dace, and Paiute sculpin (Wydoski and Whitney 1979).

Key Factors Influencing Status

The restricted distribution of the margined sculpin makes it especially vulnerable to environmental changes. Land management activities could lead to the rapid decline in margined sculpin population numbers and status. Major potential effects from these activities include reduced amounts of preferred pool habitat, habitat fragmentation, increased water temperatures, loss of streamside vegetation, increased water velocities, and increased sedimentation. Lonzarich (1993) noted the degraded condition from grazing and logging in the Touchet River, particularly the South and Wolf forks. Also, Oregon Department of Fish and Wildlife (1992) cited potential impacts from agricultural chemicals, heavy sedimentation, and toxic material spills as reasons for listing the species.



Map 4.69— Historical and current distribution of margined sculpin.

Wood River Sculpin (*Cottus leiopomus*)

The Wood River sculpin is endemic to the Wood River drainage in south-central Idaho. The Wood River sculpin is currently listed as a State of Idaho species of special concern and as a sensitive species by the BLM and Region 4 of the Forest Service.

Distribution and Status

The Wood River sculpin was first collected from the Little Wood River near Shoshone, Idaho in 1893 (map 4.70) (Gilbert and Evermann 1895). Historically, the range of Wood River sculpin consisted of all permanent, interconnected waters from the falls on the Malad River at Interstate 84 in Idaho upstream into the Little Wood and Big Wood rivers and their tributaries (Simpson and Wallace 1978). It is likely that the Wood River sculpin was the only sculpin present in the drainage. The Wood River sculpin was more widely distributed in the drainage historically than at present.[24] However, no basin-wide inventories have been conducted to determine its present range accurately. Inventories conducted on 26 streams in the drainage from the 1970s to 1990s found sculpins at 40 locations but not at 8 locations.[25] Where found, sculpin were common to abundant.

Habitat Relationships

The Wood River sculpin has similar habitat requirements as other sculpins found in Idaho (Merkley and Griffth 1993). The Wood River sculpin seems to select fast water riffles with boulder, cobble, and gravel substrate. In some streams, sculpins make limited use of overhanging banks and beaver ponds. In the Big Wood River, sculpins were found seasonally in ephemeral side channels with suitable cobble and boulder substrates.

[24]Wallace, R.L. University of Idaho. 1995. Letter dated September to B. Reininger, Idaho Department of Fish and Game, Jerome, Idaho.

[25]Personal communication. 1995. F. Partridge, Idaho Department of Fish and Game, Jerome, Idaho. Personal communication of unpublished data.

Key Factors Influencing Status

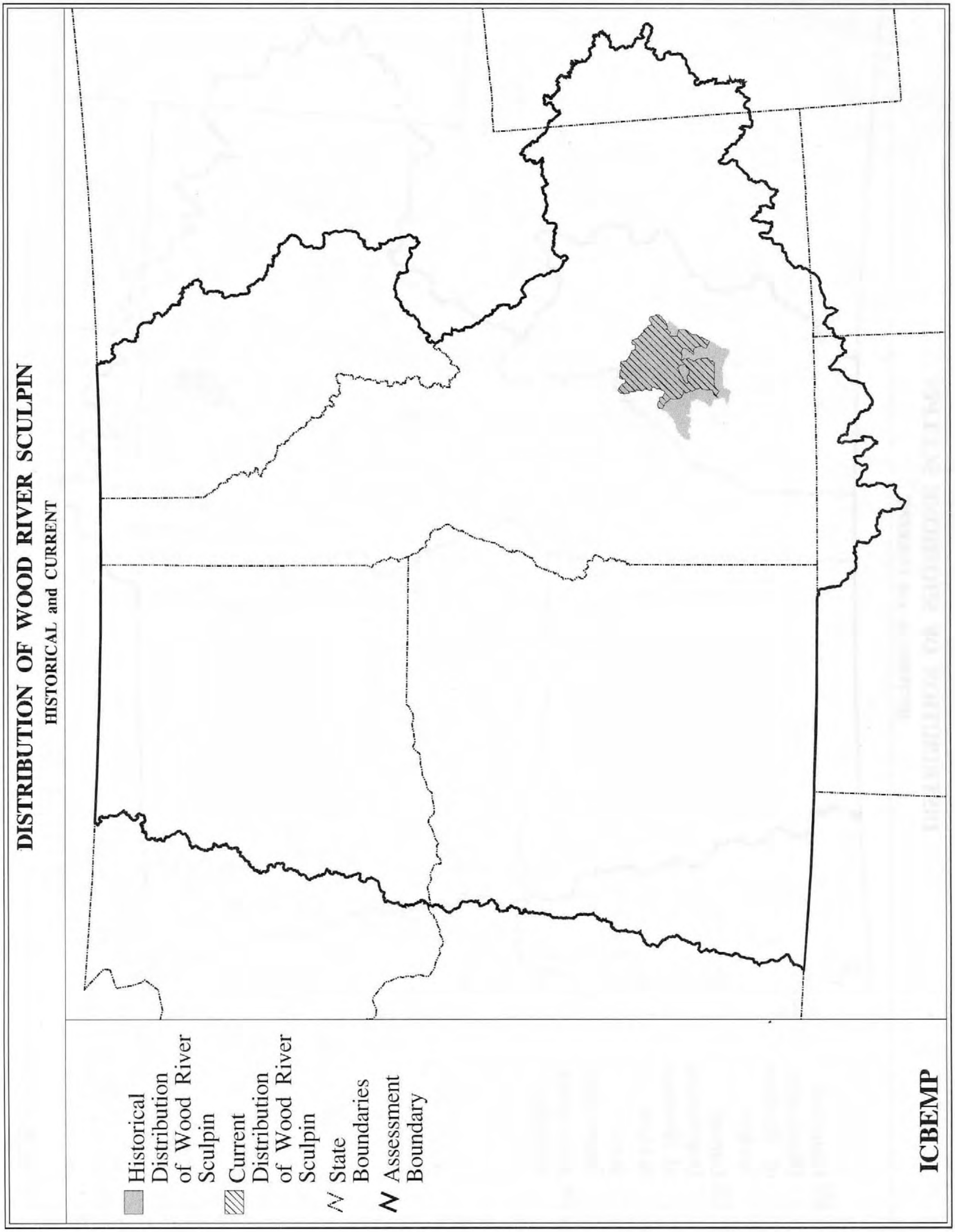
The most significant threat to Wood River sculpin is the loss of habitat caused by development of irrigation projects in the Wood River drainage and by floodplain encroachment. Habitat destruction from irrigation projects includes stream dewatering, flooding of stream channels by reservoir construction, and formation of migration barriers. Non-native fish introductions have also occurred. These introductions have created adverse effects on the population, including predation, competition, potential hybridization with Paiute sculpin, and potential introduction of disease. Most habitat loss occurred on private lands prior to the 1930s, with additional habitat loss resulting from overgrazing in the upper Big Wood River. More recently, sculpin habitat has been degraded by agriculture and residential development within the floodplain.

Shoshone Sculpin (*Cottus greenei*)

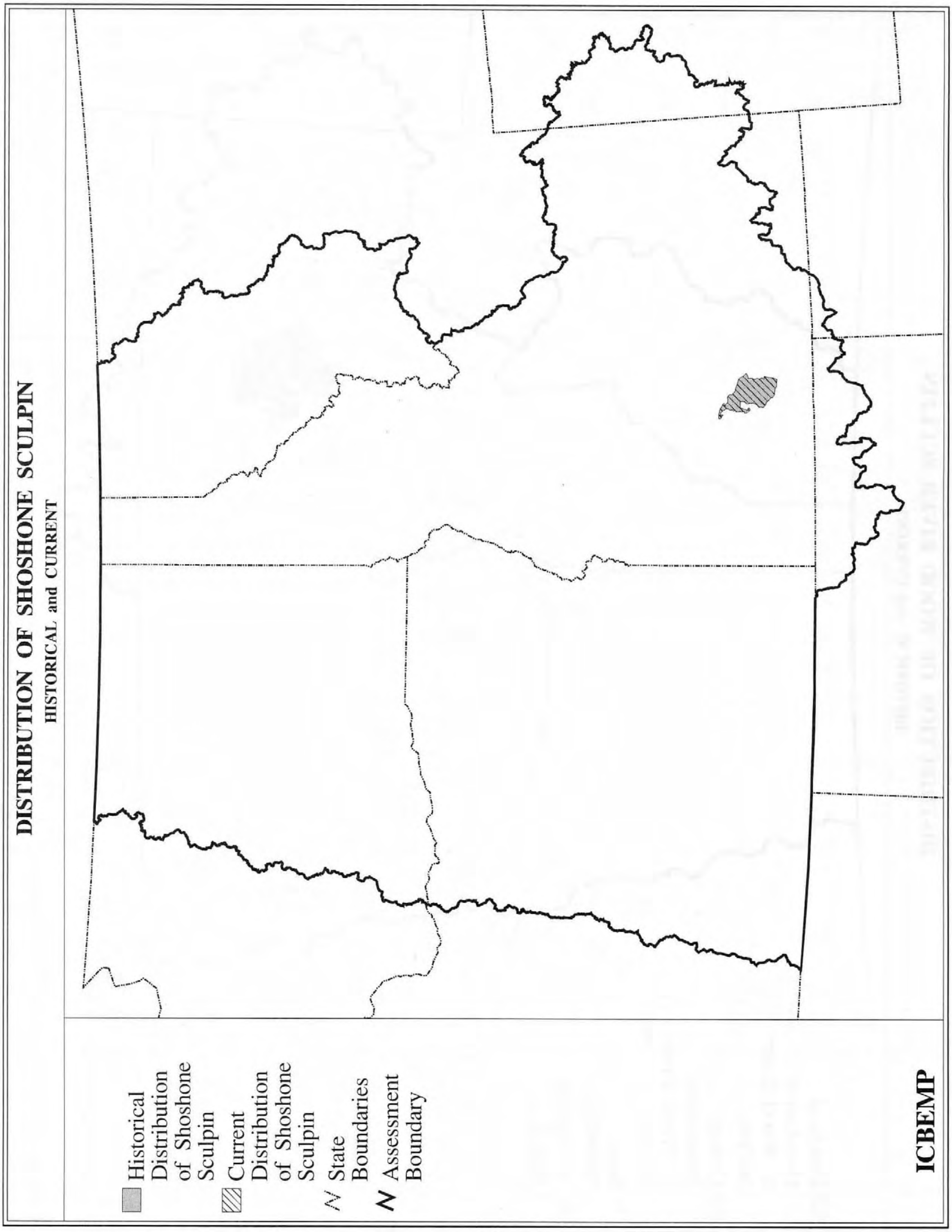
The Shoshone sculpin is endemic to springs along the Snake River in the Hagerman Valley of south-central Idaho. Because of its restricted range and the development pressures on spring systems, the Shoshone sculpin is classified as threatened by the American Fisheries Society (Williams and others 1989), as a State of Idaho species of special concern, and as a sensitive species by the BLM.

Distribution and Status

Shoshone sculpins are restricted to portions of the Snake River in south-central Idaho that contain spring systems (map 4.71). Wallace and Griffith (1982) reported the species from 49 locations within 25 spring systems in the Hagerman Valley. Most locations are along the north bank of the river in the Thousand Springs formation of Gooding County. Two localities along the south side of the river in Twin Falls County contained Shoshone sculpin. Shoshone sculpin also were collected from Billingsley Creek, a tributary of the Snake River near Hagerman. Many spring and stream systems in the region contain the more



Map 4.70— Historical and current distribution of Wood River sculpin.



Map 4.71— Historical and current distribution of Shoshone sculpin.

common mottled sculpin (*C. bairdi*). Results of a recent Idaho Chapter American Fisheries Society Workshop (IDAFS 1995) indicate some population loss but an overall "fairly stable" population status during the past decade.

Habitat Relationships

Shoshone sculpin are found only in association with spring outflows. Highest population densities occur near the headwaters of springs and lower densities occur with decreasing influence of spring water on water quality (Wallace and Griffith 1982). Younger sculpins prefer areas with more plant cover and fewer large rocks than do older Shoshone sculpins. Both juvenile and adult Shoshone sculpins prefer relatively low water velocities (less than 20 cm per second) with temperatures near 15° C (Wallace and Griffith 1982).

Key Factors Influencing Status

Shoshone sculpins are dependent on maintenance of the natural integrity of spring flows. The numerous spring systems that are located along the Snake River provide a valuable, but finite resource. Diversion of spring water has caused dewatering of Shoshone sculpin habitat in many areas (Wallace and Griffith 1982). Spring waters are used for a variety of purposes, including aquaculture facilities, hydropower operations, and agriculture. These practices have diverted, dried, and polluted springs. Various private and Federal fish hatcheries also serve as sources of non-native invertebrates, fishes, diseases, and parasites.

Malheur Sculpin (*Cottus bairdi* spp.)

The Malheur sculpin is endemic to the Harney Basin of southeastern Oregon. This undescribed subspecies is listed as a sensitive species by the State of Oregon.

Distribution and Status

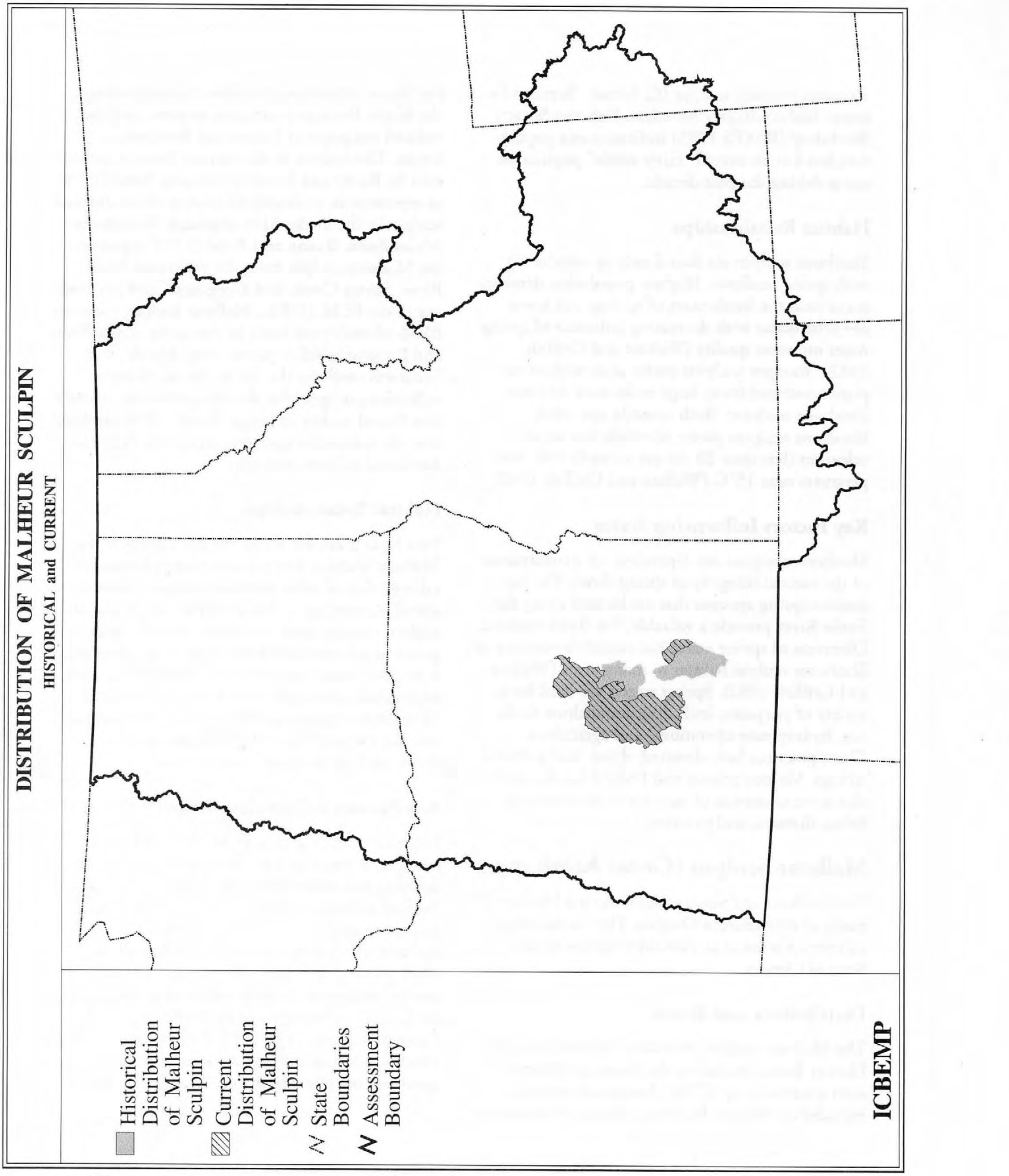
The Malheur sculpin is endemic to streams in the Harney Basin, including the Silvies and Blitzen river systems (map 4.72). Historic distribution includes the Blitzen River and tributary streams on the Steens Mountain, the Silver Creek drainage, the Silvies River and tributary streams, and the isolated drainages of Poison and Rattlesnake creeks. The sculpin in the Harney Basin is considered by Bailey and Bond (1963) and Bond (1974) to represent an undescribed relative of the mottled sculpin in the Snake River drainage. Within the Silvies Basin, Bisson and Bond (1971) reported the Malheur sculpin from the mainstem Silvies River, Scotty Creek, and Emigrant Creek. According to the BLM (1992), Malheur sculpin occur on BLM-administered lands in the upper Silvies River and Emigrant, Yellowjacket, Hay, Myrtle, and Sawtooth creeks in the Silvies Basin. Historic collections suggest that the subspecies was broadly distributed within its range. Bond (1974) reported that the subspecies has been extirpated from the Rattlesnake Creek subbasin.

Habitat Relationships

Very little is known about the life history of the Malheur sculpin, but it is assumed to be comparable to that of other mottled sculpins, *Cottus bairdi*. According to Bond (1974), the Malheur sculpin requires cool-water streams with large gravel or rubble substrates for cover and spawning. It requires water temperatures below 26° C, with high dissolved oxygen and very low turbidity. Given these characteristics, the Malheur sculpin can occupy small headwater streams and larger rivers, such as the lower Blitzen River.

Key Factors Influencing Status

Malheur sculpin appear to be very sensitive to changes in water quality, including increases in temperature, sediments, and turbidity. Aquatic habitat and water quality conditions are considered poor in the upper Silvies River because of elevated water temperature, silt loads, and livestock grazing; and poor on Hay and Yellowjacket creeks because of elevated water temperature, silt loads, and timber harvesting (BLM 1992). On Emigrant Creek, BLM (1992) rated water quality condition as fair and aquatic habitat condition as good but considers both conditions to be declin-



Map 4.72— Historical and current distribution of Malheur sculpin.

ing because of siltation, high water temperature, and upstream impacts from livestock grazing and logging. Biotic interactions are not specifically known, but the occurrence of the Malheur sculpin would appear to be negatively correlated with presence of introduced warm-water fishes such as catfish and centrarchids, which are more tolerant of turbid water conditions. Elevated water temperature, increased turbidity, and sediment transport caused by activities such as livestock grazing, road construction, and timber harvest activities are detrimental to the sculpin and have been cited by the BLM (1992) as causes for the decline of Malheur sculpin populations in the mainstem Silvies, Hay, Yellowjacket, and Emigrant creeks within the Silvies Basin.

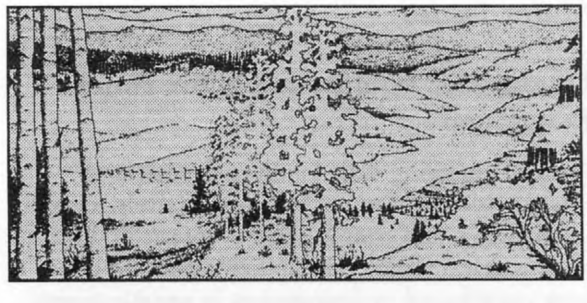
Conclusions

Approximately 50 percent of the native fish taxa in the assessment area has exhibited significant declines compared with historic levels. Presently, 45 of the 88 native fish taxa (51%) in the assessment area are identified as threatened, endangered, sensitive, or of special concern by state or Federal agencies or by the American Fisheries Society. Eleven fish taxa are formally listed as threatened or endangered under the Endangered Species Act. The following important concepts have emerged from our review of historic and current distributions of rare fish and existing literature.

1. Many of these taxa are vulnerable to anthropogenic effects and stochastic events because of their restricted range and fragile habitat requirements. Four cyprinid fishes are restricted to one or two small spring systems: Borax Lake chub, Summer Basin tui chub, Hutton tui chub, and Foskett speckled dace. Several other species appear to be extremely rare, but their precise status is unclear. These species include the Klamath lamprey, Goose Lake lamprey, Lost River sucker, and shortnose sucker. The spring-dwelling fishes, such as the Borax Lake chub and Hutton tui chub, occupy unique desert habitats that are easily disturbed by off-road vehicles, livestock grazing, or other surface disturbances. Spring aquifers and associated substrates also may be degraded from over-withdrawal of groundwater, geothermal exploration, or other subsurface drilling. Many spring habitats of the Shoshone sculpin have been lost because of aquaculture facilities and small hydropower operations. Such species require protection of remaining habitat if the species are to persist.

2. Many of these taxa are subject to increased likelihood of extinction or extirpation as environmental variability combines with low population sizes. The native lake-dwelling suckers of the Klamath Basin (shortnose, Lost River, and Klamath largescale) have declined to the extent that hybridization among the species may be occurring within restricted, remaining spawning areas. Water quality in Upper Klamath and Agency lakes has severely reduced the distribution of native fishes and has caused fish mortality in the past. Drought exacerbates problems of water quality in the lakes. In the Warner Valley, the threatened Warner sucker was nearly eliminated during the drought of the early 1990s by a combination of the desiccation of lake habitats and the diversion dams that prevented access to lower sections of remaining tributaries. The species survived largely through temporary transfer of individuals to the Summer Basin. Drought caused similar problems to the Goose Lake lamprey when Goose Lake dried in the early 1990s.

3. Many of these taxa are poorly understood and in need of study and rigorous monitoring efforts. We know very little about the lampreys (with the possible exception of the Pacific lamprey), the northern roach, the leatherside chub, most of the tui chubs, the Foskett speckled dace, the Wood River bridgelip sucker, and the Malheur sculpin. Several of these taxa, for example the Hutton tui chub and the Foskett speckled dace, have long been recognized as distinct at the subspecific level yet lack formal scientific descriptions. Monitoring of habitat and populations is essential if management is to respond to factors that threaten the persistence of the narrowly distributed endemics.



ISSUES AND OPPORTUNITIES

In the introduction to their influential paper describing the status of 214 Pacific salmon stocks from California, Oregon, Idaho, and Washington, Nehlsen and others (1991) describe Pacific salmon as being at a crossroads:

> In the 1990s, native anadromous Pacific salmon (*Oncorhynchus* spp.) are at a crossroads, the habitats of these once wide-ranging fishes are severely curtailed, many stocks are extinct, and many remaining stocks face a variety of threats.

Unfortunately, Pacific salmon are not the only aquatic resource at a crossroad. As described in earlier sections, many other fish taxa are threatened by developmental activities within the Basin that affect aquatic habitats. Aquatic ecosystem integrity is degraded in much of the Basin; however, many important elements of functioning aquatic systems remain. Native species still appear to dominate many waters, and important population strongholds and regions of high biological integrity are found in watersheds under federal management. These areas are critical to short-term conservation and long-term rehabilitation of ecologically functional aquatic systems throughout the Basin. Restoring aquatic integrity is a challenge that extends well beyond the 1990s.

The emerging paradigm of ecosystem management is a new way of doing business to solve current forest, watershed, and aquatic health issues on federal lands. An assumed goal of ecosystem management is to maintain or rehabilitate the integrity of aquatic ecosystems and to provide for the long term persistence of native and desirable non-native fishes and other species (Grumbine 1994). Achieving this goal will require the maintenance or rehabilitation of a network of well-connected, high-quality habitats that support a diverse assemblage of native species, the full expression of potential life histories and dispersal mechanisms, and the genetic diversity necessary for long-term persistence and adaptation in a variable environment. Watershed rehabilitation and the development of more ecologically compatible land-use policies are required to ensure the long-term productivity of many systems. Ecosystem management, then, implies managing to reestablish more complete or natural structure, function, and processes whenever possible. Identical goals in terrestrial ecology and the inextricable link between terrestrial and aquatic systems suggest that management efforts in one should benefit the other. The challenge is to coordinate management of terrestrial and watershed systems rather than work at cross purposes.

In this section we examine some major issues involved with ecosystem management and fishes and provide a context for conservation and rehabilitation opportunities. The four issues examined are: 1) catastrophic wildfire and active forest restoration; 2) the role of federal land management in managing anadromous fishes; 3) the effect of roads on sedimentation and fishes; and 4) stronghold watersheds relative to unroaded areas. We then provide a synthesis and discussion of a spatial context for application of the ideas emerging from our assessment, conservation biology, and the ecosystem goals. We consider several scales: the broadscale (subbasins within the context of the entire Basin), the mid-scale (within a subbasin context), and the fine-scale (riparian habitat conservation area widths and functions and cumulative watershed effects).

Major Issues

Catastrophic Wildfire and Active Forest Restoration

A major emerging issue involves the threat of catastrophic wildfire and efforts to actively restore the structure and composition of forests to diminish this threat. Since past timber harvest activities have contributed to degradation in aquatic ecosystems, emphasis on timber harvest and thinning to restore more natural forests and fire regimes represents a risk of extending the problems of the past.

Wildfire has historically been one of the most important and pervasive agents of disturbance on the landscape. The historical pattern of fire frequency and severity within the Basin was a complex mosaic that reflected variation in vegetative cover types and structure, climate and weather, and terrain. Current fire regimes may vary substantially from the historical pattern (Landscape Dynamics, Chapter 3). In forested areas, fire suppression and timber management practices are believed to be the principal causes of disrupted natural fire regimes in many areas. The most prevalent change has been a shift toward more severe fires, as fire suppression has permitted a buildup of fuels. Fire suppression has also increased connectivity of larger stands that are susceptible to insect and pathogens, further contributing to fuel loadings. Past timber management has compounded these problems in some areas by promoting large, even-aged stands, altering forest composition, and favoring higher stand densities. Active timber harvest and thinning are now proposed as primary methods for restoring and recomposing forests (Johnson and others 1995).

It can be argued that the change in fire regime may pose a new threat to the integrity and persistence of aquatic communities and species. Although we do not view wild fire as a particular threat to healthy aquatic ecosystems, depressed and strongly isolated populations could be vulnerable to the effects of intense or very large fires.

Wildfires influence aquatic ecosystems both directly and indirectly. Direct effects include heating or abrupt changes in water chemistry (Minshall and others 1989; McMahon and de Calesta 1990). Indirect effects include changes in hydrologic regime, erosion, debris flows, woody debris loading and riparian cover (Swanson and Lienkaemper 1978; Brown 1989; Megahan 1991; Bozek and Young 1994). Intense fires and related events have killed fish (Bozek and Young 1994) and caused local extinctions (Propst and others 1992; Rinne, 1996). In addition, more extreme fires may have an important influence on the diversity of habitats at watershed and subbasin scales. Since fires burning over large areas are likely to influence more habitats simultaneously, the spatial and temporal diversity in habitat condition and population dynamics we believe important to the stability and persistence of species and populations could be compromised by more extreme events. Such effects might be particularly important where populations and habitats are already degraded. Large and intense fires could threaten some populations that are seriously depressed or isolated.

Given the potential negative consequences of severe wildfires on aquatic communities, pursuing strategies that reduce risks of fire with all haste might seem prudent. But there are other facets to this story. First, historical fires were a natural and important part of the disturbance regime for terrestrial and aquatic systems (Reeves and others 1995). Large fires supplied woody debris and magnified the affect of hydrologic events and debris flows that transported coarse substrates to stream channels. These processes may well have provided the materials that maintained productive habitats for fish and other organisms (Swanson and others 1990; Reeves and others 1995). Second, proposed efforts to reduce fuel loads and stand densities often involve mechanical treatment and the use of prescribed fire. Such activities are not without their own drawbacks — long-term negative effects of timber harvest activities on aquatic ecosystems are well documented (see this chapter; Henjum and others 1994; Meehan 1991; Salo and Cundy 1987). Thus, managers face a dilemma. To do nothing invites potential loss due to uncontrolled wildfires. Reducing fuels involves methods that also may degrade aquatic systems.

We suggest that to resolve this dilemma, one must look at the mechanisms through which fish populations respond to wildfire. In a recent study, Rieman and others (in press) identify two factors that were important in fish population recovery following a large wildfire: 1) refounding of populations through dispersal from local refuges; and 2) refounding through complex life history and overlapping generations.

In studies of recolonization in fish populations following major disturbance and defaunation, internal refuges, that is, sites that protect portions of the population during the disturbance, can promote rapid recolonization of affected areas within a stream. Recovery is often evident in a few years (Meffe and Sheldon 1990; Niemi and others 1990; Yount and Niemi 1990; Lamberti and others 1991; Detenbeck and others 1992; Bayley and Osborne 1993) or even weeks (Sheldon and Meffe 1995). The benefits of spatially redundant and complex habitats to the persistence and resilience of populations are well established in theory (den Boer 1968; Poff and Ward 1990; Sedell and others 1990) and building empirical evidence (Pearsons and others 1992; Rieman and others, in press). The presence of refuges allows populations to compensate for habitat losses, in part through recolonization but also through mechanisms that may be triggered or enhanced by the disturbance event (Bisson and others 1988; Minshall and others 1989). For example, Rieman and others (in press) observed high densities for young-of-the-year redband trout in several reaches affected by intense fires or subsequent channel disruptions precipitated by fire. Although habitats have been lost or degraded through debris flows, increased sediment deposition, and scour following the fires, habitats are also being created through the influx of large debris. The preexisting and resulting complex of habitats seem to allow fishes to persist and perhaps even prosper under these dynamic conditions.

A complex of life-history patterns also provides temporal and spatial hedges against local extinction following catastrophic disruption. Maintenance of that complexity may be critical to the persistence of many populations. As an example, Rieman and others (in press) argue that refounding of a bull trout population in a tributary influenced by the fires discussed above was dependent on the presence of a migratory life history. Gross (1991) and Thorpe (1994) have proposed that the expression of multiple life-history strategies as a mechanism stabilizing populations in variable environments. That expression may also lead to full exploitation of, and be dependent upon, a complex of available habitats (Healey 1994; Lichatowich and Mobrand 1995) and historical patterns of disturbance. The diverse life histories of many native salmonids within the assessment area are discussed in preceding chapters. The existence of complex life histories, such as mixed migratory behaviors and overlapping generations, could be the expression of strategies that have emerged because of the disturbance of fire and associated hydrologic events. Species like bull trout that are associated with cold, high elevation forests have probably persisted in landscapes that were strongly influenced by low frequency, high severity fire regimes. In an evolutionary sense, many native fishes are likely well acquainted with large, stand-replacing fires. If the expression of life-history patterns does reflect the template of historically available habitats (see Healey 1994; Thorpe 1994) and if the spatial and temporal complexity of habitats is lost, the expression of complex life histories may be lost as well.

Both pathways that we have suggested for the short-term recovery of populations influenced by fires reinforce the importance of spatially diverse and complex habitats. Complex landscapes not only produce a mosaic of burn effects, they also create a mosaic of pre-fire stream habitat conditions that provide important refuges within the burn perimeter. That same pattern of stream habitats, the size of the watersheds, and the connection of the watersheds to a larger river basin are likely important in the full expression of life history. Strong, well-distributed populations appear to have a high potential for recovery following intense wildfires. Depressed populations inhabiting marginal or degraded habitat may lack the resiliency to deal with catastrophic disturbance adequately.

The chronic and widespread nature of timber harvest and other human related disturbance has led to a loss of spatial complexity of stream environments that ultimately is reflected in the loss of diversity and distribution of populations and life histories (see this chapter; Frissell and others 1993; Reeves and others 1995). Attempts to minimize the risk of large fires by expanding timber harvest risks expanding the well-established negative effects on aquatic systems as well. The perpetuation or expansion of existing road networks and other activities might well erode the ability of populations to respond to the effects of fire and large storms and other disturbances that we cannot predict or control (National Research Council 1996).

There is growing interest in use of intensive forest management to reestablish more natural landscape patterns and disturbance regimes, but the risks and benefits of that management vary across the landscape. Forest-health treatment projects have been justified from all perspectives including reducing the risk of extirpation for sensitive aquatic species. Undoubtedly, a point exists where the risk of fire outweighs the risk created by management. Management creates risks of somewhat known magnitude, timing, and extent, whereas wildfire potential is less known in each of these respects. The point at which risks are equal needs to be discovered through careful evaluation and scientific study. It appears that the consequences of large fires are dependent on habitat conditions and the inherent resiliency of local populations. Risks of fire are likely most important for aquatic ecosystems that have been seriously degraded and fragmented. Watersheds that support healthy populations may be at greater risk through disruption of watershed processes and degradation of habitats caused by intensive management than through the effects of fire.

The Role of Federal Lands in Managing Anadromous Fishes

Earlier we surveyed a number of historical changes across the Basin that have influenced aquatic ecosystems and contributed to declines in fish populations and ecological integrity. Changes are due to hydroelectric development, flood control, irrigated agriculture, hatcheries, ocean and in-river harvest, and finally, degradation of freshwater spawning and larval and juvenile rearing habitats. The cumulative effects of many of these changes are readily apparent in the distributions and status of naturally reproducing anadromous salmonids. Freshwater habitat degradation is the one most prominently influenced by the Forest Service and BLM, which administer much of the remaining habitat used for spawning and larval and juvenile rearing by anadromous fishes.

Both steelhead and stream-type chinook have most of their few remaining strong populations in subwatersheds on federal land (70% and 88% respectively). More than ninety percent of remaining bull trout and westslope cutthroat trout subwatersheds with known or predicted strong populations are on Forest Service and BLM administered lands (table 4.49). The recovery of depressed populations will depend strongly on management of federal lands as 50 to 76 percent of species with depressed populations occupy federal land. Only ocean-type chinook are less influenced by federal land management since they occupy larger stream systems that are more influenced by private lands, irrigation withdrawals, and dams.

Federal land management has an effect on anadromous salmonids. Numerous published studies describe the negative effects of land-use activities on habitat conditions and link habitat conditions to survival and productivity of anadromous fishes. Meehan (1991) and Murphy (1995), for example, provide excellent comprehensive overviews of this topic. The survival differences between pristine and degraded habitats for egg and juvenile fish can be dramatic. Scully and others (1990) show that egg-to-parr survival for chinook salmon in

Table 4.49— Current population status of seven key salmonids in the Basin and their relationship to habitat provided by Forest Service- and BLM-administered land.

Species	Percent of Historical Range Occupied	Percent of Occupied Range Classed as Strong	Percent of Strongholds in Wilderness	Percent of Strongholds on FS/BLM	Percent of Depressed on FS/BLM	Sensitive to FS/BLM Land Uses
Bull trout	44	13	55	95	82	Yes
Yellowstone cutthroat	66	35	19	70	46	Yes
Westslope cutthroat	85	25	44	94	65	Yes
Redband	64	22	8	56	58	Yes
Steelhead	46	1	9	70	61	Yes
Stream-type chinook	28	<1	50	88	77	Yes
Ocean-type chinook	29	15	0	20	25	minor influence

degraded streams with high sand content was less than one eighth that exhibited in low-sand areas. A large literature documents anthropogenic effects on other life stages (National Research Council 1996). A collection of papers in Schwiebert (1977), for example, documents numerous problems in the Columbia River Basin that still apply 20 years after publication. Our summary, described in earlier sections, supports a scientifically credible view that is emphasized repeatedly in the literature: habitat change due to land use is pervasive and at times dramatic, but impacts are not evenly distributed across the landscape. High-quality areas remain that are capable of supporting anadromous fishes at near-historical levels.

Given the other factors affecting these fishes, the precise magnitude of the loss to anadromous fishes that is due to degradation of spawning and rearing habitats is not known. Similarly, although positive responses to habitat improvement are expected, no precise estimate of the magnitude of the expected benefit is possible. The complexity of interactions across life stages confounds even the best of studies. For example, the debate over the efficacy of juvenile salmon transportation focuses not so much on the survival through the transportation process *per se*, but rather on the latent effect that transportation has on post-release smolt survival and adult returns. Similar questions could be raised regarding early rearing. For example, do juvenile fishes from degraded habitats exhibit the same migration and early marine survival as those from better habitats? Research in British Columbia suggests not (Hartman and Scrivener 1992). In the absence of empirical studies, stock-recruitment models that incorporate habitat conditions suggest that declines in habitat productivity can have a disproportionate effect on total population size. Thurow and Burns (1992) present an example for Idaho streams where a 20 percent loss in habitat productivity results in more than a 50 percent reduction in adult numbers, while a 50 percent reduction in habitat productivity causes extirpation.

The reason the effects of spawning and rearing habitat changes are difficult to measure precisely concerns the compensatory nature of fish-habitat relations. This compensation is also a reason why high-quality habitat is so vital to maintaining and rebuilding populations where they still exist or are strong. When the number of spawning adults declines as it has in recent years, the adults can choose areas offering the best conditions. Similarly, juveniles can spend their time in the better habitats with reduced competition for resources. The net result is that the number of smolts produced per adult can actually increase as the number of spawners declines. As long as the amount and distribution of high-quality habitat available remains proportional to the number of spawners and in locations used by the fishes, the apparent productivity of the population will remain fairly constant. Thus, detecting a historical decline in habitat conditions over a period when numbers of spawning adults are declining may be impossible as well, if one looks only at the number of smolts produced per adult. This is the situation in the Snake River subbasins above Lower Granite Dam. During and after construction of the federal dams in the lower Snake River (post 1970), numbers of returning adult chinook salmon declined dramatically compared to run sizes in the 1950s and 1960s (Petrosky and Schaller 1992). The declining numbers of adult salmon do not permit an adequate test of the hypothesis that habitat conditions changed during the same time period. Such a test would require a return to historical levels of spawning adults that predate the dams.

A study conducted by Lee and Rieman (1996) using the Stochastic Life-Cycle Model (SLCM) examined the effects of habitat quantity, habitat quality, and downstream passage survival on a hypothetical population of chinook salmon. The SLCM was developed by Lee and Hyman (1992) to simulate the life cycle of anadromous salmonids. It is designed to mimic the basic mechanisms regulating populations of Pacific salmon, while capturing some of the intra-annual and interannual variation inherent in these populations. The SLCM was designed for population viability assessments and has been used in recent years by the National Marine Fisheries Service and the Bonneville Power Administration.

The study illustrated the relative effects of simultaneously varying incubation success, parr carrying capacity, and downstream passage survival. Incubation success refers to the proportion of eggs produced that are successfully deposited in the redd and survive to emergence from the gravel. It can be viewed as an indicator of habitat conditions in terms of both spawning and incubation conditions (Bjornn and Reiser 1991). Parr capacity refers to the maximum number of parr or juvenile fish that an area can support. It reflects both habitat quality and quantity, but was used to measure habitat quantity only, assuming quality remained constant. Downstream passage survival refers to the proportion of the smolts that leave natal streams or rearing areas and survive migration to the estuary. Again, passage survival many reflect many things, but was of use to index the effects of changes in the hydroelectric system.

Lee and Rieman (1996) examined eleven levels of incubation success (15% to 65% in 5% increments) in combination with three levels each of parr capacity (50, 100, and 150 thousand) and downstream passage survival (35%, 45%, and 55%). All other parameters, such as fecundity, ocean survival, maturity rates, etc., were held constant at reasonable values for an upriver Columbia Basin stream-type chinook population.

Lee and Rieman (1996) found that the probability of persistence responded to changes in incubation success and passage survival in a way that was consistent across different levels of parr carrying capacity. As passage survival decreased, the level of incubation success required to ensure population persistence increased. Increasing parr capacity had no apparent effect on the relationship. Furthermore, the drop from certain persistence to certain extinction was fairly abrupt. Halving the incubation success (say from 50% to 25%) was sufficient to cause certain extinction of an apparently robust population, regardless of the passage survival or parr carrying capacity.

While all three factors affected population numbers through time, habitat quantity had a measurable effect on mean run size, but only beyond a certain threshold combination of passage survival and incubation success. In practical terms, this suggests that increasing the amount of available habitat (parr capacity) without any increase in quality (incubation success) would have no discernable effect on the chances of the population persisting through time. Alternatively, more habitat of lower quality is less advantageous in terms of population persistence than less habitat of higher quality.

Because of the habitat and population losses associated with dams, only the most productive populations may retain the resilience to persist in the face of natural and human caused disturbance. Any changes in the environment that influence survival and productivity of remaining stocks, including improvements in rearing habitats, harvest, predation, and mainstem passage, will improve chances for persistence in stochastic environments (Emlen 1996; National Research Council 1995). Simply put, with current conditions in migrant survival, many stocks are at serious risk. The differences between those that persist and those that do not will include chance events and the survival and productivity of the stocks as they are largely influenced by freshwater habitats. Without substantial improvement in migrant survival, securing and restoring the quality of freshwater habitats may make the critical difference in persistence for many of the remaining populations. In the short term, conservation and/or rehabilitation of habitats available to or directly associated with remaining populations will be key. In the long term, assuming mainstem conditions are resolved, it will be necessary to conserve and restore broader habitat networks to support the full expression of life histories and species (this chapter; Lichatowich and Mobrand 1995; National Research Council 1995). Rehabilitation of depressed populations cannot rely on habitat improvement alone but requires a concerted effort to address causes of mortality in all life stages. These include freshwater spawning, rearing, juvenile migration, ocean survival, and adult migration.

To prevent extinction of the anadromous fishes in the Snake River subbasins and maintain population resiliency until other causes of mortality are reduced, it is essential that existing high-quality habitats be maintained. To ensure recovery, the amount of high-quality habitat available must remain proportional to the number of returning adults and in appropriate areas, so that there is no net loss in productivity as adult numbers increase. Returning adult numbers can fluctuate over a broad spatial range and from year to year much more rapidly than habitat conditions can improve. Thus, to realize the benefits of improved migration and ocean survival, there must be maintenance of good quality habitats and populations as well as increases in the distribution of high-quality spawning and early rearing habitats. Improved federal land management is crucial to this task.

Effects of Roads on Sedimentation and Fishes

The relationship of roads to intensity of land use and adverse impacts on aquatic habitats has been discussed in several recent studies and publications (Naiman and others 1992; Spence and others 1995; Meehan 1991). The discussion often centers around three themes: 1) the belief that road building practices have improved in the last decade to the point we need not worry about the effects of roads on aquatic systems; 2) the legacy of past road building is so vast and road maintenance budgets so low that the problems will be with us for a long time; and 3) the belief that the correlation of road density to fish habitat and fish population is not strong.

Increases in sedimentation are unavoidable even using the most cautious logging and roading methods. Improvements in road-construction and logging methods, however, can reduce erosion rates and sediment delivery to streams. The amount of sedimentation or hydrologic alteration from roads that streams can tolerate before there is a negative response is not well known, but general effects of sediments on fishes are known. Sediment loads that exceed natural background levels can fill pools, silt spawning gravels, decrease channel stability, modify channel morphology, and reduce survival of emerging salmon fry (Burton and others 1993; Everest and others 1987; MacDonald and others 1991; Meehan 1991; Rhodes and others 1994).

Rice (1992) documented an 80 percent reduction in mass erosion from forest roads and about a 40 percent reduction in mass erosion from logged areas in northern California due to improvements in forest practices beginning in the mid 1970s. Megahan and others (1992) used the BOISED sediment yield production model to evaluate the effects of historical and alternative land management in an Idaho watershed (within the South Fork Salmon River). They reported that present day management practices, properly implemented, have the potential of reducing sediment yield by about 45 to 90 percent compared with yields caused by the historical land use in their study watershed. However, using the improved road design currently practiced by the Boise National Forest, total accelerated sediment yields were still 51 percent over natural sediment yields. These improved road designs plus maximum erosion mitigation led to 24 percent increases over natural yields. Helicopter logging resulted in 3 percent increases over natural yields, and wildfire increased sediment yield about 12 percent over natural levels (Megahan and others 1992).

Megahan and others (1995) evaluated the effects of helicopter logging and prescribed burning on south-facing slopes of headwater drainages in the Idaho batholith, using paired watersheds monitored from 1966 to 1986. Average annual sediment yields showed a statistically significant increase of 97 percent persisting for the 10 years of post-treatment study following logging and burning. Accelerated surface erosion occurred primarily as a result of the prescribed burning but not the helicopter logging, because burning resulted in the majority of bare soil exposure and connection of affected area to streams. Surface erosion rates in the logged and burned areas were about 66 times greater than those on undisturbed slopes. The conclusion is that current Best Management Practices (BMPs) can reduce sediment yields compared with historical practices. But there is a continued risk of increased sedimentation from forest management that will occur particularly if such activities as road building and timber harvest are to take place.

The legacy of past road building within the Basin is enormous. The FEMAT report (1993) noted that federally managed forest lands within the range of the northern spotted owl contain about 180,000 kilometers of roads, a major portion of which constitutes potential threats to riparian and aquatic habitats, mostly through sedimentation. An estimated 250,000 stream crossings (about 1.25 per kilometer) are associated with that road system, and a significant number of culverts are thought to be unable to withstand storms with a recurrence interval greater than 25 years (FEMAT 1993). Our road analysis indicated that over 205,000 kilometers of roads exist on Forest Service and BLM lands in the Basin. We expect that there are a large number of stream crossings, with higher densities of stream crossings in steep highly dissected terrain and lower densities in drier and flatter terrains. We also expect that many of the culverts or stream crossings may not function well in flood events with recurrence intervals greater than 25 years, like their westside counterparts identified in FEMAT. Even with adequate culvert size, lack of maintenance of a road network of this size would lead to significant road drainage problems.

The ability of the Forest Service and BLM to conduct road maintenance has been sharply reduced because funds for maintenance as well as timber purchaser conducted maintenance have declined. This is resulting in progressive degradation of road drainage structures and functions causing erosion rates and potential for erosion to increase (Furniss and others 1991). Most of the problems are with older roads that are located in sensitive terrain and roads that have been essentially abandoned but not adequately configured for longterm drainage. Applying erosion prevention and control treatments to high-risk roads can drastically reduce risks for future habitat damage and can be both effective and cost-effective. In watersheds that contain high quality habitat and have only limited road networks, large amounts of habitat can be secured with small expenditures to apply "storm-proofing" and "decommissioning" activities to roads (Harr and Nichols 1993).

Given the sheer magnitude of the area of federal forests with moderate to high road densities, the job of road maintenance will be expensive. As most road networks have not been inventoried to determine influence on riparian or aquatic resource goals and objectives, there is a need to complete inventories, especially where listed or threatened fish are of concern.

Much of our previous discussion and results point to negative consequences that often accompany roads. The effects associated with roads reach beyond their direct contribution to disruption of hydrologic function and increased sediment delivery to streams. Roads provide access, and the activities that accompany access magnify the negative effects on aquatic systems beyond those due solely to roads themselves. Activities associated with roads include fishing, recreation, timber harvest, livestock grazing, agriculture, and others. Roads also provide avenues for stocking non-native fishes. Unfortunately, we do not have adequate broad-scale information on many of these attendant effects to accurately identify their component contributions (Hicks and others 1991). Thus, we are forced to use roads as a catchall indicator of human disturbance.

In preceding sections we presented results based on two analyses. Each of these analyses supports the general conclusion that increasing road density is correlated with declining aquatic habitat conditions and aquatic integrity. Our results clearly show that increasing road densities and their attendant effects are associated with declines in the status of four non-anadromous salmonid species. They are less likely to use moderate to highly roaded areas for spawning and rearing, and if found are less likely to be at strong population levels. There is a consistent and unmistakable pattern based on empirical analysis of 3,327 combinations of known species' status and subwatershed conditions, limited primarily to forested lands managed by the Forest Service and BLM.

The declines in population status of the non-anadromous salmonids should be viewed as an indication of the types of responses that may be experienced by other native species in similar habitats. Those most like the non-anadromous key salmonids in distribution or habitat requirements would be expected to show the most similar responses. This would include the anadromous species such as steelhead, stream-type chinook salmon, and Pacific lamprey that broadly overlap in range with the non-anadromous salmonids and utilize many of the same habitats for significant portions of their life. There are no logical reasons to expect anadromous fishes to be immune to the effects of habitat change from roads evident in the non-anadromous species. Other species, including sculpins, dace, and some suckers also have considerable overlap in range and may follow similar trends in population abundance and distribution.

Stronghold Watersheds and Unroaded Areas

Most aquatic conservation strategies acknowledge the need to identify the best habitats and most robust populations to use as focal points from which populations can expand, adjacent habitat can be rehabilitated, or the last refugia of a species can be conserved. At issue is whether habitat criteria or population presence and status are better indicators for such special fish emphasis watersheds. Unroaded areas potentially represent areas in which biophysical processes are still operating unimpeded from major human disturbances. Many resource managers believe that management activities in unroaded areas will increase the risk to aquatic and riparian habitat and limit the potential to achieve aquatic conservation strategy objectives. However, not all of the unroaded areas in the Basin are located in areas that are essential to reconnecting habitats and populations.

In this section, we examine several attempts to identify different special emphasis watersheds for fish. These include the Section 7 and High Priority Watersheds in the Snake River Basin, bull trout watersheds as part of the Inland Native Fish Strategy (INFISH) implementation plan, PACFISH watersheds outside of the high priority watersheds, as well as the FEMAT key watersheds in the Northern and Southern Cascades. In addition, we discuss a population status watershed approach that incorporates the use of information on excellent habitat, strong populations, and major unroaded areas.

Special emphasis areas which provide for high quality habitat and stable populations are a cornerstone of most species conservation strategies. Concern for the continued viability of salmonids on federally managed forest lands has led to establishment of the concept of "key watersheds" in which high priority is given to protecting stream habitat (Reeves and Sedell 1992; FEMAT 1993). The goal for these watersheds is to maintain the best of habitats and fish populations, and generally watersheds are chosen that have the highest potential for rehabilitation. For instance, a total of 162 key watersheds was designated that cover 8.7 million acres or approximately one third of the federal land within the range of the northern spotted owl (FEMAT 1993). These refugia were widely distributed across the landscape and were 48 to 70 percent unroaded and/or in wilderness. The designation of the key watersheds was based on rule sets that were simple. These were watersheds that: (1) were larger than 15 square kilometers and had relatively high quality water and fish habitat, or had the potential of providing high quality water and fish habitat with the implementation of rehabilitation efforts; and (2) contained habitat for potentially threatened stocks of anadromous salmonids or other potentially threatened fish species. These watersheds function as freshwater refugia for species or stocks that are currently at low population levels and also as source areas of individuals to recolonize streams that may develop more favorable conditions.

Our assessment identified 1,693 subwatersheds with strong populations of at least one of seven key salmonids within the entire Basin regardless of ownership or management class. They occupied 27 percent of the Forest Service and BLM lands in the entire assessment area. Not unlike FEMAT, our population status review generally found strong populations to be most likely in less disturbed subwatersheds. The spatial context of remaining habitats and local populations may be critical to persistence of these remnant systems. Emphasis on maintenance and rehabilitation of habitat in the Basin is tied to the importance of maintaining biodiversity and stable populations, not just habitat and watershed conditions.

The ecological importance of unroaded areas has been highlighted in this report as well as in other reports (FEMAT 1993; Henjum and others 1995). Unroaded areas have the potential to maintain natural processes unaltered by land management activities and may be important refugia for strongholds of salmonids. We examined the overlap of areas predicted to be unroaded (both within and outside of designated wilderness areas) with stronghold subwatersheds and the other important conservation watershed efforts within the Basin (tables 4.50a-c). We further examined the proportion of strongholds by fish species (seven key salmonids) contained within each of the six efforts identifying special emphasis watersheds (table 4.51).

Designated wilderness and areas predicted to be unroaded are important anchors for strongholds throughout the Basin (map 4.73). Strongholds on Forest Service and BLM lands are 58 percent predicted unroaded. Subatersheds with strongholds in the Central Idaho Mountains and the Snake Headwaters reflect the large amounts of wilderness and National Park System lands, with the largest amounts of predicted unroaded spaces in the Basin. Many predicted unroaded areas in the Lower Clark Fork and Northern Glaciated Mountains are adjacent to isolated and fragmented strongholds.

Table 4.50a—Percent of special emphasis watersheds (identified by various assessments) containing unroaded area on Forest Service and BLM administered lands within the Interior Columbia Basin Ecosystem Management Project EIS areas.

	Eastside EIS Area			Upper Columbia Basin EIS Area		
Special Emphasis Watershed	Percent Unroaded Inside Wilderness	Percent Unroaded Outside Wilderness	Percent Unroaded	Percent Unroaded Inside Wilderness	Percent Unroaded Outside Wilderness	Percent Unroaded
Subwatersheds with Known and Predicted Strongholds [1]	28	13	41	32	37	68
Section 7 Watersheds (Snake River Basin) [2]	32	19	52	33	32	65
High Priority Watersheds (Snake River Basin) [3]	48	11	59	40	39	79
PACFISH Watersheds outside High Priority [4]	27	9	37	33	15	48
INFISH (Bull Trout) Watersheds [5]	15	26	41	32	28	60
FEMAT Key Watersheds [6]	29	11	40	NA	NA	NA

[1] Developed by the Interior Columbia Basin Ecosystem Management Project.

[2] USDC National Marine Fisheries Service. 1994. Critical habitat designation for Snake River sockeye salmon, Snake River spring/summer chinook salmon and Snake River fall chinook salmon on December 28, 1993 (58 FR 68543).

[3] Stelle, William Jr. 1995. August 4 letter to the National Marine Fisheries Service.

[4] Developed as part of the Interior Columbia Basin Ecosystem Management Project.

[5] USDA Forest Service. 1995. Inland Native Fish Strategy. Volume 12. Located in Supervisor's office at the Idaho Panhandle National Forest, Coeurd'Alene, ID.

[6] Forest Ecosystem Management Assessment Team. 1993. Forest ecosystem management: an ecological, economic, and social assessment.

Known and predicted strongholds cover 40 percent of Forest Service administered lands and 4 percent of BLM administered lands. Thus, 27 percent of Forest Service and BLM lands contain the biological building blocks necessary to maintain and rehabilitate fish populations in the Basin. Areas predicted to be unroaded occupy 41 percent of area with known and predicted strongholds in the Eastside EIS area (table 4.50a). One third of this area is outside of wilderness. Sixty eight percent of known and predicted strongholds in the Upper Columbia Basin EIS area is in unroaded condition, of which 37 percent is outside of wilderness. Our assessment at the subwatershed scale, using population presence, absence, and strength (strong and depressed), is the most comprehensive effort to characterize conditions and identify important areas for fishes to date in the Basin. More than eight million hectares of Forest Service and BLM managed land are occupied by stronghold subwatersheds which contain a large area of unroaded land (about 4.7 million hectares or 58 percent).

Table 4.50b— Hectares of unroaded area in special emphasis watersheds identified by various assessments, on Forest Service and BLM administered lands within the Interior Columbia Basin Ecosystem Management Project EIS areas.

	Eastside EIS Area			Upper Columbia Basin EIS Area		
Special Emphasis Watershed	**Unroaded Hectares Inside Wilderness**	**Unroaded Hectares Outside Wilderness**	**Total EEIS Unroaded Hectares**	**Unroaded Hectares Inside Wilderness**	**Unroaded Hectares Outside Wilderness**	**Total UCRBEIS Unroaded Hectares**
Subwatersheds with Known and Predicted Strongholds [1]	820,229	364,113	1,184,342	1,643,658	1,899,967	3,543,626
Section 7 Watersheds (Snake River Basin) [2]	280,710	165,306	446,016	1,494,053	1,441,251	2,935,304
High Priority Watersheds (Snake River Basin) [3]	193,807	44,702	238,508	730,226	726,926	1,457,152
PACFISH Watersheds outside High Priority [4]	224,508	77,403	301,911	96,504	43,702	140,205
INFISH (Bull Trout) Watersheds [5]	32,601	59,202	91,803	1,053,237	921,433	1,974,670
FEMAT Key Watersheds (Eastern Cascades) [6]	280,710	105,404	386,114	NA	NA	NA

[1] Developed by the Interior Columbia Basin Ecosystem Management Project.

[2] USDC National Marine Fisheries Service. 1994. Critical habitat designation for Snake River sockeye salmon, Snake River spring/summer chinook salmon and Snake River fall chinook salmon on December 28, 1993 (58 FR 68543).

[3] Stelle, William Jr. 1995. August 4 letter to the National Marine Fisheries Service.

[4] Developed as part of the Interior Columbia Basin Ecosystem Management Project.

[5] USDA Forest Service. 1995. Inland Native Fish Strategy. Volume 12. Located in Supervisor's office at the Idaho Panhandle National Forest, Coeurd'Alene, ID.

[6] Forest Ecosystem Management Assessment Team. 1993. Forest ecosystem management:: an ecological, economic, and social assessment.

While unroaded areas are significantly more likely to support strong populations, strong populations are not excluded from roaded watersheds. There are several possible reasons for this coexistence: 1) the inherent productivity of some areas allows fish populations to persist despite disturbances linked to roads; 2) real or detectable effects on fish populations may lag behind the initial physical effects in watersheds which have been roaded in the last several years; and/or 3) the scale of the subwatershed (8,000 hectares on average) at which strong populations are identified may mask a potential disconnect between the real locations of strongholds and roads (which are identified at one square kilometer pixels). This issue of scale would be resolved with a mid-scale or subwatershed analysis. The fact that strong salmonid populations can coexist in many roaded areas provides

Table 4.50c— Total hectares of special emphasis watersheds identified by various assessments, on Forest Service and BLM administered lands within the Interior Columbia Basin Ecosystem Management Project EIS areas

Special Emphasis Watershed	Total Eastside EIS Hectares	Total Upper Columbia Basin EIS Hectares
Watersheds with Known and Predicted Strongholds [1]	2,892,803	5,195,385
Section 7 Watersheds (Snake River Basin) [2]	853,531	4,453,258
High Priority Watersheds (Snake River Basin) [3]	402,114	1,837,465
PACFISH Watersheds outside High Priority [4]	824,129	290,210
INFISH (Bull Trout) Watersheds [5]	224,708	3,293,817
FEMAT Key Watersheds [6]	955,934	NA

[1] Developed by the Interior Columbia Basin Ecosystem Management Project.

[2] USDC National Marine Fisheries Service. 1994. Critical habitat designation for Snake River sockeye salmon, Snake River spring/summer chinook salmon and Snake River fall chinook salmon on December 28, 1993 (58 FR 68543).

[3] Stelle, William Jr. 1995. August 4 letter to the National Marine Fisheries Service.

[4] Developed as part of the Interior Columbia Basin Ecosystem Management Project.

[5] USDA Forest Service. 1995. Inland Native Fish Strategy. Volume 12. Located in Supervisor's office at the Idaho Panhandle National Forest, Coeurd'Alene, ID.

[6] Forest Ecosystem Management Assessment Team. 1993. Forest ecosystem management:: an ecological, economic, and social assessment.

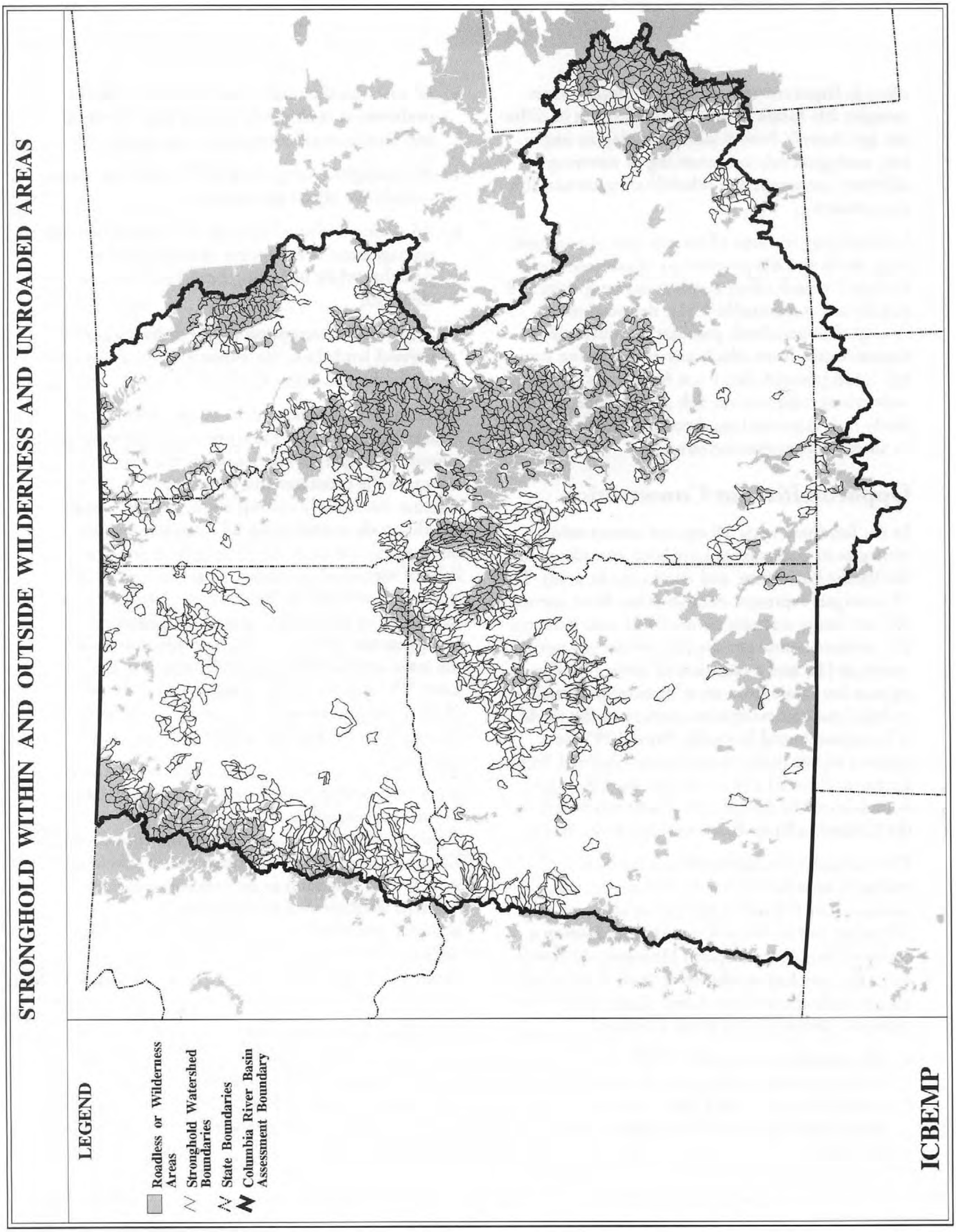
opportunities to determine the reasons why, and may prove instructive for both watershed restoration and future road building. It is not prudent, however, given current information to assume that because roads and fishes coexist in some watersheds they will in others. In general, greater short-term or long-term watershed and ecological risks are associated with entering an unroaded area than with proceeding cautiously with management activities in roaded areas to close and obliterate existing roads. The data strongly suggest a closer examination of the stronghold subwatersheds and their roaded condition.

As an outcome of Section 7 consultation and PACFISH direction, the Forest Service, BLM and National Marine Fisheries Service identified 54 high priority watersheds in the Snake River. These watersheds were intended to identify the best spawning and rearing habitats for the federally listed chinook salmon which currently has few strongholds in the Columbia River Basin, none of which are in the Snake River. These watersheds totaled approximately 2.2 million hectares accessible to anadromous fish. The Eastside EIS area contains 18 percent and the Upper Columbia Basin EIS area contains 82 percent of the total area of these watersheds. Subwatersheds with strongholds occupy 56 percent of the area in high priority watersheds.

Private land in special emphasis watersheds is usually situated in low gradient, unconstrained valley stream reaches which were historically important areas of high habitat complexity for spawning and rearing habitat (for example, approximately 12 percent of high priority watersheds are in private land). This small percentage of area belies the magnitude of its importance within forested special emphasis watersheds. Historically the best quality habitat during drought periods and winter was generally at lower elevations and

Table 4.51— Percent of special emphasis watersheds within the current range of key salmonids.

Special Emphasis Watershed	Status	Bull trout	Westslope cutthroat	Yellowstone cutthroat	Redband	Steelhead	Ocean-type Chinook	Stream-type Chinook
Section 7 Watersheds - Snake R.		58	75	0	60	39	50	60
High Priority Watersheds	Absent	25	40	0	30	24	32	31
PACFISH: outside high priority		5	2	0	10	3	10	5
INFISH (Bull Trout) Watersheds		18	10	0	15	23	1	22
FEMAT Key Watersheds		5	4	0	7	5	17	3
Section 7 Watersheds - Snake R.		53	62	0	66	81	74	83
High Priority Watersheds	Present-	39	33	0	37	47	18	56
PACFISH: outside high priority	Depressed	13	4	0	5	13	0	16
INFISH (Bull Trout) Watersheds		37	38	0	20	18	0	19
FEMAT Key Watersheds		2	1	0	3	4	14	5
Section 7 Watersheds - Snake R.		73	63	0	51	23	0	0
High Priority Watersheds	Present-	70	36	0	41	23	0	22
PACFISH: outside high priority	Strong	10	10	0	17	73	0	100
INFISH (Bull Trout) Watersheds		32	31	0	14	0	0	0
FEMAT Key Watersheds		4	11	0	4	0	30	0
Section 7 Watersheds - Snake R.						83	0	84
High Priority Watersheds	Migration					35	0	34
PACFISH: outside high priority						7	0	5
INFISH (Bull Trout) Watersheds						11	0	11
FEMAT Key Watersheds						0	0	2



Map 4.73—Designated wilderness and predicted unroaded areas overlaid on known and predicted strong populations of seven key fish species in the Basin.

often in larger streams which can support more complex life forms (large and small fishes of different age classes). Private lands can play an important ecological role in sustaining or restoring different ecosystem and rehabilitating certain fish populations.

In examining the area of various special emphasis watersheds and the proportion of subwatersheds included in each effort conditional on species and population status (table 4.51), delineation of stronghold watersheds provides a robust and extensive area from which any conservation strategy could proceed. Since too few watersheds exist with strong anadromous fish populations, watersheds with depressed anadromous fish are essential to anchor a conservation strategy.

Opportunities for Conservation

In the last three years 15 aquatic conservation strategies or assessments have been completed in the Pacific Northwest and Alaska (table 4.52). The 15 strategies represent contributions from agencies (6); university scientist panels (4), forest industry (1), environmental groups (2), private consulting contracts (1), and a coalition of tribes (1). The approaches range from an acknowledged attempt to build the best ecosystem strategy in the world (Clayoquot Sound Scientific Panel 1995) to a riparian inputs maintenance plan (National Forest Resource Council 1995). All but the Clayoquot Sound Scientific Panel (1995) were developed for the Columbia River Basin or parts of the Basin.

That at least 15 independent assessments and strategies exist for the Pacific Northwest is an indication of the difficulties of the agencies and interested parties to work together to develop a common management plan. However, the similarity in the conclusions drawn by each strategy is also an indication of the clarity about some of the problems and solutions being addressed:

- All strategies (except Fish 2000) discuss the concept of connectivity of habitats and metapopulation theory and their importance as integral components of an aquatic conservation strategy.
- All assessments emphasize the need to examine problems at multiple scales and that the social and institutional contexts are important.
- All strategies (except Fish 2000) make the same cumulative effects arguments.
- All strategies discuss moving from single species management to ecosystem management approaches while including provisions for rare elements.
- A common assumption is that federally administrated lands bear the responsibility of sustaining biological diversity.

An element common to each of the studies is an acknowledgment that the current management of aquatic and riparian ecosystems is insufficient to maintain the processes and functions of these systems. Each study attempted to provide a strategy that took conservation of aquatic resources one step further than the then-current state in light of identified problems. In general, all strategies were based on similar concepts, identified the same suite of factors and processes as concerns, and proposed mitigation that included essentially the same suite of planning and protective measures. The differences lay primarily in the specifics of what was recommended, the implementation strategy (if any), and the approach.

There appears to be a consistent assumption that activities will continue to occur within watersheds that are incompatible with maintaining biodiversity and natural processes and functions. Therefore, the common approach is to develop a strategy that mitigates or isolates the negative impacts, whether that is to allocate resources within a watershed (concept of riparian habitat conservation areas) or whether it is to dedicate whole watersheds where the highest management goal is to maintain that area of biodiversity.

With the human population of the Basin growing at about 1.9 percent annually, the challenge is readily apparent. Current and future efforts to save natural populations of both resident and anadromous salmonids, such as conservation measures,

Table 4.52— Assessments on aquatic and riparian systems in the Pacific Northwest conducted during the last five years.

Aquatic and Riparian Assessments
Botkin, D.B., K. Cummins, T. Dunne, H. Regier, M. Sobel, L. Talbot, and L. Simpson. 1995. Status and Future of Salmon of Western Oregon and Northern California: Findings and Options. Research Report 951002, The Center for the Study of the Environment, P.O. Box 6945, Santa Barbara, California 93106.
Clayoquot Sound Scientific Panel. 1995. Sustainable Ecosystem Management in Clayoquot Sound: Planning and Practices.
CRITFC. Columbia River Intertribal Fish Commision 1995. *Wy-Kan-Ush-Mi Wa-Kish-Wit*: Spirit of the Salmon, The Columbia River Anadromous Fish Restoration Plan of the Nez Perce, Umatilla, Warm Springs and Yakama Tribes. Volume I.
Henjum, M. G., J.R. Karr, D.L. Bottom, D.A. Perry, J.C. Bednarz, S.G. Wright, S.A. Beckwitt, and E. Beckwitt. 1994. Interim Protection for Late-Successional Forests, Fisheries, and Watersheds: National Forests East of the Cascade Crest, Oregon and Washington. Bethesda, MD: The Wildlife Society:129-168.
McGreer, D.J. 1995. National Forest Riparian Aquatic Habitat Management Strategy (Fish 2000). National Forest Resource Council. Lewiston, Idaho.
Moyle, P. B. 1995. Water and Life in the Sierra Nevada: Status and Trends of Aquatic Organisms and Habitats. Final Report Sierra Nevada Ecosystem Project. Davis, CA.
Murphy, M. L. 1995. Forestry impacts on freshwater habitats and anadromous salmonids in the Pacific Northwest and Alaska: requirements for protection and restoration. NOAA Coastal Ocean Program. Decision Analysis Series No.7. NOAA Coastal Ocean Office, Silver Spring, MD.
Nelson, C., ed. 1995. Key Elements for Ecological Planning: Management Principles, Recommendations, and Guidelines for Federal Lands East of the Cascade Crest in Oregon and Washington. A report to the Columbia Basin Ecosystem Management Project. May 19, 1995. Columbia River Bioregion Campaign, Science Working Group, 41 South Palouse Street, Walla Walla, Washington 99362.
Pacific Rivers Council. 1994. Management Recommendations for Aquatic Conservation in Eastern Oregon and Washington. Pacific Rivers Council, 921 SW Morrison #531, Portland, Oregon 97205.
Spence, Brian C.; Lomnicky, Gregg A.; Hughes, Robert M.; Novitzki, Richard P. 1995. An Ecosystem Approach to Salmonid Conservation, Volume I: Technical Foundation. Report to the National Marine Fisheries Service, Environmental Protection Agency, and Fish and Wildlife Service. ManTech Environmental Research Services Corp. Corvallis, OR.
USDA Forest Service; USDI Bureau of Land Management 1994. Record of Decision for Amendments to Forest Service and Bureau of Land Management Planning Documents Within the Range of the Northern Spotted Owl.
USDA Forest Service; USDI Bureau of Land Management. 1995. Decision Notice/Decision Record, FONSI, EA, Appendices for the Interim Strategies for Managing Anadromous Fish-Producing Watersheds in Eastern Oregon and Washington, Idaho, and Portions of California (PACFISH), USDA Forest Service and USDI Bureau of Land Management.
USDA Forest Service; US Fish and Wildlife Service. 1995. Environmental Assessment Inland Native Fish Strategy.
USDA Forest Service Regions 1, 4, and 6. 1991. Columbia River Basin Anadromous Fish Habitat Management Policy and Implementation Guide (PIG).
USDC National Marine Fisheries Service. 1995. Proposed Recovery Plan for Snake River Salmon.

improved land use practices, reduced competition from hatchery stocks, improved dam passage, and better riparian protection on all lands, could be undermined by continued regional population and economic growth.

A Broad-scale Context

An assumed goal of ecosystem management is to maintain or restore the integrity of aquatic ecosystems and to provide for the long term persistence of native and desirable non-native fishes and other species. Achieving this goal will require the maintenance or restoration of a network of well-connected, high-quality habitats that support a diverse assemblage of native species, the full expression of potential life histories and dispersal mechanisms, and the genetic diversity necessary for long-term persistence and adaptation in a variable environment. The concept of key watersheds has been used to identify areas that represent critical components of the mosaic that need to be conserved in the short term. In many cases, however, focus on a fixed set of high-quality watersheds will not meet the goals for healthy, functional aquatic ecosystems because they are too few and poorly distributed, and because natural succession and disturbance processes may preempt long-term productivity in fixed sites. Watershed rehabilitation and the development of more ecologically compatible land-use policies are required to ensure the long-term productivity of many systems. The next step then, implies more than a system of watershed reserves; it implies managing watersheds to reestablish more complete or natural structure, function, and processes. Again, the challenge is to work with the management of terrestrial systems rather than at cross purposes.

To assist in developing a more interdisciplinary dialogue, we developed a simple classification of subbasins throughout the Basin. The classification scheme provides a spatially explicit description of aquatic issues, needs, and opportunities that can be associated with similar descriptions for terrestrial ecosystems. It is not intended to be rigid. Rather, the classification is based on the integration of current data with local, qualitative knowledge of watershed connectivity and other conditions. This classification scheme may be useful as a tool for identifying the opportunities and conflicts that are likely to emerge from the multiple priorities and objectives inherent in ecosystem management.

Inherent in our classification is a template for prioritizing needs and opportunities for conservation and active rehabilitation. Several strategies have been proposed for the development of habitat networks designed to conserve species and aquatic biological diversity (Doppelt and others 1993; Frissell and others 1993; Moyle and Sato 1991; Reeves and Sedell 1992; Rieman and McIntyre 1993). A general consensus of these reports is that conservation and rehabilitation should focus first on the best remaining examples of aquatic biological integrity and diversity. Clearly, effectiveness of any conservation strategy improves with more detailed information regarding the distribution and composition and condition of aquatic communities. Earlier in this report we began the identification of unique or particularly important watersheds and basins. The data are most useful for considering patterns and important areas at the scale of the entire assessment area. These data provide a starting point for finer scale evaluations (such as within subbasins), but local information must be used to validate and extend these results. It will be particularly important to include information for aquatic organisms (that is, plants, invertebrates, amphibians) that could not be addressed in our analysis. We emphasize that this classification scheme is preliminary and should not be used for prescription of management activities without further refinement.

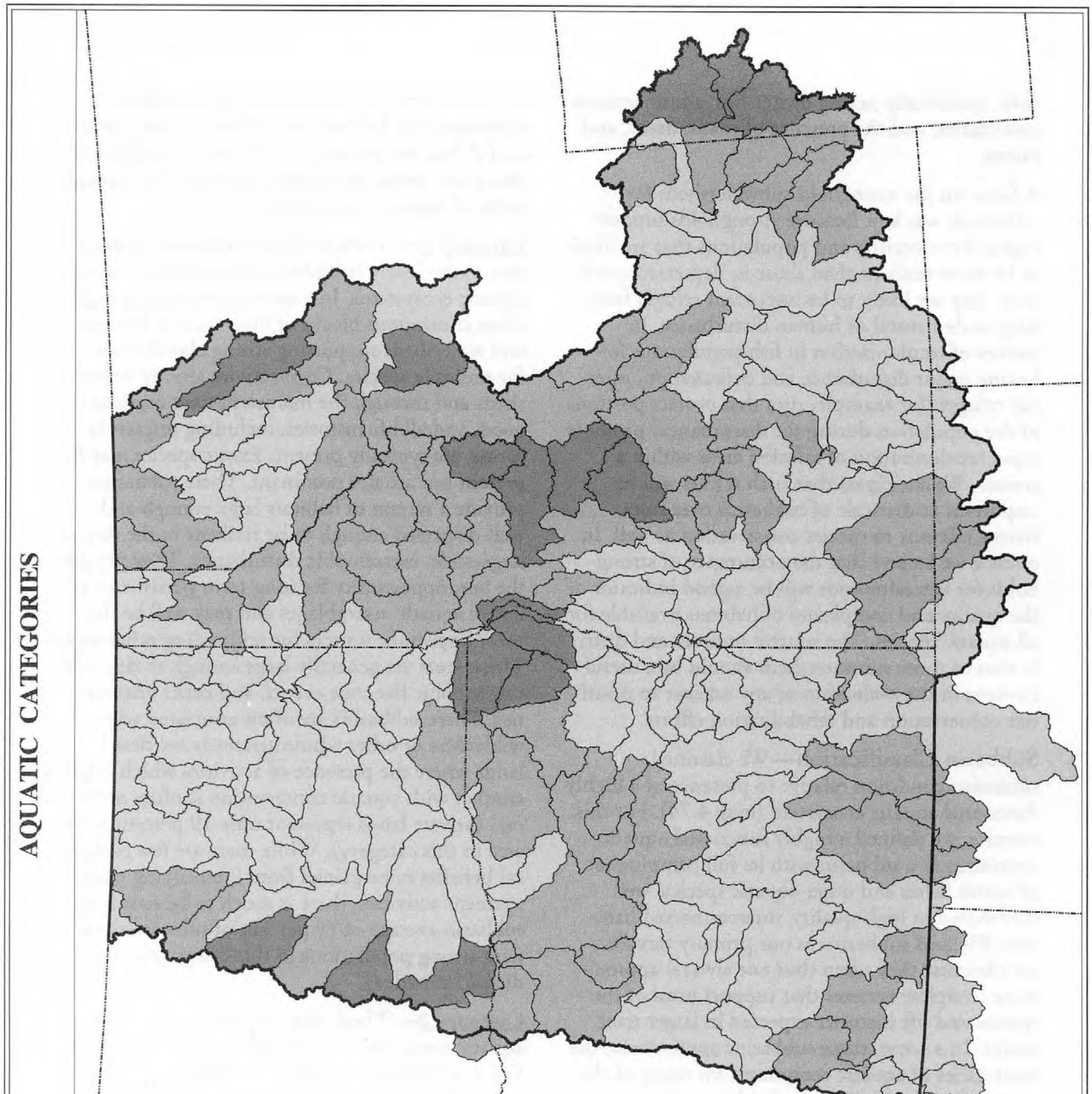
Most of the information used in our subbasin classification scheme is presented in preceding sections of this report. In particular, we relied heavily on the distributional maps of salmonid strongholds and our derived measures of community structure. We also looked closely at human influences when considering opportunities and risks, specifically noting ownership, administrative boundaries, and the presence of roads, dams, and mines.

A focus on the stronghold subwatersheds for salmonids was key. Because strongholds support higher fish densities and populations that are likely to be more resilient than those in depressed condition, they are likely to be important refugia from large-scale natural or human disturbance. In studies of recolonization in fish populations following major disturbance and defaunation, internal refuges (for example, sites that protect portions of the population during the disturbance) promote rapid recolonization of affected areas within a stream. We anticipate that such effects will be important at the scale of subbasins over time frames relevant to species conservation as well. In essence we believe that the occurrence of strongholds for key salmonids will be a good indicator of the quality and complexity of habitats available for all aquatic species. The relative number and distribution of those subwatersheds should be a useful element in the evaluation of any scheme to prioritize conservation and rehabilitation efforts.

Subbasin Classification—We classified subbasins condition relative to presence of a highly functional aquatic ecosystem (map 4.74). For this exercise, we defined a highly functional aquatic ecosystem as a subbasin with its full compliment of native fishes and other aquatic species, well distributed in high-quality, interconnected habitats. We used subbasins as our primary sample unit because they often (but not always) approximate complete systems that support most of the species and life histories expected in larger river basins. In a sense, these subbasins approximate the boundaries of aquatic ecosystems for many of the species found within them. Subbasins that support the full expression of life histories and possess productive and well-connected populations should be relatively resilient to natural disturbances anticipated over biologically important time scales. Persistence over evolutionarily important time scales may imply connection across even larger systems, and obviously anadromous species require a connection to the ocean through multiple subbasins. We believe that subbasins represent a useful, but not exclusive boundary for analysis of issues and needs and opportunities in the management of aquatic ecosystems.

Category 1— These subbasins represent systems that most closely resemble natural, fully functional aquatic ecosystems. In general they support large, often continuous blocks of high-quality habitat and watersheds supporting strong classifications for multiple species. Connectivity among watersheds and through the mainstem river corridor is good, and all life histories, including migratory forms, are typically present. Exotic species may be present but are not dominant. These subbasins provide a system of habitats large enough and well-dispersed enough to be resilient in the face of large-scale, catastrophic disturbance. They provide the best opportunity for long-term persistence of native aquatic assemblages and may well be the most important sources for refounding other areas. These areas are generally large enough to deal with catastrophic fire, rare events, and other uncertainties. These subbasins are often associated with wilderness or other administratively restricted lands where the presence of activities which might conflict with aquatic conservation is often minimal (private lands represent only 19 percent of the area in this category). While there are few ecological benefits to be gained from intensifying management activities, there is much to be risked (a subbasin average of 79 percent of subwatersheds with strong populations in this category is predicted unroaded).

Category 2— These subbasins support important aquatic resources, often with subwatersheds classified as strongholds for one or multiple species scattered throughout. The most important difference between Category 1 and Category 2 is an increased level of fragmentation that has resulted from habitat disruption or loss. These subbasins clearly have a substantial number of watersheds where native species have been lost or are at risk for a variety of reasons. Connectivity among watersheds may still exist or could be restored



Map 4.74- Classification of subbasins. Category 1 reflects the most robust populations and habitats; Category 2 reflects fewer strongholds and greater isolation within a subbasin, but with a potential to restore an interconnected network of adjacent strongholds; and Category 3 reflects few, perhaps no strongholds which have little potential for reconnection (through rehabilitation) with other strongholds to create a larger network of stronghold watersheds.

through the mainstem river system, such that maintenance or rehabilitation of life-history patterns and dispersal among watersheds is possible. Reestablishing the necessary mosaic of habitats will often require conservation of existing high quality sites as well as the rehabilitation of whole watersheds that still support remnant populations. Private lands represent 32 percent of area in this category. These subbasins often fall in some of the more intensively managed landscapes and have some of the most extensive road networks. They also have the greatest need and opportunity for restoration of structure and composition of vegetation communities.

In many cases there may be an opportunity to accomplish watershed rehabilitation while conducting forest treatments. For example, where extensive road networks already exist, treatments might be focused over a relatively short period allowing road removal upon completion (a subbasin average of 48 percent of strongholds in this category is already predicted unroaded). Because watersheds requiring very conservative approaches to protect key resources are often scattered rather than contiguous, intensive forest management might be prioritized and focused in the area around them, thereby minimizing risks. The opportunities to explore and experiment with watershed restoration through active manipulation or through attempts to produce more episodic disturbance followed by long periods of recovery (Reeves and others 1995) are most likely to exist in these subbasins. Conceivably, these subbasins offer the greatest opportunity for positive solutions across multiple resource issues.

Category 3— These subbasins may have some subwatersheds supporting key salmonids classified as strong or have other important aquatic values (such as threatened and endangered species, narrow endemics, introduced or hatchery supported sport fisheries). In general, however, these watersheds are strongly fragmented by extensive habitat loss or disruption throughout the component watersheds and most notably through disruption of the mainstem corridor. Major portions of these subbasins (43 percent of the area in this category) are often associated with private and agricultural lands not managed by the Forest Service or BLM. Although important and unique aquatic resources exist, they are most often localized. The opportunity for restoring connectivity among watersheds, the full expression of life histories, or other large-scale characteristics of fully functioning and resilient aquatic ecosystems are very limited or nonexistent. Because the remaining aquatic resources are often strongly isolated, the risks of local extirpation may be high. While conservation of the remaining strongholds or other aquatic resources is important, land use activities can occur in the rest of the subbasin with less risk to critical resources (a subbasin average of 26 percent of strongholds in this category is predicted unroaded). Because these subbasins are often associated with large areas of non-Federal land, conservation of the remaining productive areas may result in a disproportionate contribution from Federal land management agencies.

The preceding classification scheme considers both the need and opportunity for conservation and rehabilitation of more functional aquatic ecosystems at the subbasin scale. The approach is simplistic and based entirely on information regarding the status of fishes. Other important considerations that might refine judgments regarding those needs and opportunities include the risks to aquatic ecosystems from mechanical disturbance that varies with geology, climate, and topography.

The concept that risk increases in mechanically disturbed environments is supported by the findings that habitat and population conditions are inversely related to the amount of roads present. Such effects may include a general simplification and degradation of habitats but also an increase in the frequency and magnitude of disturbance events. Those effects may be more important in some landscapes than others. For example, Columbia River Basalt geology is inherently more stable and less erodible than Idaho Batholith geology (Andre and Anderson, 1961; Wallis and Willen 1963), and therefore less risk to aquatic ecosystems from mechanical disturbance can be inferred in the basalt geology. Sensitivity to disturbance may influence both the urgency and necessity of conservation and rehabilitation activities. Our classification scheme was intended only to convey a logic in determining management opportunities. The application and refinement of that scheme could include important physiographic elements as well as a finer resolution.

Application at the Mid Scale

At the mid-scale level, the focus is on single subbasins or a small group of adjacent subbasins. Many of the important issues at the broad scale are equally important at the mid scale, and vice versa. One such issue involves metapopulation dynamics. Metapopulations are a collection of populations or demes, each vulnerable to local extirpation but potentially supported or recolonized through dispersal from surrounding populations (Hanski 1991). Metapopulation ideas seem relevant for fishes, especially the salmonids (Bisson and others 1996; Rieman and McIntyre 1993, 1995; Rieman and others 1993) that show local population structuring through strong homing tendencies and the potential for dispersal through straying (Quinn and others 1991). Increased spatial structuring and associated dynamics may be particularly important for fishes persisting in increasingly fragmented environments. Metapopulation dynamics are still poorly understood, however, and there is little empirical evidence to guide conservation or active management of such processes or structures. The idea of creating or supporting spatial and temporal patterns of disturbance consistent with these larger scale dynamics is conceptually and theoretically appealing (Reeves and others 1995; Bisson and others, 1996). We still lack the understanding, however, to support broadscale and intensive manipulation of systems with that intent.

We know little about the spatial and temporal dynamics of species at scales larger than stream reaches, yet we clearly recognize processes acting among reaches, watersheds, and even river basins strongly influence the dynamics, persistence, and diversity of populations and species (Rieman and McIntyre 1993, 1995, 1996; Bisson 1995; Frissell and Bayles 1996). Conservation of genetic diversity clearly implies conservation of populations and habitats across their full range of distribution (the entire Basin area for some species) (Leary and others 1993; Lesica and Allendorf 1995). Conservation of species, life-history, and phenotypic diversity may imply conservation of populations and habitats across entire subbasins or basins (Healey 1994; Lichatowich and Mobrand 1995). For example, earlier we showed some overlap in the distribution of salmonids within 6th-field subwatersheds, but few of those subwatersheds support strong populations of more than two species while subbasins may support as many as six. Rieman and McIntyre (1995) demonstrated that the consistent occurrence of local populations of bull trout is associated with suitable watersheds larger than 5,000 hectares. Rieman and McIntyre (1996) found that local populations of bull trout distributed among a collection of similar sized watersheds throughout a much larger subbasin may fluctuate independently even though they share common migratory and rearing environments. Rieman and others (in press) argue that the persistence of bull trout in Rattlesnake Creek, an 8,000 hectare watershed, following a catastrophic fire was dependent upon a migratory life-history that required access to the much larger Boise River Basin (300,000 hectares). The full diversity of life-history forms for westslope cutthroat trout or chinook salmon are associated with areas comparable to or larger than whole subbasins (Rieman and Apperson 1979; Lichatowich and Mobrand 1995).

The implication, is that effective conservation of aquatic diversity and resilience will require the maintenance of complex habitats and networks of those habitats at multiple scales. That is not likely to happen if management activities continue to result in a chance pattern of habitats like that left by historical management practices. In many cases, the best remaining habitats are clumped in headwaters, with the most degraded conditions found at lower elevations and often on private lands. In other cases, productive areas may extend throughout a subbasin but are represented by scattered watersheds in a matrix of more severely degraded conditions.

Securing existing habitats and subwatersheds that support the strongest populations and highest native diversity and integrity is a high priority in applying any conservation principles at the subbasin scale. Within areas emphasized for aquatic systems, addressing and fixing existing threats to watershed processes without adding new ones is a next step. A third step could be the extension of favorable conditions into adjacent watersheds creating a more secure, larger, or more contiguous network of suitable or productive habitats. The fourth step could be the extension of good habitats into more poorly represented parts of a subbasin. The existing distribution of species and life histories, and their potential to colonize or support newly available or rehabilitated environments, will provide the framework for the selection or prioritization of watersheds for conservation and rehabilitation.

Further research is necessary to develop management techniques that mimic natural disturbance regimes and the temporal and spatial dynamic relevant to the evolution of aquatic ecosystems (see Reeves and others 1995). Meanwhile, however, conservation and rehabilitation of a broader network of complex habitats will preserve options for the future.

Example Within a Subbasin—As a way of better illustrating the above concepts, we provide an example of how this might be done at the subbasin level. It is our intent to provide an illustration of the logic that could be applied, not to provide a specific guide for any subbasin or subwatershed.

The hypothetical subbasin for our example is a member of our aquatic Category 2. We selected a Category 2 subbasin because we expect the most variability in management priorities as well as the highest need and opportunity for active rehabilitation in this category. The subbasin supports wild native populations of westslope cutthroat, bull, and redband trout, steelhead, and stream-type chinook salmon, several other native species, and introduced populations of brook trout and Yellowstone cutthroat trout. Two lakes that historically supported sockeye salmon retain kokanee that may be remnant, residualized sockeye. Human activities including mining, timber harvest, and roading have substantially altered watershed characteristics and the condition of fish populations. Several subwatersheds still support relatively strong local populations of the salmonids, and mainstem rivers still function as wintering areas and migratory corridors for each of these species. The subbasin still supports a core of important habitats for native species and retains important components of the connectivity and life histories found in the historical system (fig. 4.25a-c).

We indexed the location of the best remaining habitats by describing the known strong subwatersheds for the native salmonids and by describing the distribution of the rare/sensitive species. Three non-anadromous species were strong in some subwatersheds. Anadromous salmonids are not considered strong in any of the subwatersheds. Wild, indigenous stocks of both chinook salmon and steelhead spawn in the subbasin, however. We considered wild anadromous salmonid production areas to be of equal status to subwatersheds supporting strong populations of the other salmonids. Hatchery-influenced chinook salmon (progeny of hatchery reared or mixed wild and hatchery parentage that naturally spawn) also spawn in the subbasin; however, we did not consider their spawning and rearing areas to be of equal concern to wild populations because they are supported, in part, by artificial production.

Subwatersheds that supported two or more species strongholds, spawning areas for wild salmon and steelhead, or kokanee were categorized as "Type 1". Type 1 subwatersheds were clustered in three areas of the subbasin. These subwatersheds currently support much of the biological diversity in the subbasin (fig. 4.25d).

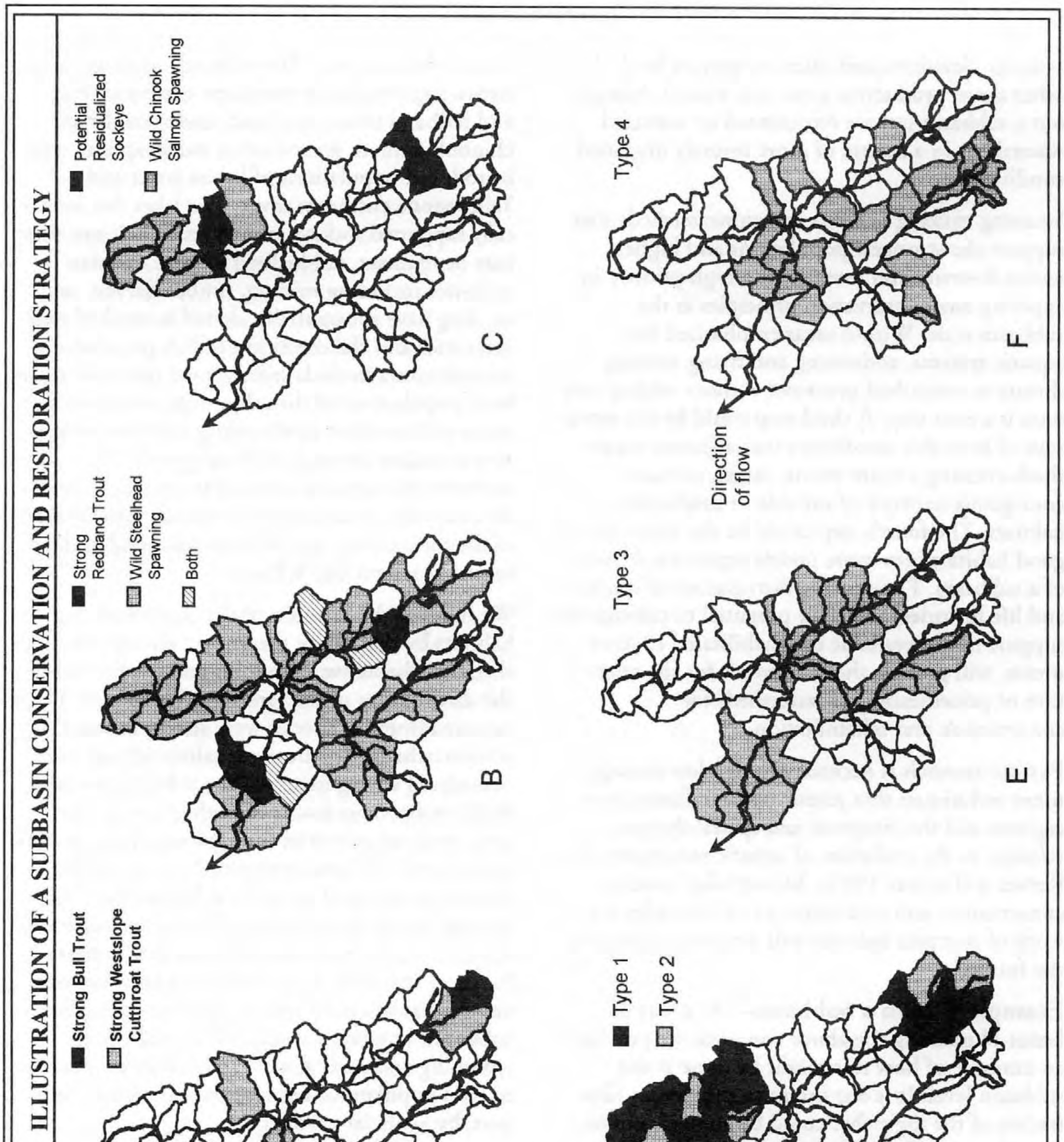


Figure 4.25—Illustration of a subbasin approach to conservation and rehabilitation of native fishes.

We considered the second type of subwatershed, Type 2, as those that supported important habitats either adjacent to or likely to influence the Type 1 subwatersheds. In this case we selected subwatersheds that created larger blocks of habitat contiguous with the Type 1 subwatersheds (fig. 4.25d).

Our third type of subwatershed, Type 3, were those that: 1) contained at least one strong population of a key salmonid, a spawning area for wild salmon or steelhead, or a rare/sensitive fish species; or 2) supported spawning and rearing habitat for two or more native key salmonids; and 3) were not closely associated with other currently productive areas (fig. 4.25e). Excepting redband trout, none of our Type 3 subwatersheds supported strong populations. Several supported populations of bull trout in headwater reaches of the subwatershed suggest that core areas for rebuilding bull trout populations are present. Most Type 3 subwatersheds supported spawning and rearing habitats for wild steelhead, natural stream-type chinook salmon, bull trout, and westslope cutthroat trout. Most Type 3 subwatersheds remained connected to larger portions of the subbasin, suggesting that rehabilitation of both fluvial and resident trouts might be feasible.

The remaining subwatersheds Type 4 (fig. 4.25f) were those that currently support no known strong populations, no spawning areas for wild anadromous salmonids, no rare/sensitive fish species, and do not directly influence subwatersheds that do. Although these areas are currently judged to support no priority populations, many retain multiple native species. Type 4 subwatersheds may be less critical to short and intermediate term persistence of key species and are less likely to contribute to the larger system than those considered above.

Potential Opportunities—Effective conservation and watershed rehabilitation efforts coupled with progress in improving salmon and steelhead migration corridors could restore a more complete and functional aquatic ecosystem. In Type 1 subwatersheds actions to secure and rehabilitate the watershed and riparian processes that maintain and create habitats in these systems would be especially important to the conservation of biological diversity and function in the subbasin. Because many of these subwatersheds may already be in relatively good condition, significant benefits may be realized with relatively small investments. Focusing conservation and rehabilitation activities in Type 2 subwatersheds could build larger networks of important habitats, add to the spatial extent and potential diversity of habitats, and influence the condition of important mainstem spawning and wintering habitats further downstream. Type 3 subwatersheds may represent important habitats in the future, and would benefit from focused management activities that allowed long-term recovery of habitat conditions. Long-term watershed rehabilitation would be slower in Type 4 subwatersheds, and initial investments might be less effective than in other areas. Type 4 systems could be good places on which to focus experimental and active management that has unproven or uncertain risks for aquatic systems.

Areas where fish information is lacking may require special treatment. An appropriate approach in areas where status of important native fishes is unknown might be to emphasize collection of adequate data prior to adding any additional risk from management actions.

The subwatershed types outlined above are presented simply to illustrate the application of conservation biology principles that might be used to develop fish/aquatic needs and opportunities to assist in the first stage of planning for ecosystem management at a subbasin scale. The classification of subwatersheds might be modified by information on distribution of other species or specific issues for the species we did consider. For example, many subbasins may not currently support strong salmonid populations, wild anadromous fish, or rare/sensitive species. In those areas, the logic for classifying subwatersheds might include those areas retaining the largest numbers of native species.

Fish conservation priorities might also be modified by risks and opportunities associated with the legacy of, and sensitivity to, natural and mechanical disturbance. Geologic and climatic setting, topographic features, existing road networks, watershed condition, anticipated risks of existing or planned activities, and conflict or convergence with other resource management goals will provide finer resolution. Watershed and habitat conditions will strongly influence conservation and rehabilitation potential and will make some systems better candidates for rehabilitation activities than others. In our analysis of subwatershed types for example, we did not incorporate trends in habitat condition, but only the existing condition as inferred from the current known distribution and status of fishes. Had we considered trends, some areas that were classified based on fish status might be elevated to higher priorities for rehabilitation if particularly important risks existed. Similarly, areas judged to be of lower emphasis for conservation and rehabilitation based solely on fish status, may become higher priorities for rehabilitation when integrated with other resources (terrestrial species concerns for example) especially if habitats are already trending toward restoration. Areas trending toward less capability to produce fish or aquatic diversity might be either too severely altered to feasibly restore or very high priority for rehabilitation, depending upon the degree of fragmentation or isolation in populations, Endangered Species Act concerns, relative risks involved, and sensitivity of the species involved.

The location and condition of a subwatershed may also be critical for assessing its importance to the overall subbasin. For example, if a lower emphasis area is severely degraded and it lies directly upstream from and influences a critical spawning and rearing area for several species, managers may choose to emphasize its rehabilitation because it will directly improve conditions in the critical habitat downstream.

Implementation of watershed conservation and rehabilitation activities will obviously be constrained by management commitment and resources. The basin and subbasin scale scheme presented here can assist planning and prioritization of activities at larger scales. We do not consider all subbasins described as "Category 2" to be the same; their inherent differences should play an important role in determining the urgency and nature of conservation and rehabilitation activities that occur within the subwatersheds. The current condition of a subbasin and the inherent environmental variability and sensitivity of the composite subwatersheds to disturbance may dictate the urgency of management actions. For example, watersheds in the Central Idaho batholith that lie in erodible soils and are subject to high intensity thunderstorms, or those in the belt geologies of northern Idaho that are subject to frequent rain on snow events represent highly variable and sensitive environments for aquatic organisms. Fragmentation and disruption of habitats in these systems could pose much greater risks for sensitive species than in more stable environments. Such risks might imply a greater urgency to secure critical habitats and to rehabilitate habitat networks and watershed processes in some subbasins than others.

Application at a Fine-Scale: Riparian Areas and Cumulative Watershed Effects

Aquatic and riparian systems are easily affected by land management activities surrounding them. In general, there is little controversy over the need to provide buffers to maintain ecological function. The controversy is over the width of the buffers, the extent and type of activities which can occur within them, and the purposes for these activities. Riparian Habitat Conservation Areas (RHCA) are portions of watersheds where riparian dependent resources receive primary emphasis and management activities are subject to specific standards and guidelines (PACFISH 1994). Forest plans and forest practice rules regulate two major features of RHCA: their width and the kind and amount of activity that can take place within them. Evaluating the effectiveness of RHCAs to protect and manage riparian areas is difficult because of the complexity of the ecological function of such areas and the extended time over which impacts can occur and ecosystems might need to recover. The

RHCA must be wide enough to maintain ecological function at the small watershed level and limit disturbance near streams. At issue is the size of RHCAs and the function they provide throughout a stream network to reduce sedimentation and limit potential losses of aquatic biodiversity and habitat. We also address cumulative watershed effects (CWE) models which purport to account for the total amount of CWE quantitatively and limit CWE that may accumulate as a result of management activities or natural events such as wildfire and floods.

Riparian Area Management—Four biophysical principles underlie any evaluation of a riparian management strategy: 1) a stream requires predictable and near-natural energy and nutrient inputs; 2) many plant and animal communities rely on streamside forests and vegetation; 3) small streams are generally more affected by hillslope activities than are larger streams; and 4) as adjacent slopes become steeper, the likelihood of disturbance resulting in discernable in-stream effects increases.

<u>Importance of Energy Inputs to Streams</u>—First, stream and riparian organisms need energy (leaves, wood, organic carbon) and nutritional inputs to sustain their biological functions. An understanding of the influence of riparian vegetation on streams is fundamental to understanding the function and effectiveness of RHCAs. Streams are intricately connected physically, chemically, and biologically to their riparian zones (Murphy and Meehan 1991; Naiman and others 1992; Gregory and others 1991). Roots of streamside vegetation stabilize banks, retard erosion, and affect nutrients in groundwater. Root systems, in combination with large woody debris, provide channel roughness elements that not only promote sediment storage but encourage the hydraulic exchange of streamflow and subsurface flows. Vegetation and downed woody debris dissipate stream energy during floods and obstruct movement of sediment and organic matter (Sedell and Bestcha 1991). The combination creates very complex habitats for aquatic organisms. The canopy provides leaves and other organic materials that are part of the energy base for the stream ecosystem, and its shade limits algal production and moderates stream temperature. Trees that fall into the stream provide the principal structural features that shape the stream's morphology, linkages to the floodplain, habitat complexity, streambed materials, and other characteristics (Salo and Cundy 1987; Meehan 1991; Naiman 1992).

<u>Protection for Riparian Dependent Plants and Animals</u>—Second, some terrestrial and semi-aquatic plant and animal communities rely on the forest and shrubs adjacent to streams (Terrestrial Ecology, Chapter 5). Animals such as beavers, otters, dippers, and some amphibians are obligate stream and riparian vegetation dependent organisms. Other bird and mammal species and many bat species need the riparian management area at crucial life history periods or seasonally for feeding or breeding. Wildlife has a disproportionally high use of riparian areas and streamside forests compared with the overall landscape. RHCAs provide habitat needs such as water; cover; food; plant community structure, composition, and diversity; increased humidity; high edge-to-area ratios; and migration routes (Carlson 1991; O'Connell and others 1993). The Washington Department of Wildlife (1992) recommended wetland buffer widths for protection of wildlife species in the state. Roderick and Milner (1991) also prescribe wildlife protection buffer requirements for wetlands and riparian habitats in Washington. These widths vary from 30 to 183 meters depending on species and habitat usage (FEMAT 1993). The variable widths of riparian areas suggest a one-size-fits-all approach will not accommodate all organisms. Hence the community ecology functions of RHCAs will need to be determined both at the site and throughout a subbasin if the organism is wide ranging.

<u>Importance of Small Streams</u>—Third, small streams are more affected by hill slope activities than are larger streams because there are more smaller than larger streams within watersheds, smaller channels respond more quickly to changes in hydrologic and sediment regimes, and stream-

side vegetation is a more dominant factor in terms of woody debris inputs and leaf litter and shading. Small perennial and intermittent non-fish bearing streams are especially important in routing water, sediment, and nutrients to downstream fish habitats (Reid and Ziemer 1994). Intermittent streams account for more than one-half the total channel length in many watersheds in the Basin and therefore strongly influence the input of materials to the rest of the channel system.

Channelized flow from intermittent and small streams into fish bearing streams is a primary source of sediment in mountainous regions (Belt and others 1992). In steep, highly dissected areas, intermittent streams can move large amounts of sediment hundreds of meters, though buffer strips, and into fish bearing streams. In-channel sediment flows are limited primarily by the amount and frequency of flow and by the storage capacity of the channel. Flows in forested, intermittent streams are generally insufficient to move the average sized wood piece, allowing large wood to accumulate in small channels (Bisson and others 1987). These accumulations increase the channel storage capacity and reduce the likelihood of normal flows moving sediment downstream.

Live vegetation plays an important role in stabilizing granitic colluvium that accumulates in small headwater basins of the Idaho batholith. Typically, these draws or hollows show little evidence of surface flow and contain deep (several meters), unconsolidated granitic colluvium. Periodically these sites are rejuvenated by floods that flush some or most of the material until another period of relative stability results in accumulation of colluvium and filling (Gray and Megahan 1981; Megahan and others 1995).

Gray (1970, 1978) identified four mechanisms by which vegetation enhances soil stability including: 1) mechanical reinforcement by roots; 2) regulation of soil moisture content; 3) buttressing between trunks or stems of plants; and 4) surcharge from the weight of trees. Gray and Megahan (1981) evaluated these hydromechanical effects in the batholith and found that the first three are highly important in stabilizing slopes, hollows, and intermittent streams. Gray and Megahan (1981) recommended using buffer zones along the margins of streams and in critical areas such as hollows and intermittent streams.

The direct influence of riparian vegetation on stream and animal and plant community declines with increasing distance from the channel and with the height of the dominant tree species (FEMAT 1993). Ecological functions provided by riparian vegetation are achieved at different distances, depending on the type of function and the width of riparian vegetation needed for the function. The maximum height of dominant trees influences the potential distance over which riparian vegetation directly affects stream channels. For instance, tall trees potentially contribute shade, particulate organic matter, and large woody debris at greater distances from streams than do short trees. Areas capable of producing large tall trees thus possess wider functional riparian zones than areas in which trees do not grow as large. For this reason, FEMAT (1993), PACFISH (1995), and INFISH (1995) used the height of dominant late-successional tree species that would naturally grow in a particular riparian zone as the basis for reconnecting streamside buffers needed to safeguard ecological functions instead of suggesting a fixed "onesize-fits-all" linear distance. Use of a fixed distance from the streambank to the outer margin of the buffer strip would not allow for differences in potential tree growth between regions.

PACFISH (1995) prescribed 90 meter minimum RHCA widths for fish bearing streams to maintain stream function from sediment inputs from non-channelized sources. A review of the literature indicates that this should also be sufficient to provide for other riparian functions with a margin for error (Gregory and others 1987, Beschta and others 1987, Brazier and Brown 1973, Steinblums and others 1984, McDade and others 1990, Sedell and Beschta 1991, Belt and others 1992). These functions include litterfall and nutrient input and retention in streams (23 to 46 meters), shade to streams for maintenance of summer stream temperatures (23 to 46 meters), woody debris delivery (30 to 46 meters), and stream bank stability (23 to 46 meters). RHCA widths for intermittent streams should protect small channels from large volumes of sediment and water that could be generated by land management activities and be channeled into fish bearing streams.

The effectiveness of riparian buffer strips in influencing sediment delivery from non-channelized flows is quite variable. Belt and others (1992), cited numerous studies conducted throughout the range of anadromous salmonids and reported that sediment travel-distances and filter strip efficiencies varied considerably from study to study. Belt and others (1992) concluded, based on studies conducted in Idaho (Haupt 1959a and 1959b, Ketcheson and Megahan 1990, Burroughs and King 1985 and 1989) and elsewhere (Trimble and Sartz 1957, Packer 1967, Swift 1986) that sediment rarely travels more than about 91 meters for non-channelized flow. Therefore, 91-meter filter strips are generally effective in controlling sediment that is not channelized. Trimble and Sartz 1957, recommended that where the highest possible water quality standard was required, this could be maintained with 100 meter buffer strips on 70 percent slopes. Recent work by Ketcheson and Megahan (1996) indicates that this may not be adequate on some lithologies and slopes.

Importance of Hill Slope Steepness—Fourth, the likelihood of disturbance resulting in discernible in-stream effects increases as adjacent slopes become steeper. Thus, greater preventive measures to avert or rehabilitate riparian function and structure on steeper slopes may be required to prevent or reduce in-stream effects. The designation of default RHCA widths can easily incorporate the major topographic driver of surface erosion and slope steepness.

Prior research on a variety of wildland and agricultural settings has demonstrated that surface erosion increases with increasing slope steepness, although the increase is not linear. The effect of slope has generally been modeled empirically, and has taken the shape of a power function where the exponent is less than 1, so that slope effects are large for gentle slopes, and decline as slopes get steeper (Foster 1982; Liebenow and others 1990; McCool and others 1987). Megahan and Ketcheson (1996) found that sediment travel distances from road cross drains in the Idaho batholith are proportional to slope gradient (in percent) raised to the 0.5 power. This study was conducted below roads on forested lands, and includes slope gradients ranging from 9 to 59 percent.

Megahan and Ketcheson (1996) and Ketcheson and Megahan (1996) present equations for estimating sediment travel distance below road fills and cross drains which incorporate sediment volume, obstructions, slope angle, and source area as significant explanatory variables. Slope is a significant predictor of distance, and it is not unreasonable to adjust an RHCA width to slope when lacking other intensive site variable information. At slopes greater than 70 percent, other screening tools that incorporate mass erosion risk are needed (Tang and Montgomery 1995). If risk varied solely as a function of slope, one could use the exceedence probability equation directly to tune a slope-directed RHCA model. However, at least three other site variables have been demonstrated to influence travel distance and therefore affect risk. Though it is erroneous to assume that the exceedence probability equations presented by Ketcheson and Megahan (1996) can be used to assign a general slope-driven risk to the RHCA width equation, at the subbasin scale a slope-driven default RHCA width is useful. It is also prudent to use for watershed analysis and planning at the subbasin and Forest project scales.

The research findings of Megahan and Ketcheson (1996) can be used to parameterize a slope-sensitive default RHCA width in the following manner: Distance can be made proportional to slope angle in percent raised to the 0.5 power to provide the proper shape. A constant can be derived from the exceedence probability function of Ketcheson and Megahan (1996) by taking the travel distance that is exceeded only one time in 20 (exceedence p=0.05), a low probability but well-defined event from their data. The travel distance of this event for all their data combined is 480 feet. This distance can then be assigned to a slope of 70 percent, which results in the equation Distance = 58 X $(\text{Slope})^{0.5}$ (fig. 4.26). Although this equation is adjusted to the 5 percent travel distance event, it is not strictly correct to assume that the relationship defines the 5 percent risk associated with operating on slopes of a given steepness.

Similarly, equations and curves that represent "10%" and "25%" risk can be derived by using the 10 percent and 25 percent probability of exceedence distance from Ketcheson and Megahan (1996; fig. 4.26). For the same reasons stated above, these equations do not directly represent 10 and 25 percent risk. They are less conservative than the 5 percent risk equation, but not necessarily by a factor of 2 and 5.

The strongest single variable affecting sediment travel distance from soil disturbing activities is the volume of material displaced, or delivered to a point on a slope from a culvert, drain, etc. Over 78 percent of the variance in sediment travel

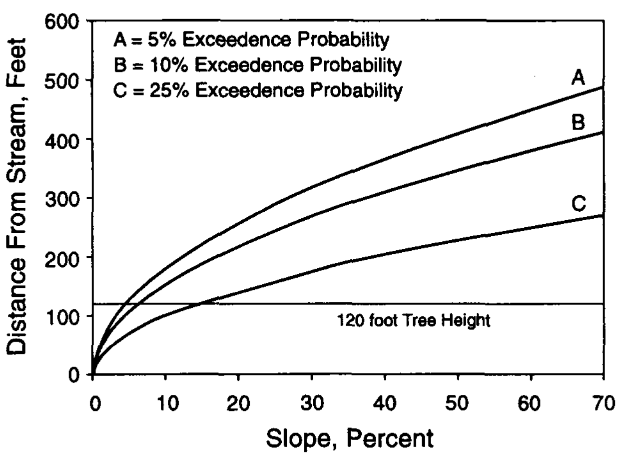


Figure 4.26—Examples of slope-sensitive adjustments to RHCA widths with associated probabilities of exceedence (0.05, 0.10, and 0.25), based on models described by Megahan and Ketcheson (1996) and Ketcheson and Megahan (1996). (A) Distance=58•$(\text{Slope})^{0.5}$; (B) Distance=49•$(\text{Slope})^{0.5}$; (C) Distance=32•$(\text{Slope})^{0.5}$.

distance is explained by volume in the culvert model of Megahan and Ketcheson (1996). Given the strong influence of this relationship, the probability density function of sediment volumes from the data set used in developing their model can be used to define various levels of risk. This is a subtle difference from defining risk using the probability exceedence function (equation 4 of Ketcheson and Megahan 1996) as above, because risk is attributable to a single, measurable attribute — sediment volume. In contrast, the probability exceedence function for travel distance includes the combined effects of all driving variables. Defining risk by volume alone allows a direct application of the Megahan-Ketcheson model for tuning travel distance on slope.

This method assumes that travel distance is strongly influenced by slope in the culvert model. We tested this by regressing the residuals of a 3-variable model [Distance = f (volume, source area, obstacles) on slope]. This regression is significant at P=0.001 and has an r^2=0.33, indicating that there is ample variance to be explained by slope gradient after accounting for the other variables in the multiple model.

A slope-gradient sensitive default RHCA can be estimated directly by setting the two variables "obstacles" and "source area" equal to their median values, allowing slope gradient to vary from 0 to 70 percent, and assigning risk by taking various volumes based on the distribution of volumes sampled in the Megahan-Ketcheson data set. The following reconfiguration of the culvert model was used to generate the curves in fig. 4.27:

$$D=3.28\cdot10^{(0.393+0.554\cdot\log_{10}\text{Vol}+0.5\cdot\log_{10}\text{Slope})}$$

The variable $\log_{10}$ Vol was set equal to 1.60, 1.57, and 1.41 corresponding to the 95, 90, and 75 percentile values of sediment volume sampled; median values of obstructions and source area were used and slope was allowed to vary from 0 to 70. This procedure results in a series of three curves that are similar, but somewhat more conservative than the curves based on the travel distance exceedence curve of Ketcheson and Megahan (1996). Again, the utility of this second set of curves is that risk is defined from the single, strongly influential variable of volume, and the effect of slope is then predicted directly using the Megahan-Ketcheson model.

The width necessary to protect stream and riparian area structure and function can be determined from watershed and site-specific analysis. The dimensions of riparian protection areas, particularly if they are to be used as interim or default standards, should also include safety factors to allow for natural disturbances, uncertainties about the riparian ecosystem of interest, and changes in public values (National Research Council 1996). If an additional margin of error is allowed (not unlike bridge design accounting for unknown factors and longevity of structure), the probability of habitat improvement becomes greater and options for future management decisions are increased (FEMAT 1993).

In general, buffer widths prescribed in FEMAT (1993) and reexamined by Murphy (1995) and Spence and others (1995) are applicable to pro-

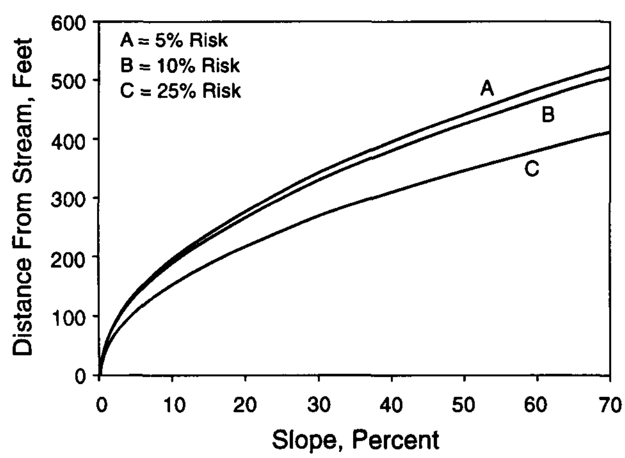


Figure 4.27—Examples of volume-driven risk associated with adjustments to RHCA widths, based on models developed and sediment volume sampled by Megahan and Ketcheson (1996) and Ketcheson and Megahan (1996).

tecting ecological functions whether streams contain fish or not (National Research Council 1996). In contrast, all national forest plans, PACFISH, and INFISH maintain a higher level of riparian protection where fish are present or strongly affected than for non-fish bearing streams. The width of streamside management zones required by state land-use laws is much smaller than the width of natural vegetation needed to provide full ecological protection (FEMAT 1993; Murphy 1995; Spence and others 1995; National Research Council 1996). For example, the forest-practices regulations of California, Washington, Oregon, and Idaho require narrower buffers that leave only an estimated 23 to 58 percent of sources of potential large woody debris present in a mature conifer forest (Murphy 1995).

Taken in aggregate, the standards for management of stream and riparian systems on forest lands are far more restrictive and ecologically more effective than requirements for riparian areas where agriculture and urban or industrial land uses are dominant. No state within the Basin has enacted an agricultural-practices act explicitly protecting riparian vegetation. Botkin and others (1995), the National Research Council (1996), and Spence and others (1995) all call for improved protection of riparian areas in agricultural lands if salmon and many native fishes are to survive in the long term. Currently, agricultural activities are limited by the water-quality requirements of the Clean Water Act, state water-quality standards, and voluntary compliances with best management practices (National Research Council 1996; Spence and others 1995). Even though state forest-practices regulations incompletely deal with maintenance of ecological function and structure, agricultural-practices regulations tend to emphasize meeting waterquality thresholds for drinking water and aquatic biota protection only, rather than addressing riparian protection. In urban and industrial areas, riparian area protection is generally left up to local ordinances (National Research Council 1996). Much of the historically most productive salmon habitat exists in lower river valleys and coastal lowlands where riparian zones are given the least protection (Sedell and other 1990).

Cumulative Watershed Effects—Without a larger-than-project-scale context, it is difficult for managers to assess the full spectrum of effects that management activities may have as well as how off-site activities and conditions may affect site conditions. Cumulative effects analyses are designed to provide this large context. Cumulative watershed effects (CWE) are those from more than one activity or disturbance event that have combined to affect a stream or riparian area more significantly than any of the single events alone. Most direct sources of cumulative effects are local in space and time. The analysis and management of cumulative watershed effects are intended in part to control watershed level sediment, hydrologic change, and biodiversity that project planning might miss (Reid 1993). Planning at the watershed or subwatershed levels can help forest managers reduce or prevent undesirable cumulative effects on fish habitat.

Evaluations of potential cumulative effects of a given project should consider watershed erosion potential; slope stability; disturbance from wildfires, timber harvest, roads, and other land uses; rate of recovery after disturbances; and overall watershed condition and stream condition. Assessments of cumulative effects account for land uses other than forestry. Grazing, mining, agriculture, hydroelectric and water development projects, recreation, and urbanization can all cause increments of cumulative effects on a watershed and its aquatic populations (Platts 1991; Nelson and others 1991; Clark and Gibbons 1991). Most important to cumulative effects is to provide a spatial and temporal context for assessing effects of a proposed project at different scales.

At the broad scale, the Biophysical Environment assessment (Chapter 2) developed three data layers which assess potential risk from roads, fire, and sensitivity of channels to the presence of vegetation (influence of vegetation on channel integrity in rangelands) for each subbasin (approximately 280,000 hectares each). The road hazard index is derived from coefficients of accelerated erosion from roads based on lithology, slope, and drainage density throughout the Basin. Risk of sedimentation from fire is derived from the potential erosivity of bare soil at the subwatershed level. These data plus stream recovery data driven primarily by channel gradient have been used to display relative risk of sedimentation to subbasins throughout the Basin (Quigley and others 1996). The utility of this basin-wide template is a spatially explicit strategic context from which one can consider watershed and aquatic risks.

At the mid-scale, within a watershed or subwatershed, several methods of cumulative effects assessments exist. Three methods are commonly used within the Basin: 1) equivalent clearcut area (ECA), 2) equivalent road area (ERA) used by Region 5 of the Forest Service (Pacific Southwest Region), and 3) the Region 1 and 4 (R1/R4) sediment model. These are the most useful models currently available for planning and evaluating the effects of management strategies.

Menning and others (in press), building on existing concepts and the work of McGurk and Fong (1995), propose a more spatially explicit model for assessing cumulative watershed effects and placing limits on watershed disturbances near streams. They propose a 45 meter-wide riparian area which would be conservatively managed for riparian and aquatic ecological processes and communities. They propose an adjacent zone of potential landscape disturbance or region of influence which is equal to 45 meters or adjusted for slope. The steeper the slope adjacent to the stream, the wider the zone. This second zone would protect the stream from sediments and provide habitat and microclimate if needed. The third zone, the uplands of the watershed, is assumed to have a small effect on streams and riparian areas. This system serves two main purposes: 1) it closely approximates the aquatic and riparian ecological regions described above and 2) it allows better consideration of the distance from a road or harvest area to the stream which they might affect. Their vegetation dynamics model is programed to limit activity in each zone up to respective levels of 5, 10, and 15 percent allowable ERA. Aquatic and riparian systems are most influenced by activity close to the stream. Hence, the strictest limits on watershed disturbance are in the first zone. An intermediate allowable ERA percent is set for the potential landscape disturbance area and a more permissive ERA percent is established for the uplands. Applying this method in the highly dissected topography of the Eldorado and Plumas national forests resulted in about 46 percent of small watersheds being within the two riparian zones (13 percent in zone one, and 33 percent in zone two). This is a more logical approach to protection of ecological function than the "one-size-fits-all" default riparian zones. The objective is to reduce disturbance near streams and to assess and plan how ecological function can be maintained. Menning and others (in press) have provided an important improvement to making cumulative watershed effects spatially explicit with their approach to riparian buffers. In their scheme, streams with adjacent steep slopes have greater protection from management, streams get a full compliment of energy inputs and large wood, in-stream aquatic biodiversity is minimally effected, and riparian habitat for transitory and obligate species is maintained.

Each of the three CWE models requires more information on streams, adjacent slopes, and soils than is currently available. They do not predict effects with precision, and have a tenuous link between upland area disturbance and in-stream effects. Each model can be modified to express near stream disturbances distinct from upslope disturbances and set lower thresholds near streams than on the upslope (McGurk and Fong 1995; Menning and others, in press). Compared with watershed level CWE treatment, this would provide a more conservative approach to maintaining ecological function of RHCAs within a watershed by partitioning disturbance which could accommodate the four basic principles listed in the beginning of this section. Use of watershed analysis, use of modified cumulative effects models, and adoption of quantifiable objectives for riparian forest stand structure and cover would allow land managers to limit disturbance close to stream channels while identifying acceptable or rehabilitative management activities in the uplands away from the stream. The absolute width of the riparian management area is less important in many ways than specifying site specific and watershed riparian goals and objectives in a way that gives clear direction as to expected ecological outcomes. The principles and tools described above attempt to provide a means to attaining clear goals, and will provide the most accountable system for restoring riparian areas, limiting disturbance, and achieving proper functioning condition.

Another strategy to protect streams from the effects of management activities and cumulative watershed effects, Riparian Management Objectives (RMOs), is currently being proposed in the Basin. While the intended goals are water quality protection and in-stream habitat and bank protection, these strategies are primarily performance or end-use oriented and not prescriptive, giving little direction or flexibility to meet these goals. Though explicit thresholds or standards are established by RMOs and must be met, how they must be met is not specified. The use of RMOs as standards and thresholds has four primary problems. Such standards usually become inflexible and are unable to accommodate the range of spatial variability within a watershed through time as well as for the rare event. Second, because it is difficult to anticipate and accommodate the temporal nature of biophysical processes operating in the watershed, a specified means of attaining standards (such as managing toward mature forest structure and cover in RHCAs and setting cumulative watershed effects thresholds lower near streams than in the uplands) is needed. Third, RMOs tend to be simplistic and often disconnected from watershed processes and adjacent forests. The complex interaction between watershed and in-channel processes are not well represented by a few measurable RMOs for sediment and large wood. Finally, RMOs often distinguish between fish-bearing and non-fishbearing streams, with more relaxed standards for the latter. Since non-fishbearing streams are generally more numerically abundant and in steeper and more erodible terrain, this can result in much of the stream network being without adequate protection.

Discussion

The acknowledgment of landscape issues in recent conservation strategies and the implementation of watershed and ecosystem analysis represent an important advance from the use of simple habitat standards (Montgomery 1995). The existence of even the most productive habitats and populations are likely to be dynamic and uncertain through time. Our ability to effectively predict and manage disturbance effects is still unproven; we cannot simply control or predict the natural disturbances that will undoubtedly challenge these systems in the future (Frissell and Bayles 1996).

Long term persistence of aquatic biological diversity will depend on more than the current distribution of productive habitats for many systems. It will likely depend on restoring watershed processes that create and maintain habitats across broad networks that will support the species, genetic, and phenotypic diversity necessary to buffer populations and communities in variable and changing environments. Therefore, while attempts to preserve existing conditions are focused in some watersheds, it is important that others be actively managed with the intent of rehabilitating more complete systems. As Bisson and others (1996) suggest, bold steps and experimentation will be necessary to make progress.

The opportunity for conservation of currently important watersheds will be dictated by the current distribution of those watersheds and the conflict with future management goals. The opportunity for rehabilitation will depend on the amount of investment in effective practices and convergence with other goals. The current management emphasis on restoring structure, composition, and the processes relevant to the condition of terrestrial ecosystems through active management has been strongly articulated in the Landscape Dynamics chapter (Chapter 3) and in the public debate regarding issues of forest health. The discussion must be broadened to integrate active rehabilitation of aquatic systems and watershed processes as well.

The assessments of aquatic, watershed, and terrestrial communities demonstrate that each has departed dramatically from historical conditions. It is striking, however, that the changes in forest communities parallel those in aquatic and watershed systems, often in the same areas and for similar reasons. Dramatic changes in these systems, for example, are associated with the homogenization and fragmentation of habitat types, selective and overly optimistic harvest regimes, artificial culture and stocking practices, and the introduction of non-native species. Road densities appear to be one of the best predictors of departure from historic conditions and indicators of current integrity for those landscapes (Landscape Dynamics, Chapter 3).

Conventional timber harvest activities and road building can threaten watershed processes and the conditions of aquatic habitats. But changing fire regimes have become a major issue in forest management that has galvanized new efforts to actively restore/manipulate forest structure and composition. In some cases this could be viewed as a threat to the persistence of aquatic systems. Restoration of structure, composition, and function of forest ecosystems more consistent with natural disturbance regimes, however, might also benefit aquatic ecosystems. We suggest that efforts to restore forests could represent an opportunity to reestablish a mosaic and more natural disturbance regime in aquatic systems without risking those that are currently productive. Three elements to this approach include: 1) an opportunity to conserve key remnant habitats and populations; 2) an opportunity to rehabilitate degraded watersheds to a more productive condition; and 3) an opportunity to restore a structure and composition in forests that reduces the risks of simultaneous events among productive or critical habitats.

Conservation

The focus on conservation of critical elements is clear from this and preceding work. The habitats supporting the most productive, diverse or otherwise critical populations provide the best opportunities for shortterm persistence. They also provide the best opportunities for rehabilitation of more complete systems in the future. An emphasis on conservation in those areas does not necessarily mean forest management activities cease. It does imply that any management must clearly minimize or eliminate risks that might compromise the ability of populations to maintain or improve their status over time.

Watershed Rehabilitation

Many of the subbasins in the Basin appear to support a patchwork of productive and degraded watersheds. The best remaining aquatic habitats are often found in higher elevation systems associated with cold forest types that also are in relatively good condition. Subwatersheds at mid and lower elevations also support important elements for aquatic systems and native species, but they are more strongly influenced by habitat loss, degradation, and watershed disturbance associated with timber harvest, grazing, and more extensive roading. Active watershed rehabilitation, particularly the obliteration of excessive road networks and use of new grazing strategies, will be an important step to conserving and expanding the network of habitats. The mid-elevation landscapes often show the greatest departure in forest conditions and potentially the greatest needs for active restoration of vegetation structure and composition (Landscape Dynamics, Chapter 3). We suspect that in many cases the need for active restoration of forest and aquatic systems will coincide. Existing road networks could represent a key to progress in both. Existing road systems provide ready access and the opportunity for active management and generally exist where recomposition of forests is most urgent. By focusing projects in individual and heavily roaded watersheds rather than dispersing them across a basin (Sensu, Franklin 1992; Reeves and others 1995) forest management needs might be accomplished quickly. This would also allow longer periods for recovery in watershed processes and eliminate the need for continued road maintenance. Existing road densities often exceed those necessary for modern, logging systems. The obliteration of unnecessary or particularly damaging roads could accompany many projects to actively recompose forests.

Active watershed restoration has not been an emphasis of past land management. There is a great deal to learn, and work will necessarily be experimental with uncertain results. Because a mosaic of watershed conditions often exists in the basins where active management might play a role, minimizing risks in subwatersheds supporting critical habitats by prioritizing new work in areas of less concern would be possible.

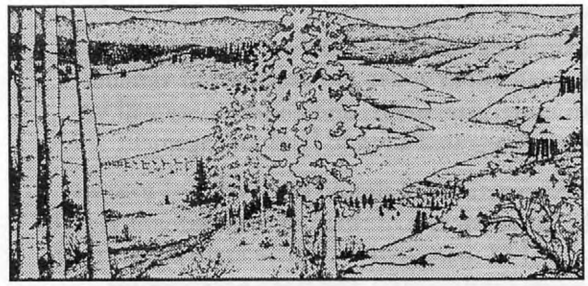
Forest Rehabilitation

Changing fire regimes have been associated with the homogenization of forest habitat types (Landscape Dynamics, Chapter 3). A major conclusion of the landscape analysis is that future wild fires are likely to burn with increased severity and over larger areas than in the past. Though wildfires were important historically to healthy aquatic ecosystems, depressed and strongly isolated populations could be vulnerable to the effects of intense or very large fires. In addition, the potential for increasing synchrony in fire related disturbance among streams and populations is of concern. Even though fires have likely played an important role in the succession and long term complexity of stream habitats, extreme fires could essentially reset the successional stage of many systems at one time. The temporal and spatial diversity of habitat conditions created and maintained through the mosaic of landscape patterns and historical disturbance regimes could be simplified or lost. Threats from such fires will be most important in aquatic systems that are already highly degraded and fragmented.

We understand little about the historical disturbance regimes structuring aquatic ecosystems. We do not believe that the natural disturbance critical to the maintenance of productive habitats can easily be restored or replaced through active management. The risks of active management may well outweigh the risks associated with large and uncharacteristic fires. There is also uncertainty about whether restoration of ecosystem structure, composition, and process can even be effective (Baker 1994; Stanley 1995). Aquatic and terrestrial systems have been strongly altered from historical conditions and in many cases natural recovery seems highly unlikely. Changing fires regimes have clearly become a dominant issue in forest management and substantial resources will be focused on the problem. Those resources could be focused in work that creates the greatest potential benefit and least possible risk to diversity and condition of aquatic ecosystems. We suggest that the logical priorities for active management lie in the watersheds and areas surrounding the population strongholds, critical habitats, and riparian zones most important in the aquatic system. Often the forest lands most in need of active forest restoration will be those that are most heavily roaded and encompass the most degraded watersheds. The opportunity and need to attempt restoration of forests without risking critical habitats may be more common than not.

Even within watersheds where forest restoration proceeds, efforts could focus to minimize disruption to aquatic processes. Active management within RHCAs is often proposed as a way to reduce fire severity by recreating a more historical-like mosaic of stands that offer natural firebreaks and less concentrated food sources for pests. The trade-off between fire risk and the risks associated with management activities is a watershed and site specific issue. Ecologically, it makes sense for treatment to begin in upland zones and work down to riparian zones, where activities occurring closer to the stream have a greater probability for adverse effects on the stream. We believe this is a prudent approach. Productivity and fuel loading may be highest in riparian zones but these zones are generally more moist and historically had longer fire return intervals (Agee 1994). Thus, in an ecological sense, fire management concerns are generally not a primary emphasis in riparian zones. Given the critical nature of the riparian management area to fish and aquatic habitat, a prudent course of treatment with benefits for both RHCAs and the entire landscape might be to treat the fire-prone portions of the landscape outside riparian areas to protect historical refuges in RHCAs.

By focusing work outside the most productive or critical aquatic habitats, we minimize the risk associated with direct disturbance but still gain experience and the potential for reestablishing a more characteristic vegetation mosaic and a broader distribution of productive watersheds. We can begin the process of experimentation and adaptive management that may ultimately lead to management more consistent with the natural processes that structure aquatic ecosystems.



INFORMATION AND RESEARCH NEEDS

To maintain a coordinated, adaptive approach to management of federal forests and rangelands in the Basin, a comprehensive inventory, research, and monitoring program is needed to assess success over time and to modify management strategies as new information becomes available. To be effective, the program must fully integrate research and management areas and must be consistently applied on Federal lands. Recently, scientists and managers identified six freshwater priorities based on scientific significance, sociopolitical relevance, and needs of decision makers (Naiman and others 1995):

- **Ecological restoration and rehabilitation** - Restoration and rehabilitation are a high priority because water quality standards under the Federal Clean Water Act cannot be met in one-third of the nation's freshwater ecosystems.
- **Maintaining biodiversity** - The goal of maintaining biodiversity focuses on preserving individual species as well as the diversity of ecological processes and the integrity of ecological systems. This includes understanding relations between species and ecological processes as well as the consequences of exotic invasions.
- **Modified hydrological flow patterns** - The hydrological regime in virtually every freshwater body in the Basin has been modified to some extent by dams, diversions, and withdrawals.
- **Ecosystem goods and services** - An improved understanding of environmental factors responsible for natural resources and their values, including the costs associated with their loss, is necessary for responsible management.
- **Predictive management** - If we are to predict the consequences of cumulative and synergistic effects of management activities, we need more information on disturbance regimes and their physical and biological legacies.
- **Solving future problems** - We must ensure that basic science and education can provide the framework for meeting future water-resource challenges.

The following section briefly describes additional information and research needed to support long-term management for productive, healthy ecosystems with a high probability of ensuring the viability of species and ecosystem processes.

Riparian Management

Riparian management areas have important roles in maintaining biological diversity and productivity (such as fish, birds, bats and frogs) and regulating energy and nutrient flow between waterbodies and uplands. These areas have important values in landscape aesthetics. The basic concerns of river and lake management related to water quality and water-based recreation can be improved through a better understanding of riparian dynamics. Evaluation of the roles of riparian buffer zones and corridors of variable structure, size, shape, and connectivity will improve integrated aquatic-land management.

Field experiments should be developed to assess the potential role of riparian management areas in watershed management. These experiments may answer fundamental questions necessary for knowledgeable watershed management. They may provide information on the spatial dimensions (that is, width, length, depth, space and continuity) necessary to achieve single and multiple ecological and/or social objectives.

Understanding the Rare Event

The structure and function of stream and river riparian areas are often related to the frequency of disturbance by extreme hydrological events. Natural systems adjust to the whole range of natural process variations. The dependence of natural systems on extreme events relates to variability in geomorphic and hydrologic processes, stream habitat structure, and to the roles of succession and biological interactions. The significance of rare events (such as droughts, floods, fires, etc.) in sustaining riparian and aquatic systems may vary with climate region and from naturally stable to highly variable systems. Rare events are important for sustaining some riparian and flood plain systems. Therefore, management and restoration options for these areas must include the variability of riparian structure and function. This is particularly important given the Basin-wide efforts to regulate and dewater rivers and improve water-level control rules on lakes and reservoirs.

Evaluating the role of hydrologic and geomorphic disturbance from effects of extreme events on the biological structure and function of streams, lakes and riparian management areas is important. Three types of studies can contribute to this evaluation: 1) field surveys to assess the impact of historical events such as fires and large floods. 2) intensive, opportunistic surveys during and following such rare events (for example, the landslide of North Fork Boise River). 3) use of human impacts as analogues to assess the resilience of riparian and aquatic ecosystems to changes in the magnitude and frequency of extreme events (such as, below regulating dams and reservoirs).

Stream Habitat Inventory

Stream habitat inventories provide the foundation for land management, watershed analysis, activity planning, monitoring, and watershed restoration. Maintenance of stream habitats and watersheds that are currently in good condition is critical to reduce degradation of aquatic systems. Physical and biological conditions and processes of aquatic systems vary greatly in area and scale across federal forest and range lands in the Basin. Programs to manage these ecosystems are best based on site-specific information.

Currently, 20,000 kilometers of streams have been inventoried using a standard protocol. This represents less than 5 percent of the estimated 444,000 kilometers of stream in the assessment area. Although a complete inventory of all stream and riparian habitats would be ideal, a comprehensive coverage across physiographies and stream types, at minimum, is needed for accurate extrapolation of information. Inventory methodologies must be standardized to facilitate broad-scale analysis.

Limited stream habitat inventory data describes conditions in ERUs. The Snake Headwaters, Upper Snake, Owyhee Uplands, and eastern sections of the Central Idaho Mountains ERUs have essentially no data on condition of stream habitat. The Northern Great Basin, most of the Columbia Plateau, the southern and eastern sections of the Northern Glaciated Mountains, the Upper Clark Fork, and the southern sections of the Lower Clark Fork have very limited stream-inventory information. Overall, stream habitat data are more available in forested regions, less available in rangelands, and practically nonexistent in valley-bottom and agricultural areas.

Aquatic habitat inventory and monitoring data need to be incorporated and integrated with other resource information and linked to a Geographical Information System (GIS). Information submitted for this assessment came in many formats and levels of quality, which made it difficult to compile and interpret. A pressing need exists for both spatially referenced information and a consistent approach to collecting, accessing, and sorting these data.

Information collected at any scale needs to have both the data structure unique to the point, line, or polygon and a spatial reference. Positional data provides for further spatial and temporal analysis as well as a framework to combine and integrate other spatially located data sets. Considerable time was spent spatially referencing data from more than 6,300 stream reaches from multiple agencies. Little information had locational components.

Specific information includes:

- Exact locations of the start and end of each stream need to be recorded using Global Positioning System Units.
- The EPA reach number at both the 1:100,000 and 1:24,000 scales for each stream reach is needed.
- River miles for each stream reach need to be accurately calculated from the USGS 1:100,000 and 1:24,000 scale.
- Unnamed streams and tributary locations, as well as stream channel gradients, need to be explicitly described using 1:100,000 and 1:24,000 USGS map layers and river miles.
- Clear interagency spatial information data collection protocols need to be developed.
- The ICBEMP Master Habitat Database needs to be maintained, periodically revised, and widely distributed.
- An interagency infrastructure to provide a data repository and vehicle to transfer multiscale spatial analyses and associated technical expertise to field units needs to be developed and implemented.

Aquatic Species

Fishes

Complete species inventories are necessary to evaluate the structure and composition of aquatic communities in the Basin fully. Because of the inherent complexity of watershed and ecological processes and their interaction and linkages with stream productivity and riparian condition, the probability of understanding these processes and interactions improves through long-term research. The evaluation of the seven key salmonids and 39 additional sensitive fish species in the study area revealed some common research needs: a basic understanding of life history and habitat requirements; species viability and taxonomic and genetic characteristics; population status and distribution; predator/prey relationships and competition from non-native introductions; and effects on survival from changes in habitat condition and water quality and quantity.

Regularly scheduled population and habitat monitoring is needed for many sensitive species. This is especially critical for many rare species with restricted distribution which are particularly vulnerable to disturbance events including the introduction of exotic species, parasites, and diseases. For several species, particularly the redband-inland rainbow complex, basic genetic information is needed to determine which populations represent native genomes, which have been modified, and which have been replaced with genes of introduced fishes. However, to meet the needs of river basin and watershed management, future research must concentrate on integrated community studies to understand the diversity and complexity of habitat needs and life-history requirements of fish, and species interactions, which will compliment the species-by-species studies. Monitoring to detect changes or trends in the structure, composition, and diversity of aquatic communities across whole systems such as the subbasins defined in this assessment should be considered as important as monitoring to detect trends in individual populations. Because large-scale and community rather than population specific monitoring imply wholly different sampling schemes, substantial effort will be needed to develop efficient and effective sampling approaches.

Predictive models were invaluable for our assessment of fish distributions and the identification of management issues and opportunities. Databases and models for similar evaluations relevant to planning and management at the scale of subbasins and watersheds are virtually nonexistent. Because of the lack of complete inventories, such models could be critical in the identification and prioritization of conservation and restoration opportunities and issues. Further development and refinement of such models will require the collaboration of research and management programs to develop and access the necessary data sets. Integration with other landscape-based assessments will be critical to provide generalized models with the potential for broad application.

There is a relatively extensive body of work regarding the distribution, dynamics, and interrelationships of fish and physical habitats at the scale of individual habitat units and stream reaches. We know far less, however, about the large-scale relationships and processes that influence the structure, composition, dynamics, and persistence of populations and communities. Effective conservation and restoration of aquatic biodiversity will require a better understanding of the spatial and temporal distribution of habitats necessary to sustain functioning systems. Large-scale spatially influenced population dynamics have been a focus of interest among conservation biologists and managers dealing with fishes. These new concepts are well based in theory and appear to have particular relevance for management of aquatic systems and especially stream fishes. The empirical basis for application in management, notably information regarding extinction dynamics, dispersal and recolonization processes, the nature and relevance of disturbance, and the mechanisms structuring populations, however, are poorly developed. If large scale patterns and processes influence the dynamics and persistence of populations as we suspect, such information will be important in guiding more efficient and effective conservation and restoration efforts.

Aquatic Invertebrates

The basic biotic components of most aquatic habitats, including algae, microinvertebrates, and macroinvertebrates, are poorly known in the Basin. Data on the composition and relative abundance of various macroinvertebrate species are widely used to indicate overall productivity, biotic integrity and health of stream systems. Studies are needed which identify aquatic species composition and track their abundance and diversity.

General patterns across the breath of the assessment area were surprisingly interpretable for stream macroinvertebrate assemblages characterized from an array of information sources compiled by Li and others (1995) in a study conducted for this assessment. Using four to five general invertebrate metrics, assemblages were classified into five general groups: riverine; low diversity from hydrologically flashy systems; low diversity from high desert or disturbed systems; moderate diversity from good stream habitat conditions; and high diversity from cool water habitats.

Differences within ERUs can be detected by macroinvertebrate metrics. The Blue Mountains ERU was the most taxa rich, with high numbers of mayflies, stoneflies, and caddisflies and low dominance by the most abundant taxa. The Columbia Plateau exhibited the lowest taxa richness, and high dominance by the most abundant taxa. These values were influenced by Lower Snake River samples that were very low in several metrics. Other streams within the Columbia Plateau ecoregion have characteristics more similar to the High Desert ecoregion of the Great Basin in southern Oregon and Idaho. These distinctions within the Columbia Plateau demonstrated the importance of examining within-region variability. The information available for the Northern

Rockies indicated that streams from the Bitteroot Mountains were distinctly more diverse than streams from upper Salmon River tributaries.

Using invertebrate metrics such as taxa richness, abundance and proportion of Ephemeroptera, Plecoptera and Trichoptera, and correlations with specific landuses were vague. For instance, lack of correlation between assemblage characteristics and road densities was due in part to data collected at very different scales. Whereas road density data was collected for the entire Basin and summarized at a relatively broad scale, invertebrate samples are site-specific to a point or reach in a stream. Though invertebrates may be very sensitive to disturbances resulting from roads, such as sedimentation and increased solar radiation, these assemblages may reflect riparian or streambank conditions relatively immediate to the stream. If roads are dense but somewhat distant from the stream, there may not be stream disturbances that result in changes within invertebrate communities. On the other hand, composite information for invertebrates, representing long stream extents, may better represent assemblages comparable to road densities at the same scale. The metrics used for this analysis were those most readily calculated from available information, representing very general assemblage characteristics. Perhaps more finely-tuned metrics in concert with finer scale resolution road data would be more appropriate for detecting the effects of road density.

Short-term Research and Data Needs (1-3 years)

- Relevant geomorphic and hydrologic data needs to be matched to the invertebrate sample sites throughout the Basin.
- In the Blue Mountains a joint National Science Foundation and Environmental Protection Agency Watershed Ecosystems study is underway in the John Day and Grande Ronde rivers. A systematic effort to sample wilderness areas, seeps, springs, and other unusual habitats with detailed description of taxa could be accomplished in conjunction with this ongoing study.

Long-term Research (1-15 years)

- Research is needed to establish more direct links between biological responses to landscape conditions. Gradients of influence such as changing riparian canopy cover and composition, sedimentation, road density, and grazing system could be established within and between watersheds with corresponding invertebrate sampling along these gradients.
- Tests for causal mechanisms controlling invertebrate assemblages can be experimentally designed at sites where human activities are least and where active restoration activities are planned.

Lake and Stream Productivity in Relation to Lithologic Variables

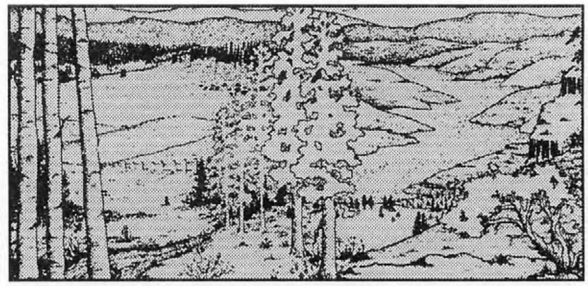
Research into lithologic variables, including geochemistry, sediment yield and texture, morphologic character, erosion rates, and their controls on species distributions and habitat suitability are needed. That three lithologic variables, simplified from the original 41 bedrock lithologic types of Johnson and Raines (1995), which were themselves simplified from over 800 lithologic units, emerged as significant components in the analysis of habitat suitability and populations suggests that linkages between aquatic and riparian conditions, stream productivity, and bedrock geology may be stronger than previously believed. Linkages between bedrock geochemical nutrient distributions (Raines and others 1995) and aquatic/riparian conditions and populations appear to be a particularly fruitful area for research. Such investigations would be appropriate at regional to watershed extents.

Monitoring

With the future likelihood of reduced Federal land-management budgets and staffs, monitoring of individual projects and inventory of all streams will not be feasible. Long-term systematic inventory and monitoring of reference watersheds, however, are likely feasible, and are necessary to assess whether standards, guidelines and objectives are met and to provide reference points for effectiveness and validation monitoring. Long-term data sets from reference watersheds will provide an essential basis for adaptive management and a gauge by which to assess trends in stream and riparian condition. Establishment of reference watersheds that contain lands administered by both the Forest Service and BLM would facilitate long-term coordination between the agencies.

Because of the variability among watersheds, study designs to facilitate broad application of reference watershed information must focus on appropriate pairing and aggregation of data based on such variables as basin groups and size, parent geology, geomorphology, discharge, vegetation types, and stream types. Monitoring results need to be documented, analyzed, reported, and stored by a method that is easily retrievable by the agencies responsible for land management in any given river basin and/or watershed.

Monitoring of water quality, riparian vegetation trends, and channel characteristics can be remotely assessed. These remote sensing techniques have the potential to provide less costly and more extensive assessment of riparian management areas. There is an urgent need to compare capabilities of field and remotely-sensed imagery at several resolutions for monitoring riparian and instream processes.



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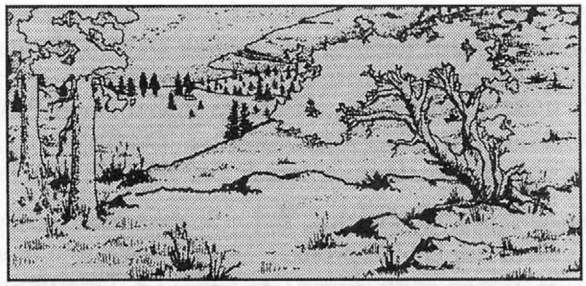

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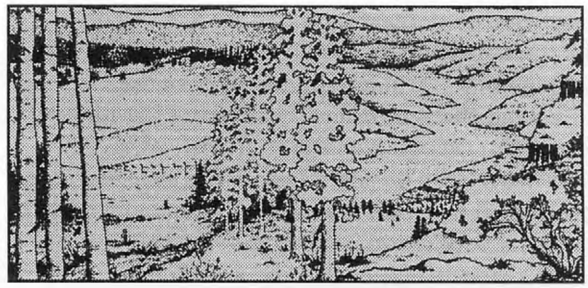


GLOSSARY

adfluvial - fishes that spawn in tributary streams but spend a significant portions of their life in a lake.

alevin - a newly-hatched salmon or trout prior to absorption of the yolk sac.

allopatric - species, taxa, or life-history forms occurring in separate or disjunct geographic areas.

allospecies - semispecies; The component species of a superspecies.

ammocoetes - larval stage of lampreys, usually lasting four to seven years.

anadromous - fishes which spawn in fresh water but spend a significant portion of their life in the ocean.

escapement - the number of adults which survive to reach the spawning grounds.

eutrophic - rich in dissolved nutrients, photosynthetically productive, and often deficient in oxygen during warm periods.

life history - the significant features of the life cycle through which a fish passes - may include timing and age of maturation, reproduction, migration, patterns of migration, etc.

parr - juvenile anadromous salmonids which normally reside for a year or more in fresh water and are not capable of tolerating saline water; may refer to steelhead trout, coho, sockeye, or chinook salmon.

phenotype - the physical expression resulting from the combined influences of genotype and environment; different phenotypes may be represented by different life-history expressions of patterns.

pool - portion of a stream with reduced current velocity, often with deeper water than surrounding areas and with a smooth surface.

population - organisms of the same species that occur in a particular place at a given time; a population may contain several discrete breeding groups or stocks.

redd - the spawning nest of salmonids; usually a scooped depression in clean gravel in which eggs are deposited and buried.

resident - spends entire life in a single stream.

riparian area - area with distinctive soils and vegetation between a stream or other body of water and the adjacent upland; includes wetlands and those portions of floodplains and valley bottoms that support riparian vegetation.

salmonid - fishes of the family Salmonidae; includes salmon, trout, char, white fishes, and grayling.

stronghold - defined by the ICBEMP Science Integration Team as directly associated with strong populations. Strong populations have the following characteristics: 1) all major life-history forms (for example: resident, fluvial, adfluvial) that historically occurred within the watershed are present; 2) numbers are stable or increasing and the local population is likely to be at half or more of its historical size or density; and 3) the population or metapopulation within the watershed, or within a larger region of which the watershed is a part, probably contains at least 5,000 individuals or 500 adults.

sucker - fishes of the family Catostomidae.

Symbiont - A species of plant or animal that lives closely associated (sometimes necessarily so) with another species.

sympatric - distinct species, taxa, or life-history forms occupying the same or overlapping geographic areas without interbreeding.

taxa - refers to more than one species, subspecies, or other taxonomic units (taxon - singular).

tributary - stream or river flowing into a lake or larger stream or river.

zooplankton - small (often microscopic) aquatic animals suspended or weakly swimming in water.

APPENDIX 4A

Lake Systems of the Upper Columbia River Basin

The Aquatic/Riparian Staff of the Interior Columbia Basin Ecosystem Management Project contracted Gary Larson and William Liss of Oregon State University and E&S Environmental Chemistry in Corvallis, Oregon to (1) characterize lakes within the assessment area and (2) evaluate the potential effects of land-use activities of lake ecosystems across the assessment area. Data was compiled from multiple sources throughout the Basin into a single database (Pacific Northwest Lakes DLG) that was spatially identified at the 1:100,000 scale (map 4A.1). A total of 9,186 was identified. Because wetlands and/or dry lakes were sometimes counted as lakes, this data may overestimate the number of lakes, particularly in some arid areas such as the Northern Great Basin.

Characterization of Lakes Within the ICBEMP Assessment Area

Lakes in the study area were initially grouped by modified Bailey Ecoregions (to section level) which roughly correspond to ICBEMP Ecological Reporting Unit boundaries. General comparisons for this report were based on conversions from Bailey Ecoregions to ERUs. The greatest number of lakes occurs in the Northern Rockies (Northern Glaciated Mountains and Lower Clark Fork; N=2,339), and Middle Rockies provinces (Upper Clark Fork, Blue Mountains, and Central Idaho Mountains; N=2,171). The Southern Rockies (Snake Headwaters ERU; N=814) and Cascades provinces (Northern and Southern Cascades; N=811) have the fewest lakes. The greatest number of lakes are small (1-5 hectares) and are located at high elevations (>1,600 meters).

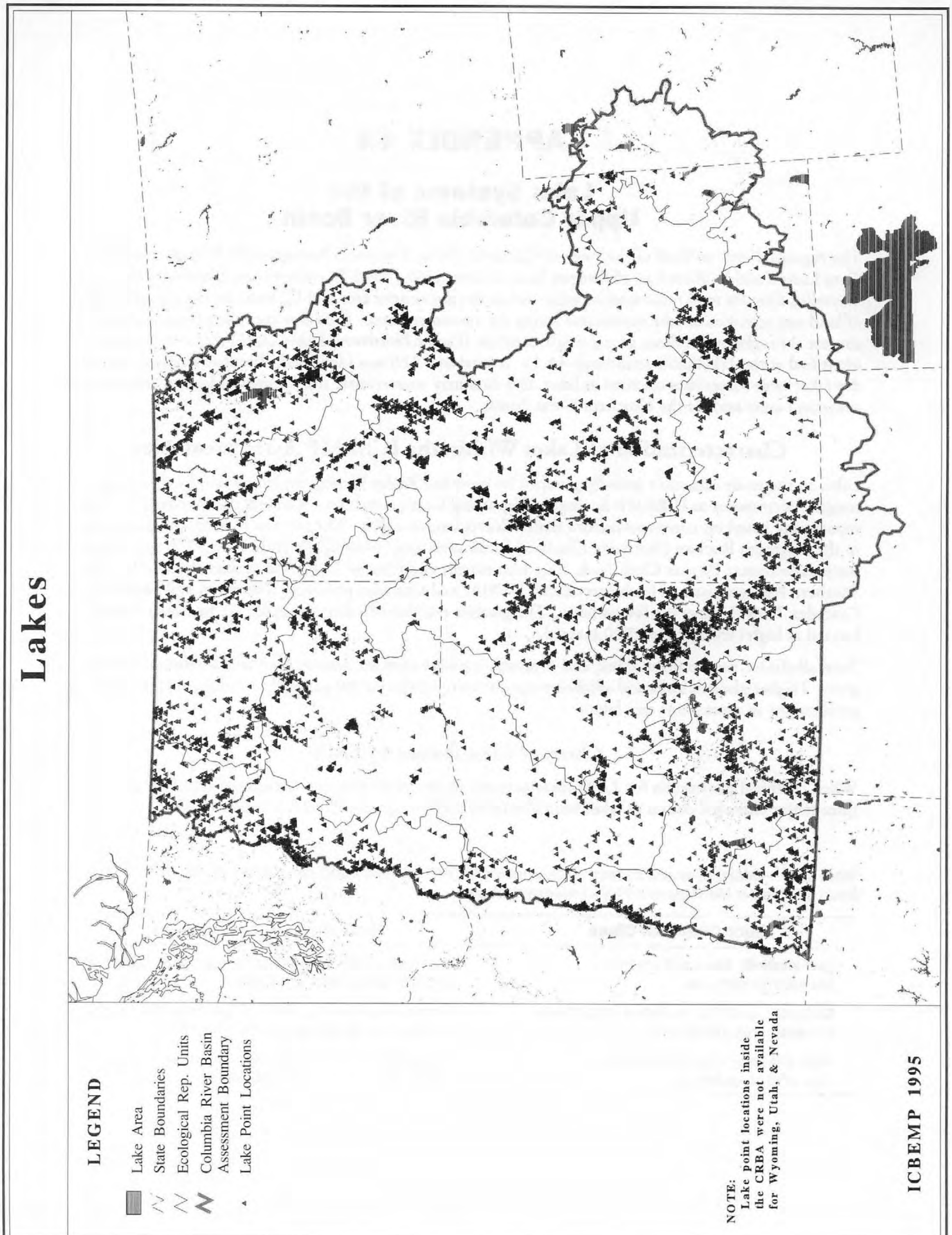
Total alkalinity, total phosphorous, and conductivity were used for determining trophic status on productivity. Higher phosphorous and alkalinity conductivity combinations generally related to higher levels of productivity in assessment area lakes.

Chemical Classification by ERUs

When chemical parameters for 1,543 lakes sampled in the study area were compared among ecoregions, three major chemical classes of lakes were identified and summarized by ERU (table 4A.1).

Table 4A.1— Major chemical classes of lakes identified from 1543 sampled lakes within the Interior Columbia Basin Ecosystem Management Project assessment area.

Major Chemical Class	Ecological Reporting Unit
Low alkalinity, low conductivity, low total phosphorus	Northern and Southern Cascades, Upper Clark Fork, Central Idaho, Blue Mountains
Moderate alkalinity, moderate conductivity, moderate total phosphorus	Snake Headwaters, Northern Glaciated Mountains (Okanagan Highlands), Lower Clark Fork
High alkalinity, high conductivity, high total phosphorus	Columbia Plateau, Northern Great Basin, Upper Snake, Owyhee Uplands



Map 4A.1—Lakes.

A major caveat in this analysis is that a "found" sample was used to make inferences on the entire population of lakes; therefore, it was not possible to assign levels of statistical confidence to differences in lake attributes. Different sampling protocols and analytical methods for the data sets used in this evaluation also confounded direct comparison. Furthermore, there was no consistency in the number of observations among parameters.

Lakes that were most sensitive to atmospheric pollution sources (such as low alkalinity lakes) were located in the Central Idaho, upper Clark Fork, Blue Mountains, and Northern and Southern Cascades. Lakes most sensitive to land-use disturbance (for example, lakes in more accessible terrain that currently have low levels of total phosphorus) are those in the eastern part of the Northern Glaciated Mountains and Lower Clark Fork ERUs. Other sections with low total phosphorus include the Northern and Southern Cascades, Upper Clark Fork, Central Idaho, and Blue Mountains. These areas would be sensitive to nutrient additions from any source.

Some areas, including the Columbia Plateau, Northern Great Basin, and Upper Klamath have very high total phosphorus values. Although these areas contain volcanically-influenced soils, a significant anthropogenic signal may be superimposed on these lakes. Likely nutrient sources would include grazing and irrigated agriculture. The Columbia Plateau and Northern Great Basin are also noteworthy because of the strong saline influence (sodic soils).

Lake Classifications by Kuchler Potential Natural Vegetation Type

There appeared to be a strong relationship between lake water chemistry and Kuchler vegetation types (table 4A.2). Productive land for forests or agriculture has productive lakes nested on it. A general relationship between elevation and lake chemistry was also noted. Higher elevation lakes generally were low alkalinity/conductivity/total phosphorus; however this relationship does not hold when comparisons are expanded to western lakes overall (Eilers and others 1987).

Total phosphorus, alkalinity, and conductivity were logarithmically transformed and statistically grouped using a disjoint Euclidean distance cluster analysis (SAS 1989). Ten clusters were identified and the maximum number of iterations from recomputing cluster seeds was ten. If a lake had a missing value for a variable, the missing value was estimated by the cluster seed value. This allowed all observations to be classified without complete data. Table 4A.3 presents the mean statistics for each cluster for selected chemical and physical parameters.

The major cluster groups (dilute, moderate, hard water, and saline) were mapped. The dilute group is located at high elevations (~2000 m) in the Cascades, the Bitterroots, and Grand Tetons (map 4A.2). The bedrock types in these areas are granites, andesites, and other rocks resistant to weathering. These lakes tend to be smaller and have lower ionic concentrations. The maximum conductivity in the dilute group is 55 µS.

The moderate group has average alkalinity of 420 µeq/L, and total phosphorus can range from near zero to 194 µg/L with average concentrations near 20 µg/L. Lakes in the moderate group are geographically more diverse than the dilute system (map 4A.3) and are located in lower elevations in the Cascades, Idaho batholith, and Wyoming. The average elevation is approximately 1700 meters, and average lake size is larger than the dilute lake group. These lakes are also generally located over granites and andesites, but because of their lower elevation, they probably have longer flow paths through deeper soils.

The hard water group has higher alkalinity (~3000 µeq/L) and higher average total phosphorus (~200 µg/L). These lakes are moderate-to-large in size and are at lower elevations (~560 m). The hard water lakes are generally located over sedimentary bedrock and/or alluvial surficial deposits. Some of these lakes are in areas of basalt overlain by loess (map 4A.4).

Table 4A.2— Attributes of lakes classified by Kuchler Potential Vegetation. Data are arrayed on a lake productivity basis, lowest to highest.

Kuchler Vegetation Type	N	pH	Alkalinity (ueq/L)	Conductivity (uS)	Total Phosphorus (ug/L)	Trophic State Index	Elevation (m)	Depth (m)	Area (ha)
Western Spruce/Fir	114-318	6.96	83	13	4	24.7	2128	8.0	4.0
Alpine Meadows	61-05	7.20	86	18	6	30.1	1702	10.0	4.8
Cedar/Hemlock/Pine	9-24	6.76	530	58	8	34.1	IS	IS	IS
Fir/Hemlock	57-78	7.10	135	18	8	35.8	1608	9.0	9.2
Grand fir/Douglas-fir	21-92	6.85	82	12	8	44.5	2161	6.5	3.0
Douglas-fir	127-252	7.09	910	99	29	48.6	841	10.4	11.3
Ponderosa/Shrub	29-42	7.61	342	41	25	50.6	IS	IS	IS
Western Ponderosa	30-139	7.12	297	26	34	54.8	2194	8.5	3.7
Sagebrush/Steppe	187-218	8.40	3850	438	95	69.8	361	5.4	20.6
Fescue/Wheat grass	87-97	7.37	5016	565	119	72.3	642	6.2	18.2
Wheat grass/Bluegrass	14-27	6.86	2280	258	191	79.7	523	5.2	13.4

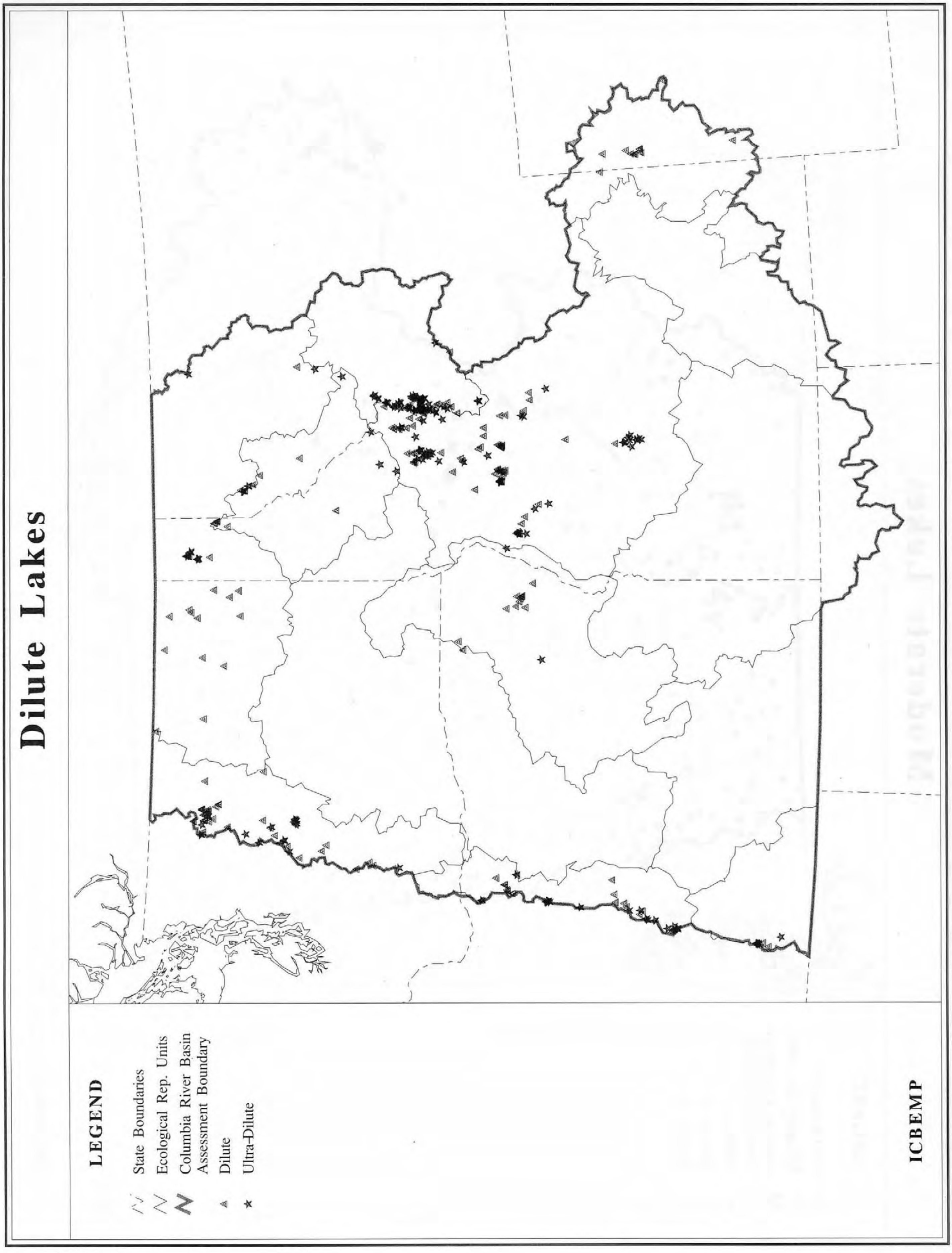
IS = Insufficient Sample

The saline group consists of lakes with high conductivity and alkalinity. The minimum alkalinity is 8000 μeq/L and the maximum alkalinity is 373,788 μeq/L (conductivity of 42,000 μS). The lakes are located in the central Washington and south central Oregon deserts (map 4A.5). The average elevation is 540 meters. The bedrock composition in these areas is typically basalt with sodic soils.

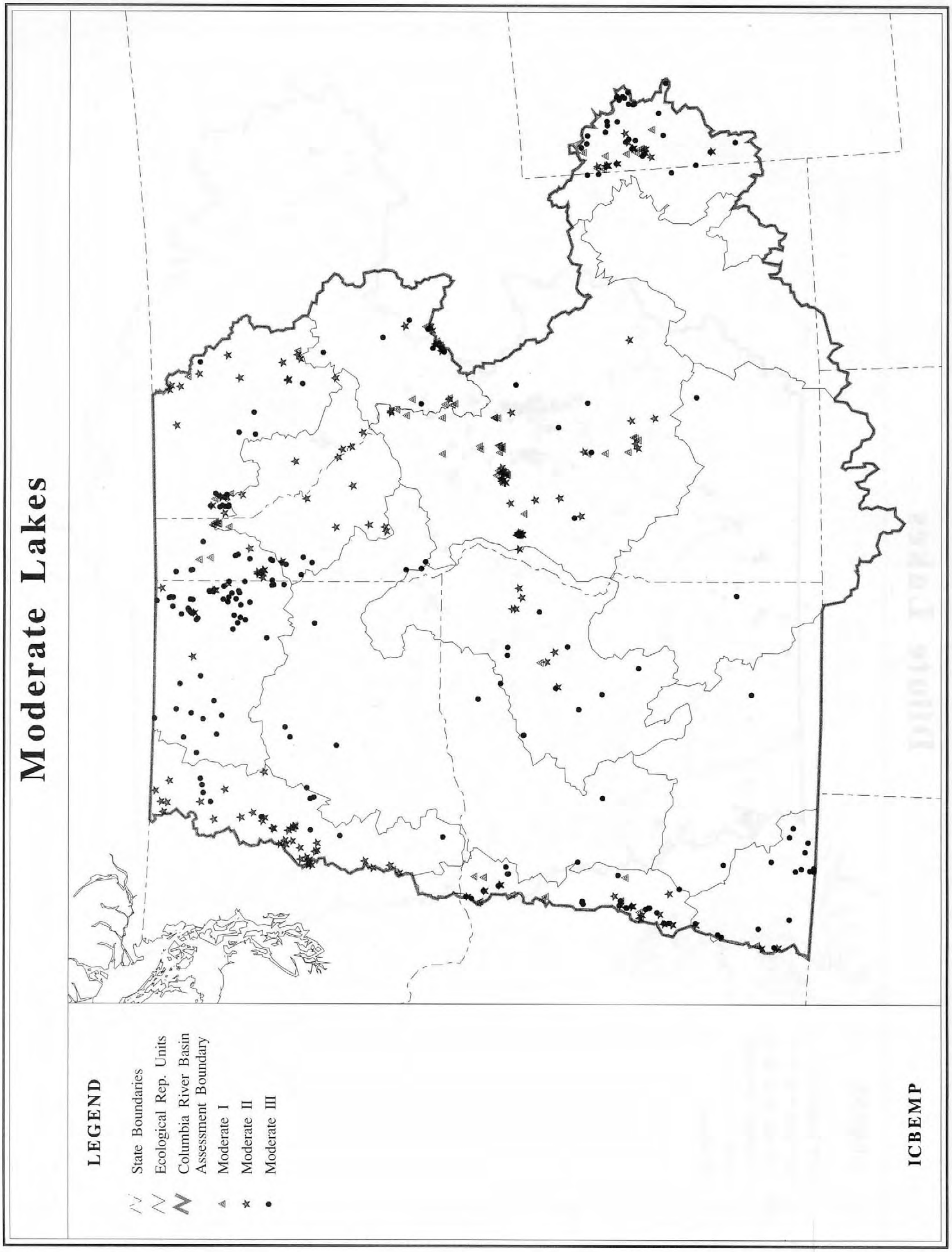
Effects of Land-use Activities on Lake Ecosystems

Evaluating the potential effects of land-use activities on lake ecosystems is difficult at a broad scale, and determinations for ERU sections rely on generalizations that may not be applicable in specific lake basins. The relative rankings of land-use effects on lakes in the study area were based largely on land use and cultural ecology described for each of the sections by McNab and Avers (1994).

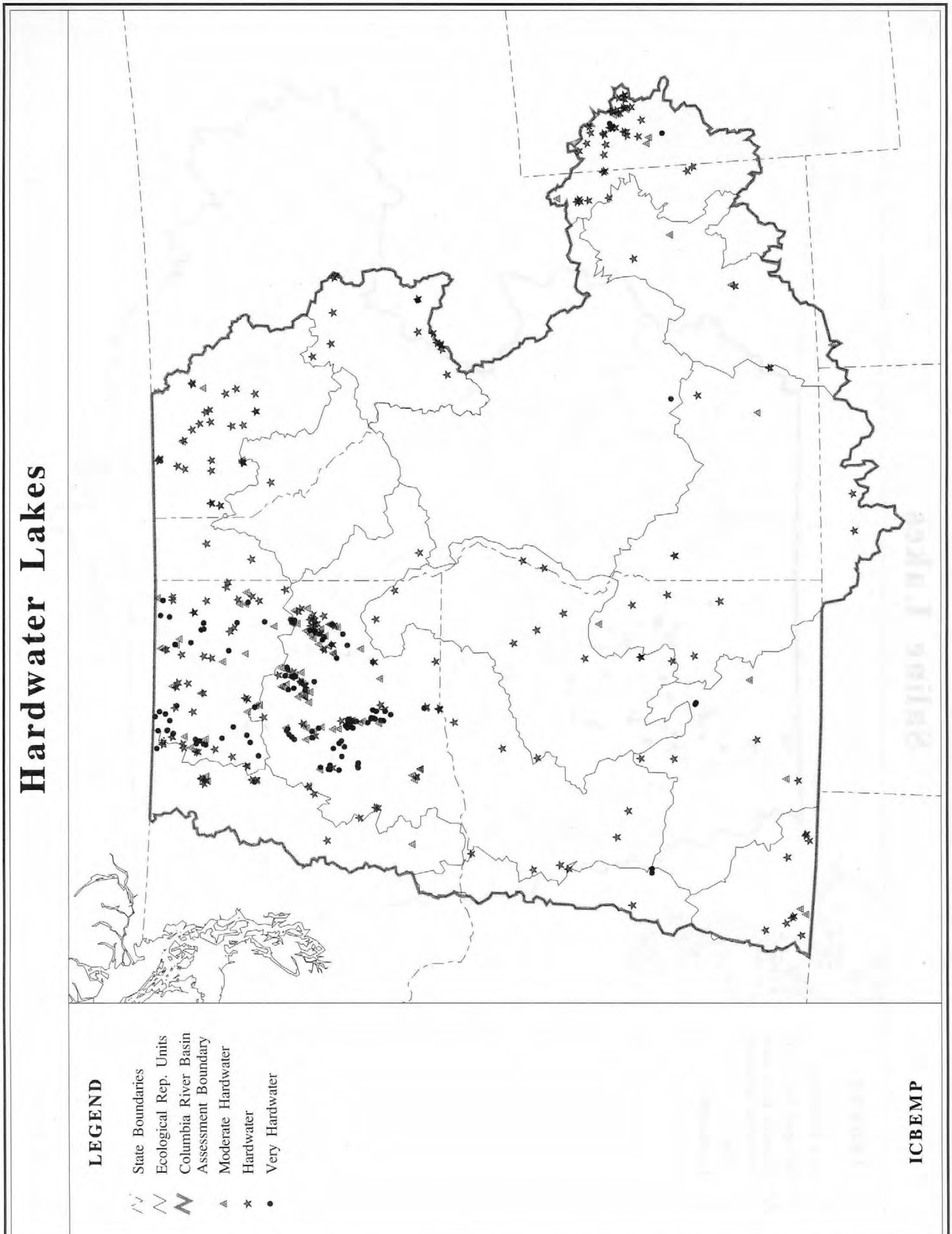
Information on the importance of timber production in the study area was supplemented by vegetation types described by Küchler (1964). The impacts from timber production will vary greatly within the sections depending on land ownership, special land use categories such as wilderness area, national parks or refuges, and vegetation zone. For example, the greatest timber production will generally occur within the montane zone and to a lesser degree the sub-alpine zone. Much of the research suggests that the damage to surface waters occurs largely because of erosion associated with logging roads. There is a low risk from timber harvest to the vast majority of lakes within forested areas because most lakes enjoy special management status and subsequent protection. A few researchers feel that timber harvest accelerates nutrient input into lakes in amounts that accelerate production in oligatrophic lakes, but this conclusion is controversial among the research community.



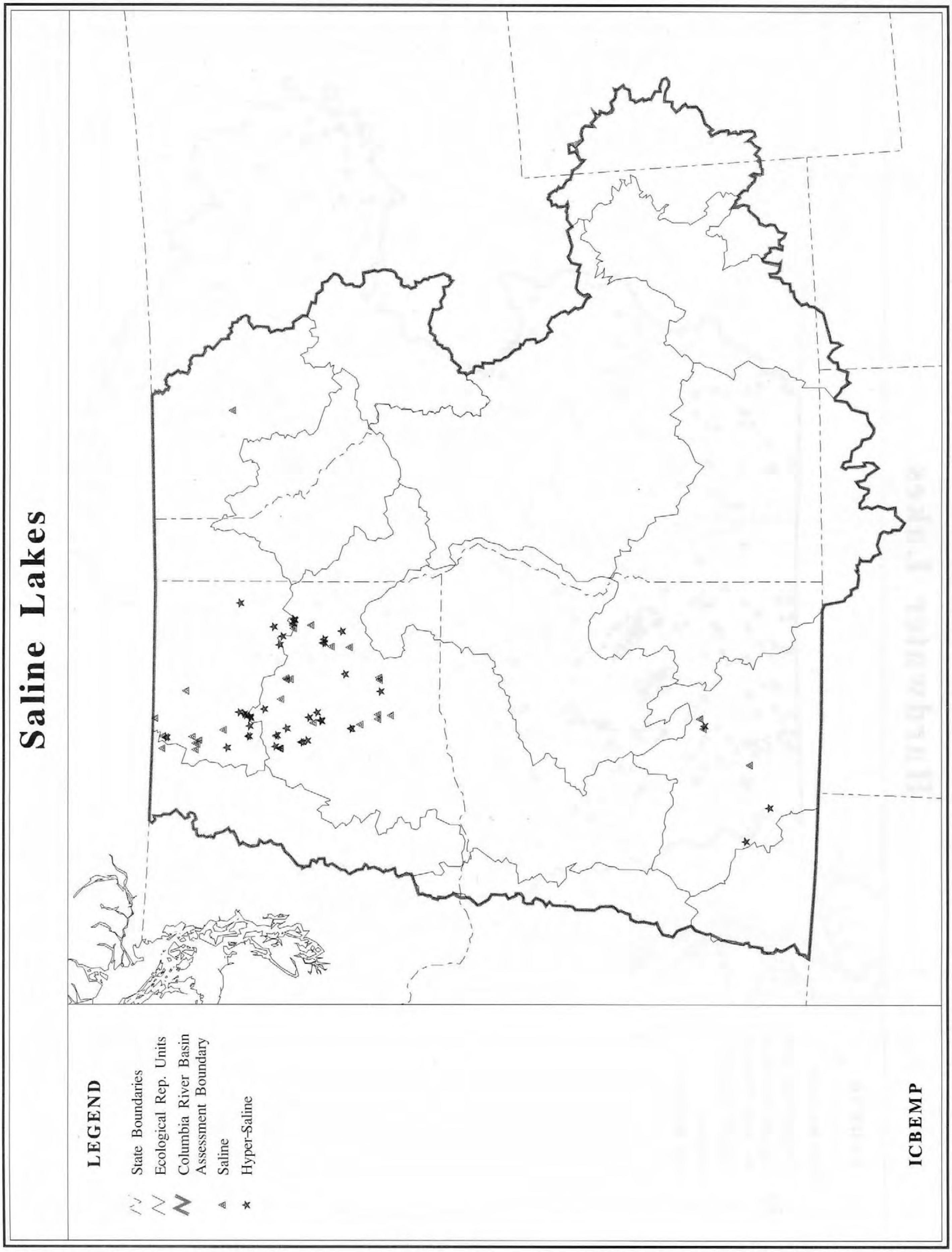
Map 4A.2—Dilute lakes.



Map 4A.3—Moderate lakes.



Map 4A.4—Hardwater lakes.



Map 4A.5—Saline lakes.

Table 4A.3—Chemistry of lake clusters, arranged from near distilled water to salt water. See maps 4A.2-4A.5 for assessment area distribution within each of the four clusters.

	Alkalinity	Conductivity	TP	Area	Elevation	Land Use Sensitivity
Dilute						Atmospheric deposition (smoke, acid rain)
Ultra Dilute	35	7	3	30	1993	
Dilute	67	12	12	15	1997	
Moderate						
Moderate I	132	17	0	10	2230	Development, Recreation(boats), Water consumption
Moderate II	350	38	4	228	1790	
Moderate III	634	70	34	207	1210	
Hardwater						
Mod-Hardwater	4516	499	469	64	507	Grazing, Mining, Development, Water consumption
Hardwater	1818	196	127	133	708	
Very Hardwater	5423	604	52	164	492	
Saline						
Saline	18247	2010	173	53	570	Irrigation, Agriculture, Grazing
Hyper Saline	57848	6526	2700	39	577	

Grazing impacts are expected to be greatest in the areas historically used as grasslands in the low elevation areas. Slight to moderate effects of grazing will occur in the higher elevation forested areas. However, because of the greater slopes in the forested areas, some localized effects of grazing may be severe.

Timber and grazing impacts are generally the most widespread impacts to the landscape. Many mining impacts remain from long-abandoned mines throughout the West. In some cases mining impacts may be restricted to relatively benign borrow pits for gravel or crushed rock. In other cases, processed ores generate considerable leachate which contaminate surface waters far from their source. Valuable ores remain scattered throughout the study area and future mining endeavors continue to look risky for lakes downstream from such activities.

Recreation pressures are increasing, causing damage to some resources, particularly beaches and littoral areas. Recreation activities include backpacking, horsepacking, use of motorized vehicles, and the road and trail development associated with greater demand to use the resources. National parks and scenic forested areas are areas where recreational impacts can be expected to increase at the fastest rate.

Development refers to urbanization/suburbanization of formerly rural areas. In some cases, the development is closely related to recreation, particularly with respect to construction of seasonal dwellings on lakeshores. Many soils in the moderate to higher elevation areas in the study area have very limited capabilities to sustain development pressures.

Lake biological communities have been seriously modified throughout the study area, and the most common modification is introduction of non-indigenous fishes. The consequences of deliberate and inadvertent introductions of fishes and invertebrates vary from changes in historic population structure and distribution to extirpation of indigenous invertebrates, amphibians, and fish (for example, bull trout and westslope cutthroat trout). In some cases, native species have been intentionally eradicated to enhance other recreational fishing for desirable "sport" species. Boating associated with angling and other recreational activities has led to the introduction of numerous exotic plants such as Eurasian watermilfoil, *Myriophyllum spicatum.*

Water transfers for potable water supply or for irrigation continue to impact lakes throughout the study area. Even where precipitation is adequate, diversions (even in wilderness areas) to more arid areas in the valleys have negative effects on the lakes and stream resources. Dozens of moderate sized lakes throughout the assessment area have their shoreline heavily influenced through regulation of their outlet streams or rivers. Regulation of the lake level for consumptive and irrigation purposes has major effects on near shore aquatic and wetland communities and spawning success of near shore spawning fishes. Additionally, interbasin water transfers promote the continued spread of non-indigenous plants and animals while often inhibiting natural migration routes of native species.

Currently, effects of atmospheric pollutants appear to be negligible for lakes in the study area. There is no evidence of anthropogenic acidification, but greater use of gas fields and coal in the region could be a long-term concern. Pesticides and various chlorinated hydrocarbons have been measured in the sediments of wilderness lakes in the region, suggesting that these compounds have been transported atmospherically. Though data do not yet exist, a potential risk to very dilute and dilute lakes could be nutrient augmentation from extensive prescribed burning (161,900 hectares/year) and wildfires in general. Spencer and Hauer (1991) found that phosphorus and nitrogen concentrations in streams increased from 5- to 60-fold over background levels within the first two days of fire, likely from leaching of ash and diffusion of smoke gases into surface waters.

The mid-elevation lakes throughout the basin have and will continue to show the greatest impacts from a growing regional population seeking to live or recreate near lakes. Lakes like Flathead, Priest, Hayden, Payette, Lake of the Woods, Odell and many others share common elements of the same theme from the past or likely trend for the future. A short story of changes to Flathead Lake, Montana will provide a glimpse of what growing lakeside development and recreational use might mean combined with the threat of wildfires and extensive prescribed burning.

Flathead Lake is one of the largest and most pristine natural lakes in the western United States and is valued for the clarity and purity of its water (Spencer and others 1991; Stanford and others 1994, 1995). Even though nearly 70 percent of the lake's watershed occurs in Glacier National Park and surrounding wilderness areas, human activities continue to threaten the quality of the lake's waters.

Cultural Eutrophication of Flathead Lake

Both point and nonpoint sources of pollution have affected water quality in Flathead Lake. The increase in primary production of algae from 1977 to 1989, a possible indicator of deteriorating water quality, was associated with high levels of phosphorous and nitrogen loading from sewage treatment plants located on tributaries to the lake (Stanford and others 1994, 1995). Water clarity, as measured by secchi depth, was negatively related to nutrient loading (Stanford and others 1995).

A program to reduce phosphorous input to the lake was instituted by the Montana Water Quality Department. The program included technological improvements in sewage treatment plants to remove phosphorous from effluents and a ban on the sale of phosphorous-containing detergents. This program has resulted in a decline in the contribution of sewage treatment plants from near 20 percent to less than 5 percent of the total phosphorous load delivered to Flathead Lake. The decline in nutrients has been accompanied by a decline in primary productivity (Stanford and others 1994, 1995).

Although nutrient inputs from sewage treatment plants have been reduced, investigators have documented high concentrations of phosphorous and nitrogen stored in the sediments of a tributary stream of Flathead Lake receiving effluent from a sewage treatment facility. They suggest that these nutrients can be released under anoxic condition and so may partially offset reductions in nutrient discharge achieved by technological improvements in the treatment plants (Stanford and others 1994).

Although input of nutrients from sources along the lake shoreline such as domestic sewage drain fields probably do not contribute extensively to the total nutrient load of the lake (Stanford and others 1994), localized shoreline sources can increase algal production along the lake margin, creating a "ring around the lake" (Stanford and others 1994, 1995).

There is concern about increased nutrient loading from nonpoint sources. In part this increase may be related to population expansion which is occurring in the Flathead watershed at a rate of 2 percent per year. Input from bulk precipitation is also a significant source of nutrients, accounting for 10-38 percent of the total phosphorous load over the time period of record (Stanford and others 1995). Smoke and dust particles in the atmosphere are thought to be responsible for the high levels of phosphorous in precipitation (Stanford and others 1995).

Exotic Species Introductions to Flathead Lake

At least seventeen species of fish and invertebrates have been introduced into Flathead Lake (Spencer and others 1991). The appearance of the oppossum shrimp, *Mysis relicta*, apparently triggered major shifts in the food web of the Flathead Lake ecosystem. In addition to *Mysis*, a complex of introduced species including kokanee salmon (*Oncorhynchus nerka*), lake trout (*Salvelinus namaycush*), and lake whitefish (*Coregonus clupeaformis*) may have been involved in the food web changes. *Mysis* prey upon zooplankton. Following the appearance of the shrimp in Flathead Lake, cladoceran zooplankton and a diaptomid copepod decreased in abundance (Beattie and Clancey 1991; Spencer and others 1991). Kokanee salmon, a valuable sport fish, also declined dramatically as *Mysis* abundance peaked. Kokanee feed extensively on zooplankton, especially cladocerans (Beattie and Clancy 1991; Reiman and Bowler 1980). *Mysis* are thought to have contributed to the decline in salmon through reduction in the salmon's food resources. Since *Mysis* and kokanee are able to coexist in some lakes (Beattie and Clancy 1991), the decline in kokanee in Flathead Lake may have been related not only to interspecific competition with *Mysis* but also to the presence of kokanee predators such as lake trout (Bowles and others 1991; Spencer and others 1991) and other planktivorous competitors such as juvenile lake whitefish (Beattie and Clancy 1991). The collapse of the kokanee population in Flathead Lake following *Mysis* introduction reduced the number of migrating bald eagles and other wildlife that congregated to feed on kokanee during their spawning migration (Spencer and others 1991).

APPENDIX 4B

A Description of the 61 Vegetative Classes Used In the Cluster Analysis.

Vegetative Class	Potential Vegetation Type	Structural Stage	Cover Type	Total Area (hectares)
1	Agricultural	Closed Herbland	Cropland / Hay / Pasture	7529100
2	Agricultural	Agricultural	Cropland / Hay / Pasture	4307200
3	Alpine	Open Low Shrub	Alpine Tundra	372400
4	Cold Forest	Stand Initiation Forest	Whitebark Pine / Alpine Larch Engelmann Spruce/Subalpine Fir Whitebark Pine Interior Douglas-fir Aspen Lodgepole Pine	1908200
5	Cold Forest	Stem Exclusion Open Canopy Forest Stem Exclusion Closed Canopy Forest	Whitebark Pine / Alpine Larch Whitebark Pine Aspen	304500
6	Cold Forest	Stem Exclusion Closed Canopy Forest	Engelmann Spruce/Subalpine Fir Interior Douglas-fir Lodgepole Pine	3253900
8	Cold Forest	Understory Reintiation Forest	Engelmann Spruce/Subalpine Fir Interior Douglas-fir Lodgepole Pine	760800
10	Cold Forest	Young Multi-strata Forest	Engelmann Spruce/Subalpine Fir Interior Douglas-fir Lodgepole Pine	1033300
13	Cold Forest	Old Multi-strata Forest	Engelmann Spruce/Subalpine Fir Lodgepole Pine	468700
14	Cold Forest	Old Single-strata Forest	Whitebark Pine	906100
17	Cold Forest	Closed Mid Shrub	Shrub or Herb/Tree Regeneration	926400
19	Cool Shrub	Stand Initiation Woodland	Juniper / Sagebrush	982500
21	Cool Shrub	Understory Reintiation Woodland	Juniper / Sagebrush	494900
23	Cool Shrub	Open Herbland Closed Herbland Open Low Shrub Closed Mid Shrub	Exotic Forbs / Annual Grass Fescue-Bunchgrass *Agropyron* Bunchgrass Fescue-Bunchgrass Chokecherry/Serviceberry/Rose Mountain Big Sagebrush	4187800
25	Dry Forest	Stand Initiation Forest	Grand Fir/White Fir Interior Douglas-fir Lodgepole Pine Interior Ponderosa Pine Sierra Nevada Mixed Conifer Pacific Ponderosa Pine	556000
26	Dry Forest	Stem Exclusion Open Canopy Forest	Interior Ponderosa Pine	1767200

Vegetative Class	Potential Vegetation Type	Structural Stage	Cover Type	Total Area (hectares)
27	Dry Forest	Stem Exclusion Closed Canopy Forest	Grand Fir/White Fir	1641200
			Interior Douglas-fir	
			Lodgepole Pine	
			Interior Ponderosa Pine	
			Pacific Ponderosa Pine	
28	Dry Forest	Understory Reintiation Forest	Grand Fir/White Fir	628100
			Interior Douglas-fir	
			Lodgepole Pine	
			Interior Ponderosa Pine	
			Pacific Ponderosa Pine	
29	Dry Forest	Young Multi-strata Forest	Grand Fir/White Fir	4028800
			Interior Douglas-fir	
			Lodgepole Pine	
			Interior Ponderosa Pine	
			Sierra Nevada Mixed Conifer	
			Pacific Ponderosa Pine	
30	Dry Forest	Old Multi-strata Forest	Grand Fir/White Fir	4114200
			Interior Douglas-fir	
			Interior Ponderosa Pine	
			Sierra Nevada Mixed Conifer	
			Pacific Ponderosa Pine	
		Old Single-strata Forest	Interior Douglas-fir	
			Interior Ponderosa Pine	
			Sierra Nevada Mixed Conifer	
			Pacific Ponderosa Pine	
32	Dry Forest	Open Herbland	Fescue-Bunchgrass	1798000
		Closed Herbland	Shrub or Herb/Tree Regeneration	
			Exotic Forbs / Annual Grass	
			Fescue-Bunchgrass	
		Open Low Shrub	Shrub or Herb/Tree Regeneration	
		Open Mid Shrub	Mountain Big Sagebrush	
		Closed Mid Shrub	Shrub or Herb/Tree Regeneration	
			Mountain Big Sagebrush	
34	Dry Grass	Stem Exclusion Woodland	Mixed Conifer Woodlands	291600
		Old Multi-strata Woodland	Mixed Conifer Woodlands	
35	Dry Grass	Open Herbland	*Agropyron* Bunchgrass	3713500
			Native Forb	
			Exotic Forbs / Annual Grass	
			Agropyron Bunchgrass	
			Exotic Forbs / Annual Grass	
			Fescue-Bunchgrass	
37	Dry Shrub	Open Herbland	*Agropyron* Bunchgrass	20937900
			Exotic Forbs / Annual Grass	
		Closed Herbland	Big Sagebrush	
			Agropyron Bunchgrass	
			Exotic Forbs / Annual Grass	

Vegetative Class	Potential Vegetation Type	Structural Stage	Cover Type	Total Area (hectares)
38	Moist Forest	Stand Initiation Forest	Pacific Silver Fir/Mountain Hemlock	14118300
			Grand Fir/White Fir	
			Mountain Hemlock	
			Engelmann Spruce/Subalpine Fir	
			Interior Douglas-fir	
			Western Larch	
			Western White Pine	
			Lodgepole Pine	
			Western Redcedar/Western Hemlock	
			Interior Ponderosa Pine	
39	Moist Forest	Stem Exclusion Closed Canopy Forest	Pacific Silver Fir/Mt. Hemlock	1766900
			Grand Fir/White Fir	
			Mt. Hemlock	
			Western Redcedar/Western Hemlock	
40	Moist Forest	Stem Exclusion Closed Canopy Forest	Engelmann Spruce/Subalpine Fir	3100500
			Interior Douglas-fir	
			Western Larch	
			Western White Pine	
			Lodgepole Pine	
			Interior Ponderosa Pine	
41	Moist Forest	Understory Reintiation Forest	Pacific Silver Fir/Mountain Hemlock	4595300
			Grand Fir/White Fir	
			Mt. Hemlock	
			Engelmann Spruce/Subalpine Fir	
			Interior Douglas-fir	
			Western Larch	
			Western White Pine	
			Lodgepole Pine	
			Western Redcedar/Western Hemlock	
			Interior Ponderosa Pine	
42	Moist Forest	Young Multi-strata Forest	Pacific Silver Fir/Mountain Hemlock	6178000
			Grand Fir/White Fir	
			Red Fir	
			Mountain Hemlock	
			Engelmann Spruce/Subalpine Fir	
			Interior Douglas-fir	
			Western Larch	
			Western White Pine	
			Lodgepole Pine	
			Western Redcedar/Western Hemlock	
			Interior Ponderosa Pine	
43	Moist Forest	Old Multi-strata Forest	Pacific Silver Fir/Mountain Hemlock	1617700
			Mt. Hemlock	
			Engelmann Spruce/Subalpine Fir	
			Lodgepole Pine	
			Western Redcedar/Western Hemlock	
		Old Single-strata Forest	Mt. Hemlock	

Vegetative Class	Potential Vegetation Type	Structural Stage	Cover Type	Total Area (hectares)
44	Moist Forest	Old Multi-strata Forest	Grand Fir/White Fir	14119800
			Red Fir	
			Interior Douglas-fir	
			Western Larch	
			Western White Pine	
			Interior Ponderosa Pine	
		Old Single-strata Forest	Grand Fir/White Fir	
			Interior Douglas-fir	
			Western Larch	
			Western White Pine	
			Interior Ponderosa Pine	
45	Moist Forest	Open Low Shrub	Shrub or Herb/Tree Regeneration	5349600
		Closed Mid Shrub	Shrub or Herb/Tree Regeneration	
		Open Tall Shrub	Shrub or Herb/Tree Regeneration	
	Riparian Shrub	Open Herbland	Herbaceous Wetlands	
			Exotic Forbs / Annual Grass	
45	Moist Forest	Closed Herbland	Herbaceous Wetlands	
		Closed Low Shrub	Shrub Wetlands	
		Open Low Shrub	Shrub Wetlands	
		Open Mid Shrub	Salt Desert Shrublands	
		Closed Tall Shrub	Shrub Wetlands	
46	Rock	Rock	Rock	228000
47	Urban	Urban	Urban	114300
48	Water	Water	Water	754800
50	Woodland	Stem Exclusion Open Canopy Forest	Limber Pine	2403800
		Stem Exclusion Closed Canopy Forest	Aspen	
		Understory Reintiation Forest	Aspen, Limber Pine	
53	Woodland	Young Multi-strata Forest	Aspen	855100
58	Woodland	Understory Reintiation Woodland	Juniper Woodlands	113400
61	Woodland	Closed Herbland	Exotic Forbs / Annual Grass	4321700
			Fescue-Bunchgrass	
		Closed Low Shrub	Mountain Mahogany	
			Mountain Big Sagebrush	
		Closed Mid Shrub	Shrub or Herb/Tree Regeneration	
			Shrub Wetlands	

APPENDIX 4C

Fish Status and Distribution Databases

The information conveyed in the section entitled "Distribution and Status of Fishes" on fish assemblages and key salmonids is available in a series of four electronic databases. The discussion below and accompanying tables supplement the information given in that section regarding derivation of the fisheries data and provide a guide to the databases. The sources for data contained therein were available from preexisting databases and from professional biologists throughout the Basin. Databases were compiled for subbasins, watersheds, and subwatersheds in the following manner.

Current-Status Database

Preexisting Data—The following databases were available at the initiation of this effort: the River Information Systems (RIS) databases for Washington (WARIS), Oregon (ORIS), Montana (MRIS) and Idaho (IRIS); the Oregon State University fish collection data base; the Coordinated Information System (CIS) database for anadromous fish maintained by the Pacific States Marine Fisheries Commission, and the Wilderness Society distributions of anadromous salmonids. The RIS and CIS databases were attributed to stream segments based on the EPA river-reach codes. The Wilderness Society database was represented by GIS coverages that were used only in development of the historical ranges. We worked with the State agencies to update RIS information for the key salmonids in Idaho and Montana; biologists in both states reviewed and updated existing information regarding known presence or absence of key species.

Data acquisition was closed in June 1995 to begin proofing and analysis. Data that were not incorporated because of delayed availability include: the Washington Department of Fisheries and Wildlife Streams and Lakes Fish Database; Oregon Department of Fish and Wildlife database for the distribution of bull trout; Oregon Department of Fish and Wildlife database for habitat and fish inventory sampling; University of Washington collection records; Oregon State University collection records; and the Plum Creek Timber Company bull trout inventories. This information is not included in the current database, but can be incorporated in future versions. Relevant portions of these data were used in a validation exercise.

Biologists' Classifications—Biologists (more than 140 participated, table 4C.1) from throughout the Basin were asked to classify the distribution and status of each of the key salmonids within subwatersheds. Possible classifications included strong or depressed in spawning and rearing areas, present but status unknown, migration corridor, absent, or unknown. Proofing of the status calls was initiated where a second opinion was possible; changes in the original classifications generally were confirmed with the biologists that made the original classifications. Corridor habitats were added in the proofing process where the oversight was obvious (for example, missing segments for anadromous or non-anadromous species that are known to move completely through entire rivers or river segments).

Merging—We merged the status classification calls with preexisting data to produce as comprehensive a database as possible. In most cases, the RIS and CIS databases were limited to presence/absence information. In some cases (MRIS all species, IRIS for bull and cutthroat trout), additional information was available for classification of status and/or life history stage. Rules for interpretation of data from IRIS and MRIS are summarized in table 4C.2. All RIS data were attributed to the corresponding subwatershed. A "present" call implies presence in any stream reach within the unit; status reflects the predominant status among all reaches. The biologists' judgements were merged with the RIS data sets using the rules in table 4C.3. In cases where species classifications differed between presence and absence, the final classification was "unknown" unless we could clearly resolve the inconsistency with recent data. Where differences were in status, the most recent information was used. The final database includes summaries of the number of key salmonids classified as strong, strong or depressed, present, and unknown, all within the potential historical range. Table 4C.4 documents the format for the current status database, CRBFISH6. In addition to the status calls, the database also includes predicted status based on the landscape data and classification trees developed by the aquatics team (see "Distribution and Status of Fishes, this chapter), and historical-range information.

Limitations of the Database—The framework of the assessment required that the species status information be summarized by subwatersheds rather than stream reaches. With limited time for development, it was necessary to classify subwatersheds directly, rather than first classifying all stream reaches and then attributing subwatersheds. This limits further analysis. Because many subwatersheds contain a combination of small, first- and second-order streams within a section of a larger river or stream, the type or quality of habitat available to a species or life stage may vary widely. In these cases, distinguishing migration corridors from spawning and rearing areas may have been confusing. This is a particular problem for westslope cutthroat trout that often use small tributaries in direct association with mainstem areas.

Extrapolating status from information limited to only a few sites was often necessary. In watersheds that vary widely in habitat type or condition, such classification may not be representative of the watershed. It was not always possible to access all or the most recent information sources, and information sources varied in quality. It is difficult to judge the status of a population if historical or basin-wide data are limited. For species like bull trout, there is often little information available to judge whether current densities are high or low. Sampling methods vary in their efficiencies making comparisons across broad scales difficult. Rare species may be missed by limited sampling, and sampling protocols rarely address that error in existing inventories (see Rieman and McIntyre 1995 for one example). As a result, classifications of a species' presence are likely to be more robust than those of absence, classification of strong and depressed may be equivocal and perhaps inconsistent from one region to another.

Despite the limitations, we believe these data provide an accurate representation of current distributions at the broad scale. We were conservative in the development of the database by cross checking sources and by encouraging biologists to rely on empirical information when making classifications. We accepted classifications of absent from RIS databases only where sampling was documented. A number of subwatersheds were classified as unknown because of conflicting information from RIS databases and biologists' designations: 23 for Yellowstone cutthroat trout, 163 for westslope cutthroat trout, 160 for redband trout, 92 for summer steelhead, 16 for ocean-type chinook salmon, 106 for stream-type chinook salmon, and 323 for bull trout. While the status calls retain an element of subjectivity and inconsistency, the professional biologists who classified watersheds generally hold the best understanding of current distributions and relative status for these species. There is simply no other way to provide a current and comprehensive assessment of fish distribution than to rely on the existing but often unpublished information that can be summarized by these people. Although some classifications may be equivocal, we anticipate no consistent bias. The overall patterns should reflect important characteristics of the species distributions and the current state of our collective knowledge. The database can be useful for interpreting broad patterns in distribution and general relationships with landscape characteristics. These data should not be used to draw detailed conclusions about extinctions where historical ranges are uncertain (for example, bull trout, redband trout, cutthroat trout), and should not be used for project-level management decisions without local review and validation.

Other Databases

Derivation of the species-assemblage database (CRBFISH5) is described in the text. It includes information on 127 species and 15 species groups at the watershed level (table 4C.5). Presence/absence for the key salmonid species at the watershed level was derived by summing over subwatersheds using the current-status database. In a similar fashion, a presence/absence database at the subbasin level (CRBFISH4, table 4C.6) was constructed by summing over watersheds within each subbasin. The subbasin-level database contains information on additional species that were not reported at smaller scales.

The measures of community integrity described in the text are available in a separate database. Values are expressed at the watershed level.

Table 4C.1— List of participants in classification of watersheds for the key salmonid status and assemblage databases, historical ranges, genetic integrity, and validation information.

Rich Torquemada	Bitterroot National Forest
Karen Kuzis	Boise Cascade Corporation
Tim Burton	Boise National Forest
Don Corley	Boise National Forest
John Augsburger	Bureau of Land Management
Dana Danzer	Bureau of Land Management
Jim Eisner	Bureau of Land Management
Scott Feldhausen	Bureau of Land Management
Lisa Healy	Bureau of Land Management
Robert House	Bureau of Land Management
Craig Johnson	Bureau of Land Management
Craig Johnson	Bureau of Land Management
Lou Jurs	Bureau of Land Management
Joe Kelly	Bureau of Land Management
Pat Koelsch	Bureau of Land Management
Mark Lacy	Bureau of Land Management
Paul McClain	Bureau of Land Management
Alan Munhall	Bureau of Land Management
Pat Olmstead	Bureau of Land Management
Brent Ralston	Bureau of Land Management
Gina Sato	Bureau of Land Management
G. Sheeter	Bureau of Land Management
Cynthia Tate	Bureau of Land Management
Todd Thompson	Bureau of Land Management
Cindy Weston	Bureau of Land Management
Jack Williams	Bureau of Land Management
David Young	Bureau of Land Management
Bruce Zoellick	Bureau of Land Management
Neal Anderson	Challis National Forest
Leon Jadlowski	Challis National Forest
Ken Meyer	Challis National Forest
Pat Murphy	Clearwater National Forest
Tom Shuhda	Colville National Forest
John Kelly	CTWSIR
S. Gerdes	Deerlodge National Forest
Brian Sanborn	Deerlodge National Forest
Tom Merritt	Deschutes National Forest
Mike Riehle	Deschutes National Forest
Tom Walker	Deschutes National Forest
Steve Robertson	Dixie National Forest
Don Hair	Flathead National Forest
Pat VanEimeren	Flathead National Forest
Darryl Gowan	Fremont National Forest
Darrell Martin	Fremont National Forest
C. Speas	Fremont National Forest
Bruce May	Gallatin National Forest
Mike Faler	Gifford-Pinchot National Forest
Len Walch	Helena National Forest
Kathy Ramsey	Humboldt National Forest
Dale Allen	Idaho Department of Fish and Game
Don Anderson	Idaho Department of Fish and Game
Bart Butterfield	Idaho Department of Fish and Game

Table 4C.1 (continued).

Tim Cochnauer	Idaho Department of Fish and Game
Chip Corsi	Idaho Department of Fish and Game
Steve Elle	Idaho Department of Fish and Game
Mark Gamblin	Idaho Department of Fish and Game
Stacy Gebhards	Idaho Department of Fish and Game
Judy Hall-Griswold	Idaho Department of Fish and Game
Jerome Hansen	Idaho Department of Fish and Game
Terry Holubetz	Idaho Department of Fish and Game
Ned Horner	Idaho Department of Fish and Game
Dwight Kilgore	Idaho Department of Fish and Game
Mark Larkin	Idaho Department of Fish and Game
Mark Leider	Idaho Department of Fish and Game
Jim Mende	Idaho Department of Fish and Game
Fred Partridge	Idaho Department of Fish and Game
Richard Scully	Idaho Department of Fish and Game
Kathy Worthen	Idaho Department of Fish and Game
Steve Yundt	Idaho Department of Fish and Game
Gwynne Chandler	Intermountain Research Station
John Gebhards	Intermountain Research Station
Lori Leatherbury	Intermountain Research Station
Danny Lee	Intermountain Research Station
Jack McIntyre	Intermountain Research Station
Deborah Myers	Intermountain Research Station
Kerry Overton	Intermountain Research Station
Mike Radko	Intermountain Research Station
Bruce Rieman	Intermountain Research Station
Russ Thurow	Intermountain Research Station
Doug Perkinson	Kootenai National Forest
Ken Furrow	Lolo National Forest
Richard Kramer	Lolo National Forest
Brian Riggers	Lolo National Forest
Rich Gritz	Malheur National Forest
John Morris	Malheur National Forest
Bill Stover	Malheur National Forest
Joel Waldo	Malheur National Forest
Janet Decker-Hess	Montana Dept. of Fish, Wildlife and Parks
Sue Ireland	Montana State University
Brad Shepard	Montana State University
Joe Moreau	Mt. Hood National Forest
Wayne Paradis	Nez Perce National Forest
Mary Ann High	Nez Perce National Forest
Kathy Moynan	Nez Perce National Forest
Scott Russell	Nez Perce National Forest
Dean Grover	Ochoco National Forest
Jim Spotts	Okanogan National Forest
Wayne Bowers	Oregon Department of Fish and Wildlife
Rich Carmichael	Oregon Department of Fish and Wildlife
Steve Preble	Oregon Department of Fish and Wildlife
Mike Gray	Oregon Department of Fish and Wildlife
Steve Marx	Oregon Department of Fish and Wildlife
Ray Perkins	Oregon Dept. of Fish and Wildlife
Amy Stuart	Oregon Dept. of Fish and Wildlife
Steve Thiesfeld	Oregon Dept. of Fish and Wildlife
Tim Unterwegner	Oregon Dept. of Fish and Wildlife
Jeff Zakel	Oregon Dept. of Fish and Wildlife
Bob Gresswell	Pacific Northwest Research Station
Dave Cross	Panhandle National Forest
John Chatel	Panhandle National Forest
Rob Daves	Panhandle National Forest
Kathy Heffner	Panhandle National Forest

Table 4C.1 (continued).

Ed Lider	Panhandle National Forest	Pete Hahn	Washington Dept. of Wildlife and Fish
Mike Owen	Panhandle National Forest	Pat Hulett	Washington Dept. of Wildlife and Fish
Dave Burns	Payette National Forest	Larry Lavoy	Washington Dept. of Wildlife and Fish
Kate Forster	Salmon National Forest	Bob Peck	Washington Dept. of Wildlife and Fish
Bruce Smith	Salmon National Forest	Mark Shuck	Washington Dept. of Wildlife and Fish
Mark Moulton	Sawtooth National Recreation Area	John Weinheimer	Washington Dept. of Wildlife and Fish
Gary Dean	Targhee National Forest	John Whalen	Washington Dept. of Wildlife and Fish
Ted Kellogg	Targhee National Forest	Philip Archibald	Wenatchee National Forest
Phil Howell	Umatilla National Forest	Judy Delavergne	Wenatchee National Forest
Mike Northrop	Umatilla National Forest	Jackie Haskins	Wenatchee National Forest
Dan Mahony	U.S. Fish and Wildlife Service	Scott Hoefer	Wenatchee National Forest
Billy Connor	U.S. Fish and Wildlife Service	Karen Lindhurst	Wenatchee National Forest
Reg Reisenbichler	U.S. Fish and Wildlife Service	Tina Mayo	Wenatchee National Forest
John Anderson	Wallowa-Whitman National Forest	Ken MacDonald	Wenatchee National Forest
Ralph Browning	Wallowa-Whitman National Forest	Dan Rife	Wenatchee National Forest
Joe Platz	Wallowa-Whitman National Forest	Amy Unthank	Wenatchee National Forest
Gretchen Sausen	Wallowa-Whitman National Forest	Rob Gipson	Wyoming Game and Fish
Wade Sims	Wallowa-Whitman National Forest	Ralph Huddelson	Wyoming Game and Fish
Greg Willmore	Wallowa-Whitman National Forest	John Kiefling	Wyoming Game and Fish
Paul Cowley	Wasatch-Cache National Forest	David Lind	Yakima Indian Nation
Larry Brown	Washington Dept. of Wildlife and Fish	Jim Matthews	Yakima Indian Nation
Joe Foster	Washington Dept. of Wildlife and Fish	Mark Teske	Yakima Indian Nation
		Bruce Watson	Yakima Indian Nation

Table 4C.2—Rules for classification of status in 6th code watersheds from the IRIS and MRIS databases.

Database	Status Call	Rules(s) used
MRIS	Present	Species was considered present if the species or a hybrid was present in the subwatershed
	Present - Migration corridor	If present and *fishuse* was designated as (A) Spawning elsewhere; (C) Migration corridor; or (F) Feeding run.
	Present - Depressed	If present and *fishabund* was not equal to (A) Abundant or (Y) Present - no further information and fishuse was not equal to (A) Spawning elsewhere; (C) Migration corridor; (F) Feeding run; or (Z) Use undetermined.
	Present - Strong	Only if non-hybrid species were present and *fishabund* was designated as (A) Abundant fishuse was not equal to (A) Spawning elsewhere; (C) Migration corridor; (F) Feeding run; or (Z) Use undetermined.
	Present - Unknown	If present and *fishabund* was designated as (Y) Present - no further information.
	Known Absent	If not present and *fishrat* was greater than or equal to 5 which represents some form of sampling has been done and the subwatershed was within the species historic range.
	Unknown	All other subwatersheds not included above and within the species historic range.
IRIS	Present	Species was considered present if it was noted as (P) Present but abundance unknown in the database.
	Present - Migration corridor	Not possible to determine for any key salmonid from this database.
	Present - Depressed	Only possible with westslope cutthroat trout which was designated as *abund* equal to (D) Depressed or (R) Remnant in the WESTCUTT database.
	Present - Strong	Again, only westslope cutthroat trout is possible where *abund* was designated as (S) Strong.
	Present - Unknown	Same as present
	Known Absent	If the species was not present and *document* was designated as (P) which denotes some form of sampling has been done.
	Unknown	All other subwatershed's not included above that are within the species historic range.

Table 4C.3— Decision matrix for merging classification of 6th code watersheds from the RIS and status classification databases. The only case where these rules were not followed was where steelhead and native rainbow/redband trout coexist. In this case the status call for steelhead was kept and native rainbow/redband trout were coded as present unknown.

RIS Calls	Status Calls					
	Present Unknown	Present Corridor	Present Strong	Present Depressed	Known Absent	Unknown
Present Unknown	Present Unknown	Present Corridor	Present Strong	Present Depressed	Unknown	Present Unknown
Present Corridor	Present Corridor	Present Corridor	Present Strong	Present Depressed	Unknown	Present Corridor
Present Strong	Present Unknown	Present Corridor	Present Strong	Present Depressed	Unknown	Present Strong
Present Depressed	Present Unknown	Present Corridor	Present Strong	Present Depressed	Unknown	Present Depressed
Known Absent	Unknown	Unknown	Unknown	Unknown	Known Absent	Known Absent
Unknown	Present Unknown	Present Corridor	Present Strong	Present Depressed	Known Absent	Unknown

Table 4C.4— Format of the database describing status of key salmonids in subwatersheds.

Variable	Field type/size[1]	Range of values	Definition
HUC4	N/8	16040201 - 18080001	subbasin identifier
HUC5	N/10	1604020102 - 1808000101	watershed identifier
HUC6	N/12	160402010201 - 180200011204	subwatershed identifier
ERU	N/2	1 - 13[2]	Ecological reporting unit
YCT_STAT	N/1	1 - 7[3]	Yellowstone cutthroat trout status
WCT_STAT	N/1	1 - 7[3]	Westslope cutthroat trout status
RBT_STAT	N/1	1 - 7[3]	Native rainbow/redband trout status
STH_STAT	N/1	1 - 7[3]	Summer steelhead status
OTC_STAT	N/1	1 - 7[3]	Ocean-type chinook salmon status
STC_STAT	N/1	1 - 7[3]	Stream-type chinook salmon status
BTR_STAT	N/1	1 - 7[3]	Bull trout status
YCT_HIST	N/1	0, 1[4]	Historical range for Yellowstone cutthroat trout
WCT_HIST	N/1	0, 1[4]	Historical range for westslope cutthroat trout
RBT_HIST	N/1	0, 1[4]	Historical range for native rainbow/redband trout
STH_HIST	N/1	0, 1[4]	Historical range for summer steelhead
OTC_HIST	N/1	0, 1[4]	Historical range for ocean-type chinook salmon
STC_HIST	N/1	0, 1[4]	Historical range for stream-type chinook salmon
BTR_HIST	N/1	0, 1[4]	Historical range for bull trout
BTR_POT	N/1	0, 1[4]	Potential range for bull trout
STH_WILD	N/1	0, 1[5]	Wild indigenous steelhead
OTC_WILD	N/1	0, 1[5]	Wild indigenous ocean-type chinook salmon
STC_WILD	N/1	0, 1[5]	Wild indigneous stream-type chinook salmon
BTR_PRB	N/8	0-1	Probability of bull trout presence from model
BTR_PRD	A/2	PA, PD, PS[6]	Predicted status of bull trout from model
BTR_PV1	N/2	0-1	Known and predicted probability of bull trout presence
BTR_PV2	A/2	A, D, S, PA, PD, PS[7]	Known and predicted bull trout status

Table 4C.4 (continued).

Variable	Field type/size[1]	Range of values	Definition
OTC_PRB	N/8	0-1	Probability of ocean-type chinook salmon presence from model
OTC_PRD	A/2	PA, PD, PS, PM[6]	Predicted status of ocean-type chinook salmon from model
OTC_PV1	N/2	0-1	Known and predicted probability of ocean-type chinook salmon presence
OTC_PV2	A/2	A, D, S, M, PA, PD, PS, M[7]	Known and predicted ocean-type chinook salmon status
RBT_PRB	N/8	0-1	Probability of native rainbow/redband trout presence from model
RBT_PRD	A/2	PA, PD, PS[6]	Predicted status of native rainbow/redband trout from model
RBT_PV1	N/8	0-1	Known and predicted probability of native rainbow/redband trout presence
RBT_PV2	A/4	A, D, S, PA, PD, PS[7]	Known and predicted native rainbow/redband trout status
STC_PRB	N/8	0-1	Probability of stream-type chinook salmon presence from model
STC_PRD	A/3	PA, PD, PS, PM[6]	Predicted status of stream-type chinook salmon from model
STC_PV1	N/8	0-1	Known and predicted probability of stream-type chinook salmon presence
STC_PV2	A/2	A, D, S, M, PA, PD, PS, M[7]	Known and predicted stream-type chinook salmon status
STH_PRB	N/8	0-1	Probability of summer steelhead presence from model
STH_PRD	A/2	PA, PD, PS, PM[6]	Predicted status of summer steelhead from model
STH_PV1	N/8	0-1	Known and predicted probability of summer steelhead presence
STH_PV2	A/2	A, D, S, M, PA, PD, PS, M[7]	Known and predicted summer steelhead status
WCT_PRB	N/8	0-1	Probability of westslope cutthroat trout presence from model
WCT_PRD	A/2	PA, PD, PS[6]	Predicted status of westslope cutthroat trout from model
WCT_PV1	N/8	0-1	Known and predicted probability of westslope cutthroat trout presence
WCT_PV2	A/2	A, D, S, PA, PD, PS[7]	Known and predicted westslope cutthroat trout status
YCT_PRB	N/8	0-1	Probability of Yellowstone cutthroat trout presence from model
YCT_PRD	A/2	PA, PD, PS[6]	Predicted status of Yellowstone cutthroat trout from model
YCT_PV1	N/8	0-1	Known and predicted probability of Yellowstone cutthroat trout presence
YCT_PV2	A/2	A, D, S, PA, PD, PS[7]	Known and predicted Yellowstone cutthroat trout status
HISTORIC	N/1	0-7	Number of key salmonids historically present
PRESENT	N/1	0-7	Number of key salmonids present within their historical range
STRONG	N/1	0-7	Number of key salmonids with strong status
STR_DEP	N/1	0-7	Number of key salmonids with strong or depressed status
STR_PRD	N/1	0-7	Number of key salmonids with strong status (known and predicted)
TOT_REM	N/1	0-6	Number of key salmonids remaining from historical within subwatershed (known and predicted)
TOT_PRD	N/1	0-6	Total number of key salmonids now present (known and predicted)

[1] - Field type/size values: N=Numeric; A=Alphanumeric

[2] - ERU range of values: 1=Northern Cascades; 2=Southern Cascades; 3=Upper Klamath; 4=Northern Great Basin; 5=Columbia Plateau; 6=Blue Mountains; 7=Northern Glaciated Mountains; 8=Lower Clark Fork; 9=Upper Clark Fork; 10=Owyhee Uplands; 11=Upper Snake; 12=Snake Headwaters; 13=Central Idaho Mountains.

[3] - Species' status range of values: 1=strong; 2=depressed; 3=known absent; 4=present but status unknown; 5=migration corridor; 6=no classification; and 7=introduced.

[4] - 0=Outside historical range, 1=within historical range.

[5] - 0=Presumed hatchery influence, 1=No known hatchery influence.

[6] - Species' predicted status range of values: PA=Predicted absent; PD=Predicted depressed; PS=Predicted strong; and PM (for anadromous species)=Predicted migratory corridor.

[7] - Species_PV2 range of values: A=Absent; D=Depressed; S=Strong; M (for anadromous species)=Migratory corridor; PA=Predicted absent; PD=Predicted depressed; PS=Predicted strong; and PM (for anadromous species)=Predicted migratory corridor.

Table 4C.5— Format of the database describing species' presence within watersheds.

Variable	Field type/size[1]	Range of values	Definition
HUC4	N/8	16040201 - 18080001	subbasin identifier
HUC5	N/10	1604020102 - 1808000101	watershed identifier
ERU	N/2	1 - 13 [2]	Ecological reporting unit
CLASS	A/1	A-P, Z	Species assemblage designation (Z indicates no information)
GRP1	N/1	0, 1[3]	Suckers
GRP2	N/1	0, 1[3]	Dace
GRP3	N/1	0, 1[3]	Sculpins
GRP4	N/1	0, 1[3]	Shiners
GRP5	N/1	0, 1[3]	Chubs
GRP6	N/1	0, 1[3]	Crappie
GRP7	N/1	0, 1[3]	bullheads
GRP8	N/1	0, 1[3]	Lampreys
GRP9	N/1	0, 1[3]	Cutthroat
GRP10	N/1	0, 1[3]	Trout
GRP11	N/1	0, 1[3]	Whitefish
GRP12	N/1	0, 1[3]	Steelhead
GRP13	N/1	0, 1[3]	Rainbow
GRP14	N/1	0, 1[3]	Chinook
GRP15	N/1	0, 1[3]	Sunfish
HD_1	N/1	0, 1[4]	Historical range for white sturgeon
HD_8	N/1	0, 1[4]	Historical range for Goose Lake sucker
HD_11	N/1	0, 1[4]	Historical range for Klamath largescale sucker
HD_13	N/1	0, 1[4]	Historical range for Warner sucker
HD_15	N/1	0, 1[4]	Historical range for shortnose sucker
HD_16	N/1	0, 1[4]	Historical range for Lost River sucker
HD_20	N/1	0, 1[4]	Historical range for Malheur sculpin
HD_21	N/1	0, 1[4]	Historical range for piute sculpin
HD_22	N/1	0, 1[4]	Historical range for slimy sculpin
HD_23	N/1	0, 1[4]	Historical range for shorthead sculpin
HD_24	N/1	0, 1[4]	Historical range for Shoshone sculpin
HD_27	N/1	0, 1[4]	Historical range for Wood River sculpin
HD_28	N/1	0, 1[4]	Historical range for margined sculpin
HD_29	N/1	0, 1[4]	Historical range for reticulate sculpin
HD_30	N/1	0, 1[4]	Historical range for pit sculpin
HD_32	N/1	0, 1[4]	Historical range for torrent sculpin
HD_33	N/1	0, 1[4]	Historical range for slender sculpin
HD_37	N/1	0, 1[4]	Historical range for Alvord chub
HD_40	N/1	0, 1[4]	Historical range for Sheldon tui chub
HD_41	N/1	0, 1[4]	Historical range for Oregon Lakes tui chub
HD_42	N/1	0, 1[4]	Historical range for Catlow tui chub
HD_43	N/1	0, 1[4]	Historical range for Hutton tui chub
HD_44	N/1	0, 1[4]	Historical range for Summer Basin tui chub
HD_48	N/1	0, 1[4]	Historical range for Goose Lake tui chub
HD_50	N/1	0, 1[4]	Historical range for leatherside chub
HD_51	N/1	0, 1[4]	Historical range for pit roach
HD_59	N/1	0, 1[4]	Historical range for foskett speckeled dace

Table 4C.5 (continued).

Variable	Field type/size[1]	Range of values	Definition
HD_64	N/1	0, 1[4]	Historical range for burbot
HD_66	N/1	0, 1[4]	Historical range for sand roller
HD_67	N/1	0, 1[4]	Historical range for River lamprey
HD_68	N/1	0, 1[4]	Historical range for Pit Klamath brook lamprey
HD_71	N/1	0, 1[4]	Historical range for Klamath River lamprey
HD_72	N/1	0, 1[4]	Historical range for Pacific lamprey
HD_73	N/1	0, 1[4]	Historical range for Goose Lake lamprey
HD_74	N/1	0, 1[4]	Historical range for Yellowstone cutthroat trout
HD_75	N/1	0, 1[4]	Historical range for Coastal cutthroat trout
HD_76	N/1	0, 1[4]	Historical range for Lahontan cutthroat trout
HD_77	N/1	0, 1[4]	Historical range for westslope cutthroat trout
HD_79	N/1	0, 1[4]	Historical range for chum salmon
HD_80	N/1	0, 1[4]	Historical range for coho salmon
HD_81	N/1	0, 1[4]	Historical range for native rainbow/redband trout
HD_82	N/1	0, 1[4]	Historical range for summer steelhead
HD_86	N/1	0, 1[4]	Historical range for sockeye (kokanee) salmon
HD_87	N/1	0, 1[4]	Historical range for ocean-type chinook salmon
HD_88	N/1	0, 1[4]	Historical range for stream-type chinook salmon
HD_89	N/1	0, 1[4]	Historical range for pygmy whitefish
HD_91	N/1	0, 1[4]	Historical range for bull trout
SPC_1	N/1	0, 13	White sturgeon
SPC_2	N/1	0, 13	Utah sucker
SPC_3	N/1	0, 13	Longnose sucker
SPC_4	N/1	0, 1[3]	Bridgelip sucker
SPC_5	N/1	0, 1[3]	Bluehead sucker
SPC_6	N/1	0, 1[3]	Largescale sucker
SPC_7	N/1	0, 1[3]	Sacramento sucker
SPC_8	N/1	0, 1[3]	Goose Lake sucker
SPC_9	N/1	0, 1[3]	Mountain sucker
SPC_10	N/1	0, 1[3]	Klamath smallscale sucker
SPC_11	N/1	0, 1[3]	Klamath largescale sucker
SPC_12	N/1	0, 1[3]	Tahoe sucker
SPC_13	N/1	0, 1[3]	Warner sucker
SPC_14	N/1	0, 1[3]	Sucker, generic
SPC_15	N/1	0, 1[3]	Shortnose sucker
SPC_16	N/1	0, 1[3]	Lost River sucker
SPC_17	N/1	0, 1[3]	Coastrange sucker
SPC_18	N/1	0, 1[3]	Prickly sculpin
SPC_19	N/1	0, 1[3]	Mottled sculpin
SPC_20	N/1	0, 1[3]	Malheur sculpin
SPC_21	N/1	0, 1[3]	Piute sculpin
SPC_22	N/1	0, 1[3]	Slimy sculpin
SPC_23	N/1	0, 1[3]	Shorthead sculpin
SPC_24	N/1	0, 1[3]	Shoshone sculpin
SPC_25	N/1	0, 1[3]	Riffle sculpin
SPC_26	N/1	0, 1[3]	Marbled sculpin
SPC_27	N/1	0, 1[3]	Wood River sculpin
SPC_28	N/1	0, 1[3]	Margined sculpin

Table 4C.5 (continued).

Variable	Field type/size[1]	Range of values	Definition
SPC_29	N/1	0, 1[3]	Reticulate sculpin
SPC_30	N/1	0, 1[3]	Pit sculpin
SPC_31	N/1	0, 1[3]	Klamath Lake sculpin
SPC_32	N/1	0, 1[3]	Torrent sculpin
SPC_33	N/1	0, 1[3]	Slender sculpin
SPC_34	N/1	0, 1[3]	Sculpin, generic
SPC_35	N/1	0, 1[3]	Pacific staghorn sculpin
SPC_36	N/1	0, 1[3]	Chiselmouth
SPC_37	N/1	0, 1[3]	Alvord chub
SPC_38	N/1	0, 1[3]	Utah chub
SPC_39	N/1	0, 1[3]	Tui chub
SPC_40	N/1	0, 1[3]	Sheldon tui chub
SPC_41	N/1	0, 1[3]	Oregon Lakes tui chub
SPC_42	N/1	0, 1[3]	Catlow tui chub
SPC_43	N/1	0, 1[3]	Hutton tui chub
SPC_44	N/1	0, 1[3]	Summer Basin tui chub
SPC_45	N/1	0, 1[3]	Warner Basin tui chub
SPC_46	N/1	0, 1[3]	XL Spring tui chub
SPC_47	N/1	0, 1[3]	Goose Lake tui chub
SPC_48	N/1	0, 1[3]	Borax Lake chub
SPC_49	N/1	0, 1[3]	Blue chub
SPC_50	N/1	0, 1[3]	Leatherside chub
SPC_51	N/1	0, 1[3]	Pit roach
SPC_52	N/1	0, 1[3]	Peamouth
SPC_53	N/1	0, 1[3]	Northern squawfish
SPC_54	N/1	0, 1[3]	Umpqua squawfish
SPC_55	N/1	0, 1[3]	Longnose dace
SPC_56	N/1	0, 1[3]	Leopard dace
SPC_57	N/1	0, 1[3]	Speckeled dace
SPC_58	N/1	0, 1[3]	Klamath speckled dace
SPC_59	N/1	0, 1[3]	Foskett speckled dace
SPC_60	N/1	0, 1[3]	Dace, generic
SPC_61	N/1	0, 1[3]	Redside shiner
SPC_62	N/1	0, 1[3]	Lahontan redside shiner
SPC_63	N/1	0, 1[3]	Shiner perch
SPC_64	N/1	0, 1[3]	Burbot
SPC_65	N/1	0, 1[3]	Three spine stickelback
SPC_66	N/1	0, 1[3]	Sand roller
SPC_67	N/1	0, 1[3]	River lamprey
SPC_68	N/1	0, 1[3]	Pit Klamath brook lamprey
SPC_69	N/1	0, 1[3]	Miller Lake Lamprey
SPC_70	N/1	0, 1[3]	Western brook lamprey
SPC_71	N/1	0, 1[3]	Klamath river lamprey
SPC_72	N/1	0, 1[3]	Pacific lamprey
SPC_73	N/1	0, 1[3]	Goose Lake lamprey
SPC_74	N/1	0, 1[3]	Yellowstone cutthroat trout
SPC_75	N/1	0, 1[3]	Coastal cutthroat trout
SPC_76	N/1	0, 1[3]	Lahontan cutthroat trout

Table 4C.5 (continued).

Variable	Field type/size[1]	Range of values	Definition
SPC_77	N/1	0, 1[3]	Westslope cutthroat trout
SPC_78	N/1	0, 1[3]	Cutthroat trout, generic
SPC_79	N/1	0, 1[3]	Chum salmon
SPC_80	N/1	0, 1[3]	Coho salmon
SPC_81	N/1	0, 1[3]	Interior redband trout
SPC_82	N/1	0, 1[3]	Summer steelhead
SPC_83	N/1	0, 1[3]	Winter steelhead
SPC_84	N/1	0, 1[3]	Catlow Valley redband trout
SPC_85	N/1	0, 1[3]	Warner Valley redband trout
SPC_86	N/1	0, 1[3]	Sockeye (kokanee) salmon
SPC_87	N/1	0, 1[3]	Ocean-type chinook salmon
SPC_88	N/1	0, 1[3]	stream-type chinook salmon
SPC_89	N/1	0, 1[3]	Pygmy whitefish
SPC_90	N/1	0, 1[3]	Mountain whitefish
SPC_91	N/1	0, 1[3]	Bull trout
SPC_92	N/1	0, 1[3]	White sucker
SPC_93	N/1	0, 1[3]	Green sunfish
SPC_94	N/1	0, 1[3]	Pumpkinseed
SPC_95	N/1	0, 1[3]	Warmouth
SPC_96	N/1	0, 1[3]	Bluegill
SPC_97	N/1	0, 1[3]	Smallmouth bass
SPC_98	N/1	0, 1[3]	Largemouth bass
SPC_99	N/1	0, 1[3]	White crappie
SPC_100	N/1	0, 1[3]	Black crappie
SPC_101	N/1	0, 1[3]	American shad
SPC_102	N/1	0, 1[3]	Goldfish
SPC_103	N/1	0, 1[3]	Finescale dace
SPC_104	N/1	0, 1[3]	Carp
SPC_105	N/1	0, 1[3]	Spottail shiner
SPC_106	N/1	0, 1[3]	Fathead minnow
SPC_107	N/1	0, 1[3]	Tench
SPC_108	N/1	0, 1[3]	Northern pike
SPC_109	N/1	0, 1[3]	Black bullhead
SPC_110	N/1	0, 1[3]	Yellow bullhead
SPC_111	N/1	0, 1[3]	Brown bullhead
SPC_112	N/1	0, 1[3]	Channel catfish
SPC_113	N/1	0, 1[3]	Tadpole madtom
SPC_114	N/1	0, 1[3]	Flathead catfish
SPC_115	N/1	0, 1[3]	Yellow perch
SPC_116	N/1	0, 1[3]	Walleye
SPC_117	N/1	0, 1[3]	Variable platyfish
SPC_118	N/1	0, 1[3]	Lake whitefish
SPC_119	N/1	0, 1[3]	Golden trout
SPC_120	N/1	0, 1[3]	Rainbow trout
SPC_121	N/1	0, 1[3]	Kamloops trout
SPC_122	N/1	0, 1[3]	Atlantic salmon
SPC_123	N/1	0, 1[3]	Brown trout

Table 4C.5 (continued).

Variable	Field type/size[1]	Range of values	Definition
SPC_124	N/1	0, 1[3]	Sunapee char
SPC_125	N/1	0, 1[3]	Brook trout
SPC_126	N/1	0, 1[3]	Lake trout
SPC_127	N/1	0, 1[3]	Arctic grayling
TOT_SPC	N/2	-1, 0-465	Total number of species present
TOT_GRP	N/2	-1, 0-145	Total number of groups present
TOT_NAT	N/2	-1, 0-295	Total number of native species present
TOT_EXOT	N/2	-1, 0-195	Total number of exotic species present
PCT_NAT	N/8	0-1	Fraction of total comprised of native species
PCT_EXOT	N/8	0-1	Fraction of total comprised of exotic species
FEDLIST	N/2	0-3	Number of Federally listed species
SENLIST	N/2	0-10	Number of designated sensitive species

1 - Field type/size values: N=Numeric; A=Alphanumeric

2 - ERU range of values: 1=Northern Cascades; 2=Southern Cascades; 3=Upper Klamath; 4=Northern Great Basin; 5=Columbia Plateau; 6=Blue Mountains; 7=Northern Glaciated Mountains; 8=Lower Clark Fork; 9=Upper Clark Fork; 10=Owyhee Uplands; 11=Upper Snake; 12=Snake Headwaters; 13=Central Idaho Mountains.

3 - 0=Not reported as present, 1=reported as present.

4 - 0=Outside historical range, 1=within historical range.

5 - A value of -1 indicates insufficient information for estimation.

Table 4C.6— Format of the database describing species' presence within subbasins.

Variable	Field type/size[1]	Range of values	Definition
ERU	N/2	1 - 13[2]	Ecological reporting unit
HUC4	N/8	16040201 - 18080001	subbasin identifier
HD_1	N/1	0, 1[3]	Historical range for white sturgeon
HD_8	N/1	0, 1[3]	Historical range for Goose Lake sucker
HD_11	N/1	0, 1[3]	Historical range for Klamath largescale sucker
HD_13	N/1	0, 1[3]	Historical range for Warner sucker
HD_15	N/1	0, 1[3]	Historical range for shortnose sucker
HD_16	N/1	0, 1[3]	Historical range for Lost River sucker
HD_20	N/1	0, 1[3]	Historical range for Malheur sculpin
HD_21	N/1	0, 1[3]	Historical range for piute sculpin
HD_22	N/1	0, 1[3]	Historical range for slimy sculpin
HD_23	N/1	0, 1[3]	Historical range for shorthead sculpin
HD_24	N/1	0, 1[3]	Historical range for Shoshone sculpin
HD_27	N/1	0, 1[3]	Historical range for Wood River sculpin
HD_28	N/1	0, 1[3]	Historical range for margined sculpin
HD_29	N/1	0, 1[3]	Historical range for reticulate sculpin
HD_30	N/1	0, 1[3]	Historical range for pit sculpin
HD_32	N/1	0, 1[3]	Historical range for torrent sculpin
HD_33	N/1	0, 1[3]	Historical range for slender sculpin
HD_37	N/1	0, 1[3]	Historical range for Alvord chub
HD_40	N/1	0, 1[3]	Historical range for Sheldon tui chub
HD_41	N/1	0, 1[3]	Historical range for Oregon Lakes tui chub
HD_42	N/1	0, 1[3]	Historical range for Catlow tui chub
HD_43	N/1	0, 1[3]	Historical range for Hutton tui chub
HD_44	N/1	0, 1[3]	Historical range for Summer Basin tui chub
HD_48	N/1	0, 1[3]	Historical range for Goose Lake tui chub
HD_50	N/1	0, 1[3]	Historical range for leatherside chub
HD_51	N/1	0, 1[3]	Historical range for pit roach
HD_59	N/1	0, 1[3]	Historical range for foskett speckeled dace
HD_64	N/1	0, 1[3]	Historical range for burbot
HD_66	N/1	0, 1[3]	Historical range for sand roller
HD_67	N/1	0, 1[3]	Historical range for River lamprey
HD_68	N/1	0, 1[3]	Historical range for Pit Klamath brook lamprey
HD_71	N/1	0, 1[3]	Historical range for Klamath River lamprey
HD_72	N/1	0, 1[3]	Historical range for Pacific lamprey
HD_73	N/1	0, 1[3]	Historical range for Goose Lake lamprey
HD_74	N/1	0, 1[3]	Historical range for Yellowstone cutthroat trout

Table 4C.6 (continued).

Variable	Field type/size[1]	Range of values	Definition
HD_75	N/1	0, 1[3]	Historical range for Coastal cutthroat trout
HD_76	N/1	0, 1[3]	Historical range for Lahontan cutthroat trout
HD_77	N/1	0, 1[3]	Historical range for westslope cutthroat trout
HD_79	N/1	0, 1[3]	Historical range for chum salmon
HD_80	N/1	0, 1[3]	Historical range for coho salmon
HD_81	N/1	0, 1[3]	Historical range for native rainbow/redband trout
HD_82	N/1	0, 1[3]	Historical range for summer steelhead
HD_86	N/1	0, 1[3]	Historical range for sockeye (kokanee) salmon
HD_87	N/1	0, 1[3]	Historical range for ocean-type chinook salmon
HD_88	N/1	0, 1[3]	Historical range for stream-type chinook salmon
HD_89	N/1	0, 1[3]	Historical range for pygmy whitefish
HD_91	N/1	0, 1[3]	Historical range for bull trout
SPC_1	N/1	0, 1[4]	White sturgeon
SPC_2	N/1	0, 1[4]	Utah sucker
SPC_3	N/1	0, 1[4]	Longnose sucker
SPC_4	N/1	0, 1[4]	Bridgelip sucker
SPC_5	N/1	0, 1[4]	Bluehead sucker
SPC_6	N/1	0, 1[4]	Largescale sucker
SPC_7	N/1	0, 1[4]	Sacramento sucker
SPC_8	N/1	0, 1[4]	Goose Lake sucker
SPC_9	N/1	0, 1[4]	Mountain sucker
SPC_10	N/1	0, 1[4]	Klamath smallscale sucker
SPC_11	N/1	0, 1[4]	Klamath largescale sucker
SPC_12	N/1	0, 1[4]	Tahoe sucker
SPC_13	N/1	0, 1[4]	Warner sucker
SPC_14	N/1	0, 1[4]	Sucker, generic
SPC_15	N/1	0, 1[4]	Shortnose sucker
SPC_16	N/1	0, 1[4]	Lost River sucker
SPC_17	N/1	0, 1[4]	Coastrange sucker
SPC_18	N/1	0, 1[4]	Prickly sculpin
SPC_19	N/1	0, 1[4]	Mottled sculpin
SPC_20	N/1	0, 1[4]	Malheur sculpin
SPC_21	N/1	0, 1[4]	Piute sculpin
SPC_22	N/1	0, 1[4]	Slimy sculpin
SPC_23	N/1	0, 1[4]	Shorthead sculpin
SPC_24	N/1	0, 1[4]	Shoshone sculpin
SPC_25	N/1	0, 1[4]	Riffle sculpin
SPC_26	N/1	0, 1[4]	Marbled sculpin
SPC_27	N/1	0, 1[4]	Wood River sculpin

Table 4C.6 (continued).

Variable	Field type/size[1]	Range of values	Definition
SPC_28	N/1	0, 1[4]	Margined sculpin
SPC_29	N/1	0, 1[4]	Reticulate sculpin
SPC_30	N/1	0, 1[4]	Pit sculpin
SPC_31	N/1	0, 1[4]	Klamath Lake sculpin
SPC_32	N/1	0, 1[4]	Torrent sculpin
SPC_33	N/1	0, 1[4]	Slender sculpin
SPC_34	N/1	0, 1[4]	Sculpin, generic
SPC_35	N/1	0, 1[4]	Pacific staghorn sculpin
SPC_36	N/1	0, 1[4]	Chiselmouth
SPC_37	N/1	0, 1[4]	Alvord chub
SPC_38	N/1	0, 1[4]	Utah chub
SPC_39	N/1	0, 1[4]	Tui chub
SPC_40	N/1	0, 1[4]	Sheldon tui chub
SPC_41	N/1	0, 1[4]	Oregon Lakes tui chub
SPC_42	N/1	0, 1[4]	Catlow tui chub
SPC_43	N/1	0, 1[4]	Hutton tui chub
SPC_44	N/1	0, 1[4]	Summer Basin tui chub
SPC_45	N/1	0, 1[4]	Warner Basin tui chub
SPC_46	N/1	0, 1[4]	XL Spring tui chub
SPC_47	N/1	0, 1[4]	Goose Lake tui chub
SPC_48	N/1	0, 1[4]	Borax Lake chub
SPC_49	N/1	0, 1[4]	Blue chub
SPC_50	N/1	0, 1[4]	Leatherside chub
SPC_51	N/1	0, 1[4]	Pit roach
SPC_52	N/1	0, 1[4]	Peamouth
SPC_53	N/1	0, 1[4]	Northern squawfish
SPC_54	N/1	0, 1[4]	Umpqua squawfish
SPC_55	N/1	0, 1[4]	Longnose dace
SPC_56	N/1	0, 1[4]	Leopard dace
SPC_57	N/1	0, 1[4]	Speckeled dace
SPC_58	N/1	0, 1[4]	Klamath speckled dace
SPC_59	N/1	0, 1[4]	Foskett speckled dace
SPC_60	N/1	0, 1[4]	Dace, generic
SPC_61	N/1	0, 1[4]	Redside shiner
SPC_62	N/1	0, 1[4]	Lahontan redside shiner
SPC_63	N/1	0, 1[4]	Shiner perch
SPC_64	N/1	0, 1[4]	Burbot
SPC_65	N/1	0, 1[4]	Three spine stickelback
SPC_66	N/1	0, 1[4]	Sand roller

Table 4C.6 (continued).

Variable	Field type/size[1]	Range of values	Definition
SPC_67	N/1	0, 1[4]	River lamprey
SPC_68	N/1	0, 1[4]	Pit Klamath brook lamprey
SPC_69	N/1	0, 1[4]	Miller Lake Lamprey
SPC_70	N/1	0, 1[4]	Western brook lamprey
SPC_71	N/1	0, 1[4]	Klamath river lamprey
SPC_72	N/1	0, 1[4]	Pacific lamprey
SPC_73	N/1	0, 1[4]	Goose Lake lamprey
SPC_74	N/1	0, 1[4]	Yellowstone cutthroat trout
SPC_75	N/1	0, 1[4]	Coastal cutthroat trout
SPC_76	N/1	0, 1[4]	Lahontan cutthroat trout
SPC_77	N/1	0, 1[4]	Westslope cutthroat trout
SPC_78	N/1	0, 1[4]	Cutthroat trout, generic
SPC_79	N/1	0, 1[4]	Chum salmon
SPC_80	N/1	0, 1[4]	Coho salmon
SPC_81	N/1	0, 1[4]	Interior redband trout
SPC_82	N/1	0, 1[4]	Summer steelhead
SPC_83	N/1	0, 1[4]	Winter steelhead
SPC_84	N/1	0, 1[4]	Catlow Valley redband trout
SPC_85	N/1	0, 1[4]	Warner Valley redband trout
SPC_86	N/1	0, 1[4]	Sockeye (kokanee) salmon
SPC_87	N/1	0, 1[4]	Ocean-type chinook salmon
SPC_88	N/1	0, 1[4]	stream-type chinook salmon
SPC_89	N/1	0, 1[4]	Pygmy whitefish
SPC_90	N/1	0, 1[4]	Mountain whitefish
SPC_91	N/1	0, 1[4]	Bull trout
SPC_92	N/1	0, 1[4]	White sucker
SPC_93	N/1	0, 1[4]	Green sunfish
SPC_94	N/1	0, 1[4]	Pumpkinseed
SPC_95	N/1	0, 1[4]	Warmouth
SPC_96	N/1	0, 1[4]	Bluegill
SPC_97	N/1	0, 1[4]	Smallmouth bass
SPC_98	N/1	0, 1[4]	Largemouth bass
SPC_99	N/1	0, 1[4]	White crappie
SPC_100	N/1	0, 1[4]	Black crappie
SPC_101	N/1	0, 1[4]	American shad
SPC_102	N/1	0, 1[4]	Goldfish
SPC_103	N/1	0, 1[4]	Finescale dace
SPC_104	N/1	0, 1[4]	Carp
SPC_105	N/1	0, 1[4]	Spottail shiner

Table 4C.6 (continued).

Variable	Field type/size[1]	Range of values	Definition
SPC_106	N/1	0, 1[4]	Fathead minnow
SPC_107	N/1	0, 1[4]	Tench
SPC_108	N/1	0, 1[4]	Northern pike
SPC_109	N/1	0, 1[4]	Black bullhead
SPC_110	N/1	0, 1[4]	Yellow bullhead
SPC_111	N/1	0, 1[4]	Brown bullhead
SPC_112	N/1	0, 1[4]	Channel catfish
SPC_113	N/1	0, 1[4]	Tadpole madtom
SPC_114	N/1	0, 1[4]	Flathead catfish
SPC_115	N/1	0, 1[4]	Yellow perch
SPC_116	N/1	0, 1[4]	Walleye
SPC_117	N/1	0, 1[4]	Variable platyfish
SPC_118	N/1	0, 1[4]	Lake whitefish
SPC_119	N/1	0, 1[4]	Golden trout
SPC_120	N/1	0, 1[4]	Rainbow trout
SPC_121	N/1	0, 1[4]	Kamloops trout
SPC_122	N/1	0, 1[4]	Atlantic salmon
SPC_123	N/1	0, 1[4]	Brown trout
SPC_124	N/1	0, 1[4]	Sunapee char
SPC_125	N/1	0, 1[4]	Brook trout
SPC_126	N/1	0, 1[4]	Lake trout
SPC_127	N/1	0, 1[4]	Arctic grayling
SPC_128	N/1	0, 1[4]	Shortnose sucker
SPC_129	N/1	0, 1[4]	Rock Bass
SPC_130	N/1	0, 1[4]	Sacramento perch
SPC_131	N/1	0, 1[4]	Redear sunfish
SPC_132	N/1	0, 1[4]	Tambaqui
SPC_133	N/1	0, 1[4]	Convict cichlid
SPC_134	N/1	0, 1[4]	Tilapia
SPC_135	N/1	0, 1[4]	Oriental weatherfish
SPC_136	N/1	0, 1[4]	Loach
SPC_137	N/1	0, 1[4]	Grass carp
SPC_138	N/1	0, 1[4]	Tiger barb
SPC_139	N/1	0, 1[4]	Grass pickerel
SPC_140	N/1	0, 1[4]	Gar
SPC_141	N/1	0, 1[4]	Striped bass
SPC_142	N/1	0, 1[4]	Rainbow smelt
SPC_143	N/1	0, 1[4]	Saddleback gunnel
SPC_144	N/1	0, 1[4]	Mosquitofish

Table 4C.6 (continued).

Variable	Field type/size[1]	Range of values	Definition
SPC_145	N/1	0, 1[4]	Green swordtail
SPC_146	N/1	0, 1[4]	Shortfin molly
SPC_147	N/1	0, 1[4]	Guppy
SPC_148	N/1	0, 1[4]	Artic char
TOT_EXOT	N/2	-1, 0-23[5]	Number of exotic species present
TOT_NAT	N/2	-1, 0-37[5]	Number of native species present
TOT_SPC	N/2	-1, 0-58[5]	Total number of species present
PCT_EXOT	N/8	0-1	Percent of species present made up of exotics
PCT_NAT	N/8	0-1	Percent of species present made up os native species
SENLIST	N/2	0-11	Number of sensitive species present
FEDLIST	N/2	0-3	Number of federally listed species present

[1] - Field type/size values: N=Numeric; A=Alphanumeric

[2] - ERU range of values: 1=Northern Cascades; 2=Southern Cascades; 3=Upper Klamath; 4=Northern Great Basin; 5=Columbia Plateau; 6=Blue Mountains; 7=Northern Glaciated Mountains; 8=Lower Clark Fork; 9=Upper Clark Fork; 10=Owyhee Uplands; 11=Upper Snake; 12=Snake Headwaters; 13=Central Idaho Mountains.

[3] - 0=Outside historical range, 1=within historical range.

[4] - 0=Not reported as present, 1=reported as present.

[5] - A value of -1 indicates insufficient information to judge.

Table 4C.7— Format of the database of aquatic integrity measures for watersheds.

Variable	Field type/size[1]	Range of values	Definition
ERU	N/2	1 - 13 [2]	Ecological reporting unit
HUC5	N/10	1604020102 - 1808000101	watershed identifier
CLASS	A/1	A-P, Z	Species assemblage designation (Z indicates no information)
FEDLIST	N/2	0-3	Number of Federally listed species
SENLIST	N/2	0-10	Number of designated sensitive species
WILD_AND	N/1	0-2	Number of genetically pure anadromous populations
ENDEMIC	N/1	0-5	Number of narrow-endemic species present
STRIDX	N/8	0-1	Relative index of strong populations of key salmonids
INT_EVN	N/8	0-1	Realtive index of the ratio of native species diversity to total species diversity, multiplied by native species eveness
RICH4	N/2	0-1	Relative index of species richness within parent subbasin
INTEG5	N/8	0-1	Composite index of fish community integrity = average of STRIDX, INT_EVN, and RICH4

[1] - Field type/size values: N=Numeric; A=Alphanumeric

[2] - ERU range of values: 1=Northern Cascades; 2=Southern Cascades; 3=Upper Klamath; 4=Northern Great Basin; 5=Columbia Plateau; 6=Blue Mountains; 7=Northern Glaciated Mountains; 8=Lower Clark Fork; 9=Upper Clark Fork; 10=Owyhee Uplands; 11=Upper Snake; 12=Snake Headwaters; 13=Central Idaho Mountains.

Table 4C.8— Format of the database of vegetation and management clusters for 6th-code watersheds.

Variable	Field type/size[1]	Range of values	Definition
ERU	N/2	1 - 13 [2]	Ecological reporting unit
HUC6	N/12	160102020101 - 180200011402	6th code watershed number
MNGCLUS	A/2	BR, FG, FH, FM, FW, NP, PA, PF, PR, TL[3]	Management cluster designation
VEGCLUS	A/1	A - L[4]	Vegetation cluster designation

[1] - Field type/size values: N=Numeric; A=Alphanumeric

[2] - ERU range of values: 1=Northern Cascades; 2=Southern Cascades; 3=Upper Klamath; 4=Northern Great Basin; 5=Columbia Plateau; 6=Blue Mountains; 7=Northern Glaciated Mountains; 8=Lower Clark Fork; 9=Upper Clark Fork; 10=Owyhee Uplands; 11=Upper Snake; 12=Snake Headwaters; 13=Central Idaho Mountains.

[3] - Designations reflect principal ownership and use: BR=BLM rangelands, FG=USFS grazing lands, FH=USFS high-impact, FM=USFS moderate impact, FW=USFS low impact and wilderness, NP=National Parks, PA=Private agriculture, PF=Private forests, TL=Tribal lands.

[4] - Clusters based on vegetative composition: A=Agriculture, B=Moist forest — understory reinitiation, C=Grasslands, D=Desert shrublands, E=Transitional areas, F=Young, dry forests, G=Aspen stands, H=Young, spruce-fir-lodgepole stands, I=Older, spruce-fir-lodgepole stands, J=Older, dry forests, K=Mountain shrublands, L=Moist forest — stem exclusion.

APPENDIX 4D

Historical Ranges Defined for the Key Salmonids Within the Basin

Key Salmonids

Our estimates of the historical ranges of the key salmonids were based on known historical distributions in published literature, available historical accounts, and speculative distributions as summarized in the Idaho, Montana, Oregon, and Washington River Information System databases (see appendix 4C), expanded to include any natural (non-introduced) occurrences in the status survey that were not included in the historical distributions.

Bull Trout

This range was based on the known historical or speculative distributions for bull trout from the individual state accounts as summarized in the 1994-95 updates for the Oregon bull trout distribution database, IRIS, and MRIS, and the Washington status review (Mongillo 1992), but expanded to include any occurrences in the status survey that were not included in the historical distributions. The historical accounts from WARIS have not been updated because the Washington stream and lakes database was not available at the time of this analysis. Where bull trout are known to have occurred within a subbasin we generally included the entire subbasin as part of the historical range if access was likely. Occasional reports from agency sampling and anecdotal accounts suggest that bull trout still occur in the mainstem Columbia and Snake rivers; we therefore included other mainstems below known historical or current occurrences as part of the historical range assuming that fish moved through these areas. There are no available records to indicate occurrence of bull trout in the Sanpoil or Colville river subbasins, but both are close to known distributions. There is no historical barrier known for the Sanpoil river but there is an impassable falls near the mouth of the Colville River that might have excluded bull trout.[1] Because both basins have streams that should have been suitable for bull trout based on models of occurrence with elevation and latitude, they are included in the historical range, but recognized as speculative. The occurrence of bull trout within the more isolated Upper Snake River/Rock Creek and Salmon Falls Creek subbasins are also apparently based on anecdotal information[2] and should also be considered speculative.

Yellowstone Cutthroat Trout

Data sources include summaries provided by Bruce May[3] and Behnke (1992). Two disjunct populations were known to exist outside the core in the headwaters of the Snake and Missouri rivers: 1) Crab Creek in Washington, and 2) Waha Lake in Idaho near Lewiston in the Clearwater drainage (Behnke 1992). The species is believed extinct in both of these areas. Above Shoshone Falls the Yellowstone cutthroat trout was known in all waters directly connected to the Snake River except those between Jackson Lake and Palisades Reservoir, where the closely allied fine-spotted Snake River cutthroat trout was thought to be native (Behnke 1992). We recognized both forms as Yellowstone cutthroat trout because the Snake River

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[1]Personal communication. 1995. T. Shuhda, Colville National Forest, Colville, WA.

[2]Personal communication. 1995. B. Horton, Idaho Department of Fish and Game, Boise, ID.

[3]Personal communication. 1995. Bruce May, Gallatin National Forest, Bozeman, MT. Personal communication of unpublished data.

form has not received formal taxonomic recognition. It is unclear which form (Yellowstone or finespotted) was native to Jackson Lake. Populations of both occur in the Gros Ventre River drainage.[4] The range of Yellowstone cutthroat trout represents most of the Snake River Basin above Shoshone Falls, but excludes the Sinks drainages and the Wood River basin. The history of the Sinks drainages is unclear; Hubbs and Miller (1948) believed cutthroat trout were native but it is not clear whether they were the westslope or Yellowstone subspecies. Thurow and others (1988) reported Yellowstone cutthroat trout in the Sinks drainages, but it is unknown if they were the result of introductions. We relied on Behnke (1992) to place the Sinks drainages outside the Snake zoogeographic basin and so did not include them in the Yellowstone cutthroat trout historical range. Yellowstone cutthroat trout have been widely introduced to high lakes and other areas. Observations outside this defined range are considered introduced.

Westslope Cutthroat Trout

The historical range of westslope cutthroat trout is based largely on current distribution and accounts from Behnke (1992) and Mullan and others (1992). Gilbert and Evermann (1894) reported "cut-throat" in the Little Spokane River, and we assume those to have been westslope cutthroat trout and have included them here. The status of cutthroat trout in the Lost rivers and in other sink drainages in Idaho are uncertain. There is no known documentation of occurrence but Behnke (1992) speculates westslope cutthroat trout may have been in some of those drainages because of the apparent headwater capture of other species such as bull trout. The historical distribution of westslope cutthroat trout in the Snake River drainage between the Salmon River and Shoshone Falls is unclear. Although Gilbert and Evermann (1894) reported cutthroat trout in the Wood River, Idaho, Behnke (1992) examined three museum specimens and reported all as redband trout. Jordan and Evermann (1902) mention cutthroat trout in the same paragraphs in which they refer to "fine trout fishing" in Payette and Redfish lakes. It is not clear whether they meant that cutthroat were found in both lakes, but other accounts make it clear that cutthroat trout did occur in the river basin of the latter. Because redband trout often exhibit a cutthroat-like red/orange slash under the jaw, and were termed "cutthroat" in other popular accounts, it is possible the reference in the Payette River was actually for redband trout. Westslope cutthroat trout have persisted well throughout much of the known portions of the historical range but have not been confirmed from any samples above the mouth of the Salmon River that cannot be attributed to recent introduction. For these reasons we did not include any of the Snake River drainage above the mouth of the Salmon River as part of the historical range for this fish. We conclude that any cutthroat trout in the Payette drainage are the results of high lake introductions, or introductions to Deadwood Reservoir (IDFG records, Nampa, Idaho). The historical distribution of westslope cutthroat trout along the Eastern slope of the Cascades is problematic. Behnke (1992) suggests that populations were native to the Wenatchee, Methow, and Entiat river drainages. Other evidence indicates historical occurrence in Ahtanum Creek, a tributary to the lower Yakima River, and the Chelan River basin (Behnke 1992). Mullan and others (1992) indicate that stocking of westslope cutthroat trout in mountain lakes led to the establishment of the species throughout many of the mid-Columbia River tributary systems. Given the very broad occurrence currently we assume that the native range included all of the previously mentioned drainages but recognize the uncertainty in how complete that distribution may have been.

Redband Trout

We identified two forms of redband trout within the Basin, allopatric and sympatric. We considered allopatric redband trout those fish outside the historical range of steelhead. We assumed the allopatric form was genetically and evolutionarily distinct from other redband trout because of isolation from steel-

[4]Personal communication. 1995. B. Gresswell, U.S. Fish and Wildlife Service, Corvallis, OR.

head. We considered sympatric redband trout to be the non-anadromous form historically derived from or associated with steelhead. Historically redband trout were widely distributed (Behnke 1992) occupying most accessible waters from the southern desert basins to the high mountain coniferous forests (Cope 1879; Cope 1889; Jordan 1892; Gilbert and Evermann 1894; Jordan and Evermann 1896; Snyder 1908; Jordan and others 1930; Behnke 1992). Because of the broad distribution and occurrence over the widest range of conditions evident for any of the salmonids, we assumed that redband trout occurred historically in all of the drainages contiguous with the current distribution. They are believed to be native in the Kootenai River basin below Kootenai Falls.[5,6] A barrier falls below Klamath Lake separated interior redband trout from coastal rainbow trout (Behnke 1992). The only major portions of the Basin not believed to support redband trout are the Snake River upstream from Shoshone Falls, tributaries to the Spokane River above Spokane Falls, Eastern Rocky Mountain basins in Montana, and portions of the northern Great Basin in Oregon. We relied on knowledge of established barriers to anadromy to define the range for the allopatric form (see the discussion of steelhead). The distribution of small populations of allopatric redband trout isolated from, but within the general range of steelhead (for example, above natural barriers in 2nd and 3rd order streams) was poorly documented and not considered here. The historical sympatric redband trout range is assumed to include everything within the range of steelhead. There is no clear distinction between steelhead and non-anadromous redband trout throughout the range of steelhead. Because redband trout that have been isolated from steelhead by geologic processes (natural barriers) may be evoluntionarily distinct while those overlapping in distribution may be different life histories of the same populations, we have identified two historical ranges (isolated from and overlapping with the historical range of steelhead). Redband trout also had access to most of the rest of the Basin with the exception of the Upper Snake zoogeographic basin. All redband trout found within the Upper Snake basin are assumed to be introduced coastal rainbow trout.

Steelhead

Data sources for Idaho included Evermann (1896), Chapman (1940), Parkhurst (1950), Gebhards (1959), Fulton (1970), NWPPC (1986), IDFG (1992), and F. Partridge[7]. Summer steelhead were found in all accessible reaches of the Snake River and tributaries in Idaho downstream from Shoshone Falls. Steelhead were also found in the Bruneau and Owyhee rivers, and Salmon Falls Creek drainages in Nevada. Cold water temperatures at high elevations may have restricted the upper limit of steelhead (Mullan and others 1992) but this limit is not well defined. Natural barriers limited the distribution in certain areas within the historical range but most inaccessible areas are at the subwatershed scale or smaller. We were conservative in the development of historical ranges for areas with little formal documentation.

Data for Washington were compiled primarily by K. MacDonald, Wenatchee National Forest, Wenatchee, Washington and T. Shuhda, Colville National Forest, Colville, Washington. Data sources included Sholz and others (1985), Ashe and Sholz (1992), Chance (1986), Fulton (1970), and Mullan (1986). Some information was based on interviews with tribal elders who fished streams in the upper Columbia River above what is now Grand Coulee Dam, including conversations with Chuck Jones, Jerry Marcot, and Joe Peone of the Colville Confederated Tribes, and Bill Touhey and Dean Osterman of the Kalispel Tribes. Other interviewees included A. Sholz, Eastern Washington Biology Department, Cheney, Washington; J. Nisbet, historian and author; and, D. Mattson, archeologist, Colville National Forest. Data for Oregon were compiled primarily by P. Howell,

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[5]Personal communication. 1996. L. Nelson, Idaho Fish and Game, Coeur d'Alene, ID. Personal communication of unpublished data.

[6]Personal communication. 1995. D. Perkinson, Kootenai National Forest, Libby, MT. Personal communication of unpublished data.

[7]Personal communication. 1995. F. Partridge, Idaho Department of Fish and Game.

Umatilla National Forest, Ukiah, Oregon. Information sources for Oregon included Fulton (1968, 1970); Howell and others (1985); Bakke and Felstner (1990); and personal communications with various biologists recorded by P. Howell, Umatilla National Forest.

Stream-Type Chinook Salmon

Data sources for Idaho included Evermann (1896), Chapman (1940), Schoning (1947), Miller and Miller (1948), Parkhurst (1950), Gebhards (1959), Fulton (1968), and IDFG (1992). Thompson (1951) reported that prior to overfishing and environmental alterations, the runs of chinook salmon in the Columbia River formed a continuum from early spring to late fall. Similar to summer steelhead, stream-type chinook salmon were found in all accessible reaches of the Snake River and tributaries in Idaho downstream from Shoshone Falls. Stream-type chinook salmon were also found in the Bruneau and Owyhee rivers, and Salmon Falls Creek drainages in Nevada. Cold water temperatures at high elevations and the presence of suitable spawning reaches may have restricted the upper limit of salmon but this limit is not well defined. Natural barriers limited the distribution in certain areas within the historical range but most inaccessible areas are at the subwatershed level or smaller.

Data for Washington were compiled primarily by K. MacDonald, Wenatchee National Forest, Wenatchee, Washington and T. Shuhda, Colville National Forest, Colville, Washington. Data sources were similar to those cited for steelhead. Data for Oregon were compiled primarily by P. Howell, Umatilla National Forest, Ukiah, Oregon. Information sources for Oregon were similar to those cited for steelhead in addition to Thompson and Haas (1960). Howell also stated that stream-type chinook were widely distributed but probably had more limited distribution and were farther downstream in watersheds compared to the distribution of steelhead.

Ocean-Type Chinook Salmon

Data sources for Idaho included Evermann (1896), Chapman (1940), Schoning (1947), Parkhurst (1950), Gebhards (1959), Fulton (1968), Irving and others (1981), IDFG (1992), and J. Chandler[8] and B. Connor.[9] Ocean-type (fall) chinook salmon spawning areas in Idaho appear to have been confined to the mainstem Snake River. Information suggests that suitable spawning areas for fall chinook salmon are restricted to mainstem reaches where at least 960 temperature units accumulate from November 15 (spawning) to a late April-early May emergence. In the Snake River near Marshing, formerly an important fall chinook salmon spawning area, reconstruction of historical temperature data suggests fry emerged primarily in April. Fry that emerge after mid-May may not be large enough to smolt in late may-early June and begin their downstream migration as age zero fish, a characteristic of fall chinook salmon. Because of these potential temperature constraints, it is likely that fall-spawning chinook salmon observed in higher elevation basins including the Clearwater and South Fork Salmon rivers were actually late spawning stream-type (summer) chinook salmon.

Data for Washington were compiled primarily by K. MacDonald, Wenatchee National Forest, Wenatchee, Washington and T. Shuhda, Colville National Forest, Colville, Washington. Data sources were similar to those cited for steelhead. Data for Oregon were compiled primarily by P. Howell, Umatilla National Forest, Ukiah, Oregon. Information sources for Oregon were similar to those cited for steelhead in addition to Thompson and Haas (1960) and USFWS-NMFS (1981).

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[8]Personal communication. 1995. J. Chandler, Idaho Power Company, Boise, ID.

[9]Personal communication. 1995. B. Connor, U.S. Fish and Wildlife Service, Orofino, ID.

APPENDIX 4E

Classification Tree Results for Six Key Salmonids

1.A. Bull Trout Status

A. Summary

Variables actually used in tree construction:

mtemp	roaddn	hucorder	vegclus	solar	alsi3
ero	con3	elev	slope	bank	eru
baseero	pprecip	con1	streams	alsi1	mngclus
slope2x	con2	vmf			

Number of terminal nodes: 57 Residual mean deviance: 0.8296 = 2207 / 2660

Misclassification error rate: 0.184 = 500 / 2717

B. Tree structure

Node) Split criterion	Sample Size	Deviance	Mode	Relative Frequencies Absent	Depressed	Strong
1) root	2717	4103.000	A	(0.70810	0.229700	0.062200)
2) mtemp<5.08	1474	2726.000	A	(0.55830	0.339900	0.101800)
4) roaddn<2.5	539	1131.000	A	(0.47500	0.304300	0.220800)
8) hucorder<35.5	500	1070.000	A	(0.43400	0.328000	0.238000)
16) vegclus:C,D,E,J,K	147	262.400	A	(0.62590	0.258500	0.115600)
32) solar<351.796	118	228.900	A	(0.55080	0.305100	0.144100)
64) alsi3<21.3151	109	199.100	A	(0.57800	0.321100	0.100900)
128) ero<0.2975	9	6.279	D	(0.11110	0.888900	0.000000)*
129) ero>0.2975	100	178.500	A	(0.62000	0.270000	0.110000)*
65) alsi3>21.3151	9	15.280	S	(0.22220	0.111100	0.666700)*
33) solar>351.796	29	14.560	A	(0.93100	0.068970	0.000000)*
17) vegclus:A,B,F,H,I,L	353	772.400	D	(0.35410	0.356900	0.289000)
34) solar<242.778	47	65.130	A	(0.51060	0.489400	0.000000)*
35) solar>242.778	306	672.300	D	(0.33010	0.336600	0.333300)
70) roaddn<0.5	123	249.700	S	(0.39840	0.154500	0.447200)
140) con3<0.03295	61	127.500	A	(0.49180	0.245900	0.262300)
280) elev<6413.5	19	7.835	A	(0.94740	0.000000	0.052630)*
281) elev>6413.5	42	91.840	D	(0.28570	0.357100	0.357100)
562) slope<30.322	13	16.050	A	(0.69230	0.307700	0.000000)*
563) slope>30.322	29	54.720	S	(0.10340	0.379300	0.517200)
1126) bank<54.7459	16	26.600	D	(0.12500	0.687500	0.187500)*
1127) bank>54.7459	13	7.051	S	(0.07692	0.000000	0.923100)*
141) con3>0.03295	62	103.000	S	(0.30650	0.064520	0.629000)
282) elev<8355	54	78.820	S	(0.22220	0.055560	0.722200)
564) slope<40.107	48	49.130	S	(0.20830	0.000000	0.791700)*
565) slope>40.107	6	12.140	D	(0.33330	0.500000	0.166700)*
283) elev>8355	8	6.028	A	(0.87500	0.125000	0.000000)*
71) roaddn>0.5	183	389.500	D	(0.28420	0.459000	0.256800)
142) elev<6000.5	38	73.840	A	(0.52630	0.131600	0.342100)
284) eru:2,6,8,9	6	0.000	S	(0.00000	0.000000	1.000000)*
285) eru:1,7,13	32	58.640	A	(0.62500	0.156200	0.218800)*
143) elev>6000.5	145	291.300	D	(0.22070	0.544800	0.234500)
286) mtemp<2.646	120	227.700	D	(0.15830	0.591700	0.250000)
572) baseero<23.807	32	38.020	D	(0.00000	0.718800	0.281200)*

573) baseero>23.807	88	176.600	D	(0.21590	0.545500	0.238600)
1146) solar<326.372	41	55.520	D	(0.12200	0.780500	0.097560)*
1147) solar>326.372	47	103.000	S	(0.29790	0.340400	0.361700)*
287) mtemp>2.646	25	49.890	A	(0.52000	0.320000	0.160000)
574) bank<43.4606	7	9.561	S	(0.00000	0.428600	0.571400)*
575) bank>43.4606	18	21.270	A	(0.72220	0.277800	0.000000)*
9) hucorder>35.5	39	0.000	A	(1.00000	0.000000	0.000000)*
5) roaddn>2.5	935	1466.000	A	(0.60640	0.360400	0.033160)
10) pprecip<644.06	252	312.900	A	(0.77780	0.190500	0.031750)
20) con1<50.5696	211	207.300	A	(0.84830	0.128000	0.023700)
40) slope<19.0185	122	23.180	A	(0.98360	0.008197	0.008197)*
41) slope>19.0185	89	137.300	A	(0.66290	0.292100	0.044940)
82) eru:1,7,10	18	0.000	A	(1.00000	0.000000	0.000000)*
83) eru:6,8,9,13	71	120.300	A	(0.57750	0.366200	0.056340)*
21) con1>50.5696	41	73.720	D	(0.41460	0.512200	0.073170)*
11) pprecip>644.06	683	1106.000	A	(0.54320	0.423100	0.033670)
22) eru:3,6,9	130	186.500	D	(0.21540	0.730800	0.053850)
44) elev<6149.5	88	132.400	D	(0.30680	0.659100	0.034090)
88) pprecip<1060.71	71	103.900	D	(0.38030	0.605600	0.014080)
176) hucorder<0.5	40	54.550	A	(0.57500	0.425000	0.000000)*
177) hucorder>0.5	31	32.400	D	(0.12900	0.838700	0.032260)*
89) pprecip>1060.71	17	12.320	D	(0.00000	0.882400	0.117600)*
45) elev>6149.5	42	35.670	D	(0.02381	0.881000	0.095240)*
23) eru:1,2,5,7,8,13	553	847.500	A	(0.62030	0.350800	0.028930)
46) pprecip<875.085	169	154.900	A	(0.82840	0.171600	0.000000)
92) elev<7132	162	132.400	A	(0.85800	0.142000	0.000000)
184) hucorder<1.5	98	39.500	A	(0.94900	0.051020	0.000000)*
185) hucorder>1.5	64	76.050	A	(0.71880	0.281200	0.000000)
370) eru:8	9	6.279	D	(0.11110	0.888900	0.000000)*
371) eru:1,7,13	55	52.160	A	(0.81820	0.181800	0.000000)*
93) elev>7132	7	5.742	D	(0.14290	0.857100	0.000000)*
47) pprecip>875.085	384	639.200	A	(0.52860	0.429700	0.041670)
94) streams<19.4	99	123.400	A	(0.75760	0.222200	0.020200)
188) alsi1<60.3367	59	91.340	A	(0.62710	0.339000	0.033900)
376) mngclus:FH,FW	16	0.000	A	(1.00000	0.000000	0.000000)*
377) mngclus:FG,FM,PF,TL	43	72.990	A	(0.48840	0.465100	0.046510)*
189) alsi1>60.3367	40	15.880	A	(0.95000	0.050000	0.000000)*
95) streams>19.4	285	486.500	D	(0.44910	0.501800	0.049120)
190) hucorder<27.5	271	463.200	D	(0.42070	0.527700	0.051660)
380) baseero<45.6534	164	286.700	A	(0.52440	0.414600	0.060980)
760) bank<61.0671	60	121.400	A	(0.43330	0.416700	0.150000)
1520) slope2x<53.4915	43	86.930	D	(0.25580	0.534900	0.209300)*
1521) slope2x>53.4915	17	12.320	A	(0.88240	0.117600	0.000000)*
761) bank>61.0671	104	151.200	A	(0.57690	0.413500	0.009615)
1522) con2<44.4019	13	0.000	A	(1.00000	0.000000	0.000000)*
1523) con2>44.4019	91	135.600	A	(0.51650	0.472500	0.010990)*
381) baseero>45.6534	107	154.700	D	(0.26170	0.700900	0.037380)
762) slope2x<26.035	23	34.040	D	(0.08696	0.739100	0.173900)*
763) slope2x>26.035	84	103.900	D	(0.30950	0.690500	0.000000)*
191) hucorder>27.5	14	0.000	A	(1.00000	0.000000	0.000000)*
3) mtemp>5.08	1243	995.000	A	(0.88580	0.098950	0.015290)
6) baseero<47.0346	1121	704.600	A	(0.91700	0.074040	0.008921)
12) vmf<0.272017	880	657.300	A	(0.89430	0.094320	0.011360)
24) eru:2,3,6,9	210	269.200	A	(0.77620	0.181000	0.042860)
48) elev<5393	198	235.200	A	(0.80810	0.146500	0.045450)
96) eru:2	46	55.010	A	(0.80430	0.043480	0.152200)
192) slope<11.805	28	0.000	A	(1.00000	0.000000	0.000000)*
193) slope>11.805	18	34.490	A	(0.50000	0.111100	0.388900)*
97) eru:3,6,9	152	162.700	A	(0.80920	0.177600	0.013160)
194) pprecip<909.063	147	144.600	A	(0.83670	0.149700	0.013610)
388) bank<74.3221	62	89.200	A	(0.69350	0.274200	0.032260)
776) baseero<24.0754	36	49.460	A	(0.55560	0.444400	0.000000)*

777) baseero>24.0754	26	22.420	A	(0.88460	0.038460	0.076920)*
389) bank>74.3221	85	38.030	A	(0.94120	0.058820	0.000000)*
195) pprecip>909.063	5	0.000	D	(0.00000	1.000000	0.000000)*
49) elev>5393	12	13.500	D	(0.25000	0.750000	0.000000)*
25) eru:1,5,7,8,10,13	670	344.800	A	(0.93130	0.067160	0.001493)
50) slope<11.575	186	12.450	A	(0.99460	0.005376	0.000000)*
51) slope>11.575	484	309.100	A	(0.90700	0.090910	0.002066)
102) slope2x<44.7999	299	255.800	A	(0.85620	0.140500	0.003344)
204) vmf<0.0543498	160	162.900	A	(0.79370	0.206200	0.000000)*
205) vmf>0.0543498	139	78.400	A	(0.92810	0.064750	0.007194)
410) streams<34.25	69	10.450	A	(0.98550	0.000000	0.014490)*
411) streams>34.25	70	53.710	A	(0.87140	0.128600	0.000000)*
103) slope2x>44.7999	185	22.090	A	(0.98920	0.010810	0.000000)*
13) vmf>0.272017	241	0.000	A	(1.00000	0.000000	0.000000)*
7) baseero>47.0346	122	211.100	A	(0.59840	0.327900	0.073770)
14) baseero<84.8319	101	142.400	A	(0.70300	0.267300	0.029700)*
15) baseero>84.8319	21	36.910	D	(0.09524	0.619000	0.285700)*

* denotes terminal node

1.B. Bull Trout Presence/Absence

A. Summary

Variables actually used in tree construction:

slope	hucorder	elev	pprecip	eru	ero
baseero	bank	alsi2	slope2x	mtemp	con1
con2	vegclus	con3	streams	drnden	hk
anadac					

Number of terminal nodes: 42

Residual mean deviance: 0.7381 = 1974 / 2675

Misclassification error rate: 0.1671 = 454 / 2717

B. Tree Structure

Node) Split Criterion	Sample Size	Deviance	Mode	Absent	Present
1) root	2717	3710.000	A	(0.57200	0.428000)
2) slope<20.6815	1325	1467.000	A	(0.75770	0.242300)
4) hucorder<0.5	621	420.600	A	(0.89370	0.106300)
8) elev<5449	495	192.100	A	(0.95150	0.048480)
16) pprecip<735.726	330	34.180	A	(0.99090	0.009091)*
17) pprecip>735.726	165	125.800	A	(0.87270	0.127300)
34) eru:6,9	9	9.535	P	(0.22220	0.777800)*
35) eru:1,2,5,7,8,13	156	94.210	A	(0.91030	0.089740)*
9) elev>5449	126	160.400	A	(0.66670	0.333300)
18) pprecip<575.577	45	22.040	A	(0.93330	0.066670)*
19) pprecip>575.577	81	112.200	A	(0.51850	0.481500)
38) eru:3,6,9	38	45.730	P	(0.28950	0.710500)*
39) eru:1,10,13	43	50.920	A	(0.72090	0.279100)
78) ero<0.256	22	30.320	P	(0.45450	0.545500)*
79) ero>0.256	21	0.000	A	(1.00000	0.000000)*
5) hucorder>0.5	704	921.800	A	(0.63780	0.362200)
10) baseero<10.7147	197	146.000	A	(0.87820	0.121800)
20) bank<76.4444	68	82.390	A	(0.70590	0.294100)*
21) bank>76.4444	129	35.660	A	(0.96900	0.031010)*

11) baseero>10.7147	507	698.900	A	(0.54440	0.455600)
22) eru:1,2,3,8,9	112	140.700	P	(0.32140	0.678600)
44) alsi2<14.8035	91	95.850	P	(0.21980	0.780200)*
45) alsi2>14.8035	21	23.050	A	(0.76190	0.238100)*
23) eru:5,6,7,10,13	395	529.200	A	(0.60760	0.392400)
46) baseero<37.9108	367	478.300	A	(0.64310	0.356900)
92) alsi2<63.8306	348	443.000	A	(0.66670	0.333300)
184) slope2x<55.2053	314	412.500	A	(0.63380	0.366200)
368) mtemp<7.2905	199	274.100	A	(0.54770	0.452300)
736) con1<38.3993	124	157.400	A	(0.66940	0.330600)*
737) con1>38.3993	75	96.800	P	(0.34670	0.653300)*
369) mtemp>7.2905	115	120.400	A	(0.78260	0.217400)
738) hucorder<13	65	30.050	A	(0.93850	0.061540)*
739) hucorder>13	50	68.030	A	(0.58000	0.420000)
1478) eru:7,10	18	0.000	A	(1.00000	0.000000)*
1479) eru:5,6,13	32	41.180	P	(0.34380	0.656200)*
185) slope2x>55.2053	34	9.023	A	(0.97060	0.029410)*
93) alsi2>63.8306	19	19.560	P	(0.21050	0.789500)*
47) baseero>37.9108	28	22.970	P	(0.14290	0.857100)*
3) slope>20.6815	1392	1868.000	P	(0.39510	0.604900)
6) hucorder<1.5	857	1187.000	A	(0.52040	0.479600)
12) elev<5274.5	370	458.600	A	(0.68920	0.310800)
24) baseero<88.4119	347	403.400	A	(0.73200	0.268000)
48) con2<95.2886	186	251.600	A	(0.59140	0.408600)
96) pprecip<790.366	36	15.450	A	(0.94440	0.055560)*
97) pprecip>790.366	150	207.900	A	(0.50670	0.493300)*
49) con2>95.2886	161	108.600	A	(0.89440	0.105600)
98) baseero<54.2576	133	54.850	A	(0.94740	0.052630)*
99) baseero>54.2576	28	36.500	A	(0.64290	0.357100)*
25) baseero>88.4119	23	8.227	P	(0.04348	0.956500)*
13) elev>5274.5	487	652.300	P	(0.39220	0.607800)
26) vegclus:C,D,E,F,G,J,K	139	186.600	A	(0.60430	0.395700)
52) slope2x<31.3738	31	37.350	P	(0.29030	0.709700)
104) con3<20.3357	25	18.350	P	(0.12000	0.880000)*
105) con3>20.3357	6	0.000	A	(1.00000	0.000000)*
53) slope2x>31.3738	108	132.900	A	(0.69440	0.305600)
106) eru:6,8,9	23	28.270	P	(0.30430	0.695700)*
107) eru:10,13	85	85.070	A	(0.80000	0.200000)*
27) vegclus:B,H,I,L	348	429.500	P	(0.30750	0.692500)
54) streams<13.9	27	34.370	A	(0.66670	0.333300)*
55) streams>13.9	321	379.000	P	(0.27730	0.722700)
110) drnden<1.1253	118	93.660	P	(0.13560	0.864400)*
111) drnden>1.1253	203	265.200	P	(0.35960	0.640400)
222) hk<0.20045	63	47.960	P	(0.12700	0.873000)
444) mtemp<2.717	51	9.844	P	(0.01961	0.980400)*
445) mtemp>2.717	12	16.300	A	(0.58330	0.416700)*
223) hk>0.20045	140	193.400	P	(0.46430	0.535700)
446) mtemp<1.712	58	74.730	A	(0.65520	0.344800)
892) con1<13.976	40	33.820	A	(0.85000	0.150000)*
893) con1>13.976	18	19.070	P	(0.22220	0.777800)*
447) mtemp>1.712	82	103.900	P	(0.32930	0.670700)*
7) hucorder>1.5	535	527.000	P	(0.19440	0.805600)
14) slope<29.1495	268	331.700	P	(0.30970	0.690300)
28) baseero<58.8053	243	312.000	P	(0.34160	0.658400)
56) anadac<0.5	142	195.800	P	(0.45770	0.542300)
112) eru:7,9	54	54.590	P	(0.20370	0.796300)*
113) eru:1,6,8,10,13	88	117.400	A	(0.61360	0.386400)
226) bank<58.4523	24	8.314	A	(0.95830	0.041670)*
227) bank>58.4523	64	88.660	P	(0.48440	0.515600)*
57) anadac>0.5	101	94.670	P	(0.17820	0.821800)*
29) baseero>58.8053	25	0.000	P	(0.00000	1.000000)*
15) slope>29.1495	267	147.100	P	(0.07865	0.921300)
30) eru:1,6	38	50.020	P	(0.36840	0.631600)*
31) eru:5,7,8,9,13	229	62.610	P	(0.03057	0.969400)*

* denotes terminal node

2.A. Westslope Cutthroat Trout Status

A. Summary

Pruned Classification tree:

Variables actually used in tree construction:

eru	mngclus	roaddn	hk	bank	alsi4
sdt2	mtemp	alsi1	elev	drnden	sdt3
con2	slope	solar	pprecip	vmf	vegclus
sdt1	hucorder				

Number of terminal nodes: 37

Residual mean deviance: 0.9993 = 1602 / 1603

Misclassification error rate: 0.197 = 323 / 1640

B. Tree structure

Node) Split criterion	Sample Size	Deviance	Mode	Relative Frequencies Absent	Depressed	Strong
1) root	1640	3108.000	D	(0.196300	0.60240	0.20120)
2) eru:7,8,9	1047	1474.000	D	(0.078320	0.75930	0.16240)
4) mngclus:FG,FH,FM,FW	620	889.400	D	(0.036860	0.70990	0.25320)
8) roaddn<3.5	249	388.900	D	(0.024100	0.55420	0.42170)
16) hk<0.3722	220	336.900	D	(0.027270	0.61360	0.35910)
32) hk<0.207	39	21.150	D	(0.000000	0.92310	0.07692)*
33) hk>0.207	181	292.200	D	(0.033150	0.54700	0.41990)
66) bank<52.972	31	38.100	S	(0.064520	0.12900	0.80650)*
67) bank>52.972	150	225.800	D	(0.026670	0.63330	0.34000)
134) alsi4<76.2131	132	191.500	D	(0.030300	0.68180	0.28790)
268) sdt2<42.6506	124	172.500	D	(0.032260	0.71770	0.25000)
536) mtemp<3.116	63	51.670	D	(0.000000	0.85710	0.14290)
1072) alsi1<84.55	45	9.591	D	(0.000000	0.97780	0.02222)*
1073) alsi1>84.55	18	24.730	D	(0.000000	0.55560	0.44440)*
537) mtemp>3.12	61	105.600	D	(0.065570	0.57380	0.36070)*
269) sdt2>42.6506	8	6.028	S	(0.000000	0.12500	0.87500)*
135) alsi4>76.2131	18	21.270	S	(0.000000	0.27780	0.72220)*
17) hk>0.3722	29	19.290	S	(0.000000	0.10340	0.89660)*
9) roaddn>3.5	375	438.600	D	(0.045330	0.81330	0.14130)
18) elev<4044.5	127	70.810	D	(0.031500	0.93700	0.03150)
36) drnden<1.10835	29	23.270	D	(0.137900	0.86210	0.00000)*
37) drnden>1.10835	98	33.420	D	(0.000000	0.95920	0.04082)*
19) elev>4044.5	248	342.600	D	(0.052420	0.75000	0.19760)
38) bank<60.8877	69	113.500	D	(0.173900	0.69570	0.13040)
76) sdt3<0.12755	58	82.360	D	(0.086210	0.75860	0.15520)*
77) sdt3>0.12755	11	14.420	A	(0.636400	0.36360	0.00000)*
39) bank>60.8877	179	202.100	D	(0.005587	0.77090	0.22350)
78) con2<99.8966	135	128.200	D	(0.007407	0.83700	0.15560)*
79) con2>99.8966	44	60.180	D	(0.000000	0.56820	0.43180)
158) slope<25.468	7	0.000	S	(0.000000	0.00000	1.00000)*
159) slope>25.468	37	46.630	D	(0.000000	0.67570	0.32430)
318) solar<238.392	17	20.600	S	(0.000000	0.29410	0.70590)*
319) solar>238.392	20	0.000	D	(0.000000	1.00000	0.00000)*
5) mngclus:NP,PA,PF,PR,TL	423	447.300	D	(0.139500	0.83220	0.02837)
10) pprecip<425.26	32	44.240	A	(0.531200	0.46880	0.00000)*
11) pprecip>425.26	391	371.200	D	(0.107400	0.86190	0.03069)
22) drnden<1.53125	310	247.100	D	(0.067740	0.89680	0.03548)*
23) drnden>1.53125	81	102.900	D	(0.259300	0.72840	0.01235)
46) mngclus:NP,PA,PF	66	69.470	D	(0.166700	0.81820	0.01515)*

47) mngclus:PR,TL	15	19.100	A	(0.666700	0.33330	0.00000)*
3) eru:1,5,6,13	593	1287.000	A	(0.404700	0.32550	0.26980)
6) vmf<0.10069	491	1072.000	D	(0.289200	0.38490	0.32590)
12) mngclus:BR,FG,FH,FM, PA,PF,PR	307	632.500	D	(0.309400	0.49510	0.19540)
24) solar<277.369	88	183.100	A	(0.465900	0.19320	0.34090)
48) pprecip<1114.58	53	79.450	A	(0.735800	0.09434	0.16980)
96) slope<23.4005	31	14.830	A	(0.935500	0.00000	0.06452)*
97) slope>23.4005	22	46.620	A	(0.454500	0.22730	0.31820)
194) solar<262.553	10	12.220	S	(0.000000	0.30000	0.70000)*
195) solar>262.553	12	10.810	A	(0.833300	0.16670	0.00000)*
49) pprecip>1114.58	35	58.590	S	(0.057140	0.34290	0.60000)*
25) solar>277.369	219	401.100	D	(0.246600	0.61640	0.13700)
50) drnden<1.31305	105	223.700	D	(0.333300	0.43810	0.22860)
100) mtemp<1.926	30	58.210	S	(0.133300	0.33330	0.53330)*
101) mtemp>1.926	75	143.400	D	(0.413300	0.48000	0.10670)*
51) drnden>1.31305	114	147.500	D	(0.166700	0.78070	0.05263)
102) vegclus:E,F,H,J	71	115.800	D	(0.239400	0.67610	0.08451)
204) sdt1<0.28845	39	50.160	D	(0.076920	0.79490	0.12820)*
205) sdt1>0.28845	32	51.580	D	(0.437500	0.53120	0.03125)*
103) vegclus:B,C,D, I,K,L	43	16.180	D	(0.046510	0.95350	0.00000)*
13) mngclus:FW,TL	184	368.900	S	(0.255400	0.20110	0.54350)
26) hucorder<20	156	288.600	S	(0.153800	0.23080	0.61540)
52) slope<27.0375	41	85.500	D	(0.219500	0.48780	0.29270)
104) solar<307.385	21	28.680	D	(0.000000	0.57140	0.42860)*
105) solar>307.385	20	40.420	A	(0.450000	0.40000	0.15000)
210) mtemp<2.297	14	24.980	A	(0.642900	0.14290	0.21430)*
211) mtemp>2.297	6	0.000	D	(0.000000	1.00000	0.00000)*
53) slope>27.0375	115	177.000	S	(0.130400	0.13910	0.73040)
106) pprecip<1046.32	70	116.900	S	(0.100000	0.22860	0.67140)
212) sdt1<14.2349	49	52.190	S	(0.000000	0.22450	0.77550)*
213) sdt1>14.2349	21	44.980	S	(0.333300	0.23810	0.42860)*
107) pprecip>1046.32	45	42.120	S	(0.177800	0.00000	0.82220)*
27) hucorder>20	28	31.280	A	(0.821400	0.03571	0.14290)*
7) vmf>0.10069	102	33.750	A	(0.960800	0.03922	0.00000)*

* denotes terminal node

2.B. Westslope Cutthroat Trout Presence/Absence

A. Summary

Pruned Classification tree:

Variables actually used in tree construction:

eru slope2x sdt3 mngclus elev ero
con2 vegclus slope hk anadac pprecip
bank drnden streams hucorder mtemp

Number of terminal nodes: 24

Residual mean deviance: 0.4153 = 671.1 / 1616

Misclassification error rate: 0.075 = 123 / 1640

B. Tree structure

Node) Split Criterion	Sample Size	Deviance	Mode	Absent	Present
1) root	1640	1470.000	P	(0.16520	0.83480)
2) eru:5	87	53.420	A	(0.90800	0.09195)
4) slope2x<35.9634	64	0.000	A	(1.00000	0.00000)*
5) slope2x>35.9634	23	29.720	A	(0.65220	0.34780)
10) sdt3<18.464	11	12.890	P	(0.27270	0.72730)*
11) sdt3>18.464	12	0.000	A	(1.00000	0.00000)*
3) eru:1,6,7,8,9,13	1553	1162.000	P	(0.12360	0.87640)
6) mngclus:BR,FM,FW,NP	852	352.300	P	(0.05282	0.94720)
12) elev<6248	621	141.300	P	(0.02415	0.97580)
24) ero<0.274	70	45.510	P	(0.10000	0.90000)
48) con2<81.2661	26	30.290	P	(0.26920	0.73080)*
49) con2>81.2661	44	0.000	P	(0.00000	1.00000)*
25) ero>0.274	551	83.600	P	(0.01452	0.98550)*
13) elev>6248	231	178.400	P	(0.12990	0.87010)
26) vegclus:B,C,E,F	50	64.100	P	(0.34000	0.66000)
52) slope<22.2285	9	0.000	A	(1.00000	0.00000)*
53) slope>22.2285	41	40.470	P	(0.19510	0.80490)*
27) vegclus:D,H,I,J,K,L	181	93.520	P	(0.07182	0.92820)
54) hk<0.20035	99	11.180	P	(0.01010	0.98990)*
55) hk>0.20035	82	68.270	P	(0.14630	0.85370)*
7) mngclus:FG,FH,PA,PF,PR,TL	701	720.000	P	(0.20970	0.79030)
14) anadac<0.5	482	366.100	P	(0.12660	0.87340)
28) pprecip<425.26	30	41.590	A	(0.50000	0.50000)
56) bank<65.6722	20	22.490	A	(0.75000	0.25000)*
57) bank>65.6722	10	0.000	P	(0.00000	1.00000)*
29) pprecip>425.26	452	297.400	P	(0.10180	0.89820)
58) drnden<1.597	390	201.400	P	(0.07179	0.92820)
116) streams<23.45	124	102.700	P	(0.14520	0.85480)*
117) streams>23.45	266	85.240	P	(0.03759	0.96240)*
59) drnden>1.597	62	74.700	P	(0.29030	0.70970)
118) hucorder<4.5	41	56.230	P	(0.43900	0.56100)*
119) hucorder>4.5	21	0.000	P	(0.00000	1.00000)*
15) anadac>0.5	219	293.400	P	(0.39270	0.60730)
30) ero<1.9505	174	208.700	P	(0.28740	0.71260)
60) vegclus:A,E,F,H,J	127	169.400	P	(0.38580	0.61420)
120) streams<38.55	88	121.800	P	(0.47730	0.52270)
240) mngclus:FG,FH,PA,PF	80	110.700	A	(0.52500	0.47500)
480) mtemp<4.9615	61	82.570	P	(0.40980	0.59020)
960) pprecip<746.659	24	18.080	P	(0.12500	0.87500)*
961) pprecip>746.659	37	49.960	A	(0.59460	0.40540)
1922) slope2x<36.5642	11	0.000	A	(1.00000	0.00000)*
1923) slope2x>36.5642	26	35.430	P	(0.42310	0.57690)*
481) mtemp>4.9615	19	12.790	A	(0.89470	0.10530)*
241) mngclus:PR,TL	8	0.000	P	(0.00000	1.00000)*
121) streams>38.55	39	36.710	P	(0.17950	0.82050)*
61) vegclus:B,C,I,K,L	47	9.679	P	(0.02128	0.97870)*
31) ero>1.9505	45	45.040	A	(0.80000	0.20000)*

*denotes terminal node

3.A. Yellowstone Cutthroat Trout Status

A. Summary

Pruned Classification tree:

Variables actually used in tree construction:

eru mtemp hucorder solar hk ero drnden pprecip sdt1

Number of terminal nodes: 11

Residual mean deviance: 0.7893 = 301.5 / 382

Misclassification error rate: 0.1501 = 59 / 393

B. Tree structure

Node) Split criterion	Sample Size	Deviance	Mode	Relative Frequencies Absent	Depressed	Strong
1) root	393	846.400	S	(0.27230	0.2952	0.43260)
2) eru:5,11	111	157.400	A	(0.71170	0.2523	0.03604)
4) mtemp<6.1345	59	105.500	D	(0.45760	0.4746	0.06780)
8) hucorder<2.5	31	37.350	A	(0.70970	0.2903	0.00000)*
9) hucorder>2.5	28	47.530	D	(0.17860	0.6786	0.14290)*
5) mtemp>6.1345	52	0.000	A	(1.00000	0.0000	0.00000)*
3) eru:10,12	282	510.200	S	(0.09929	0.3121	0.58870)
6) solar<329.144	35	51.890	A	(0.65710	0.3143	0.02857)
12) hk<0.36925	17	27.480	D	(0.29410	0.6471	0.05882)
24) ero<0.1895	5	0.000	A	(1.00000	0.0000	0.00000)*
25) ero>0.1895	12	6.884	D	(0.00000	0.9167	0.08333)*
13) hk>0.36925	18	0.000	A	(1.00000	0.0000	0.00000)*
7) solar>329.144	247	351.600	S	(0.02024	0.3117	0.66800)
14) drnden<1.1257	120	83.960	S	(0.01667	0.0750	0.90830)*
15) drnden>1.1257	127	199.100	D	(0.02362	0.5354	0.44090)
30) pprecip<217.365	23	0.000	D	(0.00000	1.0000	0.00000)*
31) pprecip>217.365	104	166.000	S	(0.02885	0.4327	0.53850)
62) hucorder<13.5	88	138.100	D	(0.02273	0.5114	0.46590)
124) sdt1<9.79875	19	12.790	S	(0.00000	0.1053	0.89470)*
125) sdt1>9.79875	69	105.500	D	(0.02899	0.6232	0.34780)*
63) hucorder>13.5	16	7.481	S	(0.06250	0.0000	0.93750)*

* denotes terminal node

3.B. Yellowstone Cutthroat Trout Presence/Absence

A. Summary

Pruned Classification tree:

Variables actually used in tree construction:

mtemp solar hk ero hucorder con1 alsi3

Number of terminal nodes: 9

Residual mean deviance: 0.1734 = 66.6 / 384

Misclassification error rate: 0.02036 = 8 / 393

B. Tree structure

Node) Split Criterion	Sample Size	Deviance	Mode	Absent	Present
1) root	393	447.900	P	(0.25700	0.74300)
2) mtemp<6.1345	340	280.400	P	(0.14410	0.85590)
4) solar<327.01	38	47.400	A	(0.68420	0.31580)
8) hk<0.36925	20	26.920	P	(0.40000	0.60000)
16) ero<0.1895	6	0.000	A	(1.00000	0.00000)*
17) ero>0.1895	14	11.480	P	(0.14290	0.85710)*
9) hk>0.36925	18	0.000	A	(1.00000	0.00000)*
5) solar>327.01	302	162.600	P	(0.07616	0.92380)
10) mtemp<4.7385	242	32.300	P	(0.01240	0.98760)*
11) mtemp>4.7385	60	76.380	P	(0.33330	0.66670)
22) hucorder<2.5	30	38.190	A	(0.66670	0.33330)
44) con1<4.6376	14	18.250	P	(0.35710	0.64290)
88) alsi3<21.3968	8	0.000	P	(0.00000	1.00000)*
89) alsi3>21.3968	6	5.407	A	(0.83330	0.16670)*
45) con1>4.6376	16	7.481	A	(0.93750	0.06250)*
23) hucorder>2.5	30	0.000	P	(0.00000	1.00000)*
3) mtemp>6.1345	53	9.922	A	(0.98110	0.01887)*

* denotes terminal node

4.A. Redband Trout Status

A. Summary

Pruned Classification tree:

Variables actually used in tree construction:

mngclus slope eru bank vmf hucorder
mtemp alsi3 anadac vegclus con3 pprecip
con1 streams elev solar baseero ero
drnden

Number of terminal nodes: 35

Residual mean deviance: 1.138 = 2001 / 1758

Misclassification error rate: 0.2426 = 435 / 1793

B. Tree structure

Node) Split criterion	Sample Size	Deviance	Mode	Relative Frequencies Absent	Depressed	Strong
1) root	1793	3578.000	A	(0.463500	0.39710	0.139400)
2) mngclus:BR,FM,PA,PR	1254	2107.000	A	(0.598100	0.34370	0.058210)
4) slope<9.9835	757	961.500	A	(0.741100	0.24170	0.017170)
8) eru:4,7,10	464	460.500	A	(0.836200	0.14870	0.015090)
16) bank<69.2355	64	103.500	A	(0.515600	0.45310	0.031250)*
17) bank>69.2355	400	312.800	A	(0.887500	0.10000	0.012500)
34) vmf<0.0142501	8	15.590	S	(0.125000	0.37500	0.500000)*
35) vmf>0.0142501	392	258.800	A	(0.903100	0.09439	0.002551)
70) hucorder<1.5	272	104.400	A	(0.952200	0.04779	0.000000)*
71) hucorder>1.5	120	131.200	A	(0.791700	0.20000	0.008333)*
9) eru:1,2,5,6,13	293	444.200	A	(0.590400	0.38910	0.020480)
18) mtemp<10.1715	232	368.900	A	(0.495700	0.47840	0.025860)
36) alsi3<17.6948	84	126.300	A	(0.690500	0.26190	0.047620)

72) mtemp<8.2175	45	53.210	A	(0.822200	0.08889	0.088890)*
73) mtemp>8.2175	39	53.830	A	(0.538500	0.46150	0.000000)*
37) alsi3>17.6948	148	216.500	D	(0.385100	0.60140	0.013510)*
19) mtemp>10.1715	61	23.920	A	(0.950800	0.04918	0.000000)*
5) slope>9.9835	497	964.000	D	(0.380300	0.49900	0.120700)
10) hucorder<144	460	887.500	D	(0.330400	0.53910	0.130400)
20) anadac<0.5	368	684.500	D	(0.255400	0.60330	0.141300)
40) vegclus:C,E,F,H,J,K	155	231.600	D	(0.109700	0.74190	0.148400)
80) con3<56.3276	114	114.600	D	(0.035090	0.85090	0.114000)
160) bank<81.3688	68	95.030	D	(0.058820	0.75000	0.191200)*
161) bank>81.3688	46	0.000	D	(0.000000	1.00000	0.000000)*
81) con3>56.3276	41	87.720	D	(0.317100	0.43900	0.243900)*
41) vegclus:A,B,D,L	213	419.700	D	(0.361500	0.50230	0.136200)
82) hucorder<0.5	119	236.000	A	(0.470600	0.39500	0.134500)
164) eru:7	18	16.220	A	(0.833300	0.00000	0.166700)*
165) eru:4,5,6,8, 10,13	101	199.100	D	(0.405900	0.46530	0.128700)
330)pprecip<420.582	51	80.140	A	(0.627500	0.33330	0.039220)*
331)pprecip>420.582	50	94.830	D	(0.180000	0.60000	0.220000)*
83) hucorder>0.5	94	168.300	D	(0.223400	0.63830	0.138300)*
21) anadac>0.5	92	158.300	A	(0.630400	0.28260	0.086960)
42) con1<24.2747	63	117.800	A	(0.507900	0.39680	0.095240)
84) streams<22.05	26	36.410	A	(0.769200	0.11540	0.115400)*
85) streams>22.05	37	64.970	D	(0.324300	0.59460	0.081080)
170) elev<2469.5	11	3.200	A	(0.818200	0.09091	0.090910)*
171) elev>2469.5	26	32.190	D	(0.115400	0.80770	0.076920)*
43) con1>24.2747	29	23.110	A	(0.896600	0.03448	0.068970)*
11) hucorder>144	37	0.000	A	(1.000000	0.00000	0.000000)*
3) mngclus:FG,FH,FW,PF,TL	539	1067.000	D	(0.150300	0.52130	0.328400)
6) anadac<0.5	420	716.600	D	(0.061900	0.58570	0.352400)
12) solar<290.698	51	83.320	S	(0.215700	0.09804	0.686300)
24) alsi3<0.4464	15	30.290	A	(0.533300	0.20000	0.266700)*
25) alsi3>0.4464	36	35.740	S	(0.083330	0.05556	0.861100)*
13) solar>290.698	369	568.900	D	(0.040650	0.65310	0.306200)
26) eru:2,3,4,5	100	130.700	D	(0.130000	0.79000	0.080000)*
27) eru:6,10,13	269	381.500	D	(0.007435	0.60220	0.390300)
54) baseero<29.5729	222	302.400	D	(0.009009	0.66220	0.328800)
108) alsi3<33.7782	148	182.000	D	(0.013510	0.75000	0.236500)
216) con1<2.1399	82	106.500	D	(0.000000	0.64630	0.353700)
432) hucorder<1.5	51	70.680	S	(0.000000	0.49020	0.509800)
864) ero<0.253	17	12.320	D	(0.000000	0.88240	0.117600)*
865) ero>0.253	34	41.190	S	(0.000000	0.29410	0.705900)*
433) hucorder>1.5	31	19.710	D	(0.000000	0.90320	0.096770)*
217) con1>2.1399	66	57.750	D	(0.030300	0.87880	0.090910)*
109) alsi3>33.7782	74	102.500	S	(0.000000	0.48650	0.513500)*
55) baseero>29.5729	47	58.870	S	(0.000000	0.31910	0.680900)*
7) anadac>0.5	119	252.400	A	(0.462200	0.29410	0.243700)
14) ero<0.3105	37	62.670	D	(0.378400	0.56760	0.054050)
28) baseero<9.6972	7	0.000	A	(1.000000	0.00000	0.000000)*
29) baseero>9.6972	30	46.190	D	(0.233300	0.70000	0.066670)*
15) ero>0.3105	82	166.300	A	(0.500000	0.17070	0.329300)
30) alsi3<27.8208	57	104.100	A	(0.631600	0.17540	0.193000)
60) eru:1,7	23	37.990	A	(0.565200	0.04348	0.391300)*
61) eru:2,5,13	34	53.240	A	(0.676500	0.26470	0.058820)
122) drnden<1.19945	19	7.835	A	(0.947400	0.00000	0.052630)*
123) drnden>1.19945	15	25.600	D	(0.333300	0.60000	0.066670)*
31) alsi3>27.8208	25	45.040	S	(0.200000	0.16000	0.640000)*

* denotes terminal node

4.B. Redband Trout Presence/Absence

A. Summary

Pruned Classification tree:
Variables actually used in tree construction:
pprecip baseero eru mtemp bank hk
hucorder slope solar alsi3 mngclus anadac
drnden sdt2 slope2x streams
Number of terminal nodes: 20
Residual mean deviance: 0.7844 = 1391 / 1773
Misclassification error rate: 0.1846 = 331 / 1793

B. Tree structure

Node) Split Criterion	Sample Size	Deviance	Mode	Absent	Present
1) root	1793	2474.00	P	(0.46010	0.53990)
2) pprecip<380.296	840	1021.00	A	(0.70360	0.29640)
4) baseero<5.637	432	372.80	A	(0.84490	0.15510)
8) eru:2,5,6,7	99	126.00	A	(0.66670	0.33330)
16) mtemp<10.1	71	98.07	A	(0.53520	0.46480)*
17) mtemp>10.1	28	0.00	A	(1.00000	0.00000)*
9) eru:4,10	333	219.60	A	(0.89790	0.10210)
18) bank<91.5758	175	160.40	A	(0.82860	0.17140)
36) hk<0.25335	110	128.90	A	(0.72730	0.27270)*
37) hk>0.25335	65	0.00	A	(1.00000	0.00000)*
19) bank>91.5758	158	37.31	A	(0.97470	0.02532)*
5) baseero>5.637	408	560.90	A	(0.55390	0.44610)
10) mtemp<8.707	254	349.50	P	(0.44880	0.55120)
20) hucorder<0.5	115	152.00	A	(0.62610	0.37390)*
21) hucorder>0.5	139	170.30	P	(0.30220	0.69780)
42) hucorder<153.5	132	152.70	P	(0.26520	0.73480)
84) slope<8.109	26	33.54	A	(0.65380	0.34620)*
85) slope>8.109	106	96.58	P	(0.16980	0.83020)*
43) hucorder>153.5	7	0.00	A	(1.00000	0.00000)*
11) mtemp>8.707	154	180.50	A	(0.72730	0.27270)*
3) pprecip>380.296	953	1062.00	P	(0.24550	0.75450)
6) solar<277.231	258	357.40	A	(0.51550	0.48450)
12) alsi3<31.4241	193	255.00	A	(0.62690	0.37310)
24) eru:1,5,7,8	172	233.90	A	(0.58140	0.41860)*
25) eru:2,6,13	21	0.00	A	(1.00000	0.00000)*
13) alsi3>31.4241	65	62.18	P	(0.18460	0.81540)*
7) solar>277.231	695	576.20	P	(0.14530	0.85470)
14) mngclus:FG,FH,FM,PA,PF	428	206.80	P	(0.06542	0.93460)
28) anadac<0.5	362	98.52	P	(0.03039	0.96960)
56) drnden<0.22875	5	0.00	A	(1.00000	0.00000)*
57) drnden>0.22875	357	60.93	P	(0.01681	0.98320)*
29) anadac>0.5	66	75.31	P	(0.25760	0.74240)
58) sdt2<14.3508	19	23.70	A	(0.68420	0.31580)*
59) sdt2>14.3508	47	27.36	P	(0.08511	0.91490)*
15) mngclus:BR,FW,PR,TL	267	313.30	P	(0.27340	0.72660)
30) slope2x<16.4708	26	25.46	A	(0.80770	0.19230)*
31) slope2x>16.4708	241	251.40	P	(0.21580	0.78420)
62) streams<18.45	27	36.50	A	(0.59260	0.40740)*
63) streams>18.45	214	193.90	P	(0.16820	0.83180)*

* denotes terminal node

5. Steelhead Status

A. Summary

Pruned Classification tree:

Variables actually used in tree construction:

hucorder eru vmf slope baseero solar
pprecip drnden vegclus alsi3 mngclus hk
sdt2 streams elev bank con1 dampass

Number of terminal nodes: 35

Residual mean deviance: 0.5769 = 761.5 / 1320

Misclassification error rate: 0.1137 = 154 / 1355

B. Tree structure

Node) Split criterion	Sample Size	Deviance	Mode	Relative Frequencies Absent	Depressed	Migration	Strong
1) root	1355	2400.00	D	(0.13140	0.68710	0.166800	0.014760)
2) hucorder<21.5	1064	1419.00	D	(0.16730	0.78480	0.029140	0.018800)
4) eru:2,6	337	243.70	D	(0.01484	0.91990	0.026710	0.038580)
8) vmf<0.0135926	58	61.72	D	(0.00000	0.77590	0.000000	0.224100)
16) slope<19.466	37	15.56	D	(0.00000	0.94590	0.000000	0.054050)*
17) slope>19.466	21	29.06	S	(0.00000	0.47620	0.000000	0.523800)
34) baseero<33.4816	13	11.16	S	(0.00000	0.15380	0.000000	0.846200)*
35) baseero>33.4816	0	0.00	D	(0.00000	1.00000	0.000000	0.000000)*
9) vmf>0.0135926	279	129.30	D	(0.01792	0.94980	0.032260	0.000000)
18) solar<284.659	53	76.38	D	(0.09434	0.75470	0.150900	0.000000)
36) pprecip<561.617	26	53.37	D	(0.19230	0.50000	0.307700	0.000000)
72) hucorder<0.5	7	11.15	A	(0.71430	0.14290	0.142900	0.000000)*
73) hucorder>0.5	19	25.01	D	(0.00000	0.63160	0.368400	0.000000)*
37) pprecip>561.617	27	0.00	D	(0.00000	1.00000	0.000000	0.000000)*
19) solar>284.659	226	12.84	D	(0.00000	0.99560	0.004425	0.000000)*
5) eru:1,5,7,13	727	1057.00	D	(0.23800	0.72210	0.030260	0.009629)
10) hucorder<0.5	377	580.30	D	(0.37670	0.60210	0.010610	0.010610)
20) drnden<0.9504	52	47.92	A	(0.82690	0.17310	0.000000	0.000000)
40) slope<30.351	45	16.36	A	(0.95560	0.04444	0.000000	0.000000)*
41) slope>30.351	7	0.00	D	(0.00000	1.00000	0.000000	0.000000)*
21) drnden>0.9504	325	479.80	D	(0.30460	0.67080	0.012310	0.012310)
42) vegclus:A,E,I,J,K	145	225.00	D	(0.44830	0.53100	0.020690	0.000000)
84) alsi3<52.6848	103	163.80	A	(0.55340	0.41750	0.029130	0.000000)
168) mngclus: FG,FM,FW,PR	49	78.22	D	(0.34690	0.61220	0.040820	0.000000)
336) pprecip<1389.84	43	63.87	D	(0.25580	0.69770	0.046510	0.000000)
672) solar<318.167	25	16.71	D	(0.04000	0.92000	0.040000	0.000000)*
673) solar>318.167	18	30.76	A	(0.55560	0.38890	0.055560	0.000000)
1346) hk<0.2403	10	16.04	D	(0.20000	0.70000	0.100000	0.000000)*
1347) hk>0.2403	8	0.00	A	(1.00000	0.00000	0.000000	0.000000)*
337) pprecip>1389.84	6	0.00	A	(1.00000	0.00000	0.000000	0.000000)*
169) mngclus: BR,FH,PA,PF,TL	54	69.01	A	(0.74070	0.24070	0.018520	0.000000)*
85) alsi3>52.6848	42	40.90	D	(0.19050	0.80950	0.000000	0.000000)*
43) vegclus:B,C,D,F,H,L	180	223.00	D	(0.18890	0.78330	0.005556	0.022220)
86) sdt2<98.9962	145	167.00	D	(0.22760	0.76550	0.006897	0.000000)
172) streams<35.5	101	135.00	D	(0.30690	0.68320	0.009901	0.000000)
344) mngclus: BR,FG,FH,PA,PF	49	67.91	D	(0.48980	0.51020	0.000000	0.000000)*
345) mngclus:							

FM,FW,PR,TL	52	50.68	D	(0.13460	0.84620	0.019230	0.000000)*
173) streams>35.5	44	16.27	D	(0.04545	0.95450	0.000000	0.000000)*
87) sdt2>98.9962	35	33.71	D	(0.02857	0.85710	0.000000	0.114300)
174) pprecip<1160.34	25	0.00	D	(0.00000	1.00000	0.000000	0.000000)*
175) pprecip>1160.34	10	18.87	D	(0.10000	0.50000	0.000000	0.400000)*
11) hucorder>0.5	350	381.50	D	(0.08857	0.85140	0.051430	0.008571)
22) elev<1428.5	14	23.25	M	(0.07143	0.28570	0.642900	0.000000)*
23) elev>1428.5	336	316.90	D	(0.08929	0.87500	0.026790	0.008929)
46) solar<247.677	15	20.19	A	(0.60000	0.40000	0.000000	0.000000)*
47) solar>247.677	321	269.40	D	(0.06542	0.89720	0.028040	0.009346)
94) baseero<24.9986	155	111.30	D	(0.11610	0.88390	0.000000	0.000000)
188) bank<78.3389	87	85.95	D	(0.19540	0.80460	0.000000	0.000000)
376) vegclus:C,D,H	29	0.00	D	(0.00000	1.00000	0.000000	0.000000)*
377) vegclus:A,B,E,F,I,K,L	58	70.17	D	(0.29310	0.70690	0.000000	0.000000)
754) con1<4.7448	37	25.35	D	(0.10810	0.89190	0.000000	0.000000)*
755) con1>4.7448	21	27.91	A	(0.61900	0.38100	0.000000	0.000000)*
189) bank>78.3389	68	10.42	D	(0.01471	0.98530	0.000000	0.000000)*
95) baseero>24.9986	166	129.20	D	(0.01807	0.90960	0.054220	0.018070)
190) baseero<42.0739	104	87.88	D	(0.02885	0.88460	0.086540	0.000000)
380) hk<0.2419	67	17.99	D	(0.00000	0.97010	0.029850	0.000000)*
381) hk>0.2419	37	55.40	D	(0.08108	0.72970	0.189200	0.000000)*
191) baseero>42.0739	62	24.02	D	(0.00000	0.95160	0.000000	0.048390)*
3) hucorder>21.5	291	369.00	M	(0.00000	0.32990	0.670100	0.000000)
6) hucorder<121	127	170.30	D	(0.00000	0.60630	0.393700	0.000000)
12) eru:1,6	46	22.18	D	(0.00000	0.93480	0.065220	0.000000)*
13) eru:5,7,13	81	110.20	M	(0.00000	0.41980	0.580200	0.000000)
26) hk<0.33435	54	73.00	D	(0.00000	0.59260	0.407400	0.000000)
52) mngclus:BR,FH,FW,PA	37	49.96	M	(0.00000	0.40540	0.594600	0.000000)*
53) mngclus:FG,FM,PF,PR	17	0.00	D	(0.00000	1.00000	0.000000	0.000000)*
27) hk>0.33435	27	14.26	M	(0.00000	0.07407	0.925900	0.000000)*
7) hucorder>121	164	117.60	M	(0.00000	0.11590	0.884100	0.000000)
14) dampass<2.5	16	19.87	D	(0.00000	0.68750	0.312500	0.000000)*
15) dampass>2.5	148	62.24	M	(0.00000	0.05405	0.945900	0.000000)*

* denotes terminal node

6. Stream-type Chinook Salmon Status

A. Summary

Variables actually used in tree construction:

hucorder	pprecip	dampass	streams	mtemp	eru	
mngclus	alsi1		solar		alsi3	con3

Number of terminal nodes: 26

Residual mean deviance: 0.8282 Misclassification error rate: 0.1672 = 211 / 1262

B. Tree structure

Node) Split criterion	Sample Size	Deviance	Mode	Relative Frequencies Absent	Depressed	Migration	Strong
1) root	1262	2718.00	A	0.43900	0.36450	0.190200	0.006339
2) hucorder < 30	991	1652.00	A	0.54990	0.41370	0.028250	0.008073
4) hucorder = 0	500	622.40	A	0.75800	0.22200	0.020000	0.000000
8) pprecip < 601	145	93.04	A	0.92410	0.03448	0.041380	0.000000 *
9) pprecip > 601	355	473.80	A	0.69010	0.29860	0.011270	0.000000
18) dampass < 4	23	13.59	D	0.08696	0.91300	0.000000	0.000000 *
19) dampass > 3	332	418.60	A	0.73190	0.25600	0.012050	0.000000
38) streams < 23	145	112.40	A	0.88280	0.11030	0.006897	0.000000 *
39) streams > 23	187	274.20	A	0.61500	0.36900	0.016040	0.000000
78) mtemp < 6.9	158	242.70	A	0.54430	0.43670	0.018990	0.000000
156) streams < 43.6	122	177.00	A	0.63110	0.35250	0.016390	0.000000
312) pprecip < 960.7	52	50.68	A	0.84620	0.13460	0.019230	0.000000 *
313) pprecip > 960.7	70	106.00	D	0.47140	0.51430	0.014290	0.000000 *
157) streams > 43.6	36	49.04	D	0.25000	0.72220	0.027780	0.000000
314) streams < 64.1	31	32.40	D	0.12900	0.83870	0.032260	0.000000 *
315) streams > 64.1	5	0.00	A	1.00000	0.00000	0.000000	0.000000 *
79) mtemp > 6.9	29	0.00	A	1.00000	0.00000	0.000000	0.000000 *
5) hucorder > 0	491	841.50	D	0.33810	0.60900	0.036660	0.016290
10) eru: 5,7	116	117.80	A	0.81900	0.17240	0.008621	0.000000
20) mngclus: BR,PA,PF,PR	106	85.95	A	0.87740	0.11320	0.009434	0.000000
40) alsi1 < 1.56	77	18.55	A	0.97400	0.02597	0.000000	0.000000 *
41) alsi1 > 1.56	29	45.20	A	0.62070	0.34480	0.034480	0.000000 *
21) mngclus: FG,FH,FM,TL	10	10.01	D	0.20000	0.80000	0.000000	0.000000 *
11) eru: 1,2,6,13	375	568.10	D	0.18930	0.74400	0.045330	0.021330
22) mngclus :BR,PA,PR	68	129.60	A	0.45590	0.44120	0.102900	0.000000
44) mtemp < 10.54	62	94.79	A	0.50000	0.48390	0.016130	0.000000 *
45) mtemp > 10.54	6	0.00	M	0.00000	0.00000	1.000000	0.000000 *
23) mngclus: FG,FH,FM,FW,PF,TL	307	394.20	D	0.13030	0.81110	0.032570	0.026060
46) dampass < 4	56	82.10	D	0.10710	0.75000	0.000000	0.142900
92) sola r< 330.6	30	8.77	D	0.03333	0.96670	0.000000	0.000000 *
93) solar > 330.6	26	53.37	D	0.19230	0.50000	0.000000	0.307700
186) pprecip < 535.2	10	13.86	A	0.50000	0.50000	0.000000	0.000000 *
187) pprecip > 535.2	16	22.18	D	0.00000	0.50000	0.000000	0.500000 *
47) dampass > 4	251	280.20	D	0.13550	0.82470	0.039840	0.000000
94) hucorder <4	130	149.20	D	0.22310	0.76920	0.007692	0.000000 *
95) hucorder > 3	121	105.00	D	0.04132	0.88430	0.074380	0.000000
190) mngclus: FG,FM	51	0.00	D	0.00000	1.00000	0.000000	0.000000*
191) mngclus: FH,FW,PF	70	88.31	D	0.07143	0.80000	0.128600	0.000000*
3) hucorder > 29	271	334.40	M	0.03321	0.18450	0.782300	0.000000
6) hucorder < 58	74	138.00	M	0.09459	0.39190	0.513500	0.000000
12) eru: 5,7	10	12.22	A	0.70000	0.00000	0.300000	0.000000 *
13) eru: 1,6,13	64	88.16	M	0.00000	0.45310	0.546900	0.000000
26) alsi3 < 59.7	52	65.73	M	0.00000	0.32690	0.673100	0.000000
52) alsi1 < 0.37	9	6.28	D	0.00000	0.88890	0.111100	0.000000 *
53) alsi1 > 0.37	43	44.12	M	0.00000	0.20930	0.790700	0.000000 *
27) alsi3 > 59.7	12	0.00	D	0.00000	1.00000	0.000000	0.000000 *
7) hucorder > 57	197	155.60	M	0.01015	0.10660	0.883200	0.000000
14) dampass < 3	16	19.87	D	0.00000	0.68750	0.312500	0.000000*
15) dampass > 2	181	99.13	M	0.01105	0.05525	0.933700	0.000000
30) con3 < 0.49	90	19.18	M	0.02222	0.00000	0.977800	0.000000*
31) con3 > 0.49	91	63.00	M	0.00000	0.10990	0.890100	0.000000*

* denotes terminal node

6. Ocean-type chinook salmon status

A. Summary

Pruned Classification tree:

Variables actually used in tree construction:

hucorder vmf dampass ero elev pprecip slope

Number of terminal nodes: 12

Residual mean deviance: 0.4289 = 90.92 / 212

Misclassification error rate: 0.07589 = 17 / 224

B. Tree structure

Node) Split criterion Migration Strong		Sample	Size	Deviance	Relative Frequencies Mode	Absent	Depressed
1) root	224	569.400	A	(0.41520	0.23210	0.26340	0.08929)
2) hucorder<1080	161	356.600	A	(0.57760	0.22360	0.08075	0.11800)
4) hucorder<5.5	82	79.090	A	(0.85370	0.02439	0.12200	0.00000)
8) vmf<0.234099	61	17.600	A	(0.96720	0.03279	0.00000	0.00000)*
9) vmf>0.234099	21	29.060	A	(0.52380	0.00000	0.47620	0.00000)
18) dampass<3.5	9	6.279	M	(0.11110	0.00000	0.88890	0.00000)*
19) dampass>3.5	12	10.810	A	(0.83330	0.00000	0.16670	0.00000)*
5) hucorder>5.5	79	187.900	D	(0.29110	0.43040	0.03797	0.24050)
10) dampass<7.5	34	75.770	S	(0.23530	0.14710	0.05882	0.55880)
20) ero<2.349	14	27.780	A	(0.50000	0.35710	0.00000	0.14290)
40) dampass<2.5	6	7.638	D	(0.00000	0.66670	0.00000	0.33330)*
41) dampass>2.5	8	6.028	A	(0.87500	0.12500	0.00000	0.00000)*
21) ero>2.349	20	20.730	S	(0.05000	0.00000	0.10000	0.85000)
42) elev<1495	5	10.550	M	(0.20000	0.00000	0.40000	0.40000)*
43) elev>1495	15	0.000	S	(0.00000	0.00000	0.00000	1.00000)*
11) dampass>7.5	45	66.050	D	(0.33330	0.64440	0.02222	0.00000)
22) elev<3145.5	30	23.320	D	(0.06667	0.90000	0.03333	0.00000)
44) pprecip<304.361	5	10.550	A	(0.40000	0.40000	0.20000	0.00000)*
45) pprecip>304.361	25	0.000	D	(0.00000	1.00000	0.00000	0.00000)*
23) elev>3145.5	15	11.780	A	(0.86670	0.13330	0.00000	0.00000)*
3) hucorder>1080	63	81.080	M	(0.00000	0.25400	0.73020	0.01587)
6) slope<24.69	47	9.679	M	(0.00000	0.00000	0.97870	0.02128)*
7) slope>24.69	16	0.000	D	(0.00000	1.00000	0.00000	0.00000)*

* denotes terminal node

APPENDIX 4F

Summary of Self-Sustaining Trout Waters in the Basin Managed With Special Angling Regulations, by State and River Basin, 1995

Idaho[1]

Kootenai: headwaters of Kootenai River, Moyie River below Meadow Creek

Pend Oreille: Pend Oreille Lake and tributaries, Priest and Upper Priest lakes and tributaries

Spokane: Coeur d'Alene Lake and tributaries, Hayden Lake and tributaries, Coeur d'Alene River and tributaries, St. Joe River and tributaries, St. Maries River and tributaries, Spokane River below Post Falls Dam.

Clearwater: Fish Lake, Little North Fork Clearwater River and tributaries, North Fork Clearwater River and tributaries, Ten Mile Creek and tributaries, Lochsa River and tributaries above Wilderness Gateway, Selway River and tributaries above Meadow Creek

Salmon: Salmon River from Little Salmon River to Horse Creek, Partridge and French creeks, Little Salmon River and tributaries to headwaters, South Fork Salmon River and tributaries, Salmon River tributaries from Horse Creek to North Fork except Panther Creek, North Fork Salmon River, Middle Fork Salmon River and tributaries, Salmon River tributaries from North Fork to Hell Roaring Creek, Salmon River and tributaries from Hell Roaring Creek to headwaters, Alturas Lake, East Fork Salmon River and tributaries, Jimmy Smith Lake, Lemhi River and tributaries except Hawley Creek, Pahsimeroi River and tributaries

Snake River below Shoshone Falls: Wildhorse River; Indian, Reynolds and Castle creeks; Billingsly, Riley, and Box Canyon creeks, Banbury Springs

Weiser: Weiser River to Little Weiser River; Little, Middle Fork, and West Fork Weiser rivers and tributaries

Payette: Payette River from Black Canyon to Middle Fork, South Fork to Deadwood River, North Fork and tributaries from Banks to Cascade Dam, North Fork from Payette Lake to headwaters, Lake Fork Creek and tributaries to Browns Pond, Middle Fork upstream from Silver Creek, Deadwood River to Dam, South Fork above Helende, Bull Trout Lake

Boise: Boise River from Eagle Road to Eagle Island, Boise River from North Fork to Atlanta, Boise River and tributaries above Wilderness Boundary, South Fork from Arrowrock Reservoir to Anderson Ranch Dam, Big and Little Smokey creeks and other tributaries to South Fork Boise above Anderson Ranch Reservoir, North Fork to Rabbit Creek and above Hunter Creek to headwaters

Bruneau: North Fork Owyhee River and tributaries; Deep, Battle, and Blue creeks and tributaries; Bruneau River and tributaries from diversion Dam to West Fork; West Fork and tributaries, Jarbidge River and tributaries

Snake River above Shoshone Falls: Devils Corral and Vinyard creeks; Snake River from Eagle Rock to American Falls Dam, Rock Creek and triutaries, Snake River from American Falls Dam to South Fork, entire South Fork and tributaries; Portneuf River from American Falls Reservoir to Chesterfield Reservoir; Rapid, Toponce, and Pebble creeks; entire Blackfoot River drainage above lower Diversion Dam except

[1]Source: IDFG 1991; and, Personal communication. 1995. B. Horton, Idaho Department of Fish and Game, Boise, ID.

Dike Lake; Willow Creek and tributaries above Ririe Reservoir; entire Henrys Fork Snake drainage except Island Park Reservoir, Sand Creek WMA, and lakes; entire Teton River drainage except lakes

Wood: Big Wood River to I-84, Big Wood River from Glendale to North Fork, Little Wood River through Bear Track Williams State Park and upstream from Little Wood Reservoir, Silver Creek and tributaries

Salmon Falls: Shoshone Creek and tributaries upstream from and including Big Creek, upper Goose and Big Cottonwood creeks, Tributaries to Sublett Reservoir

Sinks Drainages: Big Lost River from Moore Diversion to Mackay Dam and from Chilly bridge to West Fork, Little Lost River and tributaries, Medicine Lodge Creek and tributaries, Camas Creek and tributaries above the National Wildlife Refuge, Beaver Creek and tributaries

Idaho—Waters managed by the Shoshone-Bannock Tribes[2]

Snake River above Shoshone Falls: tributaries and springs to American Falls Reservoir including Spring, Clear, Jeff Cabin and Diggie creeks; Portneuf River upstream from Chesterfield Reservoir and downstream from Siphon

Montana[3]

Clark Fork: Bitterroot River from Tucker Crossing to Florence and from 1.6 km below Darby to Como Bridge, East Fork Bitterroot River to Marten Creek, West Fork Bitterroot River to Painted Rocks Reservoir; Blackfoot River and tributaries; Clark Fork River from Anaconda settling ponds to Perkins Lane Bridge; entire Rock Creek; Georgetown, Lower, Middle, and Upper Thompson lakes

Flathead: Flathead, Little Bitterroot, Mary Ronan, and Notellum lakes; entire Flathead River; entire Middle and North forks Flathead River; South Fork Flathead River from Meadow Creek to Spotted Bear bridges; Swan Lake; Swan River from Piper Creek Bridge to Swan Lake

Kootenai: Kootenai River from Libby Dam to Kootenai Falls and From Libby Dam to Hyw. 37 Bridge

Nevada[4]

Nevada has no special regulation salmonid waters in the CRB

Oregon[5]

Deschutes: Metolius River, Deschutes River from mouth to Pelton Dam and above Crane Prairie; Crooked River below Bowman Dam and Deep Creek; Billy Chinook, Hosmer, East, Paulina, Fall, and Big Lava lakes; Crane Prairie and Wickiup Reservoirs

Hood: Hood River

Klamath: Upper Klamath, Blue, and Agency lakes; Klamath River

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[2]Source: Personal communication. 1995. C. Colter, Shoshone-Bannock Tribal Fish Biologist, Fort Hall, Idaho.

[3]Source: Personal communication. 1995. J. Vashro, Montana Department of Fish, Wildlife, and Parks, Kalispell, MT.

[4]Source: Personal communication. 1995. G. Johnson, Nevada Department of Wildlife, Elko, NV.

[5]Source: Personal communication. 1995. D. Hohler, Deshutes National Forest, Bend, OR; and D. Nolte, Trout Unlimited Oregon.

Malheur-Harney: Mann Lake, Upper Malheur River and North Fork; Chickahominy Reservoir

John Day: John Day River

Snake: South Fork Burnt River, Grande Ronde River, Imnaha River, Lakes in the Eagle Cap Wilderness

Other: Blitzen River, Trout Creek (Alvord Basin); Ana River (Summer Basin); Willow Creek

Washington[6]

Lower Columbia River below Snake River: Klikitat River, White Salmon River

Mid Columbia River from Snake River to Okanogan River: Entiat River, Wenatchee River, Icicle Creek, Methow River, Twisp River, Chelan River, Stehekin River, Chelan Lake and tributaries, Okanogan River, Similkameen River, Chewuch River

Yakima: Yakima River above Rosa Dam, Bumping River, Naches River, Leech Lake

Lost: Lost River

Upper Columbia River above Okanogan River: Kettle River, Spokane River, Colville River, Calispell Creek

Snake: Asotin River, Grande Ronde River

Other: Lenore and Bayley lakes

Wyoming[7]

Salt: Salt River from Afton to Thane

Greys: Greys River from Murphy to Corral, Stump and Barstow lakes

Gros Ventre: Gros Ventre River within the upper Gros Ventre Wilderness

Snake: Snake River from Jackson Lake to Wilson, Flat Creek within the National Elk Refuge, Green and Jackson lakes, Lake in the Woods, Grassy Lake Reservoir

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[6]Source: Personal communication. 1995. D. Fletcher, Washington Department of Wildlife, Olympia, WA.

[7]Source: Personal communication. 1995. R. Huddelson, Wyoming Game and Fish Department, Jackson, WY.

APPENDIX 4G

Historical Ranges Defined for Rare and Sensitive Fishes Within the Basin

Our estimates of the historic ranges of the rare and sensitive species were based on known historic distributions in published literature, available historic accounts, and speculative distributions based on personal interviews, expanded to include any occurrences in the status survey that were not included in the historic distributions.

White Sturgeon

Data sources include Brannon and Setter (1991) and Apperson and Anders (1991). In the Basin white sturgeon were known to exist historically in the Snake River from the mouth to Shoshone Falls. Historic photos demonstrate that white sturgeon occurred in the Salmon River upstream to at least the town of Salmon.[1] It is possible that the species was anadromous in much of the Basin before the construction of many dams on the Snake River. Kootenai Falls was a natural barrier to the upstream migration of white sturgeon above Kootenai Lake on the Kootenai/Kootenay River.

Klamath River Lamprey

Data sources are Vladykov and Kott (1979), and Moyle and others (1989). The Klamath River lamprey occurs in the Klamath River and Upper Klamath Lake in Oregon and California.

River Lamprey

The data source is Kan (1975). In the Columbia River, the river lamprey has been reported from the Bonneville Dam and locations downstream.

Pacific Lamprey

Pacific lamprey were believed to have migrated into all waters that supported anadromous fish. Kan (1975) stated that the primary consideration for occurrence of lamprey was access rather than distance from the ocean. The best records are available for Clearwater, Salmon, and Snake rivers. Their distribution in these systems lends support to the probability that lamprey were also found in the other accessible Snake River tributaries in Idaho. Anecdotal information suggests they were found in the Weiser[2] and Bruneau[3] rivers. No definitive records or collections were found for the Boise, Owyhee, or Payette river basins. Wallace and Ball (1978) documented landlocked lamprey that became parasitic above Dworshak Dam and Beamish (1980) similarly reported freshwater feeding. Beamish and Northcote (1989), however, reported that landlocked forms stopped attacking fish about seven years after dam construction and a transition to a freshwater form did not occur. As a result, Pacific lamprey are most likely extinct in landlocked areas that historically supported lamprey. Information for Oregon was compiled by P. Howell of

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[1]Personal Communication. 1995. R. Thurow, U.S. Department of Agriculture, Forest Service, Intermountain Research Station, Boise, ID.

[2]Personal communication. 1995. R. Wallace, University of Idaho, Moscow, ID.

[3]Personal communication. 1995. B. Parker, Columbia River Intertribal Fish Commission, Portland, OR.

the Deschutes National Forest in Ukiah, Oregon and includes Fulton (1968, 1970), ODFW unpublished data, personal communications with various biologists, Howell and others (1985), Bakke and Felstner (1990). Information for Washington was compiled by K. MacDonald, Wenatchee National Forest, Wenatchee, Washington. Other sources include Schoning (1947), Scott and Crossman (1973), Simpson and Wallace (1978), and IDAFS (1995).

Goose Lake Lamprey

Data sources include Moyle and others (1989) and Long and Bond (1979). The Goose Lake Lamprey is unique to the Goose Lake Basin, having been isolated during the early Pleistocene from the Pacific Lamprey, of which it is a subspecies.

Pit-Klamath Brook Lamprey

Data sources are Hubbs (1971) and Kan (1975). The species is reported from the North Fork Pit River, Pit River, Fall River, and Hat Creek, in California, the Sprague, North Fork Sprague, Sycan, and Williamson rivers, and Crooked and Meryl creeks in Oregon.

Sockeye Salmon

The 10 major watersheds and 23 lakes identified as part of the historic range for this species were based primarily on Fulton 1970, Wydoski and Whitney 1979, and Waples and others 1991a. Loon and Warm lakes within the South Fork Salmon River support what are believed to be native kokanee with anecdotal accounts of anadromous adults in those drainages[4] and are included but recognized as speculative. The Wilderness Society shows the Klamath basin as historic range for this species, while other speculate that the range may have extended to the Klamath or further south (Burgner 1991; Wydoski and Whitney 1979). We could find no evidence that sockeye ever occurred in the Klamath basin and several biologists experienced in the zoogeography of that basin were confident that they did not.[5] We therefore excluded the Klamath from the historic range.

Chum Salmon

Data sources are Fulton (1970) Nehlsen and others (1991). Chum salmon were distributed in the lower portions of lower tributaries upstream to the Umatilla River in Oregon and the Walla Walla River in Washington (Nehlsen and others 1991). Former spawning areas above Bonneville Dam included lower portions of the Little White Salmon River, Hamilton, Rock, and Herman Creeks, and areas along the margin of river banks in the main Columbia River (Fulton 1970).

Coho Salmon

Historic distribution are based on information listed in Howell and others (1984), Fulton (1970), Schoning (1947), Lane and Nash (1981), Parkhurst (1950), and Richards (1967). Personal communications with B. House, Bureau of Land Management, Boise, Idaho; Chuck Huntington, Clearwater Biostudies, Inc., Portland, Oregon; Monte Richards, retired, and Stacy Gebhards, retired, Idaho Department of Fish and Game, Boise, Idaho; Bob Schoning, retired, Bureau of Fisheries, Corvallis, Oregon

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[4]Personal communication. 1995. D. Anderson, Idaho Department of Fish and Game, McCall, ID.

[5]Personal communication. 1995. J. Williams, Bureau of Land Management, Boise, ID.

further defined historic ranges from historic records or anecdotal accounts. Personal communication between B. House, Bureau of Land Management, Boise, Idaho and Chris Frissell, University of Montana, Missoula regarding the wider historical distribution of coho salmon shown on Wilderness Society maps, assembled by Frissell, revealed that distribution was based on the assumption that coho salmon distribution overlapped to a great degree with spring chinook salmon distribution. Communications with Richards and Gebhards and the historical review of chinook salmon distributions strongly discount the occurrence of coho as a late spawning salmon in the Snake River above Hells Canyon.

Coastal Cutthroat Trout

Data sources include Behnke (1979a) and Nehlsen and others (1991). Within the Basin, coastal cutthroat trout occurred in Oregon to Fifteenmile Creek and in Washington to Rock Creek, including the Willamette River basin to its headwaters. The distribution also included populations in the Wind and Klickitat rivers of Washington, which are now extinct.

Lahontan Cutthroat Trout

Data sources include the USFWS (1993a, 1993b). Historically, Lahontan cutthroat trout were widespread throughout the basins of the Pleistocene Lake Lahontan.

Pygmy Whitefish

Data sources include Brown (1971), Draft System Operation Overview (1994), and Simpson and Wallace (1978). The historic distribution includes lakes of known current distribution, with no documented reports below Grand Coulee Dam on the Columbia River. During the last ice age, this species may have been widely distributed across North America. However, receding of glaciers most likely left the species as isolated populations (Wydoski and Whitney 1979).

Burbot

Data sources include IDAFS (1995) and the Idaho Conservation Data Center (1994). In Idaho, the Burbot historic range is limited to the Kootenai River drainage. Currently we have no information on other states within the Columbia River Basin.

Sand Roller

Data sources include Reimers (1963), Gray and Dauble (1979), Lee and others (1980), Simpson and Wallace (1982), Wydoski and Whitney (1979), and Pratt and Whitt (1952). The known historic distribution was limited, but occupation of the Columbia River is documented from Horseshoe Island Slough upstream to West Bar (south of Wenatchee), as well as in the lower Clearwater River in Idaho. Other rivers included the Yakima, Umatilla, Walla Walla and Willamette. Their complete historic distribution has not been documented, and their seclusive behavior during daylight hours suggests that the sand roller could have populations which had not been detected.

Pit Roach (California Roach)

The Pit Roach is the only subspecies of the California Roach known to occur in Oregon waters. The Pit Roach is in the upper Pit River system and the tributaries to Goose lake, which last overflowed into the Pit River in 1881 (Moyle 1976; Phillips and Van Denburgh 1971). Only their present distribution is listed here as historic information is virtually unknown.

Alvord Chub

The data source is Hubbs and Miller (1972). The Alvord chub was described in 1972 based on specimens collected from Trout Creek in the Alvord Basin, Oregon. The species is restricted to the Alvord Basin of southeastern Oregon and northwestern Nevada, but occurs rather widely within springs, creeks, and lakes of this area.

Borax Lake Chub

Data sources include Hubbs and Miller (1972), and Williams and Bond (1980). This species is known only from Borax Lake and associated waters in Harney County, Oregon. The Borax Lake Chub is closely related to the Alvord chub from which it became isolated as the waters of the pluvial Lake Alvord receded.

Catlow Tui Chub

Data sources include ODFW (1992), and Kunkel (1976). The Catlow tui chub was limited to the Catlow Valley in south-central Oregon, being found in Threemile, Skull, Home and Rock creeks which flow into the Catlow Valley.

Oregon Lakes Tui Chub

Data sources are Synder (1908) and Bills (1977). The Oregon Lakes tui chub is endemic to the Abert Lake Basin of south-central Oregon.

Summer Basin Tui Chub

Data sources are Synder (1908) and Bills (1977). Summer Basin tui chubs occurred at various localities within the Summer Lake Basin, including springs at the Summer Lake Post Office, Ana River, and source springs of the Ana River.

Sheldon Tui Chub

The data source is Williams and Bond (1981). The Sheldon tui chub is restricted to isolated waters of the Guano Basin of southeastern Oregon and northwestern Nevada.

Hutton Tui Chub

Sources include Bills (1977). This species has only been found in the two surface flow areas of Hutton Spring. These areas have very limited quantities of water. Hutton tui chub are believed to have been collected from the area as early as 1908.

Leatherside Chub

The data source is IDAFS (1995). Some have questioned whether this chub was historically found in Idaho or transplanted as a bait minnow so these distributions must be considered highly speculative. The areas listed represent confirmed (tributaries to the Raft, Goose, and upper Snake rivers and the Little Wood River) and unconfirmed (the Blackfoot, Big Wood, and Bruneau rivers and the Snake River near Payette).

Foskett Speckled Dace

The data source is Bond (1974). The Foskett speckled dace historically occurred only at Foskett Spring, located along the west side of Coleman Lake bed in Lake County, Oregon.

Jenny Creek Sucker

Data sources include Hohler (1981), J. Rossa,[6] U.S.Fish and Wildlife Service as reported in the USFWS Jenny Creek Sucker status report. The Jenny Creek sucker is a dwarf form of the Klamath smallscale sucker which has not yet been given official subspecific identification. It is morphologically different and reproductively isolated from the Klamath smallscale sucker by a series of three 30-foot high rock waterfalls. Current information reports that distribution is limited to the 26 miles of Jenny Creek and four of its tributaries, and those watersheds listed above. These are the only possible historical watersheds within the Basin. All occur just outside the assessment area.

Lost River Sucker

Sources include Andreasen (1975) and USFWS (1993b). In the Columbia River basin, large sucker populations were supported by the Upper Klamath and Agency lakes, and most associated tributaries. Historical occurrences were also documented in the mainstem of the Williamson and Sprague rivers. Sucker spawning likely occurred in the tributaries of these rivers as well,[7] but these watersheds are not included here due to speculation.

Warner Sucker

Data sources include Williams and others (1990), Andreason (1975), and Coombs and others (1979). This sucker was once common throughout the Warner Basin, with residents reporting having seen large runs of Warner suckers.

Goose Lake Sucker

The Goose Lake sucker is endemic to the Goose Lake Basin or south-central Oregon and northeastern California. Currently it is found in the lake and most of the tributaries, so much of the subbasin is included.

Shortnose Sucker

Data sources include Moyle and others (1989) and USFWS (1993b). The shortnose sucker is endemic to the Upper Klamath Basin of Oregon and California, known to be historically abundant within its range. Documented distributions include Upper Klamath Lake and its tributaries and Clear Lake, indicating that the sucker is native to the Lost River system. Spawning habitat potentially included any tributary to Upper Klamath and Agency lakes, which would include the Williamson and Sprague rivers, not listed here.

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[6]Personal communication. 1995. J. Rossa, U.S. Fish and Wildlife Service, Klamath Falls, OR.

[7]Personal communication. 1995. T. Noel, U.S. Forest Service, Pacific Northwest Research Station, Corvallis, OR.

Klamath Largescale Sucker

Sources include Ford and Thomas (1993). In the Columbia River Basin, large sucker populations were present in the Upper Klamath and Agency lakes, and most associated tributaries. Historical occurrences were also documented in the mainstem of the Williamson and Sprague rivers. Sucker spawning likely occurred in the tributaries of these rivers as well (see previous footnote), but these watersheds are not included here due to speculation.

Torrent Sculpin

Data sources include Scott and Crossman (1973), Maughan (1976), Wydoski and Whitney (1979), Northcote (1954), Wallace (1980), Simpson and Wallace (1982), Page and Burr (1991), Robins and others (1991), Reimers and Bond (1967) and Eddy and Underhill (1978) as listed by IDAFS (1995). In the watersheds where specific historic occurrence was unclear the range included all of the appropriate subbasin because torrent sculpin have been found to be widely distributed within their current range.

Shorthead Sculpin

Data sources include Bailey and Bond (1963), Reimers and Bond (1967), McPhail (1967), Maughan (1976), Maughan and Saul (1979), Wydoski and Whitney (1979), Hughes and Peden (1984), Bilby and Bisson (1992), Wallace (1982), Scott and Crossman (1973), Cannamela and Gasser (1978), Johnson and others (1983), Arthur and others (1984), Reynolds and others (1986), Simpson and Wallace (1982), and IDAFS (1995). The Shorthead sculpin has been collected from many watersheds in the Basin and has been shown to occur in the upper reaches and tributaries of large river systems. They also have been found in the sinks of the Arco desert.

Pit Sculpin

Data sources include Moyle (1976), and Long and Bond (1979). Although the Pit sculpin was widely distributed throughout the Pit River system (from the Goose Lake Basin to the Sacramento River), its occurrence in the Goose Lake Basin on the Oregon side only included a few tributaries, such as Cottonwood, Drews, and Thomas creeks (Long and Bond 1979). The sculpin has never been found in the lake itself, although it is unknown how widely distributed it was in the Goose Lake Basin historically.

Slender Sculpin

Data sources include Ford and Thomas (1993). Historical distribution was restricted to the Upper Klamath Basin above Klamath Falls. This included Upper Klamath and Agency lakes, and tributaries, and parts of the Williamson and Sprague rivers.

Margined Sculpin

Data from Lonzarich (1993) produced current distribution, listed here. Historical distribution data are lacking. Some historical records are available, but are unreliable due to the sculpin's similar morphology and overlapping distribution with *C. beldingi*.

Wood River Sculpin

Data sources include IDAFS (1995) and the Idaho Conservation Data Center (1994). The suggested historic range includes all permanent connecting waters from the Malad River falls at I-84 upstream into the Little Wood and Big Wood rivers and all connecting tributaries. The Wood River sculpin was probably the only species of sculpin present historically, in 1992 the Piute sculpin *C. beldingi* was collected in Silver Creek.

Shoshone Sculpin

The data source is IDAFS (1995). This sculpin's range was limited to canyon wall springs along the Snake River between Kanaka Rapids and Bliss, Idaho. They are only found where spring water excludes Snake River water.

Malhuer Sculpin

Data sources include Bailey and Bond (1963), Bisson and Bond (1971), and Bond (1974). The Malhuer sculpin is endemic to streams in the Harney Basin including the Silvies, Silver, and Blitzen river systems. Historic data suggest broad distribution throughout this restricted range.