3 Subbasin Assessment

3.1 Subbasin Overview

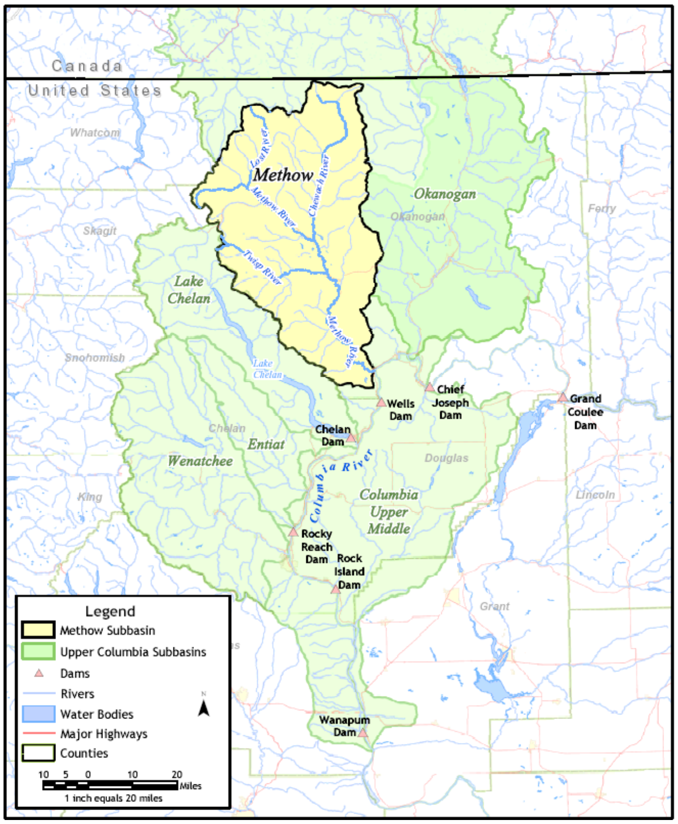
The Methow subbasin is located in north central Washington and lies entirely within Okanogan County. The subbasin comprises 12.7% of the Columbia Cascade Ecoprovince (CCP) and consists of 1,167,764 acres (1,825 $mile^2$) (**Table 1**).

Table 1 Subbasin size relative to the Columbia Cascade Ecoprovince and Washington State

Subbasin	Size		Percent of Ecoprovince	Percent of State
	Acres	Mi^2		
Entiat	298,363	466	3.2	0.7
Lake Chelan	599,925	937	6.5	1.4
Wenatchee	851,894	1,333	9.3	2.0
Methow	1,167,795	1,825	12.7	2.8
Okanogan	1,490,079	2,328	16.2	3.5
U. Mid Mainstem Columbia River	1,607,740	2,512	17.5	3.8
Crab	3,159,052	4,936	34.4	7.4
Total (Ecoprovince)	9,174,848	14,337	100	21.6

(IBIS 2003)

The Methow subbasin is one of more than 20 major Columbia Basin subbasins (seven in the CCP), its confluence being at river mile 524 near Pateros in north central Washington. The valley spans 1,667,742 acres in the northwestern segment of Okanogan County.



The Methow subbasin is characterized by large tracts of relatively pristine habitat contrasted with a close association with the growing population of subbasin citizens. Less than 2% of the subbasin's land is irrigated. Six fish species and fourteen wildlife species are as Endangered, Threatened, or as Species of Concern within the Methow subbasin.

Data Layers: Watersheds & Dams (StreamNet, TRIM), Counties & Major Rivers (WA Ecology, TRIM), Major Highways (WashDOT, ESRI) Projection: Washington State Plane North Zone NAD83. Produced by Jones & Stokes for KWA Ecological Sciences, Inc. Map Date: 5/15/2004

Figure 3 Location of Methow subbasin in relation to upper Columbia River dams and subbasins

Humans have occupied the region in and around the Methow Valley for at least 7,500 years. Ancestors of tribes that are presently part of the Yakama Nation and the Colville Tribes hunted, fished, and gathered food in the Methow subbasin area for thousands of years, and are an integral part of the heritage of the County and the Methow Valley subbasin.

Logging, mining, orcharding, farming, and grazing activities have played a substantial role in the Methow Valley for nearly a hundred years. Timber operations in the Methow watershed played an important role in the subbasin's economy through the 1980s. Activities related to timber harvest take place in the middle and upper reaches of the watershed.

Introduction of unlined irrigation agricultural canals to the Methow subbasin occurred in the 1800s as ranchers and farmers discovered that an irrigation system was required to supply consistent water for crops and livestock. The height of farming and ranching occurred in the Methow subbasin between 1940 and 1968 when 20,240 acres of land were irrigated from unlined surface diversions. Today, about 17,000 acres are under irrigation, and many of the subbasin farmers raise fresh fruit and vegetables to sell locally at the farmer's market, grocery stores and restaurants. (Methow Basin Watershed Plan, March 2004).

Farming and grazing are confined primarily to the lower and mid reaches of the subbasin. Orchards and small farms growing alfalfa and other irrigated crops constitute the majority of the subbasin's agricultural activities.

Recreation, tourism, and related development play an increasing role in the area's economy. The Methow Valley offers an extensive range of tourism- and recreational-related opportunities for the locals and tourists. Its natural setting is a destination for outdoor enthusiasts, and includes hundreds of miles of cross-country ski trails, snowmobile parks, and mountain biking, fishing, camping, and hiking areas. Dog sledding adventures, balloon rides, and llama pack tours are provided, along with many weekend get-a-way opportunities and accommodation options.

The Yakama Nation

The Yakama Nation has treaty rights to utilize Usual and Accustomed sites in the Methow subbasin. Those treaty rights give the Yakama Nation standing as a fish and wildlife co-manager under US vs. Oregon.

The Colville Tribes

The Methow Indians are a Plateau Salish people who speak a dialect of the Okanogan language very similar to the language of their close neighbors and relatives, the Entiat, Wenatchee, Okanogan and Columbia tribes.

The Methows historically relied on deer, elk, bear, mountain sheep, mountain goat, antelope, and many other animals in addition to roots, berries, and nuts for their traditional diet. The most important part of the traditional diet, however, consisted of large amounts of Pacific salmon including Chinook, sockeye, coho salmon and steelhead that were caught in the Methow River drainage and near the mouth of the river along the Columbia.

When the first European trappers arrived at the mouth of the Methow River in 1811, the Methows had at least ten villages stretching from the mouth of the river to the Chewuch. Small numbers of European trappers and travelers visited the region between 1811 and 1848 when the area became part of the United States. In 1855 the first Washington Territorial Governor, Isaac I. Stevens, attempted to involve the Methows in a treaty to cede their territory; however, the tribe chose not to participate.

The Methow tribe remained largely isolated from incoming settlers until the latter part of the 19th century, when their territory was encompassed in what was known as the Moses Columbia Reservation, a reservation set aside by executive orders of 1879 and 1880. As increasing numbers of settlers arrived, the United States negotiated an opening of the reservation amongst several Indian leaders (none of them Methow Indians).

In 1886, the reservation was opened to non-Indian settlement, and the Methows were promised a choice between taking allotments near where they lived and moving to the Colville Reservation. However, only the Methows near the mouth of the river were given the option, and almost all Methows eventually moved to the Colville Reserve where they became a constituent member of the Colville Tribes, the continuing legal representative of the tribes.

Almost all of the Methow Indian allotments in the Methow Valley were lost to non-Indians in ensuing years, and today only a few hundred acres within the Methow subbasin continue to be held in trust for the Methows of the Colville Indian Reservation. Descendents of the Methows, however, continue to hunt, gather, and fish for salmon in their usual and accustomed places, and Methows continue to assert a right to fish for salmon in their ancient ancestral lands.

Jurisdictional Authorities

Private land holdings within the Methow subbasin comprise roughly 15% of the total land. The remainder is managed by the US Forest Service (**Table 2**).

Table 2 Land ownership in the Methow Subbasin

Methow Subbasin	Federal Lands	Tribal Lands	State Lands	Local Gov't Lands	Private Lands	Water	Total (Subbasin)
Area in Acres	985,234	0	55,836	0	126,724	0	1,167,794

Source: IBIS, 2003

Over 80% of all of the lands in the watershed are managed by the U.S. Forest Service (USFS) (Methow Valley Water Pilot Planning Project Planning Committee 1994). The Pasayten Wilderness bounds the upper northern reaches of the Methow watershed, and the Lake Chelan-Sawtooth Wilderness sits along the southwest rim of the basin. Both areas range from over 5,000 feet in elevation to peaks approaching 9,000 feet, and are managed as wilderness ecosystem reserves and wildlife habitat; activities include non-motorized recreation as well as limited mining and grazing activity.

The remainder of the USFS-managed land lies in the Okanogan National Forest, and is managed for multiple use, including commercial logging, cattle grazing, mining, wildlife habitat, and recreation (Methow Valley Water Pilot Planning Project Planning Committee 1994).

The Federal Bureau of Land Management (BLM) manages approximately 1% of the land in the subbasin. BLM land consists mainly of mixed forest and grassland, and is used for commercial logging, grazing and recreation.

The State of Washington manages 5% of the land in the basin. Of this State land, 51% is managed by DNR, and 49% is managed by WDFW. Department of Natural Resources (DNR) manages their land for timber harvest, wildlife habitat, recreation, and grazing. The WDFW lands comprise the Methow Wildlife Area, which is managed for wildlife habitat, recreation, and grazing (Methow Valley Water Pilot Planning Project Planning Committee 1994).

The DNR manages more than 5 million acres of forest, range, agricultural, and aquatic lands. These lands produce income to support state services and to provide other public benefits. Nearly 3 million acres are state trust lands, most of which were given to Washington at statehood by the federal government.

Population and Growth Management

At present, approximately 5,000 people live within the 1,890 square mile Methow subbasin (2000 Census; Washington State Office of Financial Management). The population of major subbasin counties is summarized in **Table 3**.

Between 1990 and 2000, the population of Winthrop increased by 27.5% to reach its current population of approximately 385 people, and the town of Twisp had a population increase of about 13.5%, and Pateros experienced a population gain of 11.4% (Washington State Office of Financial Management).

The populations of unincorporated towns, including Carlson, Mazama, and Methow are unavailable. The County of Okanogan, including the Methow Valley, has a population density of 7.52 persons/mile^2.

Most of the population is concentrated on private lands within and near the towns of Pateros, Twisp, and Winthrop, and the unincorporated areas of Carlton, Mazama, and Methow. The unincorporated total is the tract population minus populations of Twisp and Winthrop that are already included.

Table 3 Population of major Methow subbasin counties (1990-2000)

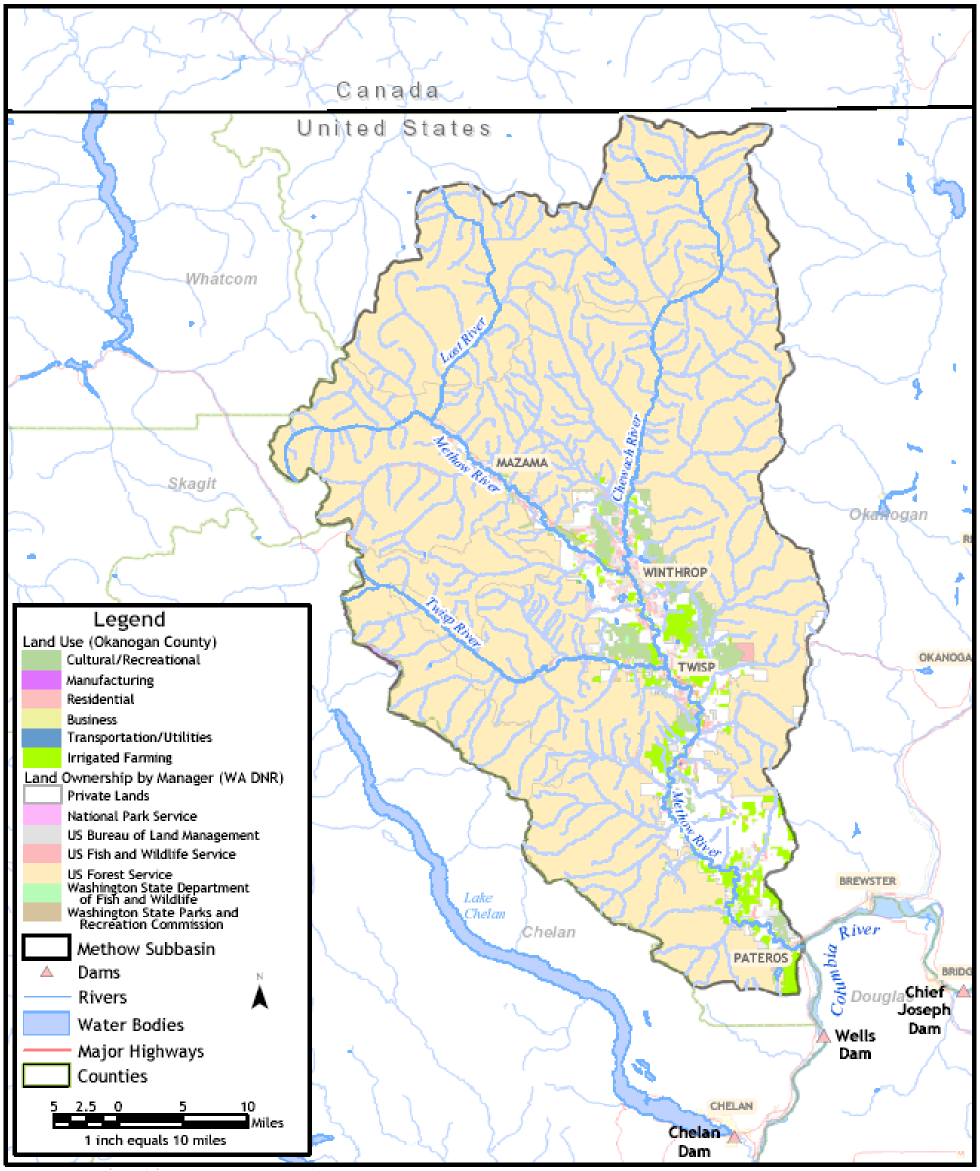
Methow Valley Subbasin	1990 Population	2000 Population	Area (mi^2)	People/mi^2	Population/mi^2
Carlton	332	567		0-15	
Mazama	115	96		0-15	
Methow	623	262		0-15	
Pateros	570	643	0.51	17-40	1261.8
Twisp	872	938	1.16	0-15	899.8
Winthrop	302	349	0.88	0-15	400.7
Methow Subbasin Total		5384			
Unincorporated Total		4097			

Source:U.S. Census Tracts ID #9709 - #9710, Washington State Office of Financial Management)

Agriculture

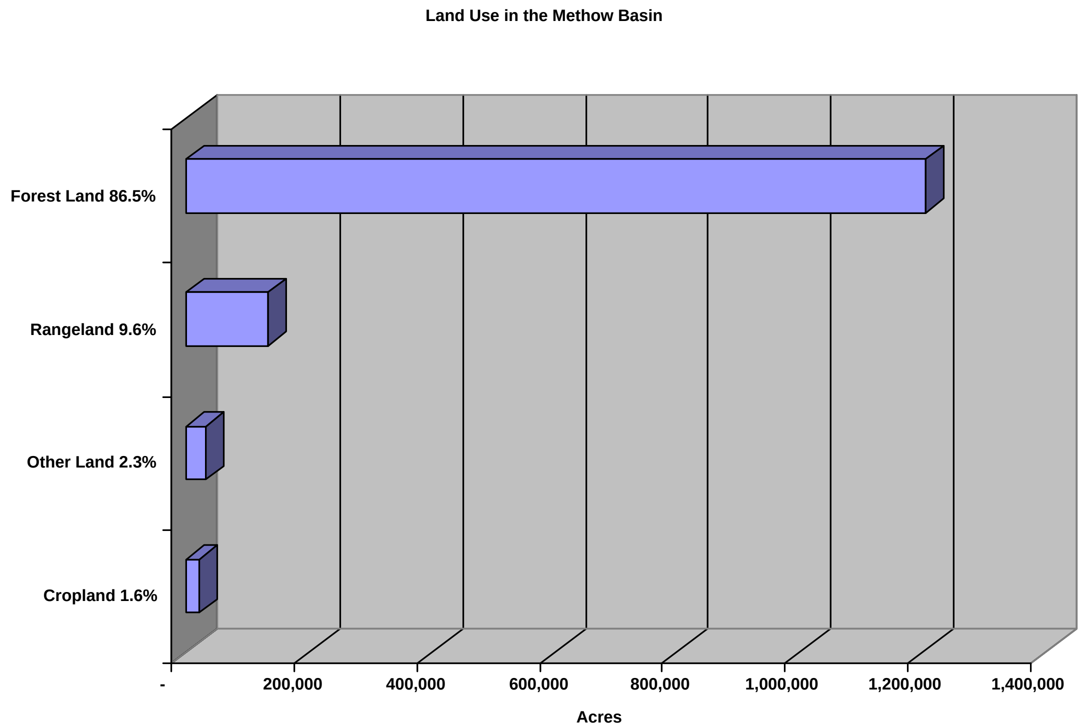
Land use includes significant rangelands, crops, and other uses (**Figure 4** and **Figure 5**) Roughly 12,800 acres of the Methow basin is cultivated (Methow Valley Water Pilot Planning Project Planning Committee 1994). Orchards and small farms growing alfalfa and other irrigated crops constitute the majority of the subbasin’s agricultural activities.

Farming and grazing are confined primarily to the lower and mid reaches of the subbasin.



Data Layers: Land Use (Okanogan County, WA DNR), Subbasins and Dams (StreamNet),
Counties & Major Rivers (WA Ecology), State Routes (WashDOT).
Projection: Washington State Plane North Zone NAD83.
Produced by Jones & Stokes for KWA Ecological Sciences, Inc. Map Date: 5/15/2004

Figure 4 Land use in the Methow Subbasin



Source: PNWRBC 1977a

Figure 5 Land use in the Methow Subbasin

It is noted, however, that not all agricultural activities result in negative impacts to fish and wildlife and their habitats. As such, each situation should be evaluated on an individual-by-individual basis. Additionally, in the U.S. portion of the Okanogan subbasin, land being converted to agriculture is not occurring at previously reported rates. In fact, agriculture as a whole is declining in the US portion of the Okanogan subbasin. In Canada, conversion of land to agriculture is occurring at an increasing rate over the past decade. (J. Dagnon 2004, pers. comm.)

Forest practices

Timber operations in the Methow watershed played an important role in the subbasin's economy through the 1980s. Years of logging have contributed to high road densities in some portions of the watershed. Timber has been harvested extensively from the Beaver Creek drainage since the 1960s (USFS 2000a).

Currently, DNR protects 12 million private and state-owned forested acres from wildfire. DNR administers Forest Practices Board rules on 12 million forested acres.

Mining

Mining activity in the Methow subbasin is currently minimal; however, historically, mining was prevalent in the subbasin.

Transportation

County roads and state highways parallel both sides of the Methow River along its entire length within the subwatershed. Road densities within the Beaver Creek drainage of the subwatershed are the highest in the Methow watershed with 41% of the drainage having road densities of 2.1 to 5 miles/mile^2 (USFS 1997).

Topographic / Physiogeographic Environment

Topography within the subbasin ranges from mountainous sub-alpine and alpine terrain along the Cascade Crest to the gently sloping wide valley found along the middle reaches of the Methow River. Elevation varies from over 8,500 feet in the headwaters of the basin along the crest of the Cascade Mountains, to approximately 800 feet at the confluence of the Methow and Columbia Rivers. Topographic features in and adjacent to the Methow Valley provide evidence of both alpine and continental ice-sheet types of glaciation (Waitt 1972 in NPPC 2002).

The western upper reaches of the Methow watershed carve deeply into the Cascade Crest's peaks. Avalanche chutes, knife-edge ridges, and cirques typify the upper elevations of the watershed following the crest. The upper Methow River valley is a U-shaped, glaciated intermountain valley. The valley margins are bounded by bedrock uplands that rise steeply, and at some locations, nearly vertically, from the valley floor to elevations over 5,000 feet.

The elevation of the valley floor within the upper valley varies from approximately 2,600 feet above Lost River to about 1,765 feet at Winthrop, a distance of roughly 21 miles. The valley floor from Lost River to Winthrop ranges between 0.5 mile to 1.5 miles wide, and consists of irregular terraces, alluvial fans, and floodplain meadows. From Winthrop downstream to the town of Twisp, the valley opens out and the slope decreases to approximately 17.0 feet/river mile (Okanogan County 1996 in NPPC 2002).

Roughly 50 to 65 million years ago, the North Cascade subcontinent docked against the Okanogan subcontinent. As the two continents collided, numerous north-to-south faults formed throughout the region that presently includes the Methow subbasin. The dominant tectonic feature distinguishing the area is the Tertiary Methow-Pasayten Graben. Over millions of years, repeated occurrences of folding transformed and redefined the Methow-Pasayten Graben, with at least four distinct episodes culminating in the present geologic composition of the region (Barksdale 1975 in NPPC 2002).

The resulting bedrock geology of the Methow Valley area is characterized by folded Mesozoic sediments and volcanic rocks down-faulted between crystalline blocks. The sediment strata include varieties of sandstones, shales, siltstones, conglomerates, and andesitic flows, breccias and tuffs. The crystalline rocks include various granitic types, igneous intrusive rocks, and high-grade metamorphic types, including gneiss, marble, and schist (Barksdale 1975 in NPPC 2002).

The valley's bedrock is overlain with a thick sequence of highly permeable unconsolidated sediment composed of pumice, ash, alluvium and glacial outwash. The majority of the subbasin's aquifers rest within this unconsolidated sediment layer, confined from below by the relative impermeability of the underlying bedrock (EMCON 1993 in NPPC 2002). Quartz and feldspar are the dominant minerals in the silt and sand fractions of sediment from the Methow River.

Soils

Methow valley soils are generally coarsely textured compositions of glacial till. The primary constituent materials are granitic, volcanic, and sedimentary (**Figure 6**). Unconsolidated materials including glacial drift, pumice and ash deposits, and alluvial plain and fan deposits, are also present. (EMCON 1973). The valley's topsoil generally consists of sandy loams with permeability ranges between 2.0 to 6.0 inches/hour.

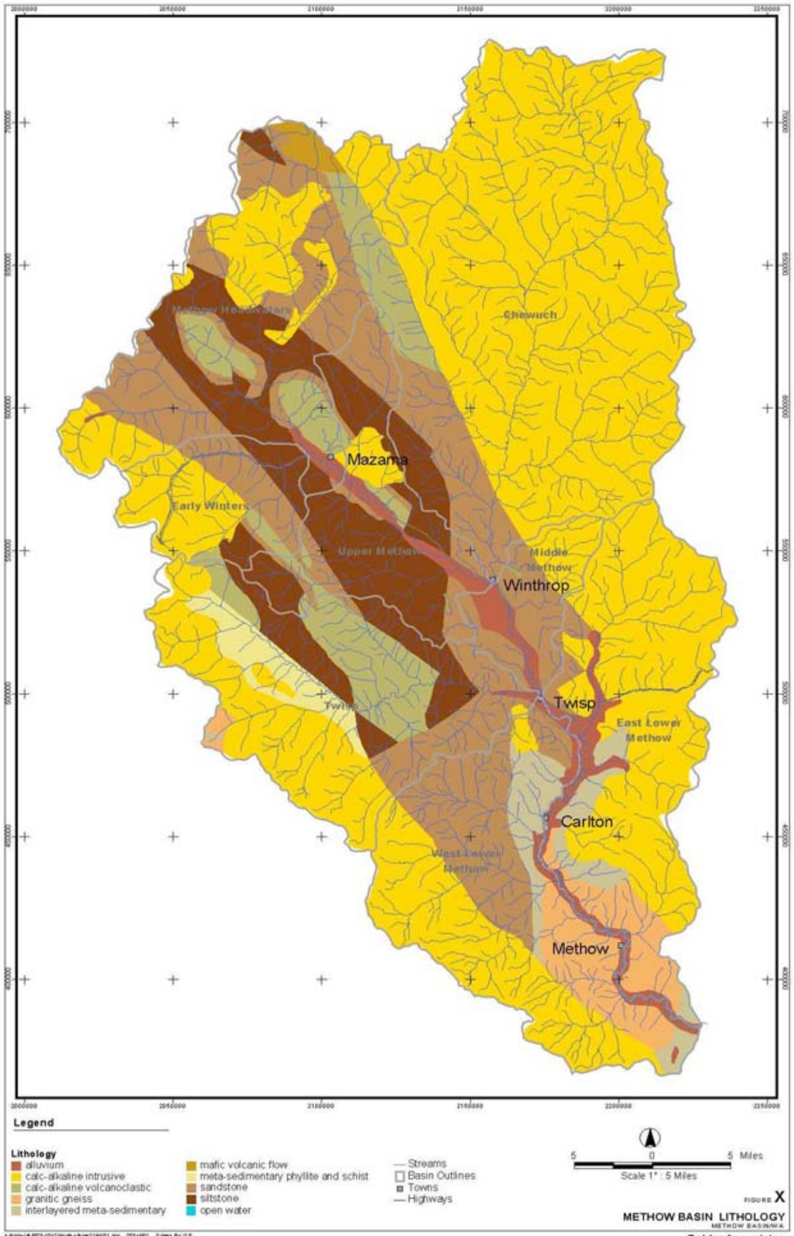


Figure 6 Methow subbasin lithology

Underneath these topsoils lie alluvium and glacial outwash materials that exhibit permeability greater than 6 inches/hour (Waitt 1972). In some areas of the valley, relatively non-porous layers of soils with permeability less than .01 inches/hour lie between the layers of alluvium (Waitt 1972).

Climate and Weather

The Methow subbasin's climate is influenced by maritime weather patterns, elevation, topography, and its location on the leeward side of the Cascade Mountains. Pacific storms driven by prevailing westerly winds are routinely interrupted by the Cascade Mountains, dropping heavy precipitation throughout the upper elevations. Precipitation falls off significantly as elevation decreases and as the distance from the Cascade Crest increases. Continental weather patterns insinuate themselves periodically throughout the winter months, forcing blasts of cold air masses southward from Canada.

The mean annual precipitation in the Methow subbasin is shown in **Figure 7**. Nearly two-thirds of the watershed's annual precipitation occurs between October and March, arriving primarily as snow. In the summer, long spells of hot, dry weather are punctuated by intense, but short-lived, thunderstorms. Fall brings increased precipitation that generally climaxes as winter snowfall between December and February. Snow usually blankets the ground from December through February at lower elevations, while at higher elevations, snow cover lingers from October through June. The upper reaches of the watershed along the Cascade Crest (at elevations of approximately 8,600 feet) receive as much as 80 inches of precipitation a year. This drops to about 60 inches in adjacent upland areas, while the town of Pateros (800 feet), at the far southern end of the subbasin, receives only about 10 inches of precipitation annually (Richardson 1976).

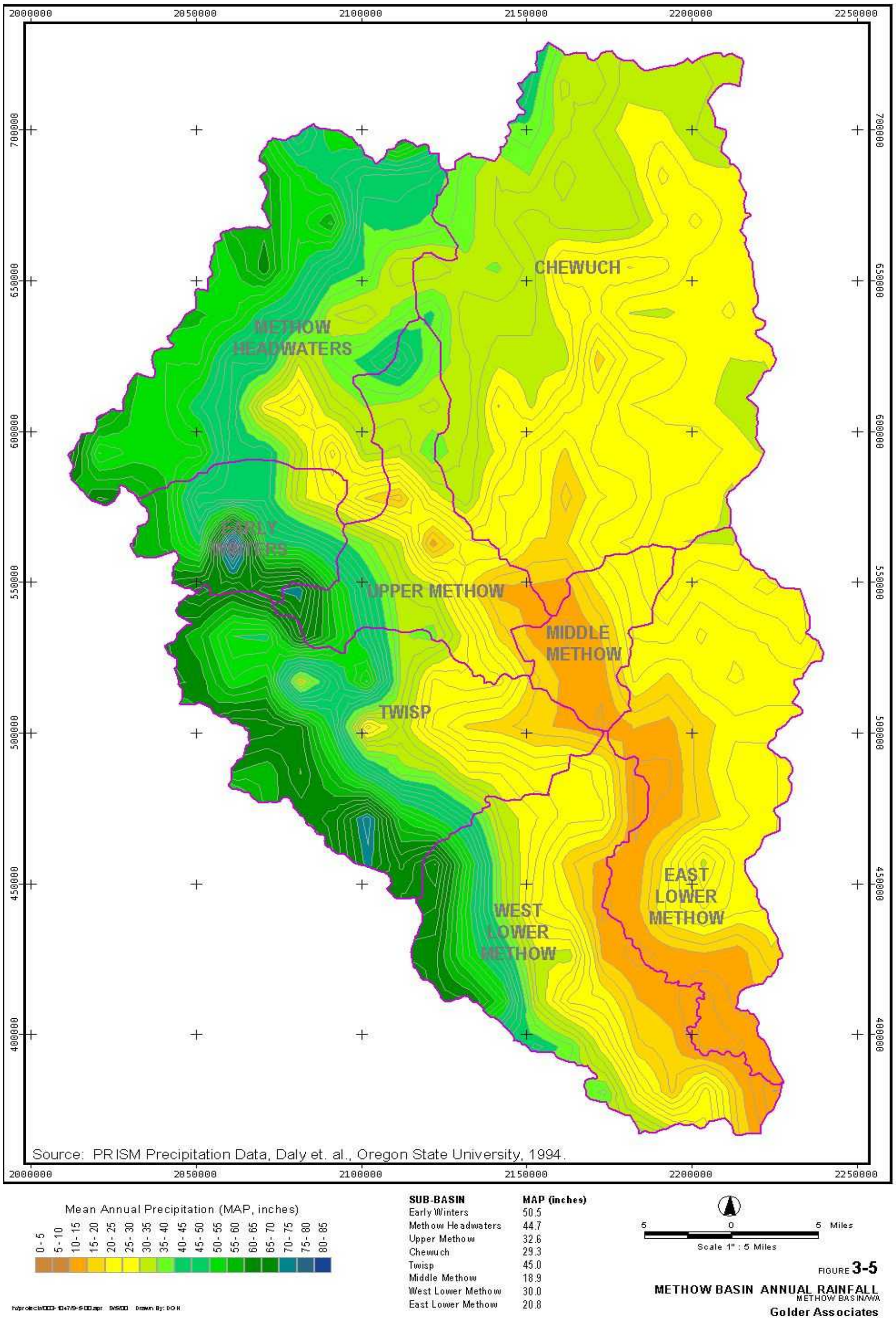
The Methow subbasin falls within the coldest of twenty-four western climate zones. The watershed is at the same latitude as Duluth, Minnesota, and Bangor, Maine. Additionally, temperatures within the basin are dictated by the fact that mean elevation within the basin is roughly a mile above sea level.

Winter low temperatures in the Methow range down to –35° F, with a monthly mean January temperatures, between 1970 and 1990 at Mazama, of 8.6° F. Average maximum temperatures in August for the upper watershed elevations range from 60° F to 70° F, with occasional highs up to 80° F. At lower elevations, August high temperatures range from 80° F to 95° F, with temperatures occasionally exceeding 100° F.

Water Resources

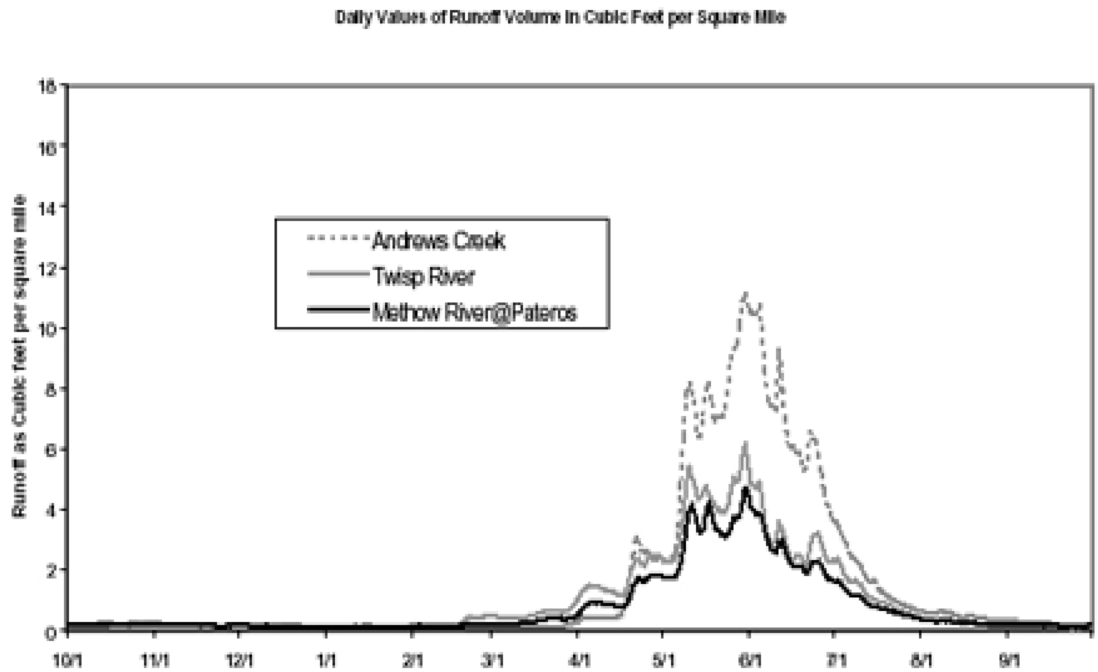
Hydrography and Watersheds

The Methow River near Pateros has a long-term mean discharge rate of 1600 cfs (45 m^3/s), or a mean annual yield of 1.2 x 106 acre-foot/year (1400 x 106 m^3/yr). Average annual runoff from the Methow basin is 12 inches (Figure 8).



Source: Golder and Associates 2003

Figure 7 Mean annual precipitation in the Methow subbasin



Source: Draft Methow River Basin Plan, 1994.

Figure 8 Daily values of runoff volume in cubic feet/mile^2

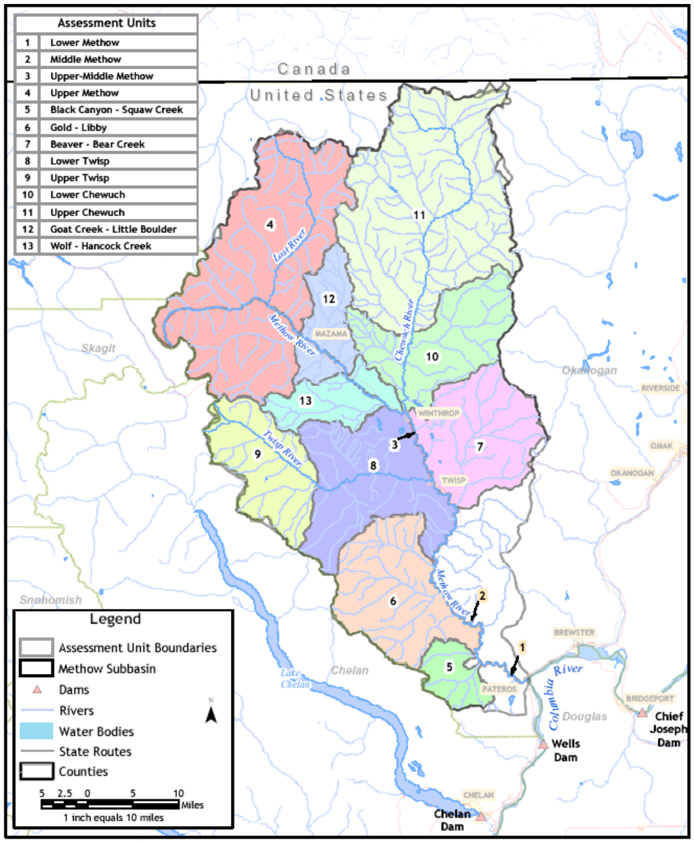
Snowmelt from the upper elevations of the Methow basin in spring and early summer generates most of the runoff in the basin, with 44-71% of the annual runoff volume occurring during May and June. Annual peak discharge occurs during May and June, as well, with the flood of record occurring on May 29, 1948 (Kimbrough et al. 2001).

The timing of spring snowmelt is triggered by a combination of seasonal temperature changes and elevation. Low summer precipitation, higher temperatures, and declining snow pack contribute to receding stream flow beginning in July and continuing through September.

The lowest stream flows occur in mid-winter (December to February) and early autumn (September) when stream flow is primarily the result of groundwater discharge, supplemented to a limited extent by snowmelt and storm runoff. During these periods, surface flow ceases in some streams and along reaches of rivers where stream flow is lost to groundwater, though the relationship between surface and ground water in the Methow subbasin is not fully understood.

Drainage Area

The Methow River drains an area of approximately 1,890 mile^2 (about 1,193,933 acres) (Golder 1993; Methow Valley Water Pilot Planning Project Planning Committee 1994; CRITFC 1995). The Methow River subbasin has seven primary subwatersheds (**Figure 9**): the Upper Methow River, Lost River, Early Winters Creek, Chewuch River, Middle Methow River, Twisp River, and Lower Methow River.



Data Layers: Assessment Units (TTFWI), Subbasins and Dams (StreamNet), Counties & Major Rivers (WA Ecology), State Routes (WashDOT).
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Figure 9 The Methow subbasin and primary subwatersheds

The Lost River subwatershed is aligned from north to south. At 107,400 acres, this subwatershed makes up roughly 9% of the Methow subbasin's total acres. Nearly 95% of that land lies within the Pasayten Wilderness. Descending steeply from its nearly pristine headwaters at elevations close to 6,900 feet, Lost River flows roughly 22.5 miles before joining the Methow River (RM 73.0) about six miles upstream from the Early Winters Creek confluence at about 2,600 feet (USFS 1999c). The main creeks and streams are shown in **Table 4**. No towns are located within this drainage.

Table 4 Creeks and streams within the Lost River Subwatershed

Lost River Subwatershed (107,400 acres)		
Drake Creek	Monument Creek	Eureka Creek

The Upper Methow River subwatershed drains an area of approximately 322,385 acres. It is the second largest subwatershed within the Methow subbasin, comprising approximately 27% of the total basin's drainage. Included within this region is the upper Methow River from its headwaters (RM 73.0) downstream to the Chewuch River's confluence (RM 50.1), with the Methow at the town of Winthrop.

Tower Mountain (elevation 8,844 feet), Mt. Hardy (8,880 feet) and Hart's Pass (6,178 feet) rim the upper edges of the Methow's headwaters along the slopes of the Cascade Crest. This stretch of the Methow takes in approximately 35 river miles from the headwaters to the southern tip of the subwatershed at town of Winthrop (1,760 feet). The town of Mazama also lies within the subwatershed about 1.5 miles upstream from Goat Creek's confluence with the Methow River.

The upper reaches of the Methow are shown in **Table 5**. The main tributaries within this drainage, Goat Creek and Wolf Creek, flow through relatively high gradient gorges and steep valleys. The river begins to meander and braid below the Goat Creek confluence where the river's gradient is much lower (approximately 0.37%, a drop of 264 feet in 13.4 miles).

Table 5 Creeks and streams within the Upper Methow River Subwatershed

The Upper Methow River Subwatershed (322,385 acres)		
Brush Creek	Trout Creek	Rattlesnake Creek
Robinson Creek	Gate Creek	Little Boulder Creek
Goat Creek	Fawn Creek	Hancock Creek
Wolf Creek	Little Falls Creek	

The Early Winters Creek drains a north-to-south oriented watershed of some 51,548 acres. The drainage, which is capped by North Gardner Mountain (8974 feet) and Cutthroat Peak (7046 feet), comprises nearly 4% of the entire Methow subbasin (USFS 1996a).

The mainstem originates near Liberty Bell Peak at 6,500 feet, and drops approximately 4,360 feet over the course of 15.7 miles before meeting the Methow River (RM 67.3) some 3.5 miles upstream from the town of Mazama. The drainage's headwaters are defined by cirques and glaciated head walls, which in turn give way to U-shaped glacial valleys and then to valley bottoms lined with glacial till. An impassable waterfall exits at RM 8 of Early Winters Creek. The main tributaries to Early Winters Creek are shown in **Table 6**. There are no towns located within the Early Winters subwatershed.

Table 6 Creeks and streams within the Early Winters Subwatershed

Early Winters Subwatershed (51,548 acres)		
Varden Creek	Cedar Creek	

The Chewuch River drainage is the largest subwatershed within the Methow subbasin. The Chewuch empties a 340,000-acre basin over the course of its 44.8-mile north-to-south journey from its headwaters to its mouth at the town of Winthrop (1,700 feet) (USFS 2000c).

Nearly 108,000 acres (34%) of the subwatershed's northern and western reaches sit within the Pasayten Wilderness. Cathedral Peak (8,601 feet), Windy Peak (8,331feet), and Andrew Peak (8301 feet) stud the subwatershed's defining crest. The U-shaped valley, in the upper reaches of the Chewuch drainage, features dramatically steep slopes often in excess of 60-70%. Upstream migration routes, along the uppermost reaches of all of the Chewuch's tributaries, are blocked by naturally occurring impediments, including waterfalls and steep gradients. The main tributaries to the Chewuch River are shown in **Table 7**.

Table 7 Creeks and streams of note within the Chewuch River Subwatershed

Chewuch River Subwatershed (340,000 acres)		
Dog Creek	Thirtymile Creek	Andrews Creek
Lake Creek	Twentymile Creek	Falls Creek
Eightmile Creek	Cub Creek	Boulder Creek

The Middle Methow River subwatershed contains 15,600 acres (about 1% of the subbasin total). This subwatershed includes the mainstem Methow River from its confluence with the Chewuch River at Winthrop (1,700 feet) downstream to the town of Carlton (1,420 feet), a distance of approximately 23 river miles.

In the lowest reaches of this subwatershed, the river meanders at a low gradient through a floodplain that is largely confined.

The main tributaries to the Middle Methow are shown in **Table 8**.

Table 8 Creeks and streams within the Middle Methow River Subwatershed

Middle Methow River Subwatershed (15,600 acres)		
Bear Creek	Alder Creek	Beaver Creek
Blue Buck Creek	Frazer Creek	Benson Creek

The Twisp River drains a subwatershed of roughly 157,000 acres, comprising approximately 13% of the Methow subbasin. Extending about 28 river miles from its headwaters in the Lake Chelan-Sawtooth Wilderness to its mouth, the river flows generally from east to west before joining the Methow River at the town of Twisp (RM 40.2).

Nearly half of the subwatershed is part of the Lake Chelan-Sawtooth Wilderness, and the upper fringe is ringed by multiple peaks and razor ridges, including Star Peak (8,680 feet) and Gilbert Mountain (8,023 feet). From these steep headwaters, the Twisp descends to an elevation of 1,600 feet at its confluence with the Methow River. In the upper reaches, natural falls block migration passage along some tributaries. Within its lower reaches, the Twisp River follows a low-gradient meander through a floodplain that is somewhat confined. The main tributaries to the Twisp River are shown in **Table 9**.

Table 9 Creeks and streams within the Twisp River Subwatershed (listed from upstream to downstream reading across the table)

Twisp River Subwatershed (157,000 acres)		
North Creek	South Creek	Reynolds Creek
Eagle Creek	War Creek	Buttermilk Creek
Canyon Creek	Little Bridge Creek	Newby Creek
Poorman Creek		

The Lower Methow River subwatershed includes a low gradient, 27-mile stretch of the Methow, starting at the town of Carlton and flowing northwest to southwest towards the town of Pateros. The least studied of the basin's subwatersheds (WSCC 2000), this area includes about 200,000 acres, with the majority of those contained in the Okanogan National Forest.

A small portion of the subwatershed falls within the Lake Chelan-Sawtooth Wilderness. Elevation ranges from 8,646 feet at Hoodoo Peak to 800 feet at the confluence of the Methow and Columbia Rivers (USFS 1999a). The upper valley is about a mile wide, narrowing in the lower reaches to less than a half-mile (USFS 1999a). State Highway 153 parallels and laces the entire stretch of the Methow River in this reach, crossing the river seven times between the towns of Methow and Carlton. The main tributaries to the Lower Methow are shown in **Table 10**.

Table 10 Creeks and streams within the Lower Methow River Subwatershed

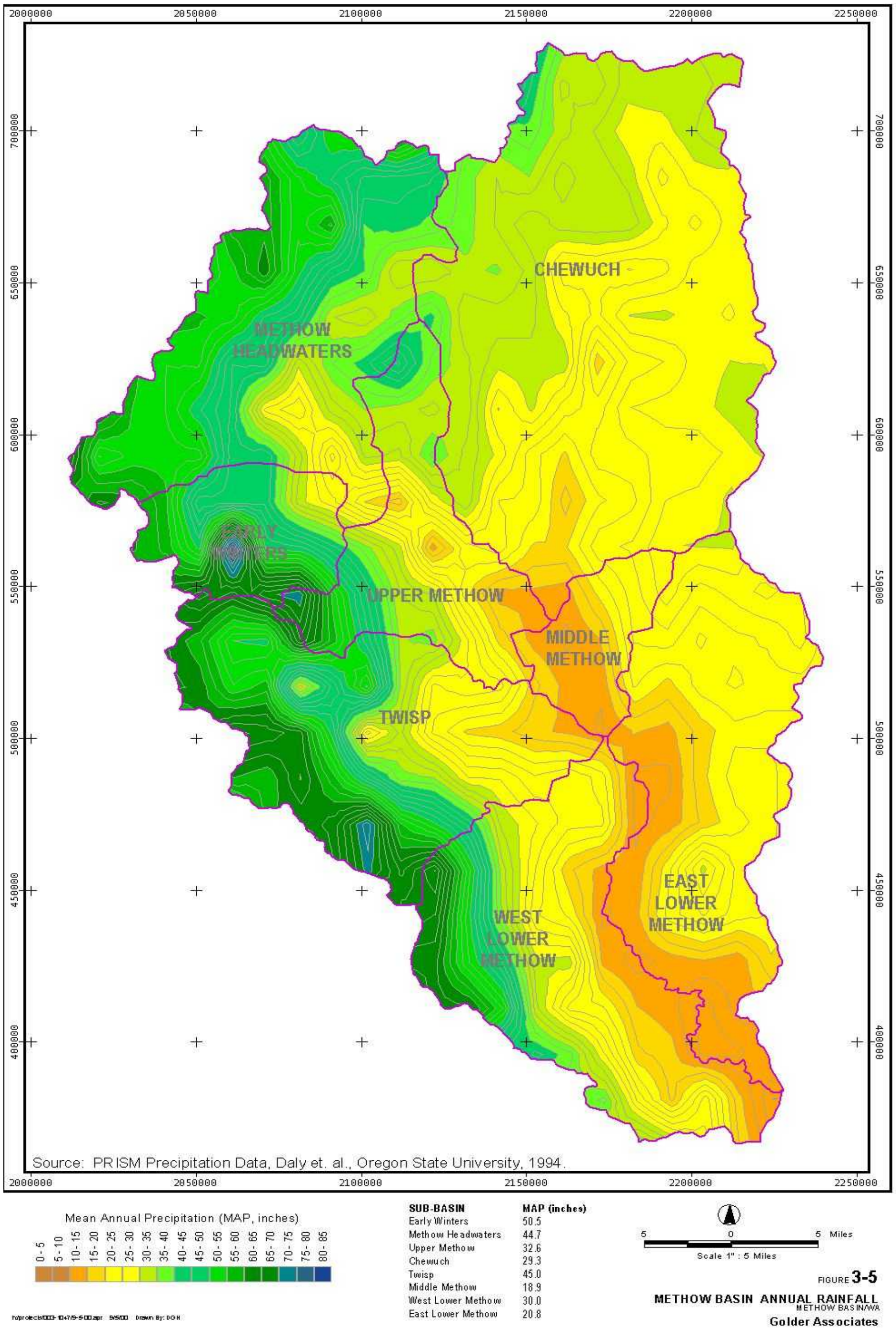
Lower Methow River Subwatershed (200,000 acres)		
Texas Creek	Libby Creek	Gold Creek
McFarland Creek	French Creek	Black Canyon Creek

Hydrologic regimes

The U.S. Geological Survey (USGS) has been collecting stream flow and other hydrologic data, and investigating water resource issues in the Methow River basin since the early 20th century. The USGS operates a network of 15 continuous stream flow gauges in the Methow River basin including eight "real-time" stations that transmit current stream flow information to the USGS's web-accessible database, the National Water Information System. The gauging network extends from the main tributaries of the Methow River to a series of gauges along the mainstem. The stream gauge at Andrews Creek serves as one of the Nation's hydrologic benchmark stations, which provide information on stream flow from basins with limited human influences.

Water resources are important to the residents and ecosystems of the Methow subbasin. People depend on reliable, high-quality water supplies for their domestic and agricultural uses, and aquatic organisms depend on stream flow from snowmelt and groundwater discharge to survive in an otherwise arid environment.

To improve the understanding of the quantity and quality of water resources of the Methow subbasin both spatially and temporally, it is important that hydrologic data are collected throughout the basin over periods spanning a range of climatic conditions. Long-term hydrologic data have been collected at some points in the basin, but generally, the information is limited. Annual precipitation varies from 10 inches annually in the valley bottom, to 70 inches annually in the valley headwaters (**Figure 10**).



Source: Golder Associates, 2001

Figure 10 Annual precipitation in the Methow subbasin

Hydrologic data of interest include long-term records of stream flow discharge, temperature, and sediment loading, irrigation diversions and application rates, and groundwater levels in the unconsolidated sediments of the basin. Currently, an extensive network of 27 stream flow gauges is operated in the Methow River Basin.

Except for seven USGS gauges that have been in operation for more than a decade, most of these gauges have been in operation for about one year. Once continuous records of hydrologic conditions have been measured throughout the basin over a period spanning wet and dry years, the records can be evaluated to determine whether some stations indicate broader conditions and, thus, provide the core physical information for a water resources management system.

The “natural-flow” watershed model in the Methow subbasin needs to be updated by including the effects of diversions. Currently no watershed management tool exists for the Methow River subbasin to estimate the cumulative effects of natural variability in stream flow and irrigation diversions and returns. The USGS recently completed a watershed model that can be used to estimate natural stream flows; however, it needs to be improved by incorporating newly collected data, and by simulating irrigation diversions and returns.

Leaking irrigation canals may return some of the diverted river water to the groundwater system. The valley-fill groundwater system is connected to streams and contributes groundwater discharge to stream flow along selected stream reaches. Increased groundwater levels that may result from leaking irrigation canals may increase groundwater contributions to stream flow.

To date, the timing and amount of the possible increase in groundwater contributions to stream flow are not known. In a current study, the USGS has instrumented part of the Twisp subwatershed to investigate the groundwater/surface-water interactions in the Twisp River. Data have been collected since the beginning of the 2001 irrigation season, and will be analyzed later in 2001 and 2002. Continued data collection in the existing study area and, potentially, other areas of the basin, would improve estimates of irrigation canal leakage and groundwater discharge to streams, particularly during non-drought years.

Forest management, including tree harvesting, road building, and fires, alter the density and type of vegetation in parts of the Methow River Basin. Cumulative effects of these land use changes may affect the accumulation and melting of the snowpack, snowmelt, and rainfall runoff patterns, and soil erosion.

Changing land use may affect stream flow temperatures by changing the quantity and timing of stream flow and by changing the degree of shading from vegetation. If stream flow temperatures are changed significantly from natural conditions, habitat may be less favorable for salmonids. Currently, no modeling has taken place in the Methow River subbasin to predict the effect of land use practices on stream flow temperatures.

Bank protection and flood control projects in the Methow River Basin have modified the development and maintenance of floodplain and off-channel habitat for salmonids.

Impoundments and Irrigation Projects

Figure 11 shows the major streams, dams, and irrigation projects for the Methow. There is currently no hydropower development within the Methow subbasin. A hydroelectric project constructed by Washington Water Power (thought to have been in 1911) blocked fish passage in the Methow River at Pateros until its removal in 1929. The dam blocked all fish passage during those years and by the time it was removed, the Methow River run of coho was extinct, and runs of spring and summer Chinook, as well as steelhead, were severely depressed.

The confluence of the Methow River is located at RM 523.9 of the Columbia River. Today, anadromous fish, migrating to the ocean, encounter Wells Dam just downstream from the Methow's confluence with the Columbia River. Beyond Wells Dam, eight more downstream dams along the Columbia River impede fish passage to the ocean.

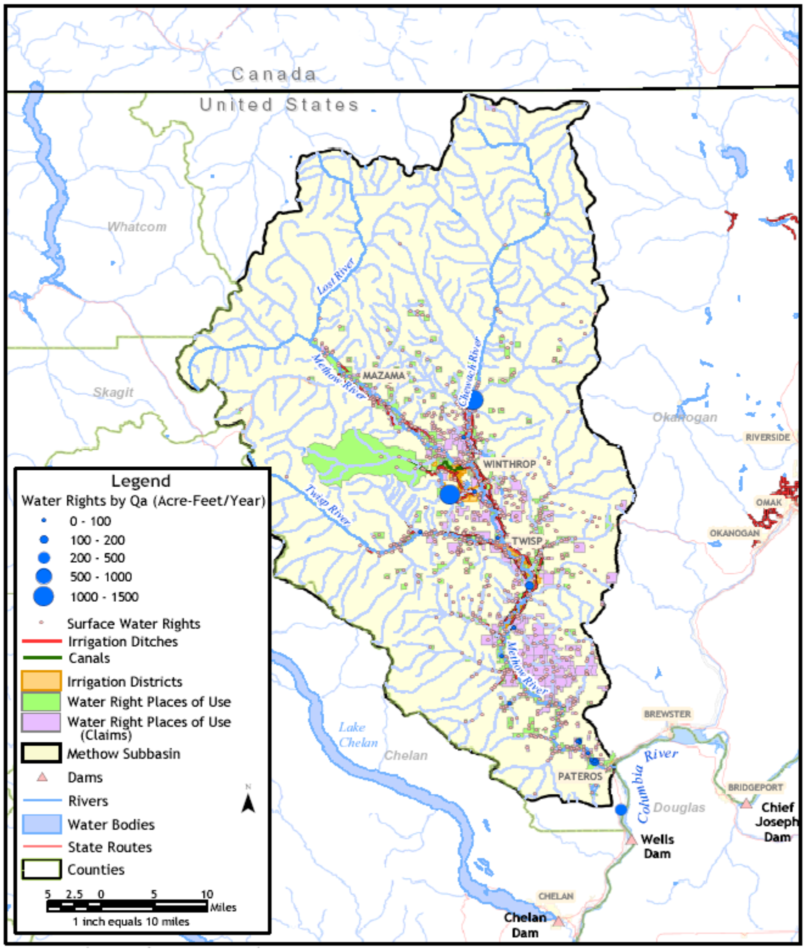


Figure 11 Major streams, dams, irrigation projects, for the Methow subbasin

There are currently two irrigation districts within the Methow subbasin; these are the Wolf Creek Reclamation District and the Methow Valley Irrigation District. All other irrigation ditches in the Methow subbasin are privately owned by their shareholders.

Historically, the majority of irrigation within the basin was delivered through a network of unlined ditches. Currently there are at least 27 irrigation canals operated by both public and private entities in the Methow subbasin (**Table 11**).

Table 11 Preliminary Methow subbasin Irrigation Canal Inventory

Ditch Name	Subwatershed	Estimated Length (Miles)	Estimated Flow (cfs)
Aspen Meadows	Twisp	2	1.3
Barkley	Middle Methow	4.2	18
Beaver	Lower Methow	NA	NA
Black Canyon	Lower Methow	NA	NA
Buttermilk	Twisp	1.2	7
Chewuch	Chewuch	12	28
Culbertson	Twisp	7000' ~1.5 miles	1
Early Winters	Upper Methow	5	12
Eightmile	Chewuch	0.1	1.6 - 2.2
Foghorn	Middle Methow	5.4	18
Foster	Beaver	1200' .0227 miles	1.2 - 3.5
Fulton	Chewuch	4	22
Gold Ck - Campbell	Lower Methow	NA	NA
Gold Ck - Krevlin	Lower Methow	NA	NA
Gold Ck - Umberger	Lower Methow	NA	NA
Hottell	Twisp	0.2	1.3 max
Kumm-Holloway	Upper Methow	2.24	4.7
Libby/ Larson	Lower Methow	NA	none
Mason	Chewuch	600'	0.5
McFarland Creek	Lower Methow	NA	NA
McKinney Mountain	Upper Methow	3.8	6 - 10 cfs
MVID East	Middle and Lower Methow	15.5	21
MVID West	Twisp and Lower Methow	12.5	20
Rockview	Chewuch	5	
Skyline	Chewuch	6.2	26
Twisp Power (TVPI)	Twisp	4	9
Wolf Creek	Middle Methow	5	<16

Many of the irrigation systems within the Methow subbasin have upgraded their facilities in recent years. Those upgrades include, among others, elimination of fish passage barriers and replacement and repair of screens. The Methow Basin (WRIA 48) Watershed Plan (March 2004) discusses water use, including that for irrigation purposes.

Irrigation Districts

Methow Valley Irrigation District

The Methow Valley Irrigation District (MVID) was organized in the early 1900s to supply water for agricultural production. The MVID currently serves roughly 900 acres. MVID facilities comprise two main canals. The West Canal diverts water from the Twisp River, and the East Canal diverts water from the Methow River. The District's east canal also carries Barkley Ditch tail water. The district has installed temporary ESA-compliant screens at its points of diversion, and is expected to complete installation of permanent screens by the end of 2004.

Wolf Creek Reclamation District

Wolf Creek Reclamation District (WCRD) has operated since 1921. WCRD supplies water for approximately 790 acres of irrigated land, including the Methow Valley School District and irrigation and domestic supply for Sun Mountain Resort. Wolf Creek Reclamation District is authorized to divert surface waters from the Wolf Creek and Little Wolf Creek drainage. The diversion structure on Wolf Creek is located approximately four miles from the stream's confluence with the Methow River. Diverted water is stored for future use in Patterson Lake Reservoir. The water right is adjudicated, with irrigation and commercial domestic supply as designated beneficial uses. In 1980, WCRD began the process of lining and making other improvements to many of its ditches. The district is continuing to upgrade its delivery system, including lining many of the remaining unlined ditches, and replacing open ditches with pressurized piping where feasible. The district has also made, and continues to make, improvements to fish screens and to other potential fish passage barriers throughout its service area.

Methow Subbasin Ditches

The Chewuch Basin Council represents three ditches, the Skyline Ditch Company, the Chewuch Canal Company, and the Fulton Ditch Company, each of which operates as a distinct company.

The Skyline Ditch Company (SDC) has operated since approximately 1900. The SDC provides irrigation water for approximately 366 acres along the west side of the Chewuch River. The source of water is a surface water diversion located at approximately RM 7.5 of the Chewuch River. The SDC serves its users through a 6.0-mile delivery system. Historically, it was unlined earthen canal, but in 2003, a multi-year process was completed, and the system became completely line and piped, replacing the diversion headgate, and installing an approved screening facility to meet NMFS and WDFW requirements.

The Chewuch Canal Company (CCC) has operated since approximately 1910. The Chewuch provides irrigation water to support a variety of agricultural, recreational, and fish recovery projects within the Methow subbasin. The CCC's source of water is a surface water diversion at approximately RM 7.0 of the Chewuch River. The CCC has a separate storage reservoir permit for storage of irrigation water within Pearygin Lake. The CCC operates approximately 20 miles of surface canals, is currently completing an efficiency audit, and has upgraded their screening facility to meet NMFS and WDFW requirements.

The Fulton Ditch Company has been in operation since approximately 1909. Fulton Ditch Company provides water for irrigation users. The ditch's source of water is a surface diversion at approximately RM 0.8 of the Chewuch River. Fulton Ditch Company is currently completing an efficiency audit, has lined approximately 1,600 linear feet of their canals, and has installed approved fish screens to meet NMFS and WDFW requirements.

Some other ditches in the Methow subbasin (not part of the Chewuch Basin Council) include Aspen Meadows, Beaver, Black Canyon, Culbertson, Early Winters, Eightmile, Foghorn, Foster, Rockview, and Twisp Power.

Water and Habitat quality

The Methow River is listed on the State of Washington 303(d) list as exceeding water quality temperature criteria at the inflow to the Winthrop National Fish Hatchery, and as supporting inadequate instream flows because of periodic dewatering (1998 303[d] list). Dewatering just upstream of the Weeman Bridge on the Methow River, and dewatering in the Popular Flats Campground area of the Twisp River, are natural seasonal occurrences (Gorman 1899).

The Twisp River is listed on the 1998 Washington State 303(d) list for inadequate instream flow and for temperature exceedences, and Beaver Creek is listed on the Washington 303(d) list for inadequate instream flows.

3.2 Habitat Areas and Quality by Subwatershed

Lost River Subwatershed

Lost River empties into the Methow from the north at RM 73.0, roughly six miles above Early Winters' confluence. About 95% of the drainage lies within the Pasayten Wilderness. Human impact in this drainage is largely restricted to the river's lower mile. Spring Chinook salmon spawn in Lost River to the confluence with Eureka Creek. Summer steelhead spawn and rear in Lost River. Bull trout spawn and rear in Lost River, as well as in several of its tributaries.

Within the channel migration zone of the first river mile, the construction of roads and dikes associated with home developments has constrained floodplain function and the channel, potentially reducing pool quality and quantity, as well as side-channel habitat.

Some riparian habitat in the lower mile has been converted to residential development and pastureland. Residential construction on the alluvial fan may lead to a constrained channel in the future. Large woody debris (LWD) has been removed from the lower mile of the river for flood control and firewood gathering; however, the potential for LWD recruitment is thought to be at natural levels. Low stream flows are a natural condition throughout the Lost River drainage, but water temperatures remain cold.

Upper Methow Subwatershed

The upper Methow River drainage includes the mainstem Methow from its headwaters to the Chewuch River confluence (RM 50.1). Other major tributaries in the drainage include Goat Creek, Wolf Creek, Hancock Creek, Little Boulder Creek, Dawn Creek, Gate Creek, Robinson Creek, Rattlesnake Creek and Trout Creek. Spring Chinook, summer Chinook, steelhead/rainbow trout, westslope cutthroat, and bull trout have all been documented in the Upper Methow River drainage. Between 1987 and 1999, approximately 40% of spring Chinook spawning in the Methow watershed occurred in the Methow River between the Lost River confluence (RM 73.0) and the Winthrop Bridge (RM 49.8) (USFS 1998).

Methow mainstem habitat between the Lost River confluence and Winthrop has been greatly affected by human activity. The river has a low gradient throughout this stretch, and a number of dikes block access to valuable side-channel spawning and rearing habitat, including sites of spring Chinook spawning redds (YN spawning ground surveys 1987-1999). Floodplains are constrained by those same dikes, as well as riprapping and bank stabilization measures.

Riparian habitat has been converted to agricultural and, more recently and increasingly, to residential use along the mainstem between the Early Winters confluence and the Mazama Bridge, which in some areas has resulted in increased bank erosion. Historic timber harvest activities, fire, livestock grazing, and construction of logging related roads throughout the lower reaches of the Goat Creek and Wolf Creek drainages have also resulted in delivery of large sediment loads to the Methow River. Improvement in grazing practices in this Subwatershed and other areas of the basin has helped lessen the current impact of livestock grazing. The amount of sediment delivered to creeks and streams from natural occurrences has not been quantified relative to the amount of sediment contributed through human use within the subbasin.

In the Wenatchee River, Don Chapman Consultants (D. Chapman 1989) described, documented and assessed both intra- and inter-species behavior and movement of juvenile Chinook and steelhead trout related to in-stream habitat factors as affected by seasonal and diurnal changes. Their work and others (Meehan 1991) emphasizes the complex and inter-related factors affecting salmonids in their environment.

There are also some studies that suggest stream habitats are not drastically altered until base flow is reduced 70-80% or more (Wesche 1974; Tennant 1976; Newcombe 1981; Mullan et al. 1992b). Some research suggests that how water fills the stream channel may be more important than the quantity of water in the channel (Binns 1982). Mullan et al. (1992b) showed wetted perimeter decreased much less rapidly than volume of flow. Other studies conclude that salmonids appear to do little to avoid the consequences of severely declining flows, although it appears larger fish are more influenced than smaller fish (Corning 1970; Kraft 1972; Bovee 1978; Randolph 1984; Mullan et al.1992b).

Goat Creek

Goat Creek, drains into the Methow from the north about a mile downstream from the town of Mazama. Portions of the upper third of the Goat Creek drainage have been heavily grazed. The lower two-thirds of the drainage have been logged, roaded and grazed (USFS 1995a). Goat Creek supports small resident and migratory bull trout populations in the upper reaches. Spring Chinook spawn in the Methow River above and below the confluence with Goat Creek and may rear in the mouth of the creek. Summer steelhead/rainbow also spawn and rear in the creek.

The Goat Creek drainage is laced with over 150 miles of roads, more than 4 miles of road per mile^2, with almost all of those located in the lower half of the drainage (USFS 2000e). Sediment from roads and slope failures is carried by Goat Creek to Chinook salmon spawning grounds in the Methow River (USFS 2000e). Livestock use has also damaged, or suppressed re-growth of riparian vegetation in some tributaries. Goat Creek exhibits both elevated water temperatures and low flows and dewatering in August and September (FWS 1998.)

Wolf Creek

Wolf Creek, a Methow River tributary, drains into the Methow about 3 miles above the town of Winthrop. Wolf Creek provides spawning and rearing habitat for resident and fluvial bull trout, westslope cutthroat trout, summer steelhead and spring Chinook. Approximately 80% of the drainage is designated wilderness with very good habitat conditions. The Forest Service manages the remainder of the drainage for multiple uses with the exception of the last 1.5 miles, which is privately owned. Impacts from timber harvest and roads are isolated primarily to the Little Wolf Creek drainage. Introduction of woody debris and pool formation projects have been completed in 2000 along the lower 0.5 miles of the creek.

Early Winters Subwatershed

Early Winters Creek enters the Methow about 3.5 miles upstream from the town of Mazama. The majority of the watershed is in relatively pristine condition. Roughly 99% of the area is managed by the USFS as a Scenic Highway Corridor with the remainder designated as Late Successional Reserve. Highway 20 follows Early Winters Creek to the Cascade Crest crossing over it in three spots. Human impacts are primarily restricted to the lower 2 miles of Early Winters Creek, including its alluvial fan.

The lower half-mile of the river has been riprapped and diked to keep the channel in a stable location in order to accommodate Highway 20 and to protect private property. Levels of LWD in the first two miles are low and pool quality and quantity is poor. Severe low flows persist in the lower 1.4 miles of the creek. Low base flows are naturally occurring during the winter months; however, low flows during late summer and early fall may be exacerbated by two irrigation diversions (USFS 1998c). In 2000 or 2001, the USFS completed a restoration project on this reach of the creek. The restoration included an increase of large woody debris, pools and quality habitat.

The Early Winters Ditch on Early Winters Creek is currently meeting NMFS and USFWS target flow of 35 cfs for spring Chinook and bull trout, and the irrigation district is using wells, that are not in continuity with groundwater and surface water to meet the remainder of its irrigation needs. Fine sediment and chemical runoff from state Route 20 may negatively impact water quality.

Chewuch River Subwatershed

The Chewuch River enters the Methow at the town of Winthrop. About 95% of the drainage is managed by the USFS, with nearly 34% falling within the Pasayten Wilderness. The majority of human impact has occurred in the lower half of the drainage, with the upper 50% remaining generally undisturbed. Spring Chinook salmon spawn in the mainstem Chewuch River (up to Thirtymile Creek), and steelhead spawn and rear in the mainstem and in the tributaries (USFS 2000c).

Bull Trout use of the Lower Chewuch is unknown with the exception as a migratory corridor, however, it is known that they use the Lower Middle Chewuch and the Lake Creek tributary for spawning and rearing. Brook trout are found in the Chewuch River and in all of the fish-bearing tributaries below Twentymile Creek (USFS 2000c). Most are isolated above natural upstream barriers, reducing their potential elimination to the existing bull trout population(s). Natural upstream barriers such as waterfalls or very steep gradients exist on the majority of the Chewuch's tributaries.

Five ditches divert water within the Chewuch subwatershed, and two roads parallel segments of the Chewuch. Low flows in late summer through winter reduce quantity of rearing habitat in the lower Chewuch River. High water temperatures in the lower river may at times cause a migration barrier. The drainage's upper reaches are also characterized by harsh winters and icing.

Roads border most of the tributaries in the lower two-thirds of the drainage. The Chewuch drainage has approximately 1,000 stream crossings, and road densities exceed 3.5 miles/mile^2 along most of the lower eight miles of the Chewuch River (USFS 1994). Skid roads in riparian areas upstream of Boulder Creek have lead to increased recreational use and resulting impacts on the stream and riparian areas. Road density, road placement, past logging activities, and grazing, in concert with highly erodible soils, have led to chronic sediment delivery to streams, particularly in Cub, Eightmile, Doe, and Boulder Creek drainages (USFS 1994). These conditions are aggravated by low levels of LWD, loss of mature riparian habitat, and channelization in the alluvial fans of numerous tributaries.

Extensive riprap for flood control associated with residential development has also occurred on the lower eight miles of the Chewuch, as well as along several tributaries; although, there is some disagreement over the effect this has had on overall habitat quality. Mullan (1992b) suggests that riprap on this section of the river may actually contribute habitat. Other studies document negative impacts on fish populations and stream channel functions associated with human-induced channel confinement and habitat simplification (Murphy and Meehan 1991, Bjornn and Reiser 1991; Leopold et al. 1992; Kohler and Hubert 1999). On the Chewuch River tributaries, Twentymile Creek and Boulder Creek, the alluvial fan has been channelized.

Middle Methow Subwatershed

The Middle Methow drainage includes the mainstem Methow from its confluence with the Chewuch River to the town of Carlton. Summer Chinook, some steelhead, some spring Chinook, and most of the remnant sockeye adults spawn in this portion of the Methow subbasin. Bull trout and westslope cutthroat trout use this portion of the mainstem as a migrational corridor and for overwintering.

County roads and state highways parallel both sides of the Methow River throughout this subwatershed. Diking, conversion of riparian areas to agriculture and residential uses, and LWD removal along the mainstem Methow River, have resulted in loss of side channel access, riparian vegetation, and overall habitat complexity. Much of the habitat within this area has not been adequately inventoried or assessed, and data gaps exist regarding the extent of habitat alterations. The Methow Valley Irrigation District diverts water to its east canal about five miles north of the town of Twisp at RM 44.8. The highest percentage of diversion from the river takes place in September. The average September diversion is 39.3 cfs, about 13% of the mean September flow in the Methow River at this point (BPA 1997). East Canal flows back into the Methow River at RM 26.6.

Beaver Creek

Beaver Creek drains into the Methow five miles downstream from the town of Twisp, and is a tributary in this subwatershed. Previously, steelhead, spring Chinook and bull trout have had limited access to Beaver Creek due to its many obstructions. Most of these obstructions have been removed or are in the process of being modified for passage. All diversions in Beaver Creek have now been screened (L. Clark, Okanogan Conservation District, e-mail communication). Road density in the Beaver Creek drainage is the highest in the Methow subbasin. In 41% of the Beaver Creek drainage, road densities vary between 2.5 and 5 miles/mile^2 (USFS 1997). Nearly 130 million board feet of timber have been harvested from the Beaver Creek drainage since the 1960s, resulting in heavy sediment loading, slope destabilization, and reduction of recruitment potential for LWD (USFS 2000a). Limited grazing activity has also slightly contributed to stream sediment delivery in this section.

In low water years, Beaver Creek goes dry in the fall, with the exception of the uppermost reaches and the lowest 0.3-mile which maintain flows via irrigation return. The subwatershed is an adjudicated drainage where water uses are provided for in excess of available water during some part of the irrigation season (USFS 1997). Eastern brook trout in the Beaver Creek drainage likely provide negative impacts on the remaining bull trout populations.

Twisp River Subwatershed

The Twisp River flows into the Methow at the town of Twisp. Like the Early Winters and Lost River subwatersheds, a substantial portion of the Twisp River subwatershed habitat rests within designated wilderness and is in nearly pristine condition. Nearly 95% of the subwatershed is federally managed, and of that, approximately 50% lies within the Lake Chelan-Sawtooth Wilderness. The remaining land is managed as Late Successional Reserves or Matrix (USFS 1995c). Spring Chinook salmon and summer steelhead spawn and rear in the Twisp River for nearly its entire length. Bull trout are found throughout the mainstem and several of its tributaries. Bull trout use the lower mainstem for overwintering and as a migrational corridor. Most of the spawning areas for bull trout are located in the upper watershed. Westslope cutthroat trout are found in these areas as well.

Most human activity and related habitat changes within the drainage have taken place within the lower 15 miles of the Twisp River. Reduced levels of LWD, road placement, diking, bank hardening, and conversion of riparian areas to agriculture and residential uses have altered habitat conditions in this area, and resulted in loss of channel complexity and floodplain function. After a flood in 1972, the U.S. Army Corps of Engineers used bulldozers to channelize and remove logjams from a tributary of the Twisp River, Little Bridge Creek (Methow Valley News, Vol.70, June 29, 1972). Some effects of these activities still linger.

There are seven irrigation diversions on the Twisp River.

The Twisp River from Buttermilk Creek to the mouth, has been diked and riprapped in places, resulting in a highly simplified channel and disconnected side channels and associated wetlands. Levels of LWD recruitment potential in the lower Twisp River are far below normal.

Little Bridge Creek, a tributary of the Twisp River, contributes large amounts of sediment to the Twisp as a result of historic logging activities. Excessive sediment delivery from both private and U.S. Forest Service (USFS) land in Poorman and Newby drainages also contributes to elevated sediment levels in the lower 15 miles of the Twisp River. The lower two-thirds of the creek have high road densities. Although some restoration activities are currently underway, construction of culverts, erosion, and grazing activities have contributed to habitat degradation in this drainage. Finally, beaver activity is very limited in the lower Twisp River where large cottonwood galleries and low gradients would once have supported beaver colonies.

Lower Methow Subwatershed

The Lower Methow River subwatershed includes the Methow mainstem and its tributaries from the town of Carlton to the mouth of the Methow River. Agriculture use in this subwatershed is primarily field crops and cattle at the upper end, with orchards along the lower end. Portions of the summer Chinook escapement spawns in the lower Methow River. In addition, this reach provides rearing habitat and acts as a migration corridor for all anadromous salmonids and fluvial bull trout.

Timber harvest, livestock grazing, and high road densities characterize much of the Libby Creek drainage, with roads running parallel to every major stream. The lower 2.9 miles of Libby Creek have been channelized. Culverts and irrigation diversion structures impede salmonid passage on a number of tributaries. Upstream passage for salmonids is also limited by heavy beaver activity in some tributaries. Libby Creek has no historical evidence of use by spring Chinook or bull trout. The lower mile is used heavily by summer steelhead for spawning and initial rearing. Ground water discharge is likely the attraction for steelhead.

Timber harvest, livestock grazing, and elevated road densities also characterize Gold Creek. The lower 3.5 miles of Gold Creek have had riprap placed along the banks. Gold and Libby Creeks are characterized by low instream flows, and Gold Creek dewaters in a lower reach between RM3 and RM2 during some low water years. The timing of dewatering may not preclude passage of adult migrants that pass through the reach prior to dewatering; however, dewatering could negatively impact movement of juvenile salmonids. A spring Chinook redd was located in 1987, an extreme drought year, and reported in Mullan et al. (1992b). Standing crop fish estimates for Gold Creek and its main tributary streams are consistently high compared to other creeks (Mullan et al. 1992b).

Fish Species/Aquatic Relationships

The Methow subbasin is considered part of the Upper Columbia River ESU, and several species of anadromous salmonids, Pacific lamprey, and resident fish stocks are considered by National Marine Fisheries Service (NMFS) and U.S. Fish and Wildlife Service (FWS) as Endangered, Threatened or locally extirpated.

An estimated 32 species of fish, including seven introduced species, are found in the Methow River subbasin (**Table 12**). Distinct Upper Columbia River population segments exist for Methow/Okanogan River summer steelhead (Endangered) and bull trout (Threatened). Methow subbasin also supports distinct population segments of summer Chinook and spring Chinook (Endangered) in the mainstem, Twisp, Lost River, and Chewuch subwatersheds (SASSI).

Table 12 Fish species of the Methow subbasin

Family & Species	Scientific Name	Habitat	Origin
Lamprey Family	**Petromyzontidae**		
Pacific Lamprey	Entosphenus tridentatus	Larvae found in backwater silt, adults in the ocean	Native
Salmon Family	**Salmonidae**		
Mountain Whitefish	Prosopium williamsoni	Riffles in summer, pools in winter	Native
Brown Trout	Salmo trutta	Streams up to 75 degrees Farenheit	Northern Europe
Cutthroat Trout	Oncorhyncus clarki	Cold water lakes and streams; some are anadromous	Native and stocked from Western states
Rainbow Trout/Steelhead	Oncorhyncus mykiss	Cold water lakes and streams, some are anadromous	Native and stocked from Western states
Chinook Salmon	Oncorhynhcus tshawytscha	Anadromous (spawn in fresh water, runs to ocean)	Native to Pacific Northwest
Sockeye Salmon	Oncorhyncus nerka	Anadromous	Native
Coho Salmon	Oncorhyncus kisutch	Anadromous	Native to Pacific Northwest
Brook Trout	Salvelinus fontinalis	Cold water lakes and streams	Eastern North America
Bull Trout	Salvelinus confluentus	cold water streams and pools; some are anadromous	Native
Minnow Family	Cyprinidae		
Carp	Cyprinus carpio	shallow, quiet water, preferring dense vegetation	Native to Asia
Longnose Dace	Rhinichthys cataractae	Among stones at the bottom of swift streams	Native
Northern Pikeminnow (Squawfish)	Ptychocheilus oregonensis	Lakes and slow streams	Native to the Columbia River
Redside shiner	Richardsonius balteatus	Warmer ponds, lakes, streams	Native to the Columbia River
Sucker Family	**Catostomidae**		
Bridgelip Sucker	Catostomus columbianus	Bottom feeder in backwaters and pools in rivers	Native
Largescale Sucker	Catostomus macrocheilus	Bottom feeder in lakes, and pools in rivers	Native
Sunfish Family	**Centrarchidae**		
Smallmouth Bass	Micropterus dolomieui	Warm streams and lakes	Eastern North America
Largemouth Bass	Micropterus salmoides	Shallow, warm weedy lakes and backwaters	Eastern North America
White Crappie	Pomoxis annularis	Lakes and streams with dense vegetation	Eastern North America
Catfish Family	**Ictaluridae**		
Brown Bullhead	Ictalurus nebulosus	Warm-water ponds, lakes, sloughs	Eastern North America
Sculpin Family	**Cottidae**		
Mottled Sculpin	Cottus bairdi	Cold rivers	Native
Shorthead Sculpin	Cottus confusus	Cold rivers	Native
Torrent Sculpin	Cottus rhotheus	Cold rivers and lakes	Native
Perch Family	**Percidae**		
Walleye	Stizostedion vitreum	Large lakes and streams	Central & Eastern North America

Source: Methow Biodiversity Project, PO Box 175, Winthrop, WA 98862

Fish species not included in the table

Westslope cutthroat trout *Oncorhynchus clarki lewisi*

Interior redband trout *Oncorhynchus mykiss gairdneri*

Additional species possible in the Methow

Western brook lamprey *Lampetra richardsoni*

Mountain sucker *Catostomus platyrhynchus* (state monitor)

Chiselmouth *Acrocheilus alutaceus*

Sandroller *Percopsis transmontana* (state monitor)

Peamouth *Mylocheilus caurinus*

Pygmy whitefish *Prosopium coulteri*

Leopard dace *Rhinichthys falcatus*

Historical anadromous production in the Methow subbasin was represented by spring Chinook (*Oncorhynchus tshawytscha*), coho (*O. kisutch*), and summer steelhead (O. *mykiss*). Craig and

Suomela (1941) found evidence only of spring Chinook salmon; although, it was possible that some summer Chinook once spawned in the lower Methow River (Mullan et al. 1992b).

The Washington Water Power Company's dam in the lower Methow River near Pateros significantly altered salmonid production in the early decades of the 20th century. Records from 1928 and 1929 indicate some Chinook salmon were dipnetted below the dam and released above it (Mullan 1987; Scribner et al. 1993), but there was no evidence steelhead and coho salmon were passed beyond the dam. When the dam was removed (circa 1929), coho salmon were extirpated, Chinook were nearly extirpated, and steelhead persisted as resident rainbow trout (Mullan et al. 1992b).

Bull trout once filled most every cold-water niche in the Methow subbasin; however, the presence of natural barriers such as waterfalls or small stream size blocked their access to many headwater streams.

Diking, conversion of riparian areas to agriculture and residential uses, and LWD removal along the mainstem Methow River have contributed to the loss of side-channel access, riparian vegetation, and overall habitat complexity. However, much of the habitat within this area has not been adequately inventoried or assessed and data gaps exist regarding the extent of habitat alterations.

Much of the watershed remains undeveloped, and large tracts of high quality fish habitat remain, particularly within the middle and upper elevations. These areas are contained in lands held largely in public ownership, and include several thousand acres managed as wilderness/roadless condition by the Okanogan National Forest. Within these management boundaries, plant communities and succession are shaped largely through such natural processes as fire, avalanches, storms, and temperature ranges.

Fish and Wildlife Focal Species Associations

The wildlife species and their habitat associations, shared with salmonids, are listed in Appendix B. The red-eyed vireo, yellow breasted chat and American beaver share riparian wetland habitats directly with salmonids.

Fish and Wildlife Species Richness

93% of the wildlife and 90% of the salmonid species that occur in the Ecoprovince, occur in the Methow subbasin (Table 13). In addition, 65% of those amphibian species and 84% of the reptile species also occur in the subbasin.

Table 13 Species richness and associations for the Methow subbasin, Washington

Class	Subbasin														
	Entiat	**%**	**Lake Chelan**	**%**	**Wena-tchee**	**%**	**Methow**	**%**	**Oka-nogan**	**%**	**Upper Middle Mainstem**	**%**	**Crab**	**%**	**Total (Eco-prov)**
Amphibians	11	65	11	65	16	94	11	65	9	53	17	100	9	53	**17**
Birds	218	93	221	94	215	92	252	94	222	95	234	100	214	91	**234**
Mammals	91	94	93	96	91	94	93	96	86	89	97	100	78	80	**97**
Reptiles	16	84	16	84	19	100	16	84	13	68	19	100	16	84	**19**
Total	**336**	**92**	**341**	**93**	**341**	**93**	**341**	**93**	**328**	**89**	**367**	**100**	**317**	**86**	**367**
Association															
Riparian Wetlands	72	92	73	94	70	90	73	94	73	94	77	99	73	94	**78**
Other Wetlands (Herbaceous and Montane Coniferous)	30	81	32	86	26	68	32	86	31	84	36	95	33	89	**38**
All Wetlands	102	89	105	91	96	83	105	91	104	90	113	97	106	92	**116**
Salmonids	77	93	75	90	76	93	75	90	71	86	81	98	72	87	**82**
Note: % = % of Total															

Source: Ibis 2003

Wildlife Species/ Terrestrial Relationships

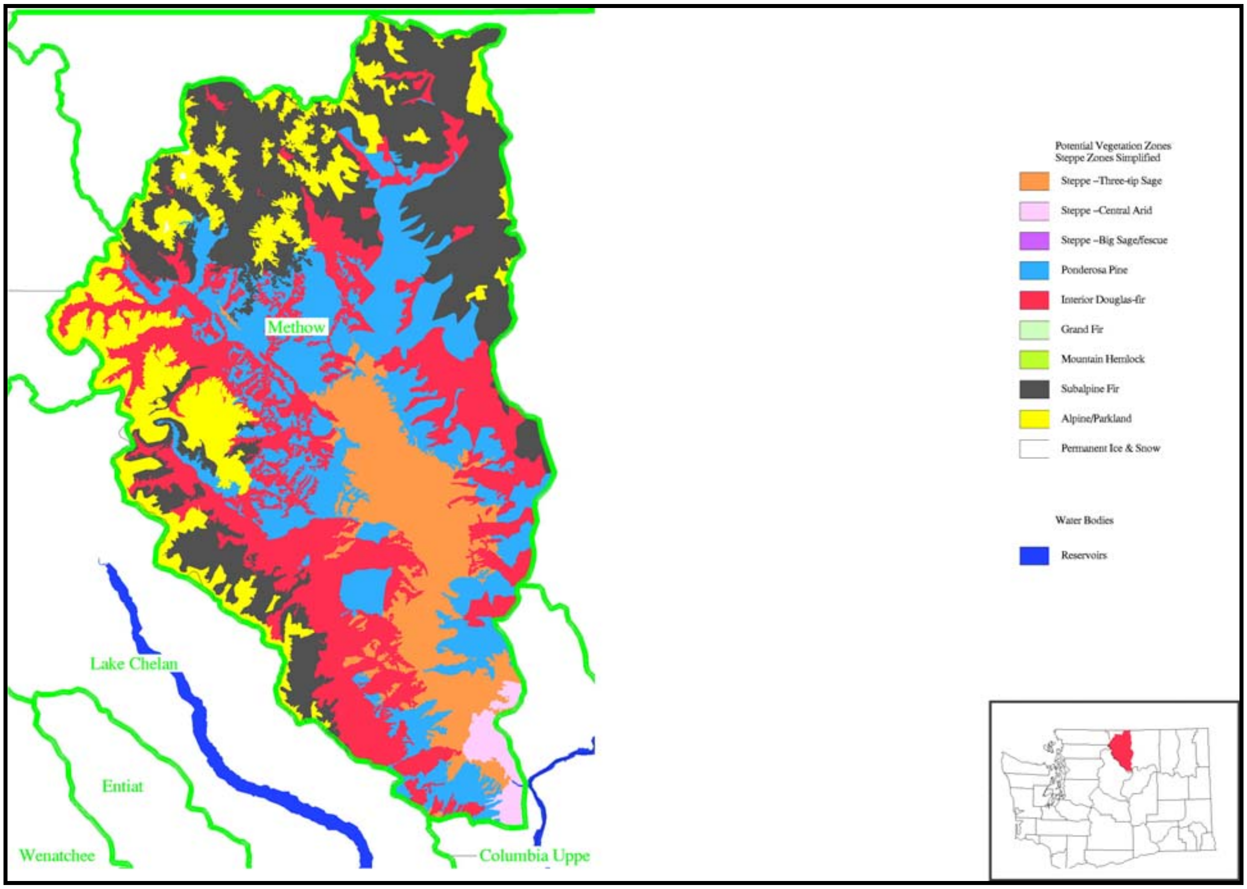
There are an estimated 341 wildlife species that occur in the Methow subbasin. These species, their assemblages, associations and relationship to the CCP are listed in Appendix B. Of those species, 105 (31%) are closely associated with riparian and wetland habitat, and 75 (22%) consume salmonids during some portion of their life cycle. Seventeen wildlife species are non-native. Eight wildlife species that occur in the subbasin are listed federally, and 38 species are listed in Washington and Idaho as Threatened, Endangered, or Candidate species. A total of 98 bird species are listed as Washington or Idaho State Partners in Flight priority and focal species. A total of 57 wildlife species are managed as game species in Washington.

The subbasin consists of 15 wildlife habitat types, which are briefly described in **Table 14** and detailed descriptions of these habitat types can be found in Appendix B of Ashley and Stovall (unpublished report, 2004).

Table 14 Wildlife Habitat Types and Vegetation Zones in the Methow subbasin

Habitat Type	Brief Description
Montane Mixed Conifer Forest	Coniferous forest of mid-to upper montane sites with persistent snowpack; several species of conifer; under-story typically shrub-dominated.
Eastside (Interior) Mixed Conifer Forest	Coniferous forests and woodlands; Douglas-fir commonly present, up to 8 other conifer species present; under-story shrub and grass/forb layers typical; mid-montane.
Lodgepole Pine Forest and Woodlands	Lodgepole pine dominated woodlands and forests; under-story various; mid- to high elevations.
Ponderosa Pine and Interior White Oak Forest and Woodland	Ponderosa pine-dominated woodland or savannah, often with Douglas-fir; shrub, forb, or grass understory; lower elevation forest above steppe, shrubsteppe.
Upland Aspen Forest	Quaking aspen (*Populus tremuloides*) is the characteristic and dominant tree in this habitat. Scattered ponderosa pine (*Pinus ponderosa*) or Douglas-fir (*Pseudotsuga menziesii*) may be present.
Subalpine Parkland	Coniferous forest of subalpine fir (*Abies lasiocarpa*), Engelmann spruce (*Picea engelmannii*) and lodgepole pine (*Pinus contorta*).
Alpine Grasslands and Shrublands	This habitat is dominated by grassland, dwarf-shrubland (mostly evergreen microphyllous), or forbs.
Eastside (Interior) Grasslands	Dominated by short to medium height native bunchgrass with forbs, cryptogamic crust.
Shrubsteppe	Sagebrush and/or bitterbrush dominated; bunchgrass under-story with forbs, cryptogamic crust.
Agriculture, Pasture, and Mixed Environs	Cropland, orchards, vineyards, nurseries, pastures, and grasslands modified by heavy grazing; associated structures.
Urban and Mixed Environs	High, medium, and low (10-29 percent impervious ground) density development.
Open Water – Lakes, Rivers, and Streams	Lakes, are typically adjacent to Herbaceous Wetlands, while rivers and streams typically adjoin Eastside Riparian Wetlands and Herbaceous Wetlands
Herbaceous Wetlands	Generally a mix of emergent herbaceous plants with a grass-like life form (graminoids). Various grasses or grass-like plants dominate or co-dominate these habitats.

Habitat Type	Brief Description
Montane Coniferous Wetlands	Forest or woodland dominated by evergreen conifers; deciduous trees may be co-dominant; under-story dominated by shrubs, forbs, or graminoids; mid- to upper montane.
Eastside (Interior) Riparian Wetlands	Shrublands, woodlands and forest, less commonly grasslands; often multi-layered canopy with shrubs, graminoids, forbs below.



Source: IBIS 2003

Figure 12. Wildlife habitat types of the Methow subbasin

The watershed contains 14 Priority Habitats as identified by WDFW. Priority Habitats are those habitat types or elements with unique or significant value to a diverse assemblage of species. A Priority Habitat may consist of a unique vegetation type or dominant plant species, a described successional stage, or a specific structural element. There are 18 habitat types currently on WDFW's Priority Habitats and Species (PHS) List. The PHS program is explained in Section 3.1 of this plan.

Ninety-three (93) % of the wildlife species that occur in the Ecoprovince occur in the Methow subbasin. In addition, sixty-five 65% of the amphibian species and eighty-four 84% of the reptile species that occur in the Ecoprovince, occur in the subbasin. Fourteen wildlife species are as Endangered, Threatened, or as Species of Concern within the Methow subbasin.

3.3 Focal Species: Population Characterization and Status

The subbasin plan used the concept of "focal species" as a way to manage both the size of the subbasin plan and the scope of the assessment, inventory and management plan. In its truest sense, this was simply a means to target our resources and cover as many species and habitats as possible.

In some limited instances this approach was also used to prioritize some actions across fish and wildlife needs or to more properly ascribe responsibilities (e.g., CWA, PCSRF, Power Act, ESA). Mitigation obligations, ESA listing status, coterminous habitat use and overlapping jurisdictions were some of the considerations used to designate focal species. However, we must clearly point out and caution the reader that it was not the intention of the subbasin planners to impart a value judgment placing an emphasis or de-emphasis on the need or responsibility to protect and/or restore a particular or species or their habitats or to decouple any species from any legal, policy or trust obligations. The subbasin plan used the concept of "focal species" as a way to manage both the size of the subbasin plan and the scope of the assessment, inventory and management plan. In its truest sense, this was simply a means to target our resources and cover as many species and habitats as possible.

A focal species has special ecological, cultural, or legal status and represents a management priority in the Methow subbasin and, by extension, in the Columbia Cascade Ecoprovince. Focal species are used to evaluate the health of the ecosystem and the effectiveness of management actions.

Criteria used in selecting the focal species build upon: a) designation as Federal Endangered or Threatened species, or management priority as designated by a management authority; b) cultural significance; c) local significance, and; d) ecological significance, or provide the ability to serve as indicators of species and ecosystem health. See Appendix C for a full classification of fish and wildlife species in this ESU. Life history summaries are provided below. See referenced literature for more detailed information.

Each of the fish and wildlife focal species, their assemblages, and their associated habitats in the Methow subbasin is summarized in **Table 15**.

Table 15 Fish and Wildlife focal species and their distribution within the Methow subbasin

Focal Species	Focal Habitat Represented		
	Ponderosa pine	Shrubsteppe	Riparian wetlands
Wildlife			
Brewer's sparrow		(shaded)	
Grasshopper sparrow		(shaded)	
Sharp-tailed grouse		(shaded)	
Mule deer		(shaded)	
Red-eyed vireo			(shaded)
Yellow-breasted chat			(shaded)
American beaver			(shaded)

Focal Species	Focal Habitat Represented		
	Ponderosa pine	Shrubsteppe	Riparian wetlands
Wildlife			
Pygmy nuthatch			
Gray flycatcher			
White-headed woodpecker			
Flammulated owl			
Fish			
Spring Chinook			
Summer/Fall Chinook			
Coho			
Steelhead			
Bull Trout			
Westslope cutthroat trout			

3.3.1 Fish Focal Species Selection

Six species in the Columbia Cascade Province are listed as Endangered or Threatened under the ESA (1972). Upper Columbia River ESU steelhead and Upper Columbia River ESU spring Chinook are listed under the ESA as Endangered, and Columbia River Population Segment bull trout are listed as Threatened. The known distribution of these species is illustrated with each species description. In addition, westslope cutthroat trout are a Species of Concern.

The Methow summer steelhead stock is listed in the Washington State Salmon and Steelhead Stock Inventory (SASSI) as Depressed based on chronically low numbers (WDF and WDW 1993). The Methow summer Chinook stocks are considered Depressed based on negative escapement trends (WDF and WDW 1993). WDF et al. (1993) classified Upper Columbia natural summer Chinook as native or mixed origin and wild production. Methow bull trout are considered an important component of Threatened Columbia River stocks. Coho salmon were once extirpated but have since been reintroduced to the Methow River.

3.3.2 Wildlife Focal Species Selection

The wildlife focal species selection process is described in Ashley and Stovall (unpublished report 2004), and important habitat attributes are summarized. An overview of focal species assemblages identified in the Methow subbasin is summarized in **Table 16**. Subbasin planners selected focal wildlife species based on their ability to serve as indicators of environmental health for other species, and in combination with several other factors, including:

1. Primary association with focal habitats for breeding;

3. Specialist species that are obligate or highly associated with key habitat elements/conditions important in functioning ecosystems;

4. Declining population trends or reduction in their historic breeding range (may include extirpated species);
5. Special management concern or conservation status such as Threatened, Endangered, Species of Concern and management indicator species; and
6. Professional knowledge on species of local interest.

Wildlife species associated with focal habitats, including agriculture, are listed in Appendix A.

Table 16 Focal wildlife species selection matrix for the Methow subbasin

Common Name	Focal Habitat[1]	Status[2]		Native Species	PHS	Partners in Flight	Game Species
		Federal	State				
Sage thrasher	SS	n/a	C	Yes	Yes	Yes	No
Brewer's sparrow		n/a	n/a	Yes	No	Yes	No
Grasshopper sparrow		n/a	n/a	Yes	No	Yes	No
Sharp-tailed grouse		SC	T	Yes	Yes	Yes	No
Sage grouse		C	T	Yes	Yes	No	No
Mule deer		n/a	n/a	Yes	Yes	No	Yes
Willow flycatcher	RW	SC	n/a	Yes	No	Yes	No
Lewis woodpecker		n/a	C	Yes	Yes	Yes	No
Red-eyed vireo		n/a	n/a	Yes	No	No	No
Yellow-breasted chat		n/a	n/a	Yes	No	No	No
American beaver		n/a	n/a	Yes	No	No	Yes
Pygmy nuthatch	PP	n/a	n/a	Yes	No	No	No
Gray flycatcher		n/a	n/a	Yes	No	No	No
White-headed woodpecker		n/a	C	Yes	Yes	Yes	No
Flammulated owl		n/a	C	Yes	Yes	Yes	No
Red-winged blackbird	HW	n/a	n/a	Yes	No	No	No

[1] SS = Shrubsteppe; RW = Riparian Wetlands; PP = Ponderosa pine; HW = Herbaceous Wetlands
[2] C = Candidate; SC = Species of Concern; T = Threatened; E = Endangered

Nine bird species and two mammalian species were selected to represent three priority habitats in the subbasin. Life-requisite habitat attributes for each species assemblage were pooled to characterize a "range of management conditions," to guide planners in development of future habitat management strategies, goals, and objectives.

General habitat requirements, limiting factors, distribution, population trends, analyses of structural conditions, key ecological functions, and key ecological correlates for individual focal species are included in Ashley and Stovall (unpublished report 2004). The reader is further encouraged to review additional focal species life history information in Appendix F in Ashley and Stovall (unpublished report 2004).

Establishment of conditions favorable to focal species will benefit a wider group of species with similar habitat requirements. Wildlife species and their association with focal habitats including agriculture are also listed in Appendix A

Assessment of Wildlife

The process used to develop wildlife assessments and management plan objectives and strategies is based on the need for a landscape-level holistic approach to protecting the full range of biological diversity at the Ecoregion scale with attention to size and condition of core areas (subbasin scale), physical connections between core areas, and buffer zones surrounding core areas to ameliorate impacts from incompatible land uses. As most wildlife populations extend beyond subbasin or other political boundaries, this "conservation network" must contain habitat of sufficient extent, quality, and connectivity to ensure long-term viability of obligate/focal wildlife species. Subbasin planners recognized the need for large-scale planning that would lead to effective and efficient conservation of wildlife resources.

In response to this need, Ecoregion planners approached subbasin planning at two scales. The landscape scale emphasizes focal habitats and associated species assemblages that are important to Ecoregion wildlife managers, while specific focal habitat and/or species needs are identified at the subbasin level.

Lambeck (1997) proposed that species requirements ("umbrella species concept") could be used to guide ecosystem management. The main premise is that the requirements of a demanding species assemblage encapsulate those of many co-occurring, less demanding, species. By directing management efforts toward the requirements of the most exigent species, the requirements of many cohabitants that use the same habitat type are met. Therefore, managing habitat conditions for a species assemblage should provide life-requisite needs for most other focal habitat obligate species.

Ecoregion/subbasin planners also assumed that by focusing resources primarily on riparian wetland, Ponderosa pine, and shrub-steppe habitats, the needs of most listed and managed terrestrial species, dependent on these habitats, would be addressed during this planning period. While other listed and managed species occur within the subbasin, primarily forested habitat obligates, needs of these species are addressed primarily through the existing land management frameworks of the federal agencies within whose jurisdiction the overwhelming majority of these habitats occur within the Okanogan subbasin (Okanogan/Wenatchee National Forest and Washington Department of Natural Resources).

Ecoprovince/subbasin planners identified a focal species assemblage for each focal habitat type and combined life requisite habitat attributes for each species assemblage to form a “recommended range of management conditions,” that, when achieved, should result in functional habitats.

The rationale for using focal species assemblages is to draw immediate attention to habitat features and conditions most in need of conservation or most important in a functioning ecosystem. The corollary is that factors that affect habitat quality and integrity within the Ecoregion and subbasins also impact wildlife species. As a result, identifying and addressing “factors that affect focal habitats” should support the needs of obligate wildlife populations as well. Planners recognize, however, that addressing factors that limit habitat does not necessarily address some anthropogenic-induced limiting factors such as affects of human presence on wildlife species.

Emphasis in this management plan is placed on the selected focal habitats and wildlife species described in the inventory and assessment. It is clear from the inventory and assessment that reliable quantification of most subbasin-level impacts is lacking; however, many anthropogenic changes have occurred, and clearly impact the focal habitats: riparian wetlands, shrub-steppe and Ponderosa pine forest habitats.

While all habitats are important, focal habitats were selected in part because they are disproportionately vulnerable to anthropogenic impacts, and likely have received the greatest degree of existing impacts within the subbasin. In particular, the majority of shrub-steppe and Ponderosa pine habitats fall within the “low” or “no” protection status categories defined above. Some of the identified impacts are, for all practical purposes, irreversible (conversion to urban and residential development, primary transportation systems); others are already being mitigated through ongoing management (i.e., USFS adjustments to grazing management).

It is impractical to address goals for future conditions within the subbasin without consideration of existing conditions; not all impacts are reversible. The context within which this plan was drafted recognizes that human uses do occur, and will continue into the future. Recommendations are made within this presumptive framework.

Landscape level vegetation information is derived from the Washington GAP Analysis Project (Cassidy 1997) and IBIS data (2003).

3.4 Fish Focal Species

3.4.1 Spring Chinook

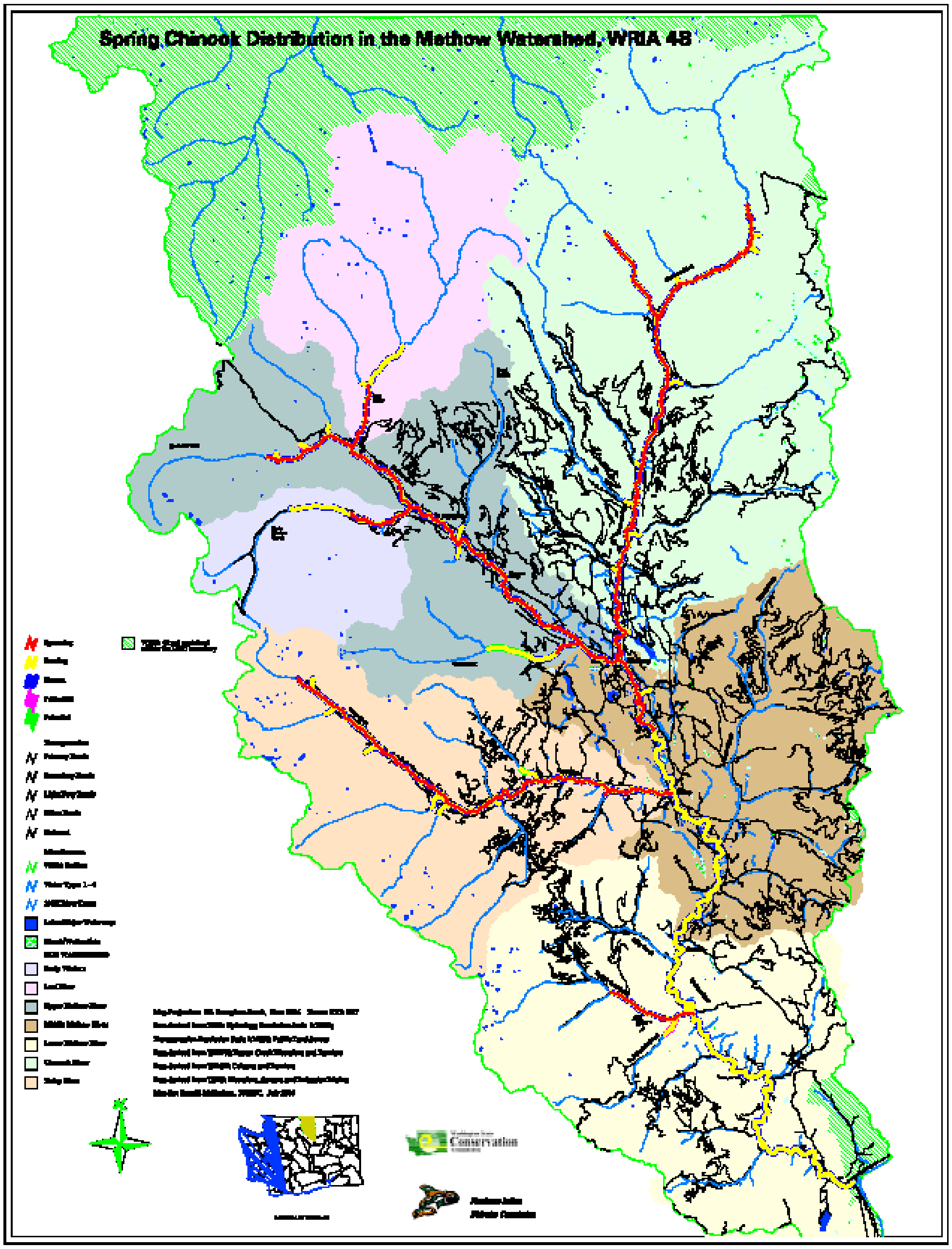
Rationale for Selection

National Marine Fisheries Service (NMFS) listed upper Columbia River spring Chinook (including the Methow Basin populations) as Endangered on March 9, 1999 (NMFS 1999). A detailed description of spring Chinook status is contained in Appendix C.

Representative Habitat

Methow River spring Chinook salmon, returning to the region, spawn primarily in the Upper Methow River and in Wolf Creek, North Fork Gold Creek, Twisp, Chewuch Early Winters and

Lost River. Juvenile rearing occurs throughout the mainstem and in key spawning tributaries. The known distribution of spring Chinook in the Methow watershed is illustrated in **Figure 13**.



Source: Washington State Conservation Commission. Salmon Steelhead and Bull Trout Habitat Limiting Factors, Water Resource Area 48, Final Report, Map Appendix (WSCC 2000).

Figure 13 Spring Chinook distribution in the Methow subbasin

Key Life History Strategies, Relationship to Habitat

The Methow spring Chinook migrate past Wells Dam and enter the subbasin in May and June, peaking after mid-May. Run timing coincides with high spring run-off. Spawning occurs late July through mid-September. Age 4 fish represent the majority of adult returns, but age 5 fish can represent 20-30% of the annual escapement (Bartlett and Bugert 1994; Bartlett 1995-1997). An average of 5% of the escapement is by age 3 fish. Fecundity averages 4,000 eggs/female for age 4 (n=93) and 5,300 eggs/female for age 5 (n=99), with a range 2,938 to 8,056 eggs/female.

Annual escapements of wild spring Chinooks are estimated to range from one to three thousand. A summary of historical spring Chinook redd counts and estimated escapement is provided in **Table 17**.

Table 17 Historical Methow subbasin spring Chinook redd counts and estimated escapement

Year	Wells Dam count	Winthrop NFH collection	Methow Hatchery collection	Wild by subtraction	Redd count	Wild run by redd expansion[1]
1962					552	3973
1963					355	2555
1964					612	4405
1965					369	2659
1966					852	6132
1967	1157			1157	377	2713
1968	4931			4931	350	2519
1969	3599			3599	292	2102
1970	2670			2670	373	2685
1971	3168			3168	319	2296
1972	3618			3618	328	2361
1973	2937			2937	502	3613
1974	3420			3420	244	1756
1975	2225	0		2225	375	2699
1976	2759	0		2759	121	871
1977	4211	0		4211	360	2591
1978	3615	38		3577	532	3829
1979	1103	102		1001	109	785
1980	1182	155		1027	91	655

[1] Index redd counts 1962-1986 (Scribner et al. 1993), total 1987-1999 (Theiss, Yakama Indian Nation, personal communication).

Year	Wells Dam count	Winthrop NFH collection	Methow Hatchery collection	Wild by subtraction	Redd count	Wild run by redd expansion[1]
1981	1935	399		1536	97	698
1982	2401	601		1800	116	835
1983	2869	755		2114	179	1288
1984	3280	900		2380	193	1389
1985	5257	1201		4056	256	1843
1986	3150	836		2315	186	1339
1987	2344	594		1750	681	1481
1988	3036	1327		1709	733	1613
1989	1740	195		1545	517	1137
1990	981	121		860	498	1060
1991	779	92		687	250	550
1992	1623	332	50	1241	738	1624
1993	2444	646	251	1547	617	1357
1994	257	29	32	196	133	293
1995	103	0	14	89	15	33
1996	335	146	318	0	NS	0
1997	971	231	328	412	150	330
1998	409	110	310	0	NS	0
1999	735	118	402	167	36	79

Fry emerge the following spring and are assumed to smolt as yearlings, although fall parr migrations from upper reaches have been observed (Hubble 1993; Hubble and Harper 1995). Juvenile Chinook have been found rearing in most of the spawning areas, mainstem margins and side channels associated with the rivers, as well as in some of the mouths of smaller tributaries (Mullan et al. 1992b; Hubble and Sexauer 1994; Hubble and Harper 1995).

Periodicity of spring Chinook salmon life history in the Methow subbasin is illustrated in **Table 18**.

Table 18 Spring Chinook life history in the Methow subbasin

Stock Group	Life history stage	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sept	Oct	Nov	Dec
Spring Chinook	Adult migration					■	■	■	■				

Stock Group	Life history stage	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sept	Oct	Nov	Dec
	Adult spawning								■	■			
	Egg incubation	■	■	■	■				■	■	■	■	■
	Juvenile rearing Smolt migration	■	■	■	■	■	■	■	■	■	■	■	■

Population Delineation and Characterization

Ford et al. (2001) concluded that there were currently three independent populations of spring Chinook within the Upper Columbia spring Chinook ESU; Wenatchee, Entiat, and Methow basins.

Four potentially distinct indigenous stocks of spring Chinook in the Methow Watershed (the upper Methow mainstem, Chewuch, Twisp and Lost River populations) exist in the Methow subbasin as identified in the SASSI process (WDFW et al. 1993a; WDFW et al. 1993b); although, the amount of genetic variability among these groups is low.

In periodic allozyme-based genetic analyses done since 1992, the Twisp, Chewuch, and Methow River populations have exhibited significant differences in allele frequencies (BAMP 1998). Some of the genetic samples, however, contained hatchery-origin fish presumably originated from the non-indigenous stock production at the Winthrop NFH.

The proportion of hatchery-origin fish in the Twisp and Chewuch populations was minimal; however, in the Methow River, above the confluence of the Chewuch River, they constituted the majority collected (BAMP 1998).

Population Status

In 1935, the Methow basin was estimated to have a run of 200 to 400 spring Chinook (Scribner et al. 1993). Although redd counts in the index reaches show a negative trend, Chapman et al. (1995a) recognized large fluctuations in redd counts between 1954 and 1994 (**Table 19**), without long-term declines in numbers.

Population Management Regimes and Activities

The most comprehensive set of spawner survey data covers years 1987 through 1999. Estimated spring Chinook migration past Wells Dam between 1987 and 1999 has ranged from 103 to 2,444 fish.

Table 19 Methow subbasin spring Chinook index redd counts (1962-1999)

Year	Wells Dam count	Winthrop NFH collection	Methow Hatchery collection	Wild by subtraction	Redd count	Wild run by redd expansion[2]
1962					552	3973
1963					355	2555
1964					612	4405
1965					369	2659
1966					852	6132
1967	1157			1157	377	2713
1968	4931			4931	350	2519
1969	3599			3599	292	2102
1970	2670			2670	373	2685
1971	3168			3168	319	2296
1972	3618			3618	328	2361
1973	2937			2937	502	3613
1974	3420			3420	244	1756
1975	2225	0		2225	375	2699
1976	2759	0		2759	121	871
1977	4211	0		4211	360	2591
1978	3615	38		3577	532	3829
1979	1103	102		1001	109	785
1980	1182	155		1027	91	655
1981	1935	399		1536	97	698
1982	2401	601		1800	116	835
1983	2869	755		2114	179	1288
1984	3280	900		2380	193	1389
1985	5257	1201		4056	256	1843
1986	3150	836		2315	186	1339
1987	2344	594		1750	681	1481
1988	3036	1327		1709	733	1613
1989	1740	195		1545	517	1137

[2] Index redd counts 1962-1986 (Scribner et al. 1993), total 1987-1999 (Theiss, Yakama Indian Nation, personal communication).

Year	Wells Dam count	Winthrop NFH collection	Methow Hatchery collection	Wild by subtraction	Redd count	Wild run by redd expansion[2]
1990	981	121		860	498	1060
1991	779	92		687	250	550
1992	1623	332	50	1241	738	1624
1993	2444	646	251	1547	617	1357
1994	257	29	32	196	133	293
1995	103	0	14	89	15	33
1996	335	146	318	0	NS	0
1997	971	231	328	412	150	330
1998	409	110	310	0	NS	0
1999	735	118	402	167	36	79

Many factors have contributed to the decline in abundance of Methow basin spring Chinook, including industrial development of the Columbia River, agricultural, forestry and private development of the Methow subbasin, and in combination with historical intensive fishing. Chapman et al. (1995a) estimated a productivity reduction of at least 43% from the 1950s to the 1980s for upper Columbia River spring Chinook salmon.

Hatchery Effects

Genetic analysis of spring Chinook in the Methow subbasin indicates that the tributary stocks in the Chewuch and Twisp Rivers are, in large part, self-recruiting populations (WDFW et al. 1993; CRITFC 2001) that have maintained or developed within the past 60 years, despite the influence of the GCFMP (WDFW et al. 1993).

Genetic data collected from samples of the Winthrop National Fish Hatchery (NFH) population in 1992 (n=100) and Winthrop Hatchery-origin adults intercepted at Methow Hatchery in 1994 (n=25), and from Twisp and Chewuch Rivers' naturally produced adults in 1992, 1993, and 1994 (n=112 and n=158 in total, respectively) showed significant genetic differentiation among the wild and hatchery populations.

Methow River mainstem natural spawners, sampled in 1993 and 1994, showed significant genetic differentiation from Twisp and Chewuch populations, but were less differentiated from the Winthrop NFH population.

Some of the Methow mainstem spawners were found to have hatchery scale patterns, and were believed to be Winthrop NFH-origin. (See also Artificial Production section). In general, the three naturally reproducing populations, prior to start-up of Methow Hatchery supplementation operations, were more closely aligned with each other than with the Winthrop NFH population, which was genetically closer to the Leavenworth, Entiat and Carson NFH populations. Twisp River spring Chinook were the most highly divergent among the three naturally reproducing Methow Basin populations.

Population divergence within a relatively short period of time has been documented in Chinook introduced in New Zealand (Quinn and Unwin 1993), and similar divergence is expected for the coho reintroduction program. Since 1992, variable broodstock collection and mating schemes of within-basin Chinook stocks (as determined by adult demographics) may have influenced the appearance of stock relationships and stock composition in the Methow subbasin.

In response to uncertainty about population structure and poor adult returns, and to a desire to spread the risk of hatchery intervention strategies, the Hatchery Working Group (HWG) developed a conceptual approach during the development of the Biological Assessment and Management Plan (BAMP) for mid-Columbia River Hatchery Programs. The approach consisted of enlarging the effective hatchery supplementation spawning populations of Methow River and Chewuch River, during periods of low adult returns, by managing them as a single gene pool.

In recent years, there has been a move to reduce the perpetuation of the Carson-origin spring Chinook in the Methow River. Agreement has been reached between the various stakeholders that the Carson stock can be used in various situations (such as in reintroduction of spring Chinook into the Okanogan Basin), and used less so for broodstock purposes in the Methow Basin (Brian Cates FWS, pers. comm).

During years of sufficient adult returns, tributary trapping locations would be utilized to obtain the broodstock components of each tributary population, and within-population mating would be made a priority in an attempt to preserve and enhance discrete population attributes that exist in the Methow Basin.

Hydroelectric Effects

Anadromous salmonids, including upper Columbia River spring Chinook depend on intact, complex and functioning habitat to support healthy populations. Perturbations in habitats throughout the Columbia Basin and Ocean environments are replete, including those associated with mainstem Columbia River hydroelectric development and operation. The development of the hydropower facilities throughout the Columbia River Basin has irrevocably altered terrestrial and aquatic habitats and is a contributor to limiting anadromous fish populations.

In attempts to mitigate for hydro-related impacts in the Mid-Columbia Region, WDFW manages a program in the Methow Basin that is funded by Chelan and Douglas PUDs as mitigation for the operation of their mainstem hydroelectric projects. The goal of the artificial production programs is to provide *no net impact* of unavoidable losses because of operation of Wells Dam, Rocky Reach Dam, and Rock Island Dam, while contributing to the rebuilding and recovery of naturally reproducing populations in their native habitats, maintaining genetic and ecological integrity, and supporting harvest operations.

Douglas will continue to fund the operation and maintenance of the Wells Hatchery and Methow Spring Chinook Supplementation Hatchery (Wells HCP 2002).

Harvest Effects

Spring Chinook were abundant in upper Columbia River tributary streams like the Methow River prior to the extensive resource exploitation in the 1860s. By the 1880s, the expanding salmon canning industry and the rapid growth of the commercial fisheries in the lower Columbia River had heavily depleted the mid and upper Columbia River spring and summer Chinook runs (McDonald 1895). The full extent of depletion in upper Columbia River spring Chinook runs is difficult to quantify because of limited historical records.

Few upper Columbia River-origin spring Chinook are currently harvested in marine or freshwater fisheries (TAC 1991). Spring Chinook from the Columbia River move northward along the continental shelf within the first few months of marine life. However, low recovery rates of upper Columbia River spring Chinook in ocean troll fisheries suggests these fish spend more time in far off-shore waters than do upper Columbia River summer Chinook.

Assuming Methow subbasin spring Chinook make similar contributions to the fishery as other upper Columbia River spring Chinook, less than 20% of the run is caught annually. Harvest is limited to incidental catches in the marine fisheries and mainstem Columbia River sport, commercial, and tribal fisheries.

3.4.2 Summer/Fall Chinook

Rationale for Selection

Summer Chinook stocks in the Methow subbasin are considered Depressed based on negative escapement trends (WDF and WDW 1993). WDF et al. (1993) classified Upper Columbia natural summer Chinook as native or mixed origin and wild production.

In the 1997 “Status Review of Chinook Salmon from Washington, Idaho, Oregon, and California,” NMFS indicated that summer/fall Chinook salmon in this ESU were not in danger of extinction, nor were they likely to become so in the foreseeable future (Myers et al. 1998). However, highly variable escapements and the desire to increase the proportion of wild origin stock in the upper Columbia River populations make the Methow River summer/fall Chinook an important stock for management attention.

Representative habitat

The known distribution of summer/fall Chinook in the Methow subbasin is illustrated in **Figure 14**.

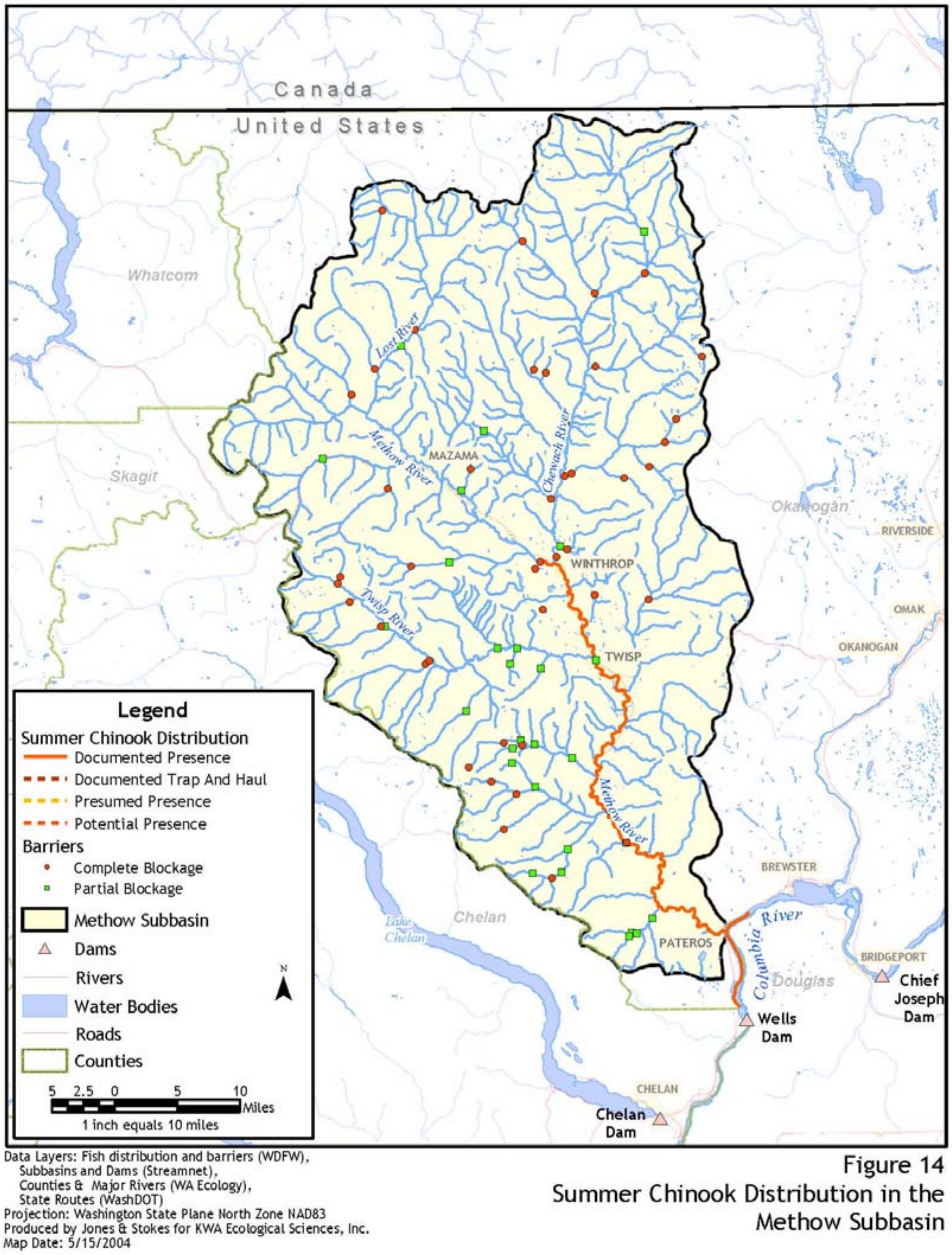


Figure 14 Summer Chinook distribution in the Methow subbasin

Key Life History Strategies, Relationship to Habitat

The dominant age class of Methow summer/fall Chinook varies between age 4 and age 5 years. Adult summer/fall Chinook enter freshwater from mid-June through late August (Wenatchee and

Methow stocks) or mid-September (Okanogan population) (WDF and WDW 1993). Methow summer Chinook, like those in the Wenatchee, begin spawning in late September.

The salmonids spawn in the lower mainstem reaches of the Methow River from the town of Winthrop down to the Methow's confluence with the Columbia River. Spawning ends in early to mid-November, with peak spawning in October (Chapman et al. 1994; WDF and WDW 1993). Methow and Wenatchee fish exhibit the same end and peak spawn timings (Chapman et al. 1994), occurring about one week later than Okanogan stocks.

A summary of spawning ground escapements from 1956-2000 is provided in **Table 20**.

Table 20 Spawning ground escapement from 1956-2000

Spawn year	Total aerial count	Total ground count	Estimated escapement
1956	109	--	605
1957	451	--	2503
1958	335	--	1860
1959	130	--	721
1960	194	--	1077
1961	120	--	666
1962	678	--	3762
1963	298	--	1654
1964	795	--	4411
1965	562	--	3119
1966	1275	--	7075
1967	733	--	4067
1968	659	--	3657
1969	329	--	1826
1970	705	--	3912
1971	562	--	3118
1927	325	--	1803
1973	366	--	2031
1974	223	--	1237
1975	432	--	2397
1976	191	--	1060
1977	365	--	2025
1978	507	--	2813
1979	622	--	3451

Spawn year	Total aerial count	Total ground count	Estimated escapement
1980	345	--	1914
1981	195	--	1082
1982	142	--	788
1983	65	--	360
1984	162	--	899
1985	164	--	910
1986	169	--	938
1987	211	--	1171
1988	123	--	683
1989	126	--	699
1990	229	--	
1990[3]	--	409	1268
1991	120	--	
1991	--	153	474
1992	91	--	
1992	--	107	331
1993	116	--	
1993	--	154	477
1994	280	--	
1994	--	310	961
1995	296	--	
1995	--	357	1107
1996[4]	151	--	
1997	173	--	
1997	--	205	636
1998	192	--	

Spawn year	Total aerial count	Total ground count	Estimated escapement
1998	--	225	698
1999	--	448	1389
2000	--	500	1550

Population Delineation and Characterization

This natural run is a mixture of strays from Wells Dam Hatchery, descendents of remnant native summer Chinook, and stocks transferred during the Grand Coulee Fish Maintenance Project (GCFMP). They are genetically homogenous with other upper- and mid-Columbia River summer and fall Chinook populations, likely because of post-GCFMP and current hatchery practices (Chapman et al. 1994a).

Population Status

The Methow summer Chinook stocks are considered Depressed based on negative escapement trends (WDF and WDW 1993). WDF et al. (1993) classified Upper Columbia natural summer Chinook as native or mixed origin and wild production. In the 1997 “Status Review of Chinook Salmon from Washington, Idaho, Oregon, and California,” NMFS indicated that summer/fall Chinook salmon in this ESU were not in danger of extinction, nor were they likely to become so in the foreseeable future (Myers et al.1998).

Population Management Regimes and Activities

Although travel distance and dam passages are essentially equal for fish from the Methow and Okanogan basins, the Methow basin summer Chinook escapement has experienced a significant decline (Chapman et al. 1994a). Chapman et al. (1994a) recommended prompt attention to studies of microhabitat, distribution, growth, egg-to-smolt survival, and pilot riparian modification. Escapement during the years 1980-2000 averaged only 36% of the total during the years 1956-1979. Since 1980, run sizes have ranged from 350 to 1,900 adults based upon redd count expansions, with an average run size of about 1,000 fish (Murdoch et al. 2001). Summer/fall Chinook typically spawn in the Methow River between RM 3.3 and RM 86.

Hatchery Effects

The U.S. Fish and Wildlife Service released summer/fall Chinook intermittently to the Methow River between 1947 and 1973 (Mullan 1987; Peven 1992). Some of these fish were obtained from adults in the Methow and Entiat rivers. From 1977 to 1982, yearling summer/fall juveniles were obtained from Wells Dam stock. The latter was a mix of Similkameen, Okanogan, and perhaps main Columbia River spawners and Wenatchee River “strays.”

In those same years, Rocky Reach Hatchery Complex produced summer/fall fish for release at Turtle Rock in Rocky Reach pool. Those releases included fall Chinook from Simpson and Elokomin hatcheries, Bonneville Dam, and Priest Rapids upriver brights, Wells summer/fall fish, and Snake River fall Chinook. A few were fingerling releases, while most were yearlings. The degree to which those releases spawned on return with summer/fall Chinook in the various tributaries, and in the Wells Dam egg-take, likely varied from year to year.

Wells Dam Hatchery production through 1991 was released at Wells Dam, except for one group of presmolts released in the Methow River. In some years, Wells Hatchery mined large portions (49% in 1969) of the summer/fall Chinook destined for the Methow River and other upstream tributaries (Mullan 1987). Upriver bright fall Chinook from Priest Rapids have entered the summer/fall Chinook broodstock complement at Wells Hatchery. We assume that they have also spawned in areas where they may mix with adults from natural spawning in various tributary and mainstem areas.

For several years, before the volunteer entrants at the Priest Rapids Hatchery trap became abundant enough to support broodstock needs, virtually all adult fall-run Chinook destined for upriver spawning areas were trapped at Priest Rapids fishway trap.

That “mining” of upriver fall-run fish probably took some summer-run fish that arrived after the cut-off date for summer Chinook, and prevented late-run Chinook from spawning upstream from Priest Rapids Dam. It, thus, may have mixed late-run Chinook from the mid-Columbia region upstream from Priest Rapids Dam with Hanford Reach late-run Chinook (Chapman et al. 1994).

Hydroelectric Effects

Anadromous salmonids, including upper Columbia River summer Chinook, depend on intact, complex and functioning habitat to support healthy populations. Perturbations in habitats throughout the Columbia Basin and in ocean environments are replete, including those associated with the mainstem Columbia River’s hydroelectric development and operation. The development of the hydropower facilities throughout the Columbia River Basin has irrevocably altered terrestrial and aquatic habitats, and is a contributor to limiting anadromous fish populations. In attempts to mitigate for hydro-related impacts in the Mid-Columbia Region, WDFW operates summer Chinook supplementation programs associated with the HCPs of Wells Dam, Rocky Reach Dam.

According to the Chelan HCP 2002 for Rocky Reach and Douglas HCP for Wells Dam, the Districts will provide hatchery compensation for Plan Species, including summer/fall Chinook salmon upstream of Rock Island and Wells Dams. This compensation may include measures to increase the off-site survival of naturally spawning fish or their progeny.

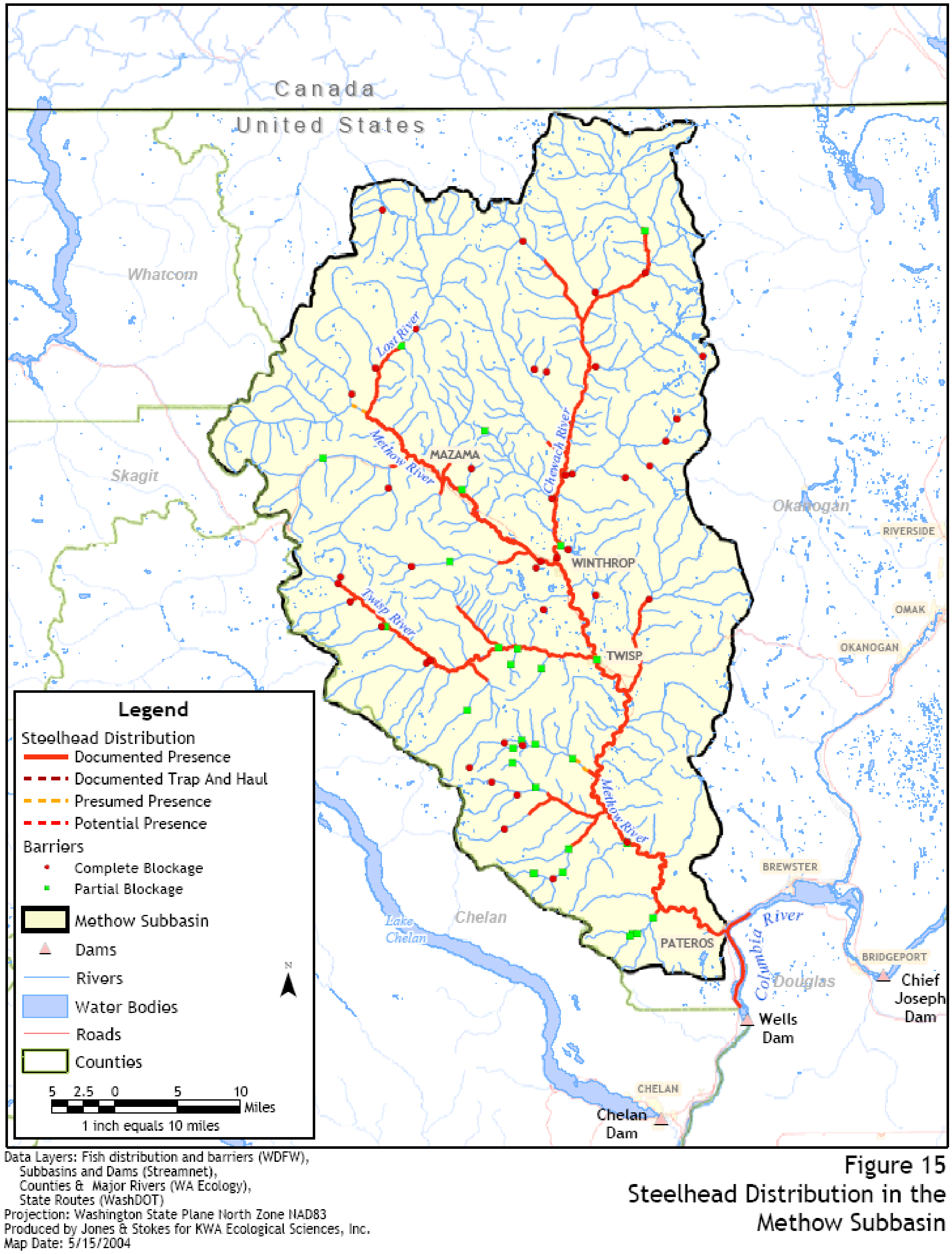
The Districts will implement the specific elements of the hatchery program consistent with overall objectives of rebuilding natural populations and achieving No Net Impact. Species-specific hatchery programs objectives developed by the Joint Fisheries Parties may include contributing to the rebuilding and recovery of naturally reproducing populations in their native habitats, while maintaining genetic and ecologic integrity, and supporting harvest.

Harvest Effects

High harvest rates in the lower Columbia River depleted populations of upper Columbia River summer Chinook by the late 1800s (McDonald 1895). In the 1930s, the fishing rate remained at almost 90% and summer Chinook escapement to Rock Island Dam hovered around 5,600 fish (Chapman et al. 1994a). Industrial development of the Columbia River system, coupled with historical over-harvest, reduced escapement. Harvest rates were reduced in 1951, and the run rebounded to an average escapement range of 20,000 to 35,000 fish at Rock Island Dam.

Summer Chinook from the region are currently harvested only incidentally in lower Columbia River fisheries that are directed at other species, and no directed commercial fisheries on upper Columbia summer-run fish have occurred in the mainstem since 1964 (BAMP 1998). During the years 1982-1989, the brood year average ocean fisheries' exploitation rate for Columbia River stocks was 39%, with a total exploitation rate of 68% estimated for the same years (Myers et al. 1998).

3.4.3 Summer Steelhead



Source: Data Layers: Fish distribution and barriers (WDFW), Subbasins and Dams (StreamNet), Counties & Major Rivers (WA Ecology), State Routes (WashDOT). Projection: Washington State Plane North Zone NAD83. Produced by Jones & Stokes for KWA Ecological Sciences, Inc. Map Date: 5/15/2004")

Figure 15 Steelhead distribution in the Methow subbasin

Rationale for Selection

The Methow summer steelhead stock is listed in the Washington State Salmon and Steelhead Stock Inventory (SASSI) as Depressed based on chronically low numbers (WDF and WDW, 1993). Upper Columbia summer steelhead were listed as Endangered under the ESA in 1997.

Key Life History Strategies, Relationship to Habitat

It is difficult to summarize one life history strategy (anadromy) without due recognition of the other (resident). The two strategies co-mingle on some continuum with certain residency at one end, and certain anadromy on the other. Upstream distribution is limited by low heat budgets (about 1,600 temperature units) (Mullan et al 1992b).

The response of steelhead/rainbow complex in these cold temperatures is residualism, presumably because growth is too slow within the time window for smoltification. However, these headwater rainbow trout contribute to anadromy via emigration and displacement to lower reaches, where warmer water improves growth rate and subsequent opportunity for smoltification.

Summer steelhead spawn in late winter, spring, and early summer in the mainstem Methow River and its tributary streams. Although steelhead are iteroparis (life after spawning), kelts represent less than 1.0% of the annual spawning population (Brown 1995). The low occurrence of repeat spawners may be related to post-spawn Columbia River discharge or spill frequency, duration, and/or sequential timing (Brown 1995). However, Chapman et al. (1994b) suggested the number of repeat spawners pre-development was never high.

Spawning grounds are not surveyed for steelhead because the adults generally spawn over a four- to five-month period that coincides with high spring flows when water visibility is low and discharge high. Preliminary surveys, conducted during the low water season in 2001, supported expected redd locations (Chapman et al. 1994b). Spawning and rearing distribution correlate closely (Mullan et al. 1992b). Unlike other species in the *Oncorhynchus* genus, steelhead eggs incubate at the same time that temperatures are increasing.

Steelhead, destined for the Methow subbasin, pass Wells Dam in July through the following May, with peak migration in September. Mullan et al. (1992b) was unable to detect a significant difference between run timing of hatchery and wild fish passing Wells Dam. Most adults hold in the mainstem Columbia River through the winter; although, some hold in large, deep pools associated with the Methow River downstream of Winthrop.

The return percentage of hatchery origin adults to and over Wells Dam is provided in **Table 22**.

Table 21 Hatchery and wild steelhead counts at Wells Dam

Year	Run to Wells Dam	Number in broodstock			Wild%	Run over Wells Dam		
		Hatchery	Wild	Total		Hatchery	Wild[5]	Total

[5] Assumes wild fish were representative of the entire run.

Year	Run to Wells Dam	Number in broodstock			Wild%	Run over Wells Dam		
1967	2199			171				2028
1968	2667			413				2254
1969	1299			530				769
1970	2023			399				1624
1971	4257			358				3899
1972	2069			354				1715
1973	2473			627				1846
1974	632			260				372
1975	732			227				505
1976	4973			337				4636
1977	5819			355				5464
1978	1831			356				1475
1979	4138			367				3771
1980	3735			372				3363
1981	4757			650				4107
1982	8395	552	38[6]	590	0.065	7298	507	7805
1983	20200	661	9	670	0.013	19276	254	19530
1984	17353	673	17	690	0.025	16246	417	16663
1985	20462	718	32	750	0.043	18864	848	19712
1986	13901	631	20	650	0.030	12853	398	13251
1987	6168	528	75	603	0.124	4875	609	5565

[6] 1982-1986 wild fish estimated by dorsal fin condition and otoliths. 1987-1999 adipose fins were clipped on all hatchery fish.

Year	Run to Wells Dam	Number in broodstock			Wild%	Run over Wells Dam		
1988	5010	581	70	651	0.108	3888	471	4359
1989	5301	629	95	724	0.131	3977	600	4577
1990	4577	644	91	735	0.124	3366	476	3842
1991	8481	588	70	658	0.107	6986	837	7823
1992	7628	599	34	633	0.054	6617	378	6995
1993	3043	534	46	586	0.079	2263	194	2457
1994	2800	581	38	619	0.062	2045	136	2181
1995	1472	521	0	521	0.123	834	117	951
1996	4523	350	19	369	0.051	3942	212	4154
1997	4534	449	11	460	0.024	3976	98	4074
1998	3083	379	31	410	0.076	2470	203	2673
1999	3958	341	47	388	0.121	3138	432	3570

Table 22 Summary of life history timing for Methow subbasin summer steelhead

Life history stage	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Adult migration	■	■	■	■	■		■	■	■	■	■	■
Adult spawning			■	■	■	■	■					
Egg incubation			■	■	■	■	■	■				
Juvenile rearing	■	■	■	■	■	■	■	■	■	■	■	■
Smolt migration				■	■	■						

Population Delineation and Characterization

Summer steelhead, native to the Methow basin, are the exclusive ecotype of the inland waters. Steelhead were not extirpated in the Methow River, as were coho, probably because of headwater resident forms sustaining the run (Mullan et al. 1992b). Anadromy is not obligatory in *O. mykiss* (Rounsefell 1958; Mullan et al. 1992b).

Progeny of anadromous steelhead can spend their entire life in freshwater, while progeny of rainbow trout can migrate seaward. Anadromy, although genetically linked (Thorpe 1987), runs under environmental instruction (Shapovalov and Taft 1954; Thorpe 1987; Mullan et al. 1992b).

Population Status

The Methow subbasin once was a productive wild steelhead system, but has declined significantly since the early 1900s. Wild summer steelhead in the Methow subbasin sustain themselves only at threshold population size today, and were listed as Endangered under the ESA on August 18, 1997.

Population Management Regimes and Activities

In addition to the effects of an obstruction constructed by Washington Water Power's dam, which blocked the Methow River at Pateros in the early 1900s, a historic network of unlined ditches grew with the settlement, along with roads and land clearing.

The mainstem barrier was removed circa 1929, and the network of ditches has now been converted to at least 27 irrigation canals operated by both public and private entities in the Methow subbasin; operations incorporate a range of fish mitigation measures. Many operators have upgraded their facilities in recent years; enhancements include elimination of fish passage barriers, replacement and repair of screens, and improvements to the overall irrigation system. Some of these have established target flows and habitat conservation plans.

Hatchery programs that have been designed to mitigate for losses in the major mainstem Columbia River hydro system, and in non-target or catch-and-release recreational fisheries combined with selective tribal fisheries, appear to have reversed declines in wild populations.

Hatchery Effects

The high hatchery return rate, the genetic homogeneity of hatchery and wild steelhead (Chapman et al. 1994b), and the maintenance of near-maximum sustained yield (MSY) levels in most years suggest a truly wild fish does not exist. Rather, natural production sustains them only at threshold levels; without hatchery supplementation, the Methow River steelhead would suffer dire consequences.

Despite the natural production sustaining them at threshold population size, the biological fitness of the hatchery spawners allows the population to meet pre-development MSY escapement and smolt production in most years (Mullan et al. 1992b).

A high percentage of hatchery males can return after one winter (Brown 1995; Bartlett 1999-2000). This does not mean that the hatchery fish are the "ecological equivalents of wild fish in all life history phases" (Chapman et al. 1994b); although, Mullan et al. (1992b) found no difference in smolt-to-adult survival for hatchery versus wild steelhead. A portion of the hatchery-released steelhead remains in the freshwater for another winter (Bartlett 1997, 1999-2000; K. Williams, pers. comm.), increasing the fitness of returning adults (Chapman et al. 1994b). In addition, the resident form contributes to anadromy at varying degrees, inversely related with the steelhead productivity.

The return percentage of hatchery verses wild origin adult summer steelhead over Wells Dam is provided in (**Table 23**).

Table 23 Hatchery versus wild origin adult summer steelhead over Wells Dam

Release year	Smolts released[7]	Adult return to Wells Dam[8]	1-salt fish[9]	2-salt fish	% to Wells Dam	Adult return over Wells Dam	1-salt fish	2-salt fish	% return over Wells Dam
1966	199720				1.19				1.06
1967	187676	2199	1319	880	1.13	2028	1217	811	0.88
1968	100644	2667	1600	1067	1.57	2254	1352	902	1.10
1969	205457	1299	779	520	1.42	769	461	308	1.23
1970	322462	2023	1214	809	1.05	1624	974	650	0.94
1971	220384	4257	2554	1703	1.02	3899	2339	1560	0.81
1972	327902	2069	1241	828	0.59	1715	1029	686	0.42
1973	170602	2473	1459	1014	0.16	1846	1089	757	0.10
1974	182111	632	145	487	0.90	372	86	286	0.76
1975	249279	732	600	132	2.14	505	414	91	2.00
1976	238405	4973	3929	1044	2.52	4636	3662	974	2.27
1977	147922	5819	4422	1397	0.29	5464	4153	1311	0.24
1978	164259	1831	256	1575	2.99	1475	207	1269	2.72
1979	268252	4138	3972	166	2.69	3771	3620	151	2.36
1980	471420	3735	2801	934	0.95	3363	2522	841	0.94
1981	358234	4757	333	4424	1.25	4107	287	3820	1.24

[7] Includes only smolts planted at or above Wells Dam.
[8] Includes broodstock plus dam count. 1967-1982 is combination of hatchery and wild. 1982-1999 is hatchery fish only.
[9] 1967-1972 ocean age unknown, but estimated by 0.6 and 0.4 for 1-salt and 2-salt, respectively. Return rates prior to 1982 were combination of hatchery and wild.

Release year	Smolts released[7]	Adult return to Wells Dam[8]	1-salt fish[9]	2-salt fish	% to Wells Dam	Adult return over Wells Dam	1-salt fish	2-salt fish	% return over Wells Dam
1982	379472	7849	3689	4160	7.54	7805	3668	4137	7.27
1983	494784	19937	19140	797	3.48	19276	18505	771	3.35
1984	466545	16919	7444	9475	3.95	16246	4148	9098	3.78
1985	413066	19582	9791	9791	1.83	18864	9432	9432	1.71
1986	452844	13484	4854	8630	1.22	12853	4627	8226	1.08
1987	564315	5403	2702	2702	0.57	4875	2437	2437	0.49
1988	826208	4469	1654	2815	0.69	3888	1439	2450	0.59
1989	623003	4607	3040	1566	0.67	3977	2625	1352	0.60
1990	740433	4009	1323	2686	1.19	3366	1111	2255	1.10
1991	656997	7574	4696	2878	0.82	6986	4331	2655	0.71
1992	541610	7216	3067	4149	0.42	6617	2812	3805	0.22
1993	511295	2803	477	2326	0.35	2263	385	1878	0.35
1994	420110	2626	945	1681	0.44	2045	1248	757	0.36
1995	450345	1355	501	840	1.19	834	309	517	1.08
1996	347950	4292	2962	1331	0.99	3942	2720	1222	0.87
1997	427900	4425	2036	2390	0.64	3976	1829	2147	0.57
1998	543030	2849	1453	1396		2470	1260	1210	
1999	843385	3479	2192	1287		3138	1977	1161	

Hydroelectric Effects

As noted in Section 2.1, steelhead populations in the subbasin were severely depressed following the removal of Washington Water Power Company's dam on the Methow River at Pateros in about 1929, with steelhead persisting as rainbow trout (Mullan et al. 1992b).

Anadromous salmonids, including upper Columbia River summer steelhead depend on intact, complex and functioning habitat to support healthy populations. Perturbations in habitats throughout the Columbia Basin and ocean environments are replete, including those associated with mainstem Columbia River's hydroelectric development and operation.

Continued development of the hydropower facilities throughout the Columbia River Basin have irrevocably altered terrestrial and aquatic habitats, and have contributed to limiting anadromous fish populations. Today, anadromous fish migrating to the ocean encounter Wells Dam just downstream from the Methow's confluence with the Columbia River. Beyond Wells Dam, eight more downstream dams along the Columbia River impede fish passage to the ocean.

Wells Dam fishway, which became operational in 1967, estimated wild run size above the dam at 1,500 to 2,000 fish in the late 1960s (**Table 23**). Hatchery fish made up an increasing fraction of the steelhead run after the 1960s, as wild runs were already depleted (Chapman et al. 1994b). Mullan et al. (1992b) spawner-recruit analysis calculated the MSY run size and escapement for the Methow subbasin at 7,234 fish and 2,212 fish, respectively.

In attempts to mitigate for hydro-related impacts in the Mid-Columbia Region, WDFW operates summer steelhead supplementation programs associated with the Wells Dam HCP. The goal of the artificial production programs is two-fold: a) to mitigate for fishery losses because of inundation and to provide No Net Impact of unavoidable losses because of operation of Wells Dam, Rocky Reach Dam and Rock Island Dam, while; b) contributing to the rebuilding and recovery of naturally reproducing populations in their native habitats, maintaining genetic and ecological integrity and supporting harvest.

Harvest Effects

Wild fish have been subjected to, and have suffered as a result of, mixed stock fisheries in the lower Columbia River that are directed at their abundant hatchery cohort.

The intensive commercial fisheries in the late 1800s, along with industrial development of the Columbia River, were largely responsible for the decline of the wild steelhead run (Mullan et al. 1992b; Chapman et al. 1994b). Curtailing the commercial fisheries resulted in a resurgence of wild steelhead productivity in the upper Columbia River region, where the run size tripled (5,000 fish to 15,000 fish) between 1941 and 1954 (Mullan et al. 1992b).

Commercial harvest of steelhead by non-tribal members was prohibited beginning in 1975. Incidental catches of steelhead do occur in present-day sockeye and fall salmon fisheries within Zones 1-5, but are minimized with time, area, and gear restrictions.

Above Bonneville, in Zone 6, only the treaty tribes conduct commercial harvest. The Zone 6 tribal commercial fishery does not selectively remove wild steelhead from gill nets, thus, both marked and unmarked fish are retained. Total catches in recent years (1985 through 1996) ranged from 86,000 in 1985 down to 5,300 in 1998. Between 1990 and 1998, tribal catches have averaged 22,100 (WDFW & ODFW 1999). Current information based on GSI analysis, however, indicates an impact of less than 10% for upper Columbia stocks (Rawding et al. 1998).

Recreational fisheries occur throughout the Columbia and Snake River watersheds. Fisheries that harvest upper Columbia steelhead occur in Zone 6 waters above the Snake River confluence, including Hanford Reach up to Chief Joseph Dam and major tributaries in Wenatchee, Entiat, Methow and Okanogan watersheds.

Since 1984, wild steelhead release has been required in these waters (i.e., steelhead with adipose fins), and since 1997, no recreational fishery targeted at steelhead has been permitted above Priest Rapids Dam. The Confederated Tribes of the Colville Indian Reservation (CCT) do take steelhead incidental to their summer Chinook snag fishery below Chief Joseph Dam, and in the Okanogan River net fishery, but Chapman et al. (1994b) concluded tribal fishing above Zone 6 was insignificant, and despite large numbers being taken in some years, the overall percentage of the catch to the total run was low.

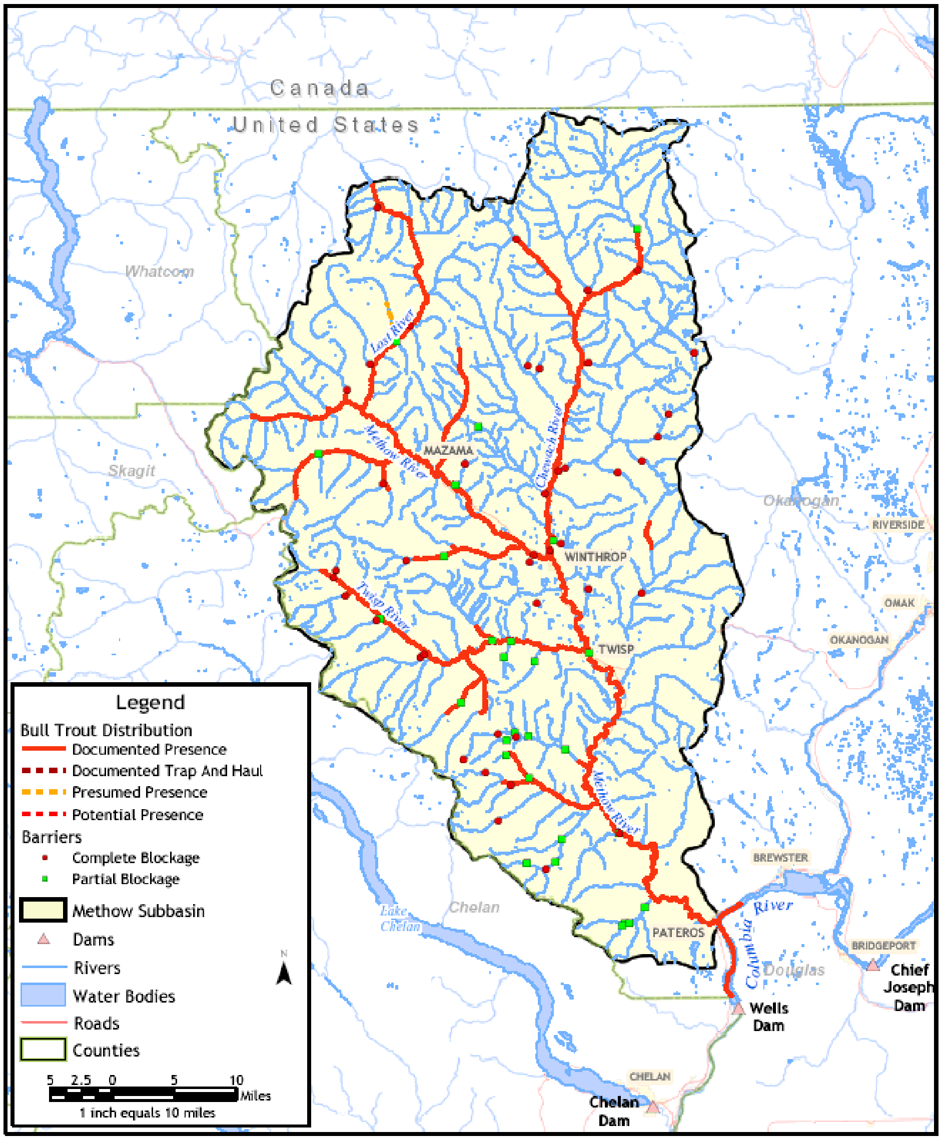
3.4.4 Bull Trout

Historically, four general forms of bull trout are recognized, each with a specific behavioral or life history pattern: anadromous, adfluvial, fluvial, and stream-resident. The Methow River subbasin is known to support fluvial, adfluvial and resident populations of bull trout. Known distribution of bull trout in the Methow subbasin are illustrated in **Figure 16**.

Adfluvial populations of bull trout are found in the Lost River and Lake Creek. Fluvial populations of bull trout are found throughout the Methow subbasin. Resident populations are found in many other streams including upstream of many natural barriers.

Rationale for Selection

The FWS listed the Columbia River Distinct Population Segment (DPS) for bull trout as Threatened under the Endangered Species Act of 1973, as amended on June 10, 1998. Methow subbasin bull trout, as a focal species, will enable subbasin-specific management prescriptions relating to the Columbia River bull trout recovery plan.



Source:Data Layers: Fish distribution and barriers (WDFW), Subbasins and Dams (StreamNet), Counties & Major Rivers (WA Ecology), State Routes (WashDOT). Projection: Washington State Plane North Zone NAD83. Produced by Jones & Stokes for KWA Ecological Sciences, Inc. Map Date: 5/15/2004")

Figure 16 Bull Trout Distribution in the Methow Subbasin

Key Life History Strategies, Relationship to Habitat

Bull trout distributions in the Methow watershed parallel the habitat conditions; the more pristine the habitat, the more robust the bull trout populations. Proebstel et al. (1998) reported that in general, bull trout were found to be persisting in small headwater populations. The Lost River and Robinson Creek Watershed Analysis (USFS 1999c) states, "Roads, access, and resultant overfishing in most waters are probably the most limiting production factors to bull trout resulting from man's influence."

Bull trout have more specific habitat requirements that do other salmonids. Their habitat components requirements are summed up by the "Four C's" – cold, clean, complex and connected. Bull trout are believed to be among the most temperature sensitive cold-water species found in western North America (Dunham et al. 2003). Water temperatures above 15 degrees Celsius (59 degrees Fahrenheit) are believed to limit bull trout distribution, a limitation that may partially explain their patchy distribution within a watershed (Fraley and Shepard 1989; Rieman and McIntyre 1995; Dunham et al. 2002).

Bull trout normally reach sexual maturity in 4 to 7 years and have a life span of 12 or more years. Repeat and alternate year spawning has been reported, although repeat spawning frequency and post spawning mortality are not well documented (Leathe and Graham 1982; Fraley and Shepard 1989: Pratt 1992; Rieman and McIntyre 1996).

Adult Methow bull trout migrate from some of the warmest water in their range in the Columbia River, back to cold headwater streams to spawn in the Methow. The coldest water is most often found in isolated headwater stream locations. After entering tributaries, most bull trout remained within them until October-November, when they migrated back to the mainstem Columbia River (BioAnalysts 2002, 2003).

Migration of bull trout from the Columbia River into the Methow subbasin occurs in May through June (BioAnalysts 2002, 2003). Spawning begins in headwater streams in mid-September and continues through October, with temperatures during spawning of 41 to 48 degrees Fahrenheit (3 to 9 degrees Celsius) (Goetz 1989; Brown 1994).

Migratory juveniles usually rear in natal streams for one to four years before emigration (Goetz 1989; Fraley and Shepard 1989; Pratt 1992). Methow subbasin juvenile bull trout rear in the coldest headwater locations until they reach a size that allows them to compete with other fish (75-100 mm) (Mullan et al. 1992 CPb).

Non-migratory forms above barrier falls probably contribute a limited amount of recruitment downstream; nevertheless, this recruitment contributes to fluvial and adfluvial productivity. The fluvial forms migrate to the warmer mainstem Methow and Columbia Rivers (e.g., Twisp River, Wolf Creek), while the adfluvial populations (e.g., Lake Creek, Cougar Lake) migrate to nearby lakes.

In Methow subbasin tributaries, bull trout spawning and early rearing is confined to streams cold enough (less than 1,600 C annual temperature units) to support them in the areas below the falls (Mullan et al. 1992 CPa). In most cases, such reaches are very short (less than 5 miles).

Repeat- and alternate-year spawning has been reported, although repeat-spawning frequency and post-spawning mortality are not well documented (Leathe and Graham 1982; Fraley and Shepard 1989; Pratt 1992; Rieman and McIntyre 1996).

A summary of bull trout spawning surveys in the Methow is provided in . After spawning, fluvial and adfluvial kelts return to their more moderate environments, while resident forms seek winter refuge (**Table 24**).

Table 24 Bull trout survey summary for the Methow subbasin (1992-2003)

Stream	'92	'93	'94	'95	'96	'97	'98	'99	'00	'01	'02	'03
Chewuch River Mainstem										9	11	6
-Lake Creek up stream of Black Lake				22	13*	9	8	0	8	21	11	10*
-Lake Creek down stream of Black Lake									4	1		4
Methow River												
-Goat Creek				0					11*		4	3
-Lost River	5*		0	0*			0					
-Monument Creek	2*	0										
-Crater Creek					2*	2	1	0		0	1	0
-Wolf Creek					3	3*	27	29	15	20	15	18*
-Early Winters Creek					9*	1*	2	0	3	5	6	0*
-Cedar Creek					1	2*		0				
-West Fork Methow River				27	15	13*	11*	1	2	19	54	
Twisp River												
-Twisp River North Fork to Barrier Falls	3*	5*	4*	18	0*	2*	67	38	72	53	67	30
-Twisp River Reynolds Creek to South Creek										19	13	16

-East Fork Buttermilk				4*	0*		0	0*	0	2	3	3
-West Fork Buttermilk											7	9
-Reynolds Creek	1*				0*					1*	0	
_North Creek				3*			19	63	33	0	2	29

*Incomplete counts as to time (single survey) and/or space (only part of index area surveyed).

summarizes redd counts of most known spawning populations. Full inventories of all streams for bull trout presence and redd counts are not complete.

Relationship with Other Species

In the Columbia River basin, bull trout occur with native cutthroat trout (*Oncorhynchus clarki* subspecies), resident (redband) and migratory (steelhead) rainbow trout (*O. mykiss*), Chinook salmon (*O. tshawytscha*), sockeye salmon (*O. nerka*), mountain whitefish (*Prosopium williamsoni*), and various sculpin (*Cottidae*), sucker (*Catostomidae*), and minnow (*Cyprinidae*) species (Mauser et al. 1988; WDF et al. 1993; WDFW 1998).

Bull trout habitat within the Methow River Basin overlaps with the range of several fishes listed as Threatened, Endangered, or proposed for listing under the Endangered Species Act (ESA), including Endangered steelhead and Endangered Upper Columbia River Spring Chinook. Because of short cold water reaches, suitable spawning habitat is largely limited in the east and west forks of Buttermilk Creek, Chewuch River, Crater Creek, Goat Creek, Wolf Creek, Lost River, Early Winters Creek, Cedar Creek, Monument Creek and Reynolds Creek. Behnke (2002) notes that the relatively smaller size of westslope cutthroat trout (WSCT) adults compared to other cutthroat subspecies may be because of their coevolution with two highly piscivorous species: bull trout and northern pikeminnow.

Non-native salmonids have been widely introduced, and have become established in numerous areas throughout the range of bull trout. These species include brook trout (*Salmo fontinalis*), lake trout brown trout (*S. trutta*), Arctic grayling (*Thymallus arcticus*), and lake whitefish (*Coregonus clupeaformis*). Kokanee (a freshwater form of *O. nerka*), non-native strains of rainbow trout, and non-native subspecies of cutthroat trout have also been introduced into areas where they did not occur naturally. Other non-native species that have been introduced into habitat occupied by bull trout include smallmouth bass, walleye, opossum shrimp, channel catfish, American shad, and yellow perch.

Population Delineation and Characterization

A summary of five surveyed bull trout spawning aggregates is illustrated in **Table 25**.

Table 25 Five potential Methow subbasin bull trout spawning aggregates with life history representation

Aggregate	Resident	Fluvial	Adfluvial
Chewuch River (including Lake Creek)			
		X	X
Upper Methow R. (including West Fork Methow, Early Winters/Cedar creeks, Wolf Creek, Goat Creek)			
	X	X	X
Lower Methow R. (including Blue Buck/Beaver creeks, Crater/Gold creeks)			
		X	
Twisp River (including North Creek, Buttermilk Creek, Reynolds Creek)			
	X	X	
Lost River (including upper Lost River, Monument Creek, Cougar and Hidden lakes)			
	X	X	X

The USFWS Draft Bull Trout Recovery Plan (2002) delineated 8 local populations of bull trout within the Methow Core Area. However; the Upper Columbia Bull Trout Recovery Team has modified their delineation to 9 populations. These populations include Gold, Beaver, Wolf, Goat, and Early Winters creeks and Twisp, Chewuch, Lost and Upper Methow rivers (Barbara Kelly-Ringel 2004, pers.comm.). Comprehensive redd surveys, coupled with preliminary radio telemetry work in the Wenatchee basin, suggests the 9 remaining spawning populations may not be complete genetic isolates of one another but rather possibly co-mingle to some degree. It is possible that the nine spawning aggregates represent the Methow subbasin, but more monitoring and DNA analysis is necessary. The Lost River aggregate gene flow occurs only in high water years and not always between all represented groups. Assumptions regarding the historic and current distribution of bull trout in the Methow subbasin as part of the QHA Analysis are summarized in electronic Appendix C.

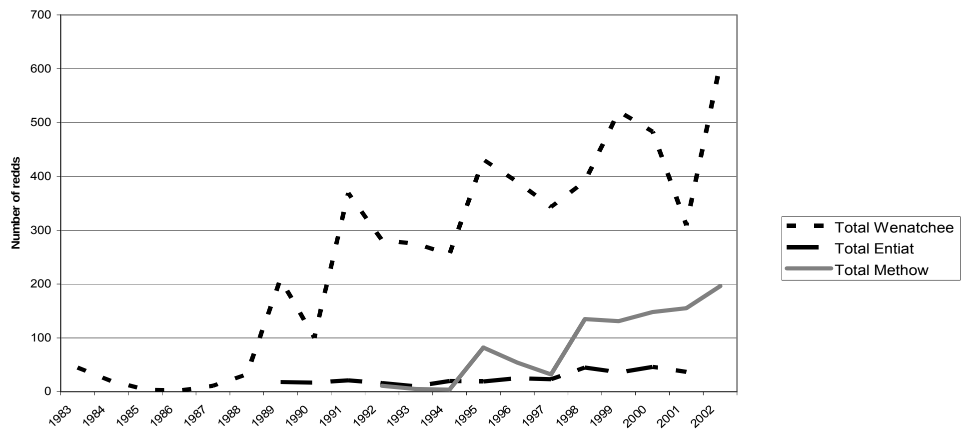
Population Status

Columbia River DPS bull trout are listed as Threatened. The FWS's Bull Trout Recovery Plan is under development and will be finalized within the next year. The FWS is currently in the process of finalizing the Critical Habitat Designation for the Columbia River DPS. This designation will be final on September 23, 2004. (Kate Terrell USFS 2004, pers. comm.). The current version of the recovery plan is available at http://pacific.fws.gov/bulltrout/recovery/Default.htm

Bull trout distribution, abundance, and habitat quality have declined rangewide (Bond 1992; Schill 1992; Thomas 1992; Ziller 1992; Rieman and McIntyre 1993; Newton and Pribyl 1994; Idaho Department of Fish and Game, in litt. 1995; McPhail and Baxter 1996). These declines result from the combined effects of habitat degradation and fragmentation, the blockage of migratory corridors, poor water quality, angler harvest and poaching, entrainment (process by which aquatic organisms are pulled through a diversion or other device into diversion channels and dams), and introduced non-native species.

The Methow River Basin has eight local populations (FWS). Of these, only the Lost River is considered healthy; the rest are listed as unknown (WDFW 1998). It appears that most of the local populations of bull trout, and in particular, the non-migratory forms, have little or no information available concerning their status. This is identified by FWS (2002) as a major need to help recover bull trout. Redd surveys began in the Methow River subbasin in the early 1990s to complement other spawning grounds surveys in the upper Columbia.

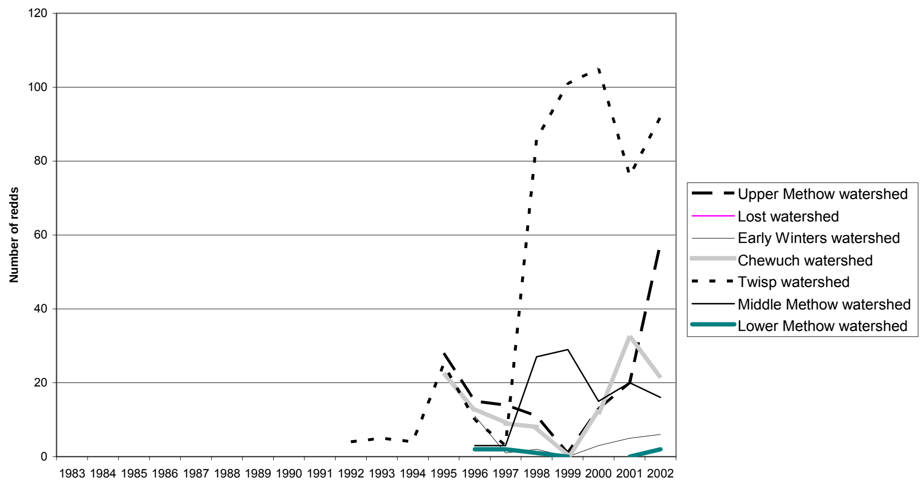
Bull trout redd counts in all subbasins within the CCP show an increase since the mid-1990s, especially within the Methow Basin (**Figure 17**); although, it should be noted that this trend may be a factor of increased effort in redd surveys in recent years (K. MacDonald, USFS, pers. comm.).



Source: Pevan 2004. Data from USFWS and USFS

Figure 17 Comparison of bull trout redd counts between the Wenatchee, Entiat, and Methow Subbasins

Within the Methow subbasin, the Twisp River basin is the largest producer of bull trout, averaging two to three times more redds than any other spawning area within the Methow Basin (**Figure 18**). The average number of redds within the basin has increased from less than 100 in the mid-1990s to greater than 150 since 1998.



Source: Pevan 2004. Data from USFWS and USFS

Figure 18 Bull trout redd counts in the Methow River Basin

Population Management Regimes and Activities

Recent comprehensive redd surveys, coupled with preliminary radio telemetry work in the CPP, suggest that remaining spawning populations are not complete “genetic isolates” of one another, but rather co-mingle to some degree (Foster et al. 2002). This comports with the belief of the prevalence of three independent populations in the CPP (in the Wenatchee, Entiat, and Methow). It is possible that there are separate, local spawning aggregates, but more monitoring and DNA analysis is necessary to be able to empirically determine this.

The chance of finding independent subpopulations within each subbasin would most likely found be in headwater areas upstream of barriers, which prevent immigration from downstream recruits, but not emigration to downstream areas during high water events.

Hatchery Effects

Introduced brook trout threaten bull trout through hybridization, competition, and possibly predation (Thomas 1992; WDW 1992; Clancy 1993; Leary et al. 1993; Rieman and McIntyre 1993; MBTSG 1996h). Hybridization results in offspring that are frequently sterile (Leary et al. 1993), although recent genetics work has shown that reproduction by hybrid fish is occurring at a higher level than previously suspected (Kanda 1998). Hybrids may be competitors; Dunsmoor and Bienz (L. Dunsmoor and C. Bienz, Klamath Tribe, in litt. 1997) noted that hybrids are aggressive and larger than resident bull trout, suggesting that hybrids may have a competitive advantage. Brook trout mature at an earlier age and have a higher reproductive rate than bull trout. This difference may favor brook trout over bull trout when they occur together, often leading to replacement of bull trout with brook trout (Clancy 1993; Leary et al. 1993; MBTSG

1995b). The magnitude of threats from non-native fishes is highest for resident bull trout because they are typically isolated and exist in low abundance.

Non-native brook trout may have an adaptive advantage over sympatric bull trout in degraded habitats where seasonal water temperatures or fine sediments levels, for example, are elevated above historical levels (Clancy 1993; Rich 1996; Dunsmoor and Bienz, in litt. 1997; Adams 1994; MBTSG 1998, 1996h). Because elevated water temperatures and sediments are often indicative of degraded habitat conditions, bull trout may be subject to stresses from both interactions with brook trout and degraded habitat (MBTSG 1996h).

Bull trout are present in Blue Buck and the mainstem of Beaver Creek; however, populations in Eightmile Creek are of unknown status. Cold water is not a deterrent for brook trout, and maturation of brook trout occurs at ages two to four, whereas maturation for bull trout occurs at ages six to nine (Mullan et al. 1992b). Since there are no barriers to block their passage, brook trout found in the Twisp River can easily invade the bull trout zone upstream; competition with other species, however, has probably limited brook trout productivity.

Hydroelectric Effects

Dams affect bull trout by: altering habitats; flow, sediment, and temperature regimes; migration corridors, and by creating additional well-coordinated interactions, mainly between bull trout and non-native species (Rode 1990; WDW 1992; Craig and Wissmar 1993; Rieman and McIntyre 1993; Wissmar et al. 1994; T. Bodurtha, FWS, in litt. 1995; USDA and USDI 1996, 1997). Impassable dams have caused declines of bull trout by preventing migratory fish from reaching spawning and rearing areas in headwaters and recolonizing areas where bull trout have been extirpated (Rieman and McIntyre 1993; MBTSG 1998).

Some of the major effects to bull trout resulting from the Federal Columbia River Power System, and from operation of other hydropower, flood control, and irrigation diversion facilities, include the following: a) fish passage barriers; b) entrainment of fish into turbine intakes and irrigation canals; c) inundation of fish spawning and rearing habitat; d) modification of stream flows and water temperature regimes; e) dewatering of shallow water zones during power peaking operations; f) reduced productivity in reservoirs; g) periodic gas supersaturation of waters downstream of dams; h) water level fluctuations interfering with retention of riparian vegetation along reaches affected by power peaking operations; i) establishment of non-native riparian vegetation along reaches affected by power peaking operations, and; j) severe reductions in reservoir levels to accommodate flood control operations. Recent studies indicate that adult bull trout are passing the Mid-Columbia dams at rates similar to their anadromous salmonid counter parts (BioAnalysts, 2003).

The Chief Joseph and Grand Coulee dams were built without fish passage facilities, and are barriers to bull trout migration. These barriers have contributed to the isolation of local populations of migratory bull trout. The middle Columbia, and lower Columbia River hydropower projects have both adult and juvenile fish passage facilities, but these fishways were designed specifically for anadromous salmonids, not for resident fish such as bull trout. The designs, therefore, address the migration needs of anadromous salmonids, primarily semelparous (i.e., fish that spawn only once in a lifetime) of the genus *Oncorhynchus* (except steelhead, that, in some instances, can spawn more than once in a lifetime). They do not include consideration for iteroparous fish (i.e., those that can spawn more than once), or fish that merely wander both upstream and downstream as adults to forage. Bull trout have been observed using upstream fish passage facilities at many of the hydropower projects on the Columbia River. However, as indicated above, even dams with fish passage facilities may be a factor in isolating bull trout local populations if they are not readily passable by bull trout and/or if the dams do not provide an adult downstream migration route.

Entrainment of bull trout may also occur at various projects in the Columbia River basin, including Rocky Reach, Rock Island, Wells, and Bonneville dams. Fish can be killed or injured when passing the dams. Potential passage routes include through spill, turbines, or the juvenile bypass systems, but the relative passage success of these routes for adult salmonids has not been thoroughly investigated. One study conducted in the early 1970s, however, revealed that passage through turbines resulted in a 22-41% mortality rate for adult steelhead (Wagner and Ingram 1973). Additionally, a 40-50% injury rate for adult salmonids passing through the juvenile fish bypass system at McNary Dam has been noted (Wagner 1991; Wagner and Hilson 1993). Adult bull trout may experience similar mortality rates. Moreover, those adult fish that survive passage at projects that do not have upstream passage facilities are isolated in downstream reaches away from their natal (native) streams. As indicated above, the loss of these larger, more fecund migratory fish is detrimental to their natal populations. A three year radio telemetry study was initiated in 2001 to track bull trout movement within the Upper Columbia region. A total of 79 bull trout were tag at the three Mid-Columbia Dams (Rock Island, Rocky Reach and Wells). During this study, no mortalities of bull trout associated with the dams were documented (BioAnalysts 2002, 2003).

The creation of mainstem Columbia River pools (i.e., the areas of slow moving water behind the dams) combined with introductions of piscivorous species (e.g., bass, walleye) has also affected the habitat of bull trout and other salmonids. An increase in predator populations, both native (e.g., northern pikeminnow) and non-native, as a result of creating artificial habitat and concentrating prey, is discussed as a factor for the decline of each listed Snake River salmon species (NMFS 1991a, b, and c). Ideal predator foraging environments have been created in these pools, particularly for warm water species in the summer. Smolts that pass through the projects are subjected to turbines, bypasses, and spillways that may result in disorientation and increased stress, reducing the smolts' ability to avoid predators below the dams. Creation of the pools above the dams has resulted in low water velocities that increase smolt travel time and increase predation opportunity. Increased water temperatures, also a result of the impoundment of the river, have also been shown to increase predation rates on salmonid smolts (Vigg and Burley 1991). Because bull trout are apex (top) predators of other fish, negative effects to the salmonid smolt prey base, and the resulting decline in adult returns, are likely to affect bull trout negatively as well. Additionally, increased water temperatures, influenced by the presence of dams, also decreases the suitability of the lower Snake and Columbia river pools for bull trout in the late spring through early fall.

Uncontrolled spill, or even high levels of managed spill, at hydropower projects can produce extremely high levels of total dissolved gas that may impact bull trout and other species. These high levels of gas supersaturation can cause gas bubble disease trauma in fish. Gas bubble disease is caused by gas being absorbed into the bloodstream of fish during respiration. Effects can range from temporary debilitation to mortality, and supersaturation can persist for several miles below dams where spill occurs. The states of Oregon and Washington have established a 111% total dissolved gas level as state water quality standards. However, total dissolved gas levels of up to 120% have been experienced during recent years of managed spill in the Federal Columbia River Power System, with involuntary spill episodes resulting in total dissolved gas levels of as high as 140% at some sites (NMFS 2000). At levels near 140%, gas bubble disease may occur in over 3% of fish exposed. At levels of up to 120%, the incidence of gas bubble disease decreases to a maximum of 0.7% of fish exposed (NMFS 2000).

Manipulated flow releases from storage projects alter the natural flow regime, affect water temperature, have the potential to destabilize downstream streambanks, alter the natural sediment and nutrient loads, and cause repeated and prolonged changes to the downstream wetted perimeter (MBTSG 1998). Power peaking operations, that change the downstream flow of the river on a frequent basis, cause large areas of the river margins to become alternately wet and then dry, and adversely affect aquatic insect survival and production (Hauer and Stanford 1997). Changes in water depth and velocity as a result of rapid flow fluctuations and physical loss or gain of wetted habitat, can cause juvenile trout to be displaced, thus, increasing their vulnerability to predation. Additionally, rapid flow reductions can strand young fish if they are unable to escape over and through draining or dewatered substrate. These effects also indirectly adversely affect bull trout by degrading the habitat of their prey (small fish) and the food upon which they depend (aquatic insects).

Most bull trout pass counting windows at mainstem dams on the Columbia during May and June (Chelan PUD, unpublished data). Diel timing of migration at the dams indicates that fish pass primarily during day light hours (Figure CP28).

At mainstem dams on the Columbia River within the CCP, very low numbers of juvenile bull trout have been documented passing through the project between April and August, with the lowest numbers primarily seen in June (Chelan PUD, unpublished data). This may be due to the limited sampling periods of juveniles in the by-pass facilities (Chelan PUD, unpublished data).

Harvest Effects

Currently, the harvest of bull trout is prohibited on all stocks in the Methow subbasin with the exception of the Lost River. Fishing may have been a leading factor in the decline of bull trout. In streams currently open to fishing of other species, bull trout are vulnerable to take due to misidentification, hooking mortality, poaching, and disturbance. Schmetterling and Long (1999) found that 44 percent of anglers correctly identified bull trout and anglers frequently confused similar species. Incidental hooking mortality varies from less than 5% to 24% for salmonids caught on artificial lures, and between16% and 58% for bait caught salmonids (Taylor and White 1992; Pauley and Thomas 1993; Lee and Bergersen 1996; Shcill 1996; Schill and Scarpella 1997). Eggs and alevins in redds are vulnerable to wading-related mortality which can cause mortality of up to 46% from a single wading event (Roberts and White 1992).

The Lost River, above Drake Creek, is open to bull trout harvest. It is thought that the strength of the healthy population and the remote location will keep harvest within a sustainable level. This fishery should continually be monitored for the effects of this fishery on the population.

3.4.5 Westslope cutthroat trout

Westslope cutthroat trout generally exhibit three main life history forms: fluvial (migrate between smaller spawning streams and larger streams to grow); adfluvial (migrate between spawning streams and a lake, where growth occurs); and non-migratory (generally spend their entire lives in the stream they were born). Much of the life history of WSCT in the Methow River is unknown at this time.

WSCT use many of the smaller streams within the Methow Basin as well as the mainstem Methow. Most reside in the upper reaches of higher order streams within the basin, as well as in some of the Alpine lakes.

Limiting factors for WSCT in the Methow River may be channel stability, habitat diversity, obstructions, temperatures, and riparian conditions. These factors need to be considered in relation to the life history of WSCT (e.g., temperatures probably always limited WSCT distribution within Methow River streams; however, conservation of known areas of abundance would increase the likelihood that they could persist in high quality habitats).

Rationale for Selection

Currently, WSCT are thought to be distributed widely within the CCP. Assumptions regarding the historic and current distribution of WSCT in the Methow subbasin as part of the QHA Analysis are summarized in electronic Appendix C.

Thurow et al. (1997) used predictive models to estimate the range and status of WSCT throughout the Interior Columbia Basin. Their models suggest that WSCT populations within the CCP headwater areas are currently "strong" in most areas; however, they are currently listed under ESA as a species of special concern, and, thus, elevated in importance to that of an important stock refuge.

Management Description of Focal Species/Populations

The FWS (1999) identified various factors that may be affecting the WSCT habitat or range in the CCPO. These factors included channelization or stream alteration within the mainstem of the Methow River, increased sediment loading, erosion, and irrigation withdrawals.

Other factors listed include past wild fire activity, flash flooding, timber harvest, and fragmentation of subpopulations by either man-made habitat alterations or natural barriers. Another potential threat mentioned was the introduction of non-native species within each drainage; introductions of brook trout and non-native *O. mykiss* were identified as being of particular concern (K. MacDonald, pers. comm.).

Key Life History Strategies, Relationship to Habitat

Differing potamodromous forms of WSCT may be found together in sympatry throughout their range (reviewed in Behnke 2002; Wydoski and Whitney 2003). Historically, most populations within the CCP were strictly fluvial (non-migratory) or fluvial-adfluvial ecotypes; although, lacustrine-adfluvial forms existed in the Lake Chelan Basin (Williams 1998). Current lacustrine populations (primarily high mountain lakes) are largely a result of hatchery plantings (Williams 1998).

From Foster et al. 2002 (edited), allopatric cold-water species such as cutthroat can flourish in much warmer environments than in sympatry, but they are vulnerable to displacement by species better suited to warmer temperatures, such as the rainbow trout (Mullan et al. 1992 CPa). Westslope cutthroat trout reside in cold-water refugia where interactive threats from other species are absent because: a) many populations are protected from invasion by barrier falls and, b) most invaders are competitively debilitated by cold temperature. The brook trout is the lone exception. Brook trout, a cold-water species itself, may replace cutthroat in low gradient streams with sandy substrates. The threat from brook trout results from stocking them above an existing cutthroat trout population.

Howell et al. (2003) found that genetically "pure" WSCT were found in suspected allopatric zones, which were usually limited to a few [miles] in the upper reaches of WSCT distribution.

Adult migration

WSCT may migrate long distances, depending on the ecotype (Schmetterling 2001; Wydoski and Whitney 2003). Fish in the St. Joe River in Idaho were found to migrate up to 214 kilometres (132 miles) (Trotter 1987). In the Blackfoot River, Schmetterling (2001) found WSCT moved an average of 39 kilometres (24 miles), and ranged from 12 to 72 kilometres (20 to 45 miles) in 1998.

No information is available for WSCT in the CCP for adult migration. However, given the size of some WSCT in the Methow River in recent years (> 500 mm (20 inches) (Mazama Fly Shop, pers. comm., photos), it seems reasonable to assume that these fish are most likely adfluvial ecotypes that probably spawn in the Upper Methow, Twisp and Chewuch rivers, and other tributaries. If this assumption is true, then fish may be easily migrating the average 39 kilometres (24 miles) that Schmetterling (2001) found in the Blackfoot River.

Non-migratory ecotypes usually do not migrate over 1 kilometre (0.6 mile) within the Blackfoot River (Schmetterling 2001), and usually appear in the CCP in areas upstream of physical or temperature barriers (Williams 1998; Wydoski and Whitney 2003).

Depending on life history, juveniles may move to a lake shortly after emergence if adfluvial-lacustrine (Behnke 2002), or may reside in tributaries for up to four years (Wydoski and Whitney 2003). Fluvial-adfluvial ecotypes may either move quickly, or spend up to three years in a stream before moving to a larger stream (Shepard et al. 1984; Liknes and Graham 1988; Behnke 2002). For juveniles that had reared in streams for extended time periods (years), most moved to either lakes or larger streams during high stream flows (Wydoski and Whitney 2003).

WSCT usually mature at four or five years of age (Downs et al. 1997; FWS 1999), and the maximum life span is typically six to eight years (Behnke 2002). Most fluvial-adfluvial ecotypes appear to mature at an earlier age than non-migratory forms (Downs et al. 1997; Schmetterling 2001; Wydoski and Whitney 2003). The oldest fish ever recorded was 13 years old in Wolf Creek, a tributary of the Methow River (Mullan et al. 2002 CPa); although, Downs et al. (1997) cite personal communication with N. Horner of IDFG stating that they have found fish to this age in Idaho as well.

Juveniles may reside for very short time periods in their natal area before migrating to larger streams or lakes, or they may spend up to four years there prior to migrating. While empirical information is limited, if the hypothesis that non-migratory ecotypes may give rise to migratory ecotypes, there may be occasions when fish may begin their migratory life style after four to five years as has been observed in steelhead (Peven 1990; Mullan et al. 1992 CPa).

Non-migratory adult fish are generally 150-250 mm (6-10 inches; Mullan et al. 1992 CPa; Downs 1997; Behnke 2002). Fluvial-adfluvial forms generally reach maximum sizes of between 410-470 millimetres (16-18.5 inches); Schmetterling 2001; Behnke 2002; Wydoski and Whitney 2003); although, Wydoski and Whitney have observed larger lacustrine forms (introduced) over 510 millimetres (20 inches).

Behnke (2002) notes that the relatively smaller size of adults of the WSCT compared to other cutthroat subspecies may be because of their coevolution with two highly piscivorous species: bull trout and northern pikeminnow. WSCT are rarely piscivorous and usually consist on aquatic and terrestrial insects.

Wydoski and Whitney (2003) reviewed length-at-age information for WSCT. At the end of their second year of life, WSCT ranged between 74 and 145 millimetres (3 and 5.8 inches). By the end of their fifth year, WSCT ranged from 140 to 320 millimetres (5.5 to 12.6 inches). WSCT from the CCP (Methow Basin) were consistently smaller at age (represented by the low end of the range at each age class) than WSCT lengths reported elsewhere in the literature.

Downs et al. (1997) found the average sex ratio for WSCT in headwater streams in Montana to be 1.3 males per female across streams (n=8) that they sampled. In the CCP, Mullan et al. (1992 CPa) found 0.9 males per female in the 412 fish sampled in the Methow River, comporting well with values of 0.2 and 0.9 males per female reported in other studies (Bjornn 1957; Johnson 1963; Lukens 1978; Thurow and Bjornn 1978; May and Huston 1983; and Shepard et al. 1984).

Downs et al. (1997) postulated that the differences in their findings compared to others may have been because of angling pressure (males were more readily removed from the population), and because their samples were from non-targeted populations. This may be true; however, Mullan et al.'s samples were primarily from fish that experience very little, if any, angling pressure. Another potential explanation is that it is possible that there are environmental differences that dictate the variation observed between sex ratios of different populations.

Average fecundity, reported in Downs et al. (1997) for Montana headwater populations, ranged from 227 to 459 eggs per female, and showed a relationship to length-at-maturity (length ranged from 162 to 218 millimetres [6.3 to 8.9 inches]). Brown (1984) reported fecundity of WSCT taken in the early hatchery on Lake Chelan for years 1916 through 1927. Fecundity ranged from 667 to 1,107 for fish that were estimated to be between 221 and 363 mm (8.7 and 14.3 inches) long. The probable reason for the difference observed in average size is most likely because of the differing life histories of fluvial-lacustrine Chelan fish and the fluvial ecotype from Montana.

WSCT spawn generally from March to July, when water temperatures rise in the range of 6°C to 9°C (43°F to 48°F) (Behnke 2002; Wydoski and Whitney 2003). Spawning and rearing streams tend to be cold and nutrient- poor.

Individual fish may spawn in alternative years (Shepard et al. 1984; Liknes and Graham 1988). Schmetterling (2001) found that WSCT entered spawning tributaries when the flow began to increase. He found that, while spawning, the fish did not move more than 200 metres (220 yards) within the spawning tributary.

When adults are migrating upstream to spawning areas, they associate with cover: debris, deep pools, and undercut banks. The availability and number of deep pools and cover is important to offset potential prespawning mortality. Adult cutthroat trout need deep, slow moving pools that do not fill with anchor ice, in order to survive the winter. Intact riparian habitat will increase the likelihood of instream cover, and normative channel geofluvial processes will increase the occurrence of deeper pools.

Important habitat needs for redd building include the availability of clean gravel at the appropriate size, as well as proper water depth and velocity. FWS (1999) state that WSCT redds are usually found in water that is about 0.7 feet deep with mean velocities of 1.0 to 1.3 fps.

Eggs incubate for several weeks and emergence occurs several days after hatching (FWS 1999).

Stream conditions (e.g., frequency of flooding, extreme low temperatures) may affect egg survival as well. Floods can scour eggs from the gravel by increasing sediment deposition that reduces oxygen percolation through the redd. Healy (1991) cites Shaw and Maga (1943) as showing that siltation may be more lethal earlier in the incubation period than in later phases.

In the Methow, flooding has a high frequency of occurrence. Westslope cutthroat trout are spring spawners, therefore fall flooding is not an issue with eggs in the gravel. Road building, grazing and mining activities in the upper watersheds may also increase siltation. All three factors were once more prevalent than they are now in the basin; conditions have improved in most watersheds.

After emergence, fry are usually found in shallow, slow backwater side channels or eddies, and in association with fine woody debris.

Conservation and restoration of natural geofluvial processes and riparian areas of natal streams within the Methow Basin would increase the type of habitat that fry utilize.

Juvenile cutthroat trout overwinter in the interstitial spaces of large stream substrate. Hillman and Miller (2002) state that most juvenile WSCT are consistently found in multiple channels and pools.

Downstream movement of juveniles from natal streams probably occurs within the Methow Basin. Movement of juvenile WSCT within streams is most likely related to changing habitat requirements as the fish grow or winter refuge.

Conservation of high functioning habitat in natal tributaries, and restoration of riparian and geofluvial processes in or near known and potential juvenile rearing areas, will have the highest likelihood of increasing parr survival.

Relationship with Other Species

Competition with rainbow and brook trout is another factor that is limiting WSCT production in the Methow Basin. Rainbow trout cross-breed with WSCT, as well as compete for food and space. Rainbow and brook trout are found in many areas that WSCT are found (Mullan et al. 1992).

WSCT are listed as a Species of Concern under the ESA. Additional information on the status of WSCT and non-migratory redband trout is needed. For management purposes, habitat improvement and conservation of tributary spawning and rearing habitat will increase the likelihood of improving and sustaining populations of westslope cutthroat trout.

Population Delineation and Characterization

The primary historic distribution of WSCT occurred in the upper Columbia and Missouri River basins (FWS 1999). WSCT were originally believed to occur in three river basins within Washington State: Methow, Chelan, and Pend O'Reille. They were, however, only abundant in the Lake Chelan Basin (Williams 1998).

Apart from Lake Chelan and the Pend O'Reille River, where an abundance of relatively large cutthroat commanded the attention of pioneers, cutthroat trout in streams were obscured by their headwater location and small body size. Accordingly, the ethnohistorical record is mostly silent on the presence or absence of cutthroat.

The picture is further blurred by the early scattering of cutthroat from the first trout hatchery in Washington (Stehekin River Hatchery, 1903) by entities (Department of Fisheries & Game and County Fish Commissions) dissolved decades ago along with their planting records. The undocumented translocation of cutthroats by interested non-professionals, starting with pioneers, is another confusing factor that challenges determination of historical distribution. Behnke (1992) believed that the disjunct populations in Washington State probably were transported here through the catastrophic ice-age floods.

Recent information, based on further genetic analyses (Trotter et al. 2001; Behnke 2002; Howell et al. 2003), indicates that the historic range of WSCT in Washington State is now believed to be broader. Historic distribution now includes the headwaters of the Wenatchee and Yakima River basins (Behnke 2002).

Mullan et al. (1992 CPa) indicated pure or essentially pure westslope cutthroat trout have been found above natural rainbow/cutthroat hybridization zones and in alpine lakes that have no history of non-native introductions in the Methow Basin.

Westslope cutthroat appear to have expanded their range within the CCP, from historic distribution, primarily from hatchery plants. Currently WSCT are found throughout the Methow Basin (Williams 1998). WSCT are found within streams and lakes throughout the basins, but spawning (for stream populations) usually occurs in the upper portions of each basin.

Population Status

WSCT appear to be more widely distributed now than they were historically. Since no census data are available, it is not possible to determine the status of these local (and independent) populations that occur in the various watersheds of the CCP.

Numerical abundance has not been documented or estimated for WSCT. Westslope cutthroat were not thought to have been very abundant where they occurred in the headwater locations within the Methow, Entiat, and Wenatchee basins (Williams 1998; FWS 1999; Behnke 2002).

Population Management Regimes and Activities

Hatchery Effects

In the Chelan Basin, the establishment of a hatchery near Stehekin in 1903 was devastating to a lake population of WSCT (Brown 1984). This hatchery was a good example of an “egg mining” hatchery, where many gametes were extracted from the population, but either few fish were planted back into the system, or aquaculture methods were so unsophisticated that few fish survived and, therefore, did not replenish the founding population. The result was the eventual collapse of the population (Brown 1984). It appears that fluvial populations remained in the small feeder tributaries of the lake and the headwaters of the Stehekin River.

The replacement of native westslope cutthroat trout in Eightmile Creek (Methow Basin) was because of stocking brook trout in a small, flat stream, ideally suited to the latter species. Brook trout co-inhabit a number of streams with cutthroat, but the effect in production decreases for both species, not in the elimination of either. Hybridization with steelhead/rainbow trout results from the natural spawning interaction of cutthroat and steelhead/rainbow at their distributional point of contact where water temperature favors neither species (Mullan et al. 1992; Williams 1998). These hybridization zones are short, limiting the negative impact to either species.

While most hatchery stocking of WSCT in the CCP has been from the Twin Lakes strain (originally Lake Chelan), there has been some stocking of other sub-species of cutthroat (FWS 1999).

Through stocking programs that began with Washington State’s first trout hatchery in the Stehekin River valley in 1903 (WSCT-targeted), WSCT have been transplanted in almost all available stream and lake habitat (Williams 1998). WSCT are found within streams and lakes throughout these basins, but spawning (for stream populations) usually occurs in the upper portions of each basin.

Extensive stocking of Twin Lake cutthroat trout in alpine lakes and mountain streams for decades has vastly increased the distribution of cutthroat in the Methow subbasin (Williams 1998). Furthermore, the hatchery brood stock (indigenous Lake Chelan stock) used was felt by Behnke (1992) to be an excellent representation of pure westslope cutthroat trout. Another factor that most likely affected WSCT in the Chelan Basin was the introduction of *O. mykiss* in 1917.

3.4.6 Coho

Coho salmon had been extirpated in the Upper Columbia River (Fish and Hanavan 1948, Mullan 1984), but have been reintroduced by the Yakama Nation. Mullan (1984) estimated that upstream of the Yakima River, the Methow River and Spokane River historically produced the most coho, with lesser runs into the Wenatchee and Entiat.

Currently the Yakama Nation is leading feasibility plans to reintroduce coho salmon to the Methow by, and in cooperation with Washington Department of Fish and Wildlife and the U.S. Fish and Wildlife Service.

Chapman (1986) estimated that the peak run of coho entering the Columbia River in the 1880s was about 560,000 fish. Mullan (1984) pointed out that most coho spawned in the lower Columbia River tributaries. The furthest upstream that coho were known to migrate in the Columbia River was the Spokane River (Fulton 1970).

Mullan (1984) estimated that between 120,000 and 166,500 coho historically ascended the mid-upper Columbia. Of those numbers, he estimated that: 50,000 to 70,000 spawned in the Yakima Basin; 6,000 to 7,000 in the Wenatchee; 9,000 to 13,000 in the Entiat; 23,000 to 31,000 in the Methow, and; 32,000 to 45,000 in the Spokane river basins.

Mid-Columbia basins historically occupied by coho include the Wenatchee, Methow, Entiat, and Okanogan basins. Mullan (1983) estimated historical mid-Columbia River adult coho populations as follows:

- Wenatchee: 6,000-7,000
- Methow: 23,000-31,000
- Entiat: 9,000-13,000
- Okanogan: Although their presence was documented, numbers were not identified.

Long-run coho are unique among a species that usually migrates very short distances to spawn in freshwater. Historical pictures of the native Methow coho indicate the fish were equal in size to the spring Chinook (Mullan et al. 1992b).

Coho (from Fish and Hanavan 1948 and Mullan 1984):

- 1938 – Normal spawning occurred; most juveniles go to sea in 1940.
- 1939 – Fish and Hanavan report only 13 coho counted over Rock Island Dam. No report of their fate (i.e., whether they were used in the program or not).
- 1940 – A few adults received from Rock Island Dam, six of which are spawned at Leavenworth station.
- 1941 – Ten adults spawned of mixed origin (count at Rock Island = 29) at Leavenworth station.
- 1942 – Coho from Lewis River (count at Rock Island = 1) incubated at Leavenworth. Fish released from 1940 brood.
- 1943 - Coho from Lewis River (count at Rock Island = 22) incubated at Leavenworth. Fish released from 1940 brood in Icicle Creek.
- 1944 – River open to migration. Coho from Lewis River (count at Rock Island = 186) incubated at Leavenworth and Entiat. Fish released from 1942 brood in Icicle Creek and Entiat River. Coho from Carson Hatchery reared at Winthrop. 128 fish return to Leavenworth, 123 of which are spawned.
- 1945 – Mullan (1984) reports just under 2,000 fish raised from coho returning to the Icicle (Rock Island count = 166; Fish and Hanavan note that these fish are descendants of Lewis River stock).
- 1946 – No fish raised. Fish released from 1945 brood from Leavenworth.
- 1947 – Fish returning to Leavenworth and Winthrop are raised and released from these stations in 1948 and 1949, respectively

Rationale for Selection

Historically, the Methow River produced more coho than chinook or steelhead (Craig and Suomela 1941). Mullan (1984) estimated that 23,000-31,000 annually returned to the Methow River. Upstream of the Yakima River, the Methow River and Spokane River historically produced the most coho, with lesser runs into the Wenatchee and Entiat (Mullan 1984). Today, coho reintroduction is identified as a priority in the *Wy-Kan-Ush-Mi Wa-Kish-Wit* document (Tribal Restoration Plan) and has been affirmed as a priority by the Northwest Power and Conservation Council.

Coho salmon prefer and occupy different habitat types, selecting slower velocities and greater depths than the other focal species; Habitat complexity and off-channel habitats such as backwater pools, beaver ponds, and side channels are important for juvenile rearing making coho good biological indicators for these areas.

While the historic stock of coho salmon are considered extirpated in the Upper Columbia River (Fish and Hanavan 1948, Mullan 1984), the species has since been reintroduced to the Methow River Basin. In cooperation with the Washington Department of Fish and Wildlife, and the U.S. Fish and Wildlife Service, the Yakama Nation is currently leading coho salmon recovery efforts in the basin.

Representative Habitat

Currently, coho salmon returning to the Methow Basin are spawning in the mainstem Methow River and small tributaries such as Gold Creek. As the recovery program continues, reintroduction of coho to tributaries within the Methow Basin will help to aid in species dispersal. A map of known coho salmon distribution is not currently available.

Key Life History Strategies, Relationship to Habitat

Coho salmon enter the Methow River in mid-to-late September through late November. Adults ascended the tributaries in the fall and spawning occurred between mid-October and late December, although there is historical evidence of an earlier run of coho salmon (Mullan 1984). As cold water temperatures at that time of year preclude spawning in some areas, it is likely that coho salmon spawn in areas where warmer ground water up-wells through the substrate.

Coho entering in September and October hold in larger pools prior to spawning, later entering fish may migrate quickly upstream to suitable spawning locations. The availability and number of deep pools and cover is important to off set potential pre-spawning mortality. Intact riparian habitat will increase the likelihood of instream cover, and normative channel geofluvial processes will increase the occurrence of deeper pools.

Important habitat needs for redd building include the availability of clean gravel at the appropriate size, and proper water depth and velocity. Burner (1951) reported the range of depths for coho spawning to be between 8 and 51 cm. Coho salmon spawn in velocities ranging from 0.30 to 0.75 m/s and may seek out sites of groundwater seepage (Sandercock 1991).

The length of time required for eggs to incubate in the gravel is largely dependent on temperature. Sandercock (1991) reported that the total heat requirement for coho incubation in the gravel (spawning to emergence) was 1036 (±138) degree (°C) days over zero. The percentage of eggs and alevins that survive to emergence depends on stream and streambed conditions. Fall and winter flooding, low flows, freezing of gravel, and heavy silt loads can significantly reduce survival. Fall flooding may negatively affect incubation and emergence success, especially in years of extreme flow. Road building activities in the upper watersheds, as well as grazing and mining activities, may also increase siltation. All three factors were once more prevalent than they are now in the basin and the conditions have improved in most watersheds. In the Wenatchee subbasin, coho fry emerge from the gravel in April or May; it is likely that emergence timing is similar in the Methow River.

Juvenile coho salmon generally distribute themselves downstream shortly after emergence and seek out suitable low gradient tributary and off channel habitats. They congregate in quiet backwaters, side channels, and shady small creeks with overhanging vegetation (Sandercock 1991). Conservation and restoration of riparian areas and off channel habitat in natal streams within the Methow Basin would increase the type of habitat fry use.

Coho salmon prefer slower velocity rearing areas than chinook salmon or steelhead (Lister and Genoe 1970; Allee 1981; Taylor 1991) Recent work completed by the Yakama Nation supports these findings (Murdoch et al. 2004). Juvenile coho tend to overwinter in riverine ponds and other off channel habitats. Overwinter survival is strongly correlated to the quantity of woody debris and habitat complexity (Quinn and Peterson 1996). Conservation of and restoration of high functioning habitat in natal tributaries and restoration of riparian and geofluvial processes in or near known and potential parr rearing areas will have the highest likelihood of increasing parr survival.

Naturally produced coho smolts in the Wenatchee Basin emigrate between March and May (Murdoch et al. 1994). It is likely that naturally produced coho smolts in the Methow River have similar emigration timing. Suspected or potential impediments to migration and sources of injury or mortality should be identified and investigated. If areas are shown to unnaturally impede emigration or injure or kill fish, they should be fixed.

Population Delineation and Characterization

Coho salmon were once extirpated from mid-Columbia tributaries but have since been reintroduced. Reintroduction initially relied on transfers of coho pre-smolts or eggs from Lower Columbia River hatcheries, but is currently transitioning to reliance upon a developing locally adapted broodstock. The developing broodstock is genetically homogeneous with the Wenatchee River broodstock.

Long-run coho are unique among a species that usually migrates very short distances to spawn in freshwater. Historical pictures of the native Methow coho indicate the fish were equal in size to the spring chinook (Mullan et al. 1992b).

Population Status

Washington Water Power blocked the Methow River at Pateros between 1915 and 1929 preventing all fish passage during those years and by the time it was removed, the Methow River run of coho was extinct. By the 1930s, the coho run into the mid- upper Columbia was virtually extirpated (see Rock Island Dam counts above). Tributary dams on the Wenatchee, Entiat, and Methow rivers appeared to be more destructive to coho than either chinook or steelhead (where genetic “storage” presided in resident forms).

Because the native stock of coho salmon no longer occur in the Upper Columbia River system, the Methow basin coho are not addressed under the ESA or by the WDFW (1994) Salmon and Steelhead Stock Inventory. Coho salmon returning to the Methow Basin are primarily hatchery origin, but include an increasing naturally produced component as a result of ongoing reintroduction efforts (YN et. al. 2002). It is likely that continued broodstock development and hatchery supplementation will be necessary to prevent coho salmon from becoming extirpated in the future.

Population Management Regimes and Activities

In the early 1940s and the mid-1970s, the USFWS raised and released coho as part of their mitigation responsibilities for the construction of Grand Coulee Dam (Mullan 1984).

Recently the Yakama Nation (YN) has begun a more concerted effort to reintroduce coho into the Upper Columbia (Scribner et al. 2002); results so far are promising. Current efforts to rebuild coho populations in the Upper Columbia are concentrated in the Wenatchee and Methow Basins.

The ideal result would be to restore coho populations in these basins to their historical levels. Because of varying degrees of habitat degradation in each of these basins, historical numbers are unlikely ever to be achieved, but remain a goal towards which to strive.

The current coho reintroduction plan, still in the feasibility stage through 2004, relies on existing or temporary facilities. Currently, coho smolts are acclimated and released in the Methow River from the WNFH for the sole purpose of broodstock development, although some natural production does occur. This phase of the program is expected to last through 2004 or 2005, after which the reintroduction program will expand to included acclimated releases in natural production areas of the basin in order to reach the tribal natural production goal.

Coho salmon are collected as volunteers into the Winthrop National Fish hatchery and from the run-at-large at Wells Dam west bank and/or east bank fish traps to support a 250,000 smolt program (YN et al. 2002). Methow basin coho broodstock may be supplement with eyed-eggs transferred from Wenatchee Basin incubation facilities or from hatcheries on the lower Columbia River (Cascade FH, Eagle Creek NFH, or Willard NFH) in years where broodstock collection falls short of production goals. Coho reared at Winthrop NFH are volitionally released into the Methow River or transferred to the Wenatchee River for acclimation and release. Under the current feasibility program, coho releases from the Winthrop National Fish Hatchery are design to contribute to the broodstock development process. Details on mating protocols, rearing and acclimation strategies, size at release and monitoring and evaluation can be found in the Yakama Nation's Mid-Columbia Coho HGMP (YN et al.2002).

Hatchery Effects

The first hatchery in the Methow Basin was built in 1889 (Craig and Suomela 1941) and raised primarily coho salmon. Releases of fish from non-indigenous sources began in the 1940s (Peven 1992CPb).

Between 1904 and 1914, an average of 360 females was used for broodstock from the Methow hatchery annually (Mullan 1984). With the building of a non-passable dam at the Methow River mouth in 1915, this hatchery was moved more towards the confluence with the Columbia.

Between 1915 and 1920, an average of only 194 females was taken, suggesting a 50% decline in the run between this and the previous period. After 1920, no coho were taken from this hatchery and it closed in 1931 (in Mullan 1984).

No further releases of coho into the Methow River occurred until the GCFMP in 1945. The broodstock originated from the Methow River (which were admixtures of various stocks originally captured at Rock Island Dam; Mullan 1984) in only 4 of the 17 years of coho releases from the Winthrop NFH between 1945 and 1969. Most of the coho released at Winthrop originated from Lower Columbia River stocks from the Eagle, Lewis, and Little White Salmon hatcheries (Mullan 1984).

Chelan PUD also had a coho hatchery program until the early 1990s. While some natural production may have occurred from these releases, the programs overall were not designed to re-establish a naturally spawning populations and relied upon lower Columbia River stocks.

Current coho reintroduction efforts focus on local broodstock development to select for traits which are successful in mid-Columbia tributaries with the long-term goal of restoring naturally reproducing populations. The mid-Columbia coho reintroduction feasibility study has a substantial monitoring and evaluation program to determine if the reintroduction of coho salmon into the upper Columbia basin may affect the production of chinook and steelhead. The results of extensive predation and competition studies indicate that a negative effect is unlikely to occur. Similarly, other researchers have found that the introduction of coho did not negatively affect the abundance or growth of naturally produced chinook or steelhead (Spaulding et. al. 1989; Mullan et al. 1992)

Hydroelectric Effects

Habitat alteration, especially tributary dams in the Methow River mainstem, reduced the viability and capability of coho to rebuild themselves locally.

Prior to the 1940's, runs of Methow River coho salmon were essentially destroyed as a result of over-harvest, early hatchery practices, habitat degradation and impassable downstream dams. Much of the failure of the GCFMP to re-establish self-perpetuating populations may have been related to reliance upon stocks lacking genetic suitability (Mullan et al. 1992b).

Recent (after GCFMP) programs to restore coho in the mid-upper Columbia began in the 1960's with releases from WDFW hatcheries for Rocky Reach Dam mitigation. Although this program did produce some initial promising results, (Figure CP15), naturally producing runs were not established, primarily because the program was not designed to re-establish naturally producing runs. The coho were released from the Turtle Rock fish hatchery, located in the middle of the Columbia River above Rocky Reach Dam. The release location likely contributed to the inability to produce a naturally spawning coho run. This reach of the Columbia River does not provide suitable coho spawning and rearing habitat. In the early 1990s, this program was abandoned.

According to the Chelan 2002 HCP, Rocky Reach Hatchery compensation for Methow River coho will be assessed in 2006 following the development of a continuing coho hatchery program and/or the establishment of a Threshold Population of naturally reproducing coho in the Methow Basin (by an entity other than the District and occurring outside this Agreement). The Hatchery Committee shall determine whether a hatchery program and/or, naturally reproducing population of coho is present in the Methow Basin. Should the Hatchery Committee determine that such a program or population exists, then (1) the Hatchery Committee shall determine the most appropriate means to satisfy the 7% hatchery compensation requirement for Methow Basin coho, and (2) the District shall have the next juvenile migration to adjust juvenile protection Measures to accommodate Methow Basin coho. Thereafter, Coordinating Committee shall determine the number of valid studies (not to exceed three years) necessary to make a juvenile phase determination.

Programs to meet NNI for Methow Basin coho may include but are not limited to: (1)provide operation and maintenance funding in the amount equivalent to 7% project passage loss, or (2) provide funding for acclimation or adult collection facilities both in the amount equivalent to 7% juvenile passage loss at the Project. The programs selected to achieve NNI for Methow Basin coho will utilize an interim value of project survival, based upon a Juvenile Project Survival estimate of 93%, until juvenile project survival studies can be conducted on Methow Basin coho.

Harvest Effects

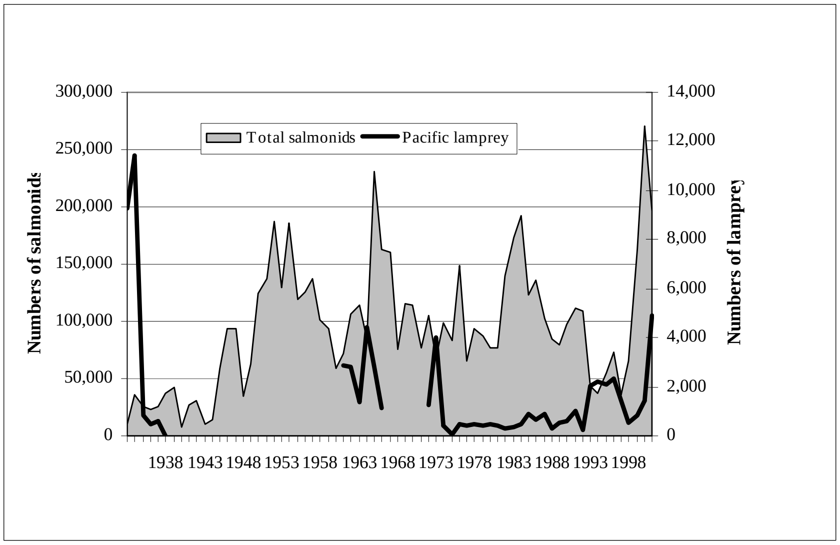
Coho were relatively abundant in upper Columbia River tributaries streams prior to extensive resource exploitation in the 1860's. By the 1880's, the expanding salmon canning industry and rapid growth of the commercial fisheries in the lower Columbia River had heavily depleted the mid and upper Columbia Rive spring and summer chinook runs (McDonald 1895), and eventually the steelhead, sockeye, and coho (Mullan 1984, 1986, 1987; Mullan et al. 1992 CPa).

The runs of coho that ascended the Columbia River were initially reduced from over-harvest in the mainstem and habitat degradation associated with watershed development.

3.5 Other fish species important to management in the Methow subbasin

3.5.1 Pacific Lamprey

Historical distribution of Pacific lamprey in the Columbia and Snake Rivers was coincident wherever salmon occurred (Simpson and Wallace 1978). A record of migration trends illustrates a significant decline in lamprey abundance over the last 50 years (**Figure 19**).



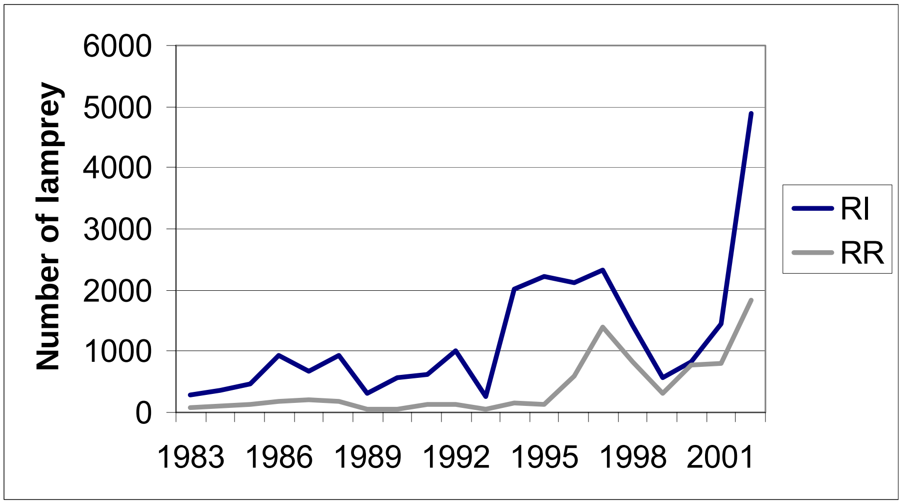
Source: Pevan 2004

Figure 19 Comparison of salmonids adn Pacific lamprey ascending Rock Island Dam (1933–2002)

It is likely that Pacific lamprey occurred historically throughout the Methow subbasin in association with anadromous salmon (Clemens 1939). In the upper Columbia, counts over Rock Island and Rocky Reach dams show a precipitous drop from the 1960s through the 1980s (Close et al. 1995), and appear to be rebuilding once again.

There is little information on the abundance of Pacific lamprey in the upper Columbia region. Abundance estimates are limited to counts of adults and juveniles at dams or juvenile salmonid traps. There are no estimates of redd counts nor juvenile and adult counts in tributaries.

Large declines of adults occurred at most mainstem dams during the late 1960s and early 1970s. During the period between about 1974 and 1993, numbers of adult lamprey counted at Rock Island Dam was quite low (**Figure 20**). Counts of adults have increased since that time; however, this increase corresponds closely with the time that the projects began day and night counts, perhaps having some effect on the comparison. Recent increases in the last few years, however, are far greater than those in the last 10, suggesting that a true increase in abundance is occurring.



Source: Pevan 2004

Figure 20 Numbers of lamprey ascending Rock Island and Rocky Reach Dams since 1983

Counts of adult lamprey at dams cannot be considered total counts because there was no standardized sampling across years and counting was restricted to certain hours (BioAnalysts 2000). For example, fish counters in the past counted for a 16-hour-day shift for the main part of the salmon runs (Close et al. 1995). Because the highest movement of lamprey occurs at night (Close et al. 1995), these day counts should be considered conservative estimates. Currently, fish counting occurs throughout the 24-hour period at most dams. At Rocky Reach and Rock Island dams, videotape or digital video record fish passage over 24 hours per day. This counting method began at Rock Island in 1992 and at Rocky Reach in 1996.

Additional problems with adult counts exist because some lamprey pass dams undetected. For example, adult lamprey can move near the bottom of the fish counting chamber making it difficult to detect them (Jackson et al. 1996). They can also bypass counting station windows by traveling behind the picketed leads at the crowder (Starke and Dalen 1995). Because of these shortcomings, adult counts at dams should only be viewed as crude indices of abundance.

Counts of juvenile lamprey at dams also suffer from sampling inconsistencies. Collection of juvenile lamprey at mainstem dams is incidental to sampling juvenile salmonids. Thus, numbers of migrants outside the juvenile salmonid migration period are unknown, since most of the literature suggests that migration occurs between fall and spring (Pletcher 1963; Beamish 1980; Richards and Beamish 1981). In addition, unknown guidance efficiencies of juvenile lamprey, and unknown spill passage to turbine passage ratios, reduce precise estimates of abundance (BioAnalysts 2000). Juveniles also tend to hide in various locations in the bypass systems (Jackson et al. 1997). These problems, combined with highly variable sampling rates during periods of juvenile salmonid passage, confound estimates of juvenile lamprey abundance (BioAnalysts 2000). Juvenile counts at dams should also be viewed as crude indices of abundance.

Comparing counts among different projects is problematic because of sampling inconsistencies, the behavior of lamprey in counting stations, and the ability of lamprey to bypass counting stations undetected (BioAnalysts 2000).

In summary, while it is difficult to determine the historical abundance of lamprey in the Columbia Basin and the CCP, circumstantial evidence suggests that they have declined. Counts of juvenile and adult lamprey fluctuate widely. It is unknown whether these fluctuations represent inconsistent counting procedures, actual population fluctuations, or both. Although these factors may make actual comparisons difficult, it appears that lamprey in the upper Columbia are increasing.

More information needs to be gathered for Pacific lamprey before any determinations of extinction risks can be made.

The American Fisheries Society's Western Division reviewed the FWS's petition to list four species of lamprey in 2001, and found strong evidence to support listing of Pacific lamprey on the Columbia, Umqua and Snake Rivers (WDAFS, 2001).

3.5.2 White Sturgeon

Historically, white sturgeon moved throughout the mainstem Columbia River from the estuary to the headwaters; although passage was probably limited at times by large rapids and falls (Brannon and Setter 1992). Beginning in the 1930s, with construction of Rock Island, Grand Coulee, and Bonneville dams, migration was disrupted because sturgeon do not pass upstream through fishways that were built for salmon, although they apparently can pass downstream (S. Hays, pers. comm.).

Current populations in the Columbia River Basin can be divided into three groups: fish below the lowest dam, with access to the ocean (the lower Columbia River); fish isolated (functionally but not genetically) between dams; and fish in several large tributaries. In the CCP, construction of Wells, Rocky Reach, Rock Island, and Wanapum Dam have disrupted upstream movement of sturgeon.

Peven (2003) concluded that white sturgeon distribution has been affected by construction of mainstem Columbia River dams. What was believed to be a relatively continuous population, traveling the length of the mainstem Columbia River below migrational barriers, is now a number of potentially disjunct populations between hydroelectric projects. There does, however, appear to be immigration and emigration from downstream recruitment.

3.5.3 Rainbow Trout

Rainbow trout are the freshwater variety of steelhead trout (*O. mykiss*). They are represented in the river and tributaries by both fluvial and adfluvial varieties.

They are present in most of the headwater tributaries, where year-round flows are hospitable, and in the mainstem Methow. The headwater fluvial varieties appear to have one life history pattern: to spawn and rear in upper tributaries. The population size and distribution of rainbow trout in these streams is not known (NMFS, 1998).

3.5.4 Redband trout

Redband trout (*Oncorhynchus mykiss gairdneri*) are indistinguishable from steelhead in the CCP; they are an exclusive ecotype of inland waters (Behnke 2002). For example, steelhead were not extirpated in the Methow River, as were coho, when a dam was constructed near its confluence with the Columbia, probably because headwater resident forms sustained the run (Mullan et al. 1992 CPa).

Anadromy is not obligatory in *O. mykiss* (Rounsefell 1958; Mullan et al. 1992). Progeny of anadromous steelhead can spend their entire life in freshwater, while progeny of rainbow trout can migrate seaward. Anadromy, although genetically linked (Thorpe 1987), runs under environmental instruction (Shapovalov and Taft 1954; Thorpe 1987; Mullan et al. 1992). It is difficult to summarize one life history strategy (anadromy) without due recognition of the other (non-migratory).

The two strategies appear to co-mingle on some continuum with certain residency at one end, and certain anadromy on the other (see further discussion in Life History section). Upstream distribution is limited by low heat budgets (about 1,600 temperature units) (Mullan et al 1992). The response of steelhead/rainbow complex in these cold temperatures is they are "thermally fated" to a nonanadromous ecotype, presumably because growth is too slow within the time window for smoltification. However, these headwater rainbow trout contribute to anadromy via emigration and displacement to lower reaches, where warmer water improves growth rate and subsequent opportunity for smoltification.

Historic distribution

Redband trout originally occurred in the Fraser and Columbia River drainages east of the Cascade Mountains to barrier falls on the Pend O'Reille, Spokane, Snake, and Kootenai rivers (Behnke 1992). It is reasonable to assume that the historical distribution of redband trout was potentially wider than that of steelhead in the CCP because populations would have, and still do, occur in areas upstream of anadromous barriers. This would include all areas (downstream of temperature barriers; Mullan et al. 1992) in the Wenatchee, Entiat, Methow, and upper reaches of the Okanogan River basins.

Current distribution

Currently, because of the admixture with hatchery fish, *O. mykiss* is widespread throughout the CCP. *Oncorhynchus mykiss* is found virtually everywhere in each major subbasin in the CCP, below thermal barriers in the headwater areas. To reiterate, in most areas of occurrence, it is not possible to distinguish between non-migratory and anadromous forms.

In conclusion, because it is not possible to distinguish anadromous from nonanadromous forms of redband trout, it is difficult to determine changes in distribution over historic times (regardless of hatchery plants, which have played an influence also).

3.5.5 Eastern Brook Trout

Eastern Brook trout are an introduced species that is present throughout the basin. In drainages where brook trout and bull trout are both present, they hybridize. Brook trout appear to be more tolerant to disturbed habitat conditions than bull trout. The introduction of brook trout, and resulting hybridization of the two species, has increased inter-species competition with bull trout in the subbasin (NMFS, 1998).

3.6 Focal Wildlife Species

3.6.1 Brewer's Sparrow

General Habitat Requirements

Brewer's sparrow is a sagebrush obligate species that prefers abundant sagebrush cover (Altman and Holmes 2000). Vander Haegen et al. (2000) determined that Brewer's sparrows were more abundant in areas of loamy soil than in areas of sandy or shallow soil, and on rangelands in good or fair condition than those in poor condition. Knopf et al. (1990) reported that Brewer's sparrows are strongly associated throughout their range with high sagebrush vigor.

Brewer's sparrow is positively correlated with shrub cover, above-average vegetation height, bare ground, and horizontal habitat heterogeneity (patchiness). Brewer's sparrows prefer areas dominated by shrubs rather than grass. They prefer sites with high shrub cover and large patch size (Knick and Rotenberry 1995). In southwestern Idaho, the probability of habitat occupancy by Brewer's sparrows increased with increasing percent shrub cover and shrub patch size; shrub cover was the most important determinant of occupancy (Knick and Rotenberry 1995).

Brewer's sparrow abundance in Washington increased significantly on sites where sagebrush cover approached the historic 10% level (Dobler et al. 1996).

In contrast, Brewer's sparrows are negatively correlated with grass cover, spiny hopsage, and budsage (Larson and Bock 1984; Rotenberry and Wiens 1980; Wiens 1985; Wiens and Rotenberry 1981). In eastern Washington, abundance of Brewer's sparrows was negatively associated with increasing annual grass cover; higher densities occurred in areas where annual grass cover (i.e., cheatgrass) was less than 20% (Dobler 1994). Removal of sagebrush cover to less than 10% has a negative impact on populations (Altman and Holmes 2000).

Recommended habitat objectives include the following: patches of sagebrush cover 10-30%; mean sagebrush height greater than 24 inches; high foliage density of sagebrush; average cover of native herbaceous plants greater than 10%, bare ground greater than 20% (Altman and Holmes 2000).

Limiting Factors

Habitat loss and fragmentation, livestock grazing, introduced vegetation, fire, and predators are the primary factors affecting Brewer's sparrows. Direct habitat loss because of conversion of shrublands to agriculture, coupled with sagebrush removal/reduction programs and residential development, have significantly reduced available habitat and contributed towards habitat fragmentation of remaining shrublands. Within the entire Interior Columbia Basin, over 48% of watersheds show moderately or strongly declining trends in source habitats for this species (Wisdom et al. in press) (from Altman and Holmes 2000).

Livestock grazing can trigger a cascade of ecological changes, the most dramatic of which is the invasion of non-native grasses escalating the fire cycle and converting sagebrush shrublands to annual grasslands. Historical heavy livestock grazing altered much of the sagebrush range, changing plant composition and densities. West (1988, 1996) estimates less than 1% of sagebrush steppe habitats remain untouched by livestock; 20% is lightly grazed, 30% moderately grazed with native understory remaining, and 30% heavily grazed with understory replaced by invasive annuals. The effects of grazing in sagebrush habitats are complex, depending on intensity, season, duration, and extent of alteration to native vegetation. Rangeland in poor condition is less likely to support Brewer's sparrows than rangeland in good and fair condition.

Introduced vegetation such as cheatgrass readily invades disturbed sites, and has come to dominate the grass-forb community of more than half the sagebrush region in the West, replacing native bunchgrasses (Rich 1996). Cheatgrass has altered the natural fire regime in the western range, increasing the frequency, intensity, and size of range fires.

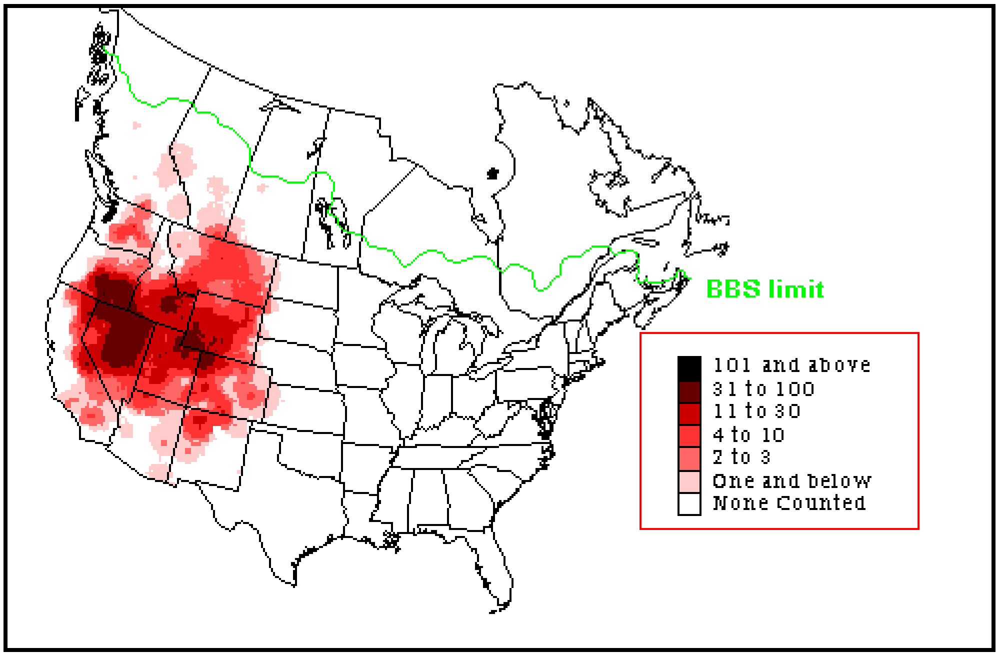
Fire kills sagebrush; as the fire cycle escalates, where non-native grasses dominate, the landscape can be converted to grasslands dominated by introduced vegetation, removing preferred habitat (Paige and Ritter 1998). Crested wheatgrass and other non-native annuals have also fundamentally altered the grass-forb community in many areas of sagebrush shrubsteppe, altering shrubland habitats.

Predators (of eggs and nestlings) include gopher snake (*Pituophis melanoleucus*), Townsend's ground squirrel (*Spermohpilus townsendii*); other suspected predators include loggerhead shrike (*Lanius ludovicianus*), common raven (*Corvus corax*), black-billed magpie (*Pica pica*), long-tailed weasel (*Mustela frenata*), least chipmunk (*Eutamias minimus*), western rattlesnake (*Crotalus viridis*), and other snake species. Nest predation is the most significant cause of nest failure.

The American kestrel (*Falco sparverius*), prairie falcon (*Falco mexicanus*), coachwhip (*Masticophis flagellum*) have been observed preying on adult sparrows (Rotenberry et al. 1999). Wiens and Rotenberry (1981) observed significant negative correlation between loggerhead shrike and Brewer's sparrow density.

Current Distribution

Undoubtedly, the Brewer's sparrow was widely distributed throughout the lowlands of southeast Washington when it consisted of vast expanses of shrubsteppe habitat. Large-scale conversion of shrubsteppe habitat to agriculture has resulted in populations becoming localized in the last vestiges of available habitat (Smith et al. 1997). Washington is near the northwestern limit of breeding range for Brewer's sparrows (**Figure 21**). Birds occur primarily in Okanogan, Douglas, Grant, Lincoln, Kittitas, and Adams Counties (Smith et al. 1997).

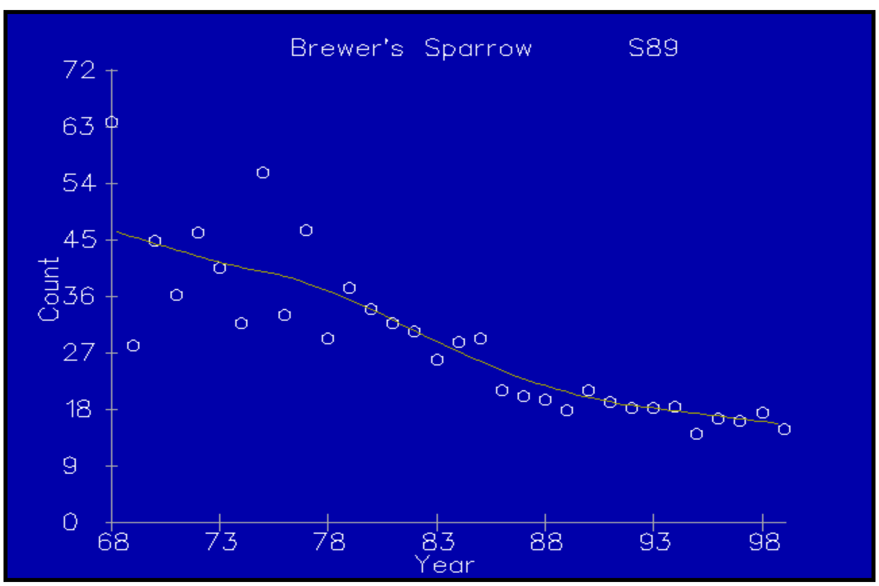


Sauer et al. 2003

Figure 21 Brewer's sparrow breeding range and abundance

Population Trend Status

Brewer's sparrow is often the most abundant bird species in appropriate sagebrush habitats **Figure 22**); however, widespread long-term declines and threats to shrubsteppe breeding habitats have placed it on the Partners in Flight Watch List of conservation priority species (Muehter 1998). Saab and Rich (1997) categorize it as a species of high management concern in the Columbia River Basin.



Source: BBS data; Sauer et al. 2003

Figure 22 Brewer's sparrow trend results for the Columbia Plateau

Historically, the Brewer's sparrow may have been the most abundant bird in the Intermountain West (Paige and Ritter 1998), but BBS trend estimates indicate a range-wide population decline during the last twenty-five years (Peterjohn et al. 1995).

Brewer's sparrows are not currently listed as Threatened or Endangered on any state or federal list. Oregon-Washington Partners in Flight consider the Brewer's sparrow a focal species for conservation strategies for the Columbia Plateau (Altman and Holmes 2000).

Breeding Bird Survey data for the period of 1966 to 1996 show significant and strong survey-wide declines averaging -3.7% per year (n = 397 survey routes). Significant declines in Brewer's sparrow are evident in California, Colorado, Montana, Nevada, Oregon, and Wyoming, with the steepest significant decline evident in Idaho (-6.0% average per year; n = 39). These negative trends appear to be consistent throughout the 30-year survey period. Only Utah shows an apparently stable population. Sample sizes for Washington are too small for an accurate estimate.

Note that, although positively correlated with presence of sage thrashers (*Oreoscoptes montanus*), probably because of similarities in habitat relations (Wiens and Rotenberry 1981), thrashers are not exhibiting the same steep and widespread declines evident in BBS data (see Sauer et al. 1997).

3.6.2 Grasshopper Sparrow

General Habitat Requirements

Grasshopper sparrows prefer grasslands of intermediate height and are often associated with clumped vegetation interspersed with patches of bare ground (Bent 1968; Blankespoor 1980; Vickery 1996). Other habitat requirements include moderately deep litter and sparse coverage of woody vegetation (Smith 1963; Bent 1968; Wiens 1969, 1970; Kahl et al. 1985; Arnold and Higgins 1986). In east central Oregon, grasshopper sparrows occupied relatively undisturbed native bunchgrass communities dominated by *Agropyron spicatum* and/or *Festuca idahoensis*, particularly on north-facing slopes on the Boardman Bombing Range in the Columbia Basin (Holmes and Geupel 1998). Vander Haegen et al. (2000) found no significant relationship with vegetation type (i.e., shrubs, perennial grasses, or annual grasses), but did find a relationship with the percent cover perennial grass.

In portions of Colorado, Kansas, Montana, Nebraska, Oklahoma, South Dakota, Texas, Wisconsin, and Wyoming, abundance of grasshopper sparrows was positively correlated with percent grass cover, percent litter cover, total number of vertical vegetation hits, effective vegetation height, and litter depth; abundance was negatively correlated with percent bare ground, amount of variation in litter depth, amount of variation in forb or shrub height, and the amount of variation in forb and shrub heights (Rotenberry and Wiens 1980).

Grasshopper sparrows occasionally inhabit cropland, such as corn and oats, but at a fraction of the densities found in grassland habitats (Smith 1963; Smith 1968; Ducey and Miller 1980; Basore et al. 1986; Faanes and Lingle 1995; Best et al. 1997).

Limiting Factors

The principal post-settlement conservation issues affecting grasshopper sparrow populations include: habitat loss and fragmentation resulting from conversion to agriculture, habitat degradation and alteration from livestock grazing, invasion of exotic vegetation, and alteration of historic fire regimes.

Fragmentation resulting from agricultural development, or large fires fueled by cheatgrass, can have several negative effects on land birds. These include: insufficient patch size for area-dependent species, and increases in edges and adjacent hostile landscapes that can result in reduced productivity through increased nest predation, nest parasitism, and reduced pairing success of males. Additionally, habitat fragmentation has likely altered the dynamics of dispersal and immigration necessary for maintenance of some populations at a regional scale. In a recent analysis of neotropical migratory birds within the Interior Columbia Basin, most species identified as being of "high management concern" were shrubsteppe species (Saab and Rich 1997); this list included the grasshopper sparrow.

Making this loss of habitat even more severe is that the grasshopper sparrow like other grassland species shows a sensitivity to the grassland patch size (Herkert 1994; Samson 1980; Vickery 1994; Bock et al. 1999). Herkert (1991) found that grasshopper sparrows in Illinois were not present in grassland patches smaller than 74 acres despite the fact that their published average territory size is only about 0.75 acres. Minimum requirement size in the Northwest is unknown.

Grazing can trigger a cascade of ecological changes, the most dramatic of which is the invasion of non-native grasses escalating the fire cycle and converting sagebrush shrublands to annual grasslands. Historical heavy livestock grazing altered much of the sagebrush range, changing plant composition and densities. West (1988, 1996) estimates less than 1% of sagebrush steppe habitats remain untouched by livestock; 20% is lightly grazed, 30% moderately grazed with native understory remaining, and 30% heavily grazed with understory replaced by invasive annuals. The effects of grazing in sagebrush habitats are complex, depending on intensity, season, duration and extent of alteration to native vegetation. Extensive and intensive grazing in North America has had negative impacts on this species (Bock and Webb 1984).

The grasshopper sparrow has been found to respond positively to light or moderate grazing in tallgrass prairie (Risser et al. 1981); however, it responds negatively to grazing in shortgrass, semi-desert, and mixed grass areas (Bock et al. 1984).

The degree of degradation of terrestrial ecosystems is often diagnosed by the presence and extent of alien plant species (Andreas and Lichvar 1995); frequently, their presence is related to soil disturbance and overgrazing. Increasingly, however, aggressive aliens are becoming established wherever their seed can reach, even in ostensibly undisturbed bunchgrass vegetation.

Cheatgrass has altered the natural fire regime in the western range, increasing the frequency, intensity, and size of range fires. Fire kills sagebrush, and where non-native grasses dominate, the landscape can be converted to annual grassland as the fire cycle escalates, removing preferred habitat (Paige and Ritter 1998).

Studies on the effects of burns on grassland birds in North American grasslands have shown similar results as grazing studies, namely, that bird response is highly variable. Confounding factors include timing of burn, intensity of burn, previous land history, type of pre-burn vegetation, presence of fire-tolerant exotic vegetation (that may take advantage of the post-burn circumstances and spread even more quickly), and grassland bird species present in the area.

It should be emphasized that much of the variation in response to grassland fires lies at the level of species, but that even at this level, results are often difficult to generalize. For instance, mourning doves have been found to experience positive (Bock and Bock 1992; Johnson 1997) and negative (Zimmerman 1997) effects by fire in different studies. Similarly, grasshopper sparrows have been found to experience positive (Johnson 1997), negative (Bock and Bock 1992; Zimmerman 1997; Vickery et al. 1999), and no significant (Rohrbaugh 1999) effects of fire. Species associated with short and/or open grassy areas will most likely experience short-term benefits from fires. Species that prefer taller and denser grasslands most likely will demonstrate a negative response to fire (CPIF 2000).

Mowing and haying affects grassland birds directly and indirectly. It may reduce height and cover of herbaceous vegetation, destroy active nests, kill nestlings and fledglings, cause nest abandonment, and increase nest exposure and predation levels (Bollinger et al. 1990). Studies of the grasshopper sparrow have indicated higher densities and nest success in areas not mowed until after July 15 (Shugaart and James 1973; Warner 1992); grasshopper sparrows are vulnerable to early mowing of fields, while light grazing, infrequent and post-season burning or mowing can be beneficial (Vickery 1996).

Grasshopper sparrows may be multiply-parasitized (Elliott 1976, 1978; Davis and Sealy 2000). In Kansas, cowbird parasitism cost grasshopper sparrows about two young/parasitized nest; there was a low likelihood of nest abandonment occurring because of cowbird parasitism (Elliott 1976, 1978).

Current Distribution

Grasshopper sparrows are found from North to South America, Ecuador, and in the West Indies (Vickery 1996; AOU 1957). They are common breeders throughout much of the continental United States, ranging from southern Canada, south to Florida, Texas, and California. Additional populations are locally distributed from Mexico to Colombia, and in the West Indies (Delany et al. 1985; Delany 1996; Vickery 1996).

The subspecies breeding in eastern Washington is *Ammodramus savannarum perpallidus* which breeds from northwest California, where it is uncommon, into eastern Washington, northeast and southwest Oregon, where it is rare and local, into southeast British Columbia, where it is considered Endangered, east into Nevada, Utah, Colorado, Oklahoma, Texas, and possibly Illinois and Indiana (Vickery 1996).

Grasshopper sparrow structural conditions and association relationships (IBIS 2003) are shown in **Table 26**.

Table 26 Grasshopper sparrow structural conditions and association relationships (IBIS 2003)

Common Name	Focal Habitat	Structural Condition (SC)	SC Activity	SC Assoc.
Grasshopper Sparrow	Shrubsteppe	Grass/Forb-Closed	B	C
		Grass/Forb-Open	B	C
		Low Shrub-Open Shrub Overstory-Mature	B	A
		Low Shrub-Open Shrub Overstory-Old	B	A

Common Name	Focal Habitat	Structural Condition (SC)	SC Activity	SC Assoc.
		Low Shrub-Open Shrub Overstory-Seedling/Young	B	A
		Medium Shrub-Open Shrub Overstory-Mature	B	A
		Medium Shrub-Open Shrub Overstory-Old	B	A
		Medium Shrub-Open Shrub Overstory-Seedling/Young	B	A

3.6.3 Sharp-tailed Grouse

General Habitat Requirements

The Columbian sharp-tailed grouse (CSTG) is one of six subspecies of sharp-tailed grouse and the only one found in Washington. Native habitats important for CSTG include grass-dominated nesting habitat and deciduous shrub-dominated wintering habitat, both of which are critical for sharp-tailed grouse (Giesen and Connelly 1993; Connelly et al. 1998).

Residual grasses and forbs are necessary for concealment and protection of nests and broods during spring and summer (Hart et al. 1952; Parker 1970; Oedekoven 1985; Marks and Marks 1988; Meints 1991; Giesen and Connelly 1993). Preferred nest sites are on the ground in relatively dense cover provided by clumps of shrubs, grasses, and/or forbs (Hillman and Jackson 1973). Fields enrolled in agricultural set-aside programs are often preferred. Giesen (1987) reported density of shrubs less than three feet tall was five times higher at nest sites than at random sites, or at sites 33 feet from the nest.

Meints (1991) found that mean grass height at successful nests averaged less than one foot, while seven inches was the average at unsuccessful nests. Hoffman (2001) recommended that the minimum height for good quality nesting and brood-rearing habitat is eight inches, with one foot being preferred. Bunchgrasses, especially those with a high percentage of leaves-to-stems, such as bluebunch wheatgrass, is preferred over sod-forming grasses such as smooth brome by nesting sharp-tailed grouse

Columbian sharp-tailed grouse are able to tolerate considerable variation in the proportion of grasses and shrubs that comprise suitable nesting habitat; the most important factor is that a certain height and density of vegetation is required. Canopy coverage and visual obstruction are greater at nest sites than at independent sites (Kobriger 1980; Marks and Marks 1987; Meints 1991).

After hatching, hens with broods move to areas where succulent vegetation and insects can be found (Sisson 1970; Gregg 1987; Marks and Marks 1987; Klott and Lindzey 1990). In late summer, riparian areas and mountain shrub communities are preferred (Giesen 1987).

Food items in the spring and summer include wild sunflower (*Helianthus* spp.), chokecherry, sagebrush, serviceberry, salsify (*Tragopogon* spp.), dandelion (*Taraxacum* spp.), bluegrass, and brome (Hart et al. 1952; Jones 1966; Parker 1970). Although juveniles and adults consume insects, chicks eat the greatest quantity during the first few weeks of life (Parker 1970; Johnsgard

1973). In winter, CSTG commonly forage on persistent fruits and buds of chokecherry, serviceberry, hawthorn, snowberry, aspen, birch, willow, and wild rose (Giesen and Connelly 1993; Schneider 1994).

Columbian sharp-tailed grouse numbers have drastically declined in Washington over the past 100 years, and they are now a federally and state listed species. The breeding population of sharp-tailed grouse in Washington is currently estimated at 380. Shrubsteppe and riparian habitat are critical habitat for sharp-tailed grouse, and both have been heavily manipulated in the basin (OWSAC 2000). The FWS recently issued a 90-day Finding on a petition to list sharp-tailed grouse as Threatened under the ESA (FWS, 1999).

According to early explorers sharp-tails used to be plentiful in Eastern Washington. A total of 112 sharp-tailed grouse leks (courtship areas) were documented between 1954 and 1994. Lek counts are used to estimate population size and stability. The number of males per lek and active leks also indicate stability of the population. Males per lek declined from 13 in 1954 to five in 1994. In Douglas County, from 1954 to 1994, 46% of active leks disappeared, 65% disappeared in Okanogan County, and 61% disappeared in Lincoln County.

Limiting Factors

The primary factors affecting the continued existence of sharp-tailed grouse in Washington relate to habitat loss and alteration and the precarious nature of small, geographically isolated subpopulations. Three of the major factors that contributed to the decline of sharp-tailed grouse and their habitat in Washington are still threats today: conversion to agriculture, conversion to residential development, and overgrazing. The removal of shrubs reduces the quantity and quality of winter habitat, and the degradation of shrub and meadow steppe habitat, as a result of livestock management, reduces the quality of breeding habitat. The remaining subpopulations are small and isolated from one another, increasing the risk of extirpation.

Population isolation is potentially a major factor influencing the continued existence of sharp-tailed grouse in Washington. As grouse populations naturally fluctuate, because of environmental conditions, the lower the population level, the greater the risk of extirpation. The isolation of populations may have important ramifications for their genetic quality and recruitment (Lacy 1987). It may require human transport of individuals to counteract loss of fitness because of genetic drift.

It is not clear if the Washington populations are declining because of their isolation, or because of a combination of other factors. Initial evidence (M. Schroeder, pers. comm.) indicates that most movements of radio-marked birds are insufficient to allow interchange of individuals among populations in north-central Washington. Although current estimates of the total population range up to 1000 individuals, it is divided among eight small isolated subpopulations. Four of these populations are estimated to contain fewer than 25 birds. These populations are under immediate threat of extirpation (Reed et al. 1986).

Near-term extirpation risks because of population size are present for two of three other populations remaining outside the Colville Indian Reservation (Gilpin 1987); less than 100 individuals are estimated at each site (M. Schroeder, pers. comm.). These populations are likely much less tolerant of environmental changes, such as habitat degradation and weather extremes, than are populations in Lincoln County and the Colville Indian Reservation. Predation is more of a concern for these very small populations than it would be for larger populations in good habitat.

A wide variety of genetic problems can occur with small populations, and these genetic problems can interact with demographic and habitat problems and lead to extinction (Gilpin and Soule 1986). Overall threats to sharp-tailed grouse are greater when individuals are spread through small subpopulations rather than being one larger population.

Sharp-tails in Douglas and Okanogan counties, and to a lesser degree in Lincoln County, are now restricted to high-elevation areas, and specifically, in those areas that have both shrubs and grasses (Schroeder 1996). High winter mortality, resulting from declining quantity and quality of winter habitat, is likely the most significant factor causing the decline in the sharp-tail population in Washington (Schroeder 1996). Protecting and enhancing high quality habitat where sharp-tails continue to concentrate, and restoring key low-elevation winter sites is vital to conservation of sharp-tailed grouse in Washington.

Habitat quality overall is improving for sharp-tailed grouse in Lincoln County, where WDFW and the Bureau of Land Management (BLM) are actively managing habitat for sharp-tailed grouse. Continuation of lands enrolled in the Conservation Reserve Program is also important to improve habitat quality in Lincoln and Douglas Counties. WDFW acquisition of lands in Okanogan County near Tunk Valley, Chesaw, and Conconully should also result in improved habitats. Private and tribal lands that are grazed change in habitat quality with the intensity of grazing. Trends on these grazed lands are not predictable.

Increases in grazing pressure on currently occupied sharp-tailed grouse habitat are a principal threat to the continued existence of populations. In general, when grazing by livestock reduces the grass and forb component, sharp-tailed grouse are excluded (Hart et al. 1950, Brown 1966b, Parker 1970, Zeigler 1979). Loss of deciduous cover is especially severe near riparian areas that attract livestock in summer because of water and shade; this cover provides critical foraging areas and escape cover for sharp-tails throughout the year (Zeigler 1979, Marks and Marks 1987a). Trampling, browsing, and rubbing decrease the annual grass and forbs, deciduous trees, and shrubs needed for food and shelter in winter (Parker 1970, Kessler and Bosch 1982, Marks and Marks 1987a). Mattise (1978) found overgrazing very detrimental in nesting and brood-rearing habitat.

In Montana, Brown (1968) reported that the reduction in habitat, because of intensive livestock grazing, resulted in the elimination of sharp-tails in particular areas. Sharp-tails were observed shifting use to ungrazed areas following livestock use of traditional sites (Brown 1968). Marks and Marks (1988) also found sharp-tails in western Idaho selecting home ranges that were least modified by livestock grazing.

The reported effects of grazing on sharp-tailed grouse vary and appear to depend primarily on intensity, duration of grazing, type of livestock, site characteristics, precipitation levels, and past and present land use practices. Grazing systems currently used in range management include seasonal, deferred, and rotation grazing (Stoddard, et al. 1975). Hart et al. (1950) found light to moderate grazing benefiting landowners and sharp-tails on the foothills and benchlands of Utah. Weddell (1992) concluded that rest rotation and deferred grazing were less detrimental to sharp-tailed grouse than season-long grazing, and suggested the disadvantages of increasing grazing under any of these systems outweigh the advantages for sharp-tailed grouse. Even light to moderate grazing can be detrimental in areas with a history of overgrazing, as it may prevent recovery of the native vegetation.

Kessler and Bosch (1982) surveyed sharp-tailed grouse management practices and concluded that grazing, and the resulting habitat loss, are the most serious threats to sharp-tailed grouse survival. Their survey of states and provinces with past or present Columbian sharp-tailed grouse populations found respondents regarded low intensity grazing as beneficial, and high intensity grazing to be negative in its effects on sharp-tails (Kessler and Bosch 1982). Twenty percent more respondents found moderate grazing negative in its effects, and twice as many preferred deferred and rest rotation over continuous grazing. Five of the seven states or provinces with Columbian sharp-tailed grouse listed overgrazing as a major issue/problem related to maintaining this species and its habitat (Braun 1991).

Grazing is a continuing threat to sharp-tailed grouse because of unpredictable changes in land ownership, grazing economics, and the needs of private landowners. Grazing pressure is increasing in several important sharp-tail areas in Washington (M. Schroeder, pers. comm.).

The removal of CRP habitat in Lincoln, Douglas, and Okanogan Counties could cause further declines in sharp-tailed grouse numbers. Contracts for approximately 318,000 hectares expired in 1997. Washington farmers submitted applications for new contracts on 239,000 hectares, and nearly 196,000 hectares were accepted. CRP lands placed back into grain production could cause further declines in the number of sharp-tailed grouse, depending upon how the sharp-tailed grouse use these areas. CRP land and other habitat enhancement areas must be near existing sharp-tail populations to be beneficial (Meints et al. 1992). Although the WDFW is assisting landowners in applying for CRP funding, the long-term status of these areas is uncertain.

The loss of deciduous trees and shrubs by chemical control was associated with declining sharp-tail populations in Washington (Zeigler 1979) and Utah (Hart et al. 1950). Chemical treatment of vegetation in sharp-tailed grouse habitat is detrimental because of the direct loss of vegetation (McArdle 1977, Blaisdell et al. 1982, Oedekoven 1985, Klott 1987). Kessler and Bosch (1982) found most biologists regarded chemical brush control as a negative management practice for sharp-tails. However, in Michigan, herbicidal treatment was used to open dense areas, and to provide more adequate sharp-tailed grouse habitat (Van Etten 1960). In Washington, continued use of herbicides to control sagebrush and other vegetation may cause additional reductions in sharp-tailed grouse habitat.

Fire is a continual threat to sharp-tailed grouse populations. Fire has become a major tool for altering large blocks of sagebrush rangelands. In Lincoln County, three large prescribed fires and one chemical control of sagebrush in the 1980s, in areas containing active leks, were believed to be directly responsible for the decline of both sharp-tailed and sage grouse populations (Merker 1988). McArdle (1977) found less use by sharp-tails in burned areas compared to when other vegetation manipulations had occurred. Likewise, Hart et al. (1950) reported Columbian sharp-tails abandoning a lek site following a fire; the fire also caused accelerated erosion, loss of nests, and loss of winter food and cover.

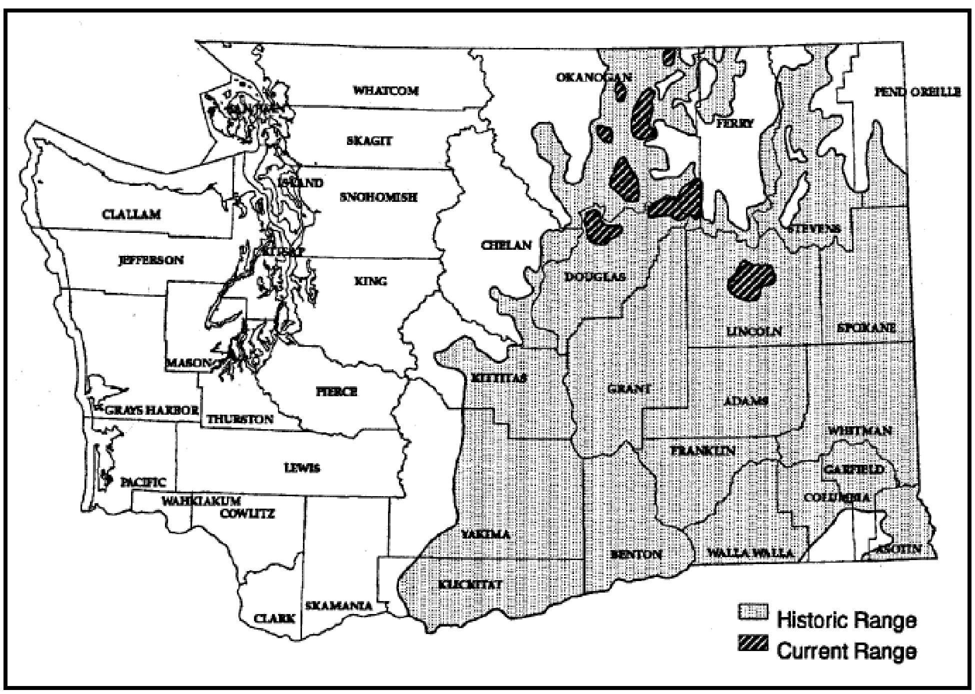
Under some circumstances, burning can help improve sharp-tailed grouse habitat. Burning dense sagebrush and thickly wooded areas was found to improve sharp-tailed grouse habitat in Utah (Hart et al. 1950), North Dakota (Kirsh et al. 1973), Colorado (Rogers 1969), and Wyoming (Oedekoven 1985). In Manitoba and British Columbia, a large movement of sharp-tailed grouse occurred from a high-use lek site to a burned area following a fire that eliminated all residual grass and forbs, but did not greatly affect shrub or tree cover.

Modern fire suppression policies have allowed conifers to invade bunchgrass-prairie habitats in some areas, to the detriment of sharp-tailed grouse populations. In these situations, prescribed burning may be effective in maintaining suitable habitats (Giesen and Connelly 1993). In Washington, prescribed fire is not recommended in shrub/meadow steppe, but may be acceptable for creating habitat where conifers have invaded traditional shrub/meadow steppe areas.

Current Distribution

Currently, Columbian sharp-tails occupy <10% of their historic range in Idaho, Montana, Utah, Wyoming, and Washington, approximately 50% in Colorado, and 8% in British Columbia (Oedekoven 1985; Sullivan 1988; Ritcey 1995). Columbian sharp-tailed grouse are extirpated from California and possibly Oregon and Nevada (Wick 1955; Evanich 1983; Oedekoven 1985). Possible sightings in Nevada (Goose Creek south of Twin Falls, Idaho) and Oregon (Baker County) were recently reported (Braun 1991). Columbian sharp-tails are being reintroduced in Oregon (Starkey and Schnoes 1979; Crawford 1986).

The current range of Columbian sharp-tailed grouse in Washington consists of eight small, severely fragmented populations in Douglas, Lincoln, and Okanogan Counties (**Figure 23**). Sightings of sharp-tails were reported in Asotin County in the mid-1980s; however, the Idaho Department of Fish and Game (IDFG) transplanted sharp-tails in Idaho at that time, some likely dispersing to Asotin County. Sharp-tailed grouse found outside Douglas, Lincoln, and Okanogan Counties are likely transient birds that periodically occupy pockets of remaining shrub/meadow steppe. They contribute little to the statewide population in terms of reproduction or genetics.



Source: Hays et al. 1998

Figure 23 Historic and current range of sharp-tailed grouse in Washington

Population Trend Status

The 1997 breeding population of sharp-tailed grouse in Washington has been estimated through lek counts and a population model. During spring surveys, 358 grouse were counted on 44 leks in three counties (**Table 27**). A model, based on scientific literature, input, and survey data from WDFW biologists and current research in Washington, was used to estimate the size of the 1997 breeding population.

Table 27 Results of 1997 sharp-tailed grouse lek counts in Washington (Hays et al. 1998)

County	Birds	Leks	Birds/lek
Okanogan	169	17	9.9
Lincoln	88	10	8.8
Okanogan (off Colville Reservation)	59	9	6.5
Douglas	42	8	5.3
TOTAL	358	44	8.1

The model assumed all leks were known and surveyed, all males were on leks during counts, and the male to female sex ratio was 1:1. This model would underestimate actual population size if some leks were not located, if all males were not on leks during counts, if the sex ratio was not 1:1, and if surveys were flawed (e.g., bad weather, incomplete counts, etc.).

The model would overestimate actual population size if lek counts included females (which are difficult to distinguish). The population estimate, based on the model, is 716 sharp-tailed grouse in Washington in 1997 (**Table 28**). Allowing for additional unsurveyed habitat, M. Schroeder (pers. comm.) suggests that as many as 1000 sharp-tailed grouse may remain in Washington.

Table 28 Estimated size of the Washington sharp-tailed grouse breeding population

Sex	Population Estimate	Estimate Source
Male	358	Statewide lek counts
Female	358	1:1 sex ratio
TOTAL	716	Males + Females

The remaining sharp-tailed grouse in Washington are distributed in eight fragmented subpopulations. Of these, the subpopulation on the Colville Indian Reservation is the largest remaining in the state (**Table 28**). It is estimated to include 352 grouse and is considered self-sustaining. Of the subpopulations outside of the Reservation, the largest population is in western Lincoln County (177 birds).

The subpopulation south of Bridgeport in Douglas County contains approximately 31 birds. Outside the reservation, Okanogan County supports a total of only 138 birds. This includes four subpopulations that each support less than 25 grouse; these are likely unstable and near extirpation. Sharp-tailed grouse in each of the eight geographic areas appear to be isolated (Schroeder 1996).

Structural Condition Associations

Several environmental and habitat changes appear to have led to improved sage grouse and sharp-tailed grouse populations. Sharp-tails are present in Douglas, Lincoln, and Okanogan counties. Areas supporting the most sharp-tails include: West Foster Creek, East Foster Creek, Cold Springs Basin, and Dyer Hill in Douglas County; Swanson Lakes Wildlife Area in Lincoln County; and the Tunk Valley and Chesaw Units of the Scotch Creek Wildlife Area in the Okanogan Basin. Ziegler (1979) documented a 51% decline in waterbirch and aspen from 1945 to 1977 in Johnson Creek.

Waterbirch buds are the primary food of sharp-tailed grouse during the winter (Hays et al., 1988). In addition, 13% of landowners contacted in Okanogan County were planning to remove waterbirch or aspen (OWSAC 2000). Much winter habitat in Okanogan County has been lost to residential development. One lek was destroyed by a recreational subdivision (OWSAC 2000). Hofmann and Dobler (1988a) also reported the loss of waterbirch in two locations in Okanogan County in less than three months of observation. Sharp-tails no longer used these areas after waterbirch was removed (Hofmann and Dobler, 1988a).

WDFW has an active survey and management program for sharp-tailed grouse because of their state-listed status, and the Okanogan population is considered to be one of the last strongholds for the species. There is an augmentation program underway. Populations and habitat are surveyed annually. Birds are transplanted from elsewhere, research is underway, and WDFW is pursuing land acquisition for habitat.

The CCT is currently managing sharp-tailed grouse within the Reservation boundaries to eliminate the habitat alteration, fragmentation, and human-caused events that put these populations at risk. The CCT has recently begun a study of this species, in coordination with Washington State University, to address limiting factors and habitat restoration within the region.

Sharp-tailed grouse structural conditions and association relationships (IBIS 2003) are shown in **Table 29**.

Table 29 Sharp-tailed grouse structural conditions and association relationships (IBIS 2003)

Common Name	Focal Habitat	Structural Condition (SC)	SC Activity	SC Assoc.
Sharp-tailed Grouse	Shrubsteppe	Grass/Forb-Closed	B	C
		Grass/Forb-Open	B	C
		Low Shrub-Open Shrub Overstory-Mature	B	C
		Low Shrub-Open Shrub Overstory-Old	B	A
		Low Shrub-Open Shrub Overstory-Seedling/Young	B	C
		Medium Shrub-Open Shrub Overstory-Mature	B	A
		Medium Shrub-Open Shrub Overstory-Old	B	P
		Medium Shrub-Open Shrub Overstory-Seedling/Young	B	C

3.6.4 Mule Deer

General Habitat Requirements

Mule deer occupy a variety of habitat types across eastern Washington. Consequently, habitat requirements vary with vegetative and landscape components contained within each herd range. Forested habitats provide mule deer with forage, snow intercept, thermal, and escape cover.

Mule deer occupying mountain-foothill habitats live within a broad range of elevations, climates, and topography that includes a wide range of vegetation; many of the deer using these habitats are migratory. Mule deer are found in the deep canyon complexes along the major rivers and in the channeled scablands of eastern Washington; these areas are dominated by native bunch grasses or shrubsteppe vegetation. Mule deer also occupy agricultural areas that once were shrubsteppe.

Limiting Factors

Mule deer and their habitats are being impacted in a negative way by dam construction, urban and suburban development, road and highway construction, over-grazing by livestock, inappropriate logging operations, competition by other ungulates, drought, fire, over-harvest by hunters, predation, disease, and parasites.

Weather conditions can play a major role in the productivity and abundance of mule deer. Drought conditions can have a severe impact on mule deer because forage does not replenish itself on summer or winter range, and nutritional quality is low. Drought conditions during the summer and fall can result in low fecundity in does, and poor physical condition going into the winter months. Severe winter weather can result in high mortality, depending on severity. Severe weather can result in mortality of all age classes, but the young, old, and mature bucks usually sustain the highest mortality. If mule deer are subjected to drought conditions in the summer and fall, followed by a severe winter, the result can be high mortality rates and low productivity the following year.

Habitat conditions in the Ecoprovince have deteriorated in some areas and improved dramatically in others. The conversion of shrubsteppe and grassland habitat to agricultural croplands and residential development has resulted in the loss of thousands of acres of mule deer habitat. This has, however been mitigated to some degree by the implementation of the CRP. Noxious weeds have invaded many areas resulting in a tremendous loss of good habitat for mule deer.

Fire suppression has resulted in a decline of habitat conditions in the mountains and foothills of the Cascade Mountains. Browse species need to be regenerated by fire in order to maintain availability and nutritional value to big game. Lack of fire has allowed many browse species to grow out of reach for mule deer (Leege 1968; 1969; Young and Robinette 1939).

The reservoirs created by dams on the Columbia River inundated prime riparian habitat that supported many species of wildlife, including mule deer. This riparian zone provided high quality habitat (forage/cover), especially during the winter months. The loss of this important habitat, and the impact it has had on the mule deer population along the breaks of the Columbia River, may never be fully understood.

Current Distribution

Deer damage is a chronic problem in the Omak district. During severe winters, deer are often forced onto low elevation private property in close proximity to human development. At such times, damage to orchards, haystacks, and landscaping can be significant" (OWSAC 2000).

The WDFW conducts annual mule deer and whitetail deer population surveys, and manages its wildlife areas for winter mule deer range. The USFS and WDNR also manage portions of their lands for winter deer range.

The CCT is a major financial contributor to, and is involved in, an ongoing long-term mule deer study with WFWD, Chelan Co. PUD, U.S. Forest Service, Inland NW Wildlife Council, WSU, UW, and UI. The CCT is actively monitoring habitat, limiting factors and population trends, and performs annual aerial surveys, regulates tribal hunting seasons and manages hunter check stations.

Population Trend Status

Mule deer structural conditions and association relationships (IBIS 2003) are shown in **Table 30**. Mule deer populations have varied dramatically throughout recorded history of the region. In the 1800s, mule deer populations were reported to be extremely low (OWSAC 2000). In the 1900s, deer populations fluctuated widely, with historic highs in the 1950s and 1960s.

Population lows are because of a number of factors, including severe weather conditions, overused winter range, and hunting pressure. Severe winter weather conditions have significantly reduced mule deer populations since 1992. The winter of 1996/1997 was especially hard on the local herds.

"Qualitative observations from land managers, biologists, and long time residents, as well as harvest figures, suggest the populations may be half of what it was in the mid 1980s and early 1990s" (OWSAC 2000). A shorter season and reduced number of hunters in 1997 along with easier overwintering conditions during the 1997/98 winter has been beneficial to the herds (OWSAC 2000).

Mule deer on the reservation are suffering long-term declines attributed to habitat changes, habitat fragmentation, severe weather conditions, and overgrazing. Data from Colville Tribes aerial trend counts indicate severe declines in both mule deer and whitetail populations (Snappily Subbasin Summary). Mule deer are important for cultural and subsistence reasons.

Table 30 Mule deer structural conditions and association relationships (IBIS 2003)

Common Name	Focal Habitat	Structural Condition (SC)	SC Activity	SC Assoc.
Mule Deer	Shrubsteppe	Grass/Forb-Closed	B	A
		Grass/Forb-Open	B	A
		Low Shrub-Closed Shrub Overstory-Mature	B	A
		Low Shrub-Closed Shrub Overstory-Old	B	A
		Low Shrub-Closed Shrub Overstory-Seedling/Young	B	A
		Low Shrub-Open Shrub Overstory-Mature	B	A
		Low Shrub-Open Shrub Overstory-Old	B	A
		Low Shrub-Open Shrub Overstory-Seedling/Young	B	A
		Medium Shrub-Closed Shrub Overstory-Mature	B	A
		Medium Shrub-Closed Shrub Overstory-Old	B	A
		Medium Shrub-Closed Shrub Overstory-Seedling/Young	B	A
		Medium Shrub-Open Shrub Overstory-Mature	B	A
		Medium Shrub-Open Shrub Overstory-Old	B	A
		Medium Shrub-Open Shrub Overstory-Seedling/Young	B	A
		Tall Shrub-Closed Shrub Overstory-Mature	B	A
		Tall Shrub-Closed Shrub Overstory-Old	B	A
		Tall Shrub-Closed Shrub Overstory-Seedling/Young	B	A
		Tall Shrub-Open Shrub Overstory-Mature	B	A
		Tall Shrub-Open Shrub Overstory-Old	B	A
		Tall Shrub-Open Shrub Overstory-Seedling/Young	B	A

3.6.5 Red-eyed Vireo

General Habitat Requirements

Partners in Flight established biological objectives for this species in the lowlands of western Oregon and western Washington. These include providing habitats that meet the following definition: mean canopy tree height greater than 50 feet, mean canopy closure greater than 60%, young (recruitment) sapling trees greater than 10% cover in the understory, and riparian woodland greater than 64 feet wide (Altman 2001). Red-eyed vireos are closely associated with riparian woodlands and black cottonwood stands, and may use mixed deciduous stands.

The patchy distribution in Washington for this species correlates with the distribution of large black cottonwood (*Populus trichocarpa*) groves, which are usually limited to riparian areas. The red-eyed vireo is one of the most abundant species in northeastern United States, but is much less common in Washington because of limited habitat.

The Methow subbasin is host to some of Eastern Washington's best remaining tracts of cottonwood gallery forests, which are found in the wide floodplain portions of the Methow River valley and its major tributaries. Almost all of this habitat type is in private ownership, of which, much has been converted to residential development or agriculture;significant forest parcels remain along the Methow River between Winthrop and Lost River.

Additional significant stands are located along the Twisp and Chewuch rivers, and more fragmented pockets can be found along the Methow between Winthrop and Carlton. Below Carlton, a higher stream gradient and a more constrained channel preclude the development of large patches of this habitat type (J. Foster, WDFW, pers. comm.). Because of its proximity to roads and other developed areas, much of the remaining riparian/floodplain habitat may be at risk of conversion to housing development.

Limiting Factors

Habitat loss because of hydrological diversions and control of natural flooding regimes (e.g., dams) has resulted in an overall reduction of riparian habitat for red-eyed vireos through the conversion of riparian habitats and inundation from impoundments.

Like other neotropical migratory birds, red-eyed vireos suffer from habitat degradation resulting from the loss of vertical stratification in riparian vegetation, lack of recruitment of young cottonwoods, ash (*Fraxinus latifolia*), willows (*Salix* spp.), and other subcanopy species.

Streambank stabilization (e.g., riprap) narrows stream channels and reduces the flood zone and extent of riparian vegetation. The invasion of exotic species such as canarygrass (*Phalaris* spp.) and blackberry (*Rubus* spp.) also contributes to a reduction in available habitat for the red-eyed vireo. Habitat loss can also be attributed to overgrazing, which can reduce understory cover. Reductions in riparian corridor widths may decrease suitability of riparian habitat, and may increase encroachment of nest predators and nest parasites to the interior of the stand.

Hostile landscapes, particularly those in proximity to agricultural and residential areas, may have a high density of nest parasites, such as brown-headed cowbirds and domestic predators (cats), and can be subject to high levels of human disturbance. Recreational disturbances, particularly during nesting season, and particularly in high-use recreation areas, may have an impact on red-eyed vireos.

Increased use of pesticide and herbicides may reduce the insect food base for red-eyed vireos.

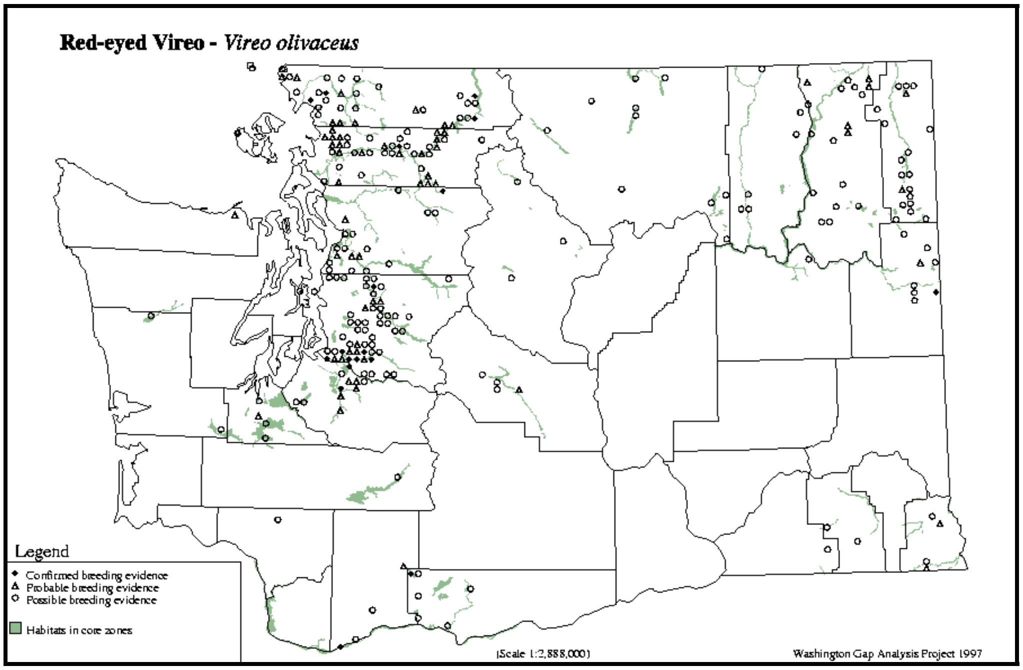
Current Distribution

The North American breeding range of the red-eyed vireo extends from British Columbia to Nova Scotia, north through parts of the Northwest Territories, and throughout most of the lower United States ((Washington GAP Analysis Project 1997)

Figure 24). The birds migrate to the tropics for the winter.

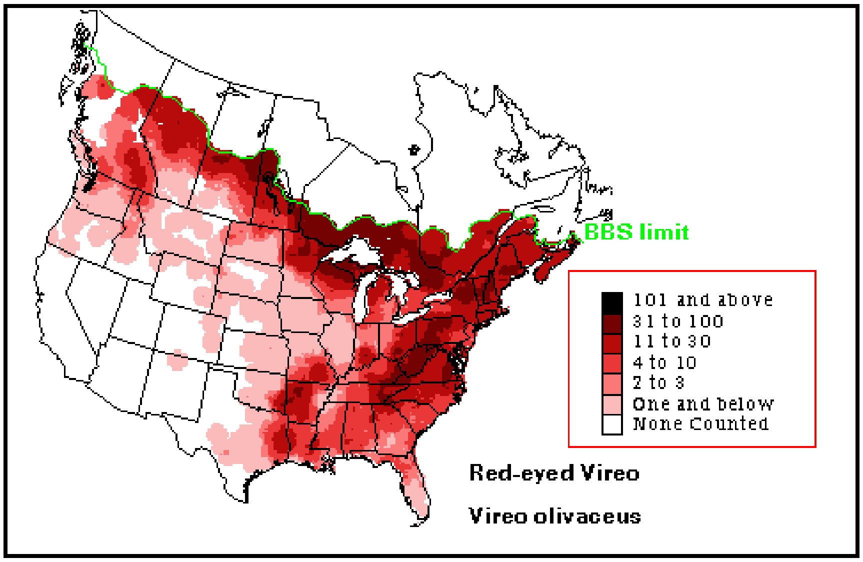
The patchy distribution in Washington for this species correlates with the distribution of large black cottonwood (*Populus trichocarpa*) groves that are usually limited to riparian areas. The red-eyed vireo is one of the most abundant species in the northeastern United States, but is much less common in Washington because of limited habitat. Red-eyed vireo breeding and summer distribution are illustrated in **Figure 25** and (Sauer et al. 2003)

Figure 26 respectively.



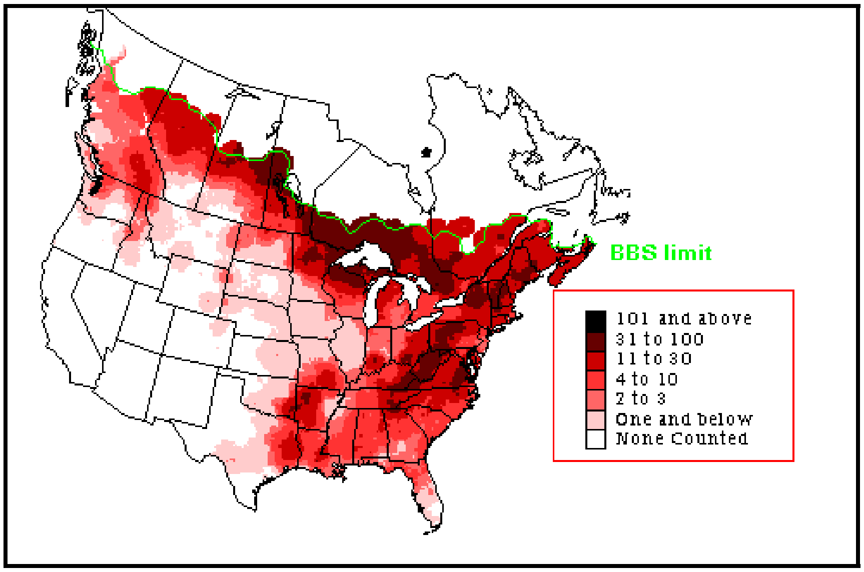
(Washington GAP Analysis Project 1997)

Figure 24 Breeding bird atlas data (1987-1995) and species distribution for red-eyed vireo



(Sauer et al. 2003)

Figure 25 Red-eyed vireo breeding distribution



(Sauer et al. 2003)

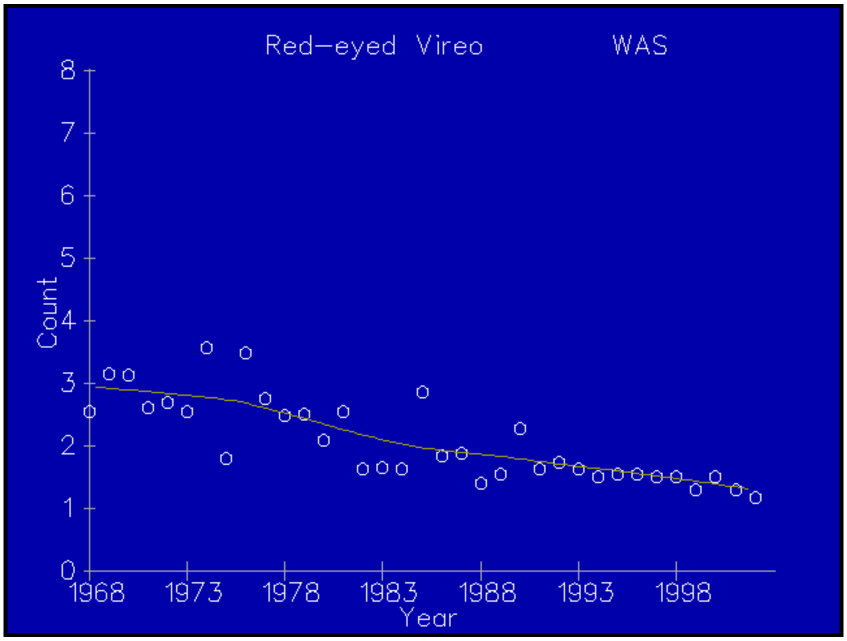
Figure 26 Red-eyed vireo summer distribution

Population Trend Status

The red-eyed vireo is secure, particularly in the eastern United States. Within the state of Washington, the red-eyed vireo is locally common, more widespread in northeastern and southeastern Washington, and not a conservation concern (Altman 1999).

Red-eyed vireos are currently protected throughout their breeding range by the Migratory Bird Treaty Act (1918) in the United States, the Migratory Bird Convention Act (1916) in Canada, and the Convention for the Protection of Migratory Birds and Game Mammals (1936) in Mexico.

In Washington, BBS data show a significant population increase of 4.9% per year from 1982 to 1991 (Peterjohn 1991). However, in the long-term results, a population decline in Washington of 2.6% per year has been observed (Figure 27), although the change is not statistically significant largely because of scanty data (Sauer et al. 2003). Because the BBS dates back only about 30 years, population declines in Washington resulting from habitat loss dating prior to the survey would not be accounted for by that effort.



Source: BBS data, Washington; Sauer et al. 2003

Figure 27 Red-eyed vireo counts (1968-1998)

3.6.6 Yellow-breasted Chat

General Habitat Requirements

Yellow-breasted chats are found in second growth, shrubby old pastures, thickets, bushy areas, scrub, woodland undergrowth, and fence rows, including low wet places near streams, pond edges, or swamps. They have been found in thickets with few tall trees, early successional stages of forest regeneration, and commonly, in sites close to human habitation. In winter, yellow-breasted chats establish territories in young second-growth forest and scrub (Dennis 1958, Thompson and Nolan 1973, Morse 1989).

Limiting Factors

Threats include habitat loss because of successional changes and clearing of land for agricultural or residential development. Frequently parasitized by the brown-headed cowbird (*Molothrus ater*), but it is not well known whether this has a significant impact on reproductive success.

Current Distribution

Yellow-breasted chat breeding range includes southern British Columbia across southern Canada and the northern U.S. to southern Ontario and central New York, south to southern Baja California, to Sinaloa on Pacific slope, to Zacatecas in interior over plateau, to southern Tamaulipas on Atlantic slope, and to Gulf Coast and northern Florida (AOU 1998).

Yellow-breasted chat non-breeding range includes southern Baja California, southern Sinaloa, southern Texas, southern Louisiana, and southern Florida south (rarely north to Oregon, Great Lakes, New York, or New England) to western Panama (AOU 1998).

Population Trend Status

North American Breeding Bird Survey (BBS) data indicate a significant population decline in eastern North America from 1966 to 1988, and a significant increase in western North America from 1978 to 1988 (Sauer and Droege 1992). In North America overall, from 1966 to 1989, there was a non-significant decline, averaging 0.8% per year from 1966 to 1989 (Droege and Sauer 1990), a non-significant 9% decline from 1966 to 1993, and a barely significant increase of 8% from 1984 to 1993 (Price et al. 1995).

Yellow-breasted chats may have declined in south-central and southeastern New York between the early 1900s and mid-1980s (Eaton, in Andrle and Carroll 1988). Numbers have steadily declined in some areas of Ohio, though the range has not changed much since the 1930s (Peterjohn and Rice 1991).

Yellow-breasted chats have declined in Indiana and Illinois since the mid-1960s; they have declined along the lower Colorado River with the loss of native habitat (Hunter et al. 1988). In Canada, they are thought to be slowly declining because of habitat destruction in British Columbia; populations in Alberta and Saskatchewan appear to be stable; population has declined at Point Pelee National Park in Ontario, which contains a considerable proportion of the province's small population, and; there no longer are breeds at Rondeau Provincial Park (Ontario). The population on Pelee Island (Ontario), however, appears to be stable (Cadman and Page 1994).

Washington trends are illustrated in Figure 28.Yellow-breasted chat breeding season abundance (from BBS data) is illustrated in Figure 29, and winter season abundance (from CBC data) is illustrated in Figure 30.

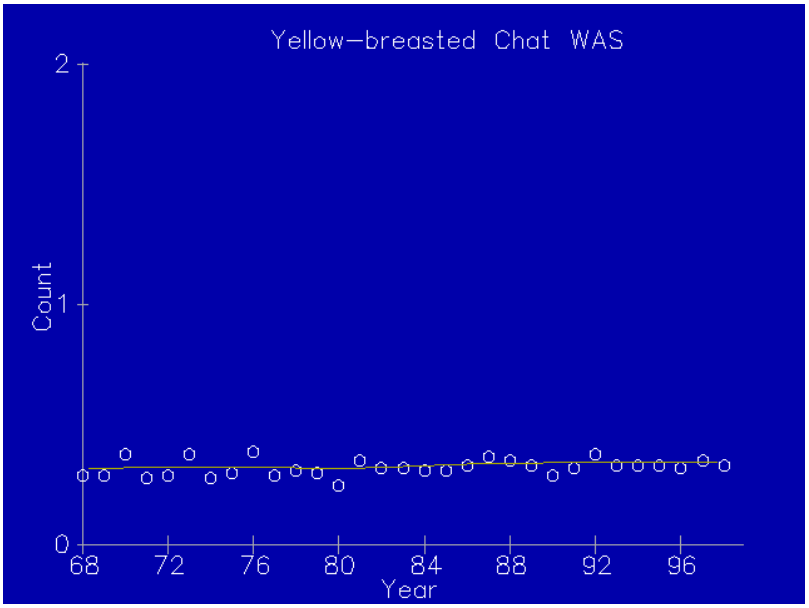
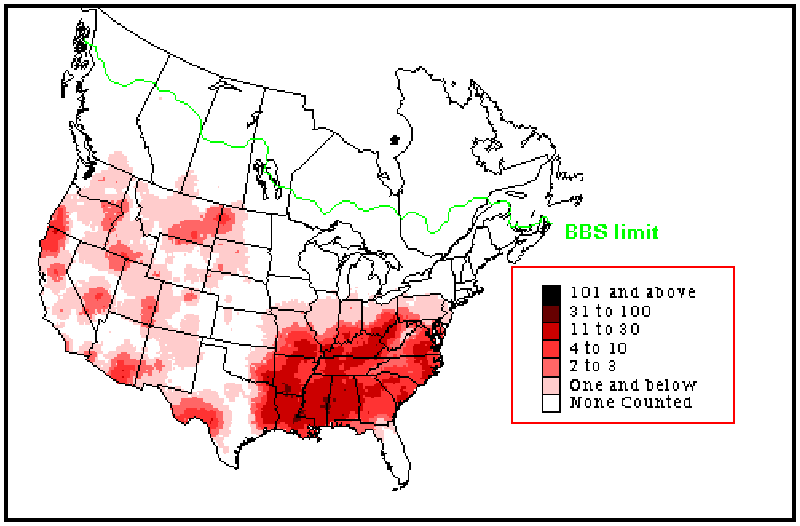
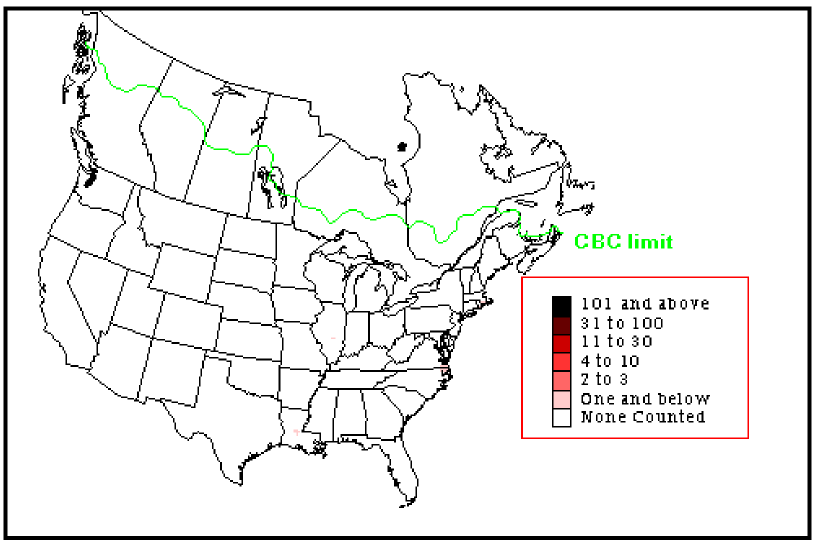


Figure 28 Population trends for Yellow-breasted Chat in Washington State



Source: BBS data; Sauer et al. 2003

Figure 29 Seasonal abundance of Yellow-breasted Chat in Washington State from the BBS



Source: CBC data

Figure 30 Winter abundance of Yellow-breasted Chat in Washington State from CBC Data

3.6.7 American Beaver

General Habitat Requirements

Suitable beaver habitat in all wetland cover types (e.g., herbaceous wetland, riparian wetland, and deciduous forested wetland) must have a permanent source of surface water with little or no fluctuation (Slough and Sadleir 1977). Lakes and reservoirs that have extreme annual or seasonal fluctuations in the water level will be unsuitable habitat for beaver. Similarly, intermittent streams, or streams that have major fluctuations in discharge (e.g., high spring runoff) or a stream channel gradient of 15% or more, will have little year-round value as beaver habitat. Assuming that there is an adequate food source available, small lakes less than 20 acres in surface area are assumed to provide suitable habitat. Large lakes and reservoirs greater than 20 acres in surface area must have irregular shorelines (e.g., bays, coves, and inlets) in order to provide optimum habitat for beaver.

Beavers are generalized herbivores and appear to prefer herbaceous vegetation such as duck potato (*Sagittaria* spp.), duckweed (*Lemna* spp.), pondweed (*Potamogeton* spp.), and water weed (*Elodea* spp.) to woody vegetation during all seasons of the year, if it is available (Jenkins 1981). The leaves, twigs, and bark of woody plants are eaten, as well as many species of aquatic and terrestrial herbaceous vegetation.

Beaver show strong preferences for particular woody plant species and size classes (Jenkins 1975; Collins 1976a; Jenkins 1979). Denney (1952) reported that beavers preferred, in order of preference, aspen, willow, cottonwood, and alder. Woody stems cut by beavers are usually less than three to four inches diameter at breast height (DBH) (Bradt 1947; Hodgdon and Hunt 1953; Longley and Moyle 1963; Nixon and Ely 1969). Jenkins (1980) reported a decrease in mean stem size cut and greater selectivity for size and species with increasing distance from the water's edge. Food preferences may vary seasonally, or from year to year, as a result of variation in the nutritional value of food sources (Jenkins 1979). Specific habitat attributes are shown in **Table 31**.

Table 31 Focal Species, Focal Habitat Types, and Key Habitat Relationships

Focal Species	Focal Habitat Type	Key Habitat Relationships		Comments	Life Requisite	Selection Rationale
		Conservation Focus	Habitat Attribute (Vegetative Structure)			
Sage thrasher	Shrub-steppe	sagebrush height	sagebrush cover 5-20%	not area-sensitive (needs > 40 ac); not impacted by cowbirds; high moisture sites w/ tall shrubs	Food, Reproduction	The sage thrasher is a shrubsteppe obligate species and an indicator of healthy, tall sagebrush dominated shrubsteppe habitat.
			sagebrush height > 80 cm		Food, Reproduction	
			herbaceous cover 5-20%		Food, Reproduction	
			other shrub cover > 10%		Food, Reproduction	
			non-native herbaceous cover < 10%		Food, Reproduction	
Brewer's sparrow	Shrubsteppe	sagebrush cover	sagebrush cover 10-30%		Food, Reproduction	The Brewer's sparrow is a shrubsteppe obligate species and is an indicator of healthy sagebrush dominated shrubsteppe habitat.

Focal Species	Focal Habitat Type	Key Habitat Relationships: Conservation Focus	Key Habitat Relationships: Habitat Attribute (Vegetative Structure)	Comments	Life Requisite	Selection Rationale
			sagebrush height > 60 cm		Food, Reproduction	
			herbaceous cover > 10%		Food, Reproduction	
			open ground > 20%		Food, Reproduction	
			non-native herbaceous cover < 10%		Food, Reproduction	
Grasshop per sparrow	Shrubsteppe	Native steppe/ grasslands	native bunchgrass cover > 15% and comprising > 60% of the total grass cover		Food, Reproduction	The grasshopper sparrow is an indicator of healthy steppe habitat dominated by native bunch grasses.
Sharp-tailed grouse	Shrubsteppe	Deciduous trees and shrubs	mean VOR > 6"		Reproduction	Sharp-tailed grouse is a management priority species and an indicator of healthy steppe/shrubstep pe habitat w/ healthy imbedded mesic draws.
			> 40% grass cover		Reproduction	
			> 30% forb cover		Reproduction	
			< 5% cover introduced herbaceous cover		Reproduction	

Focal Species	Focal Habitat Type	Key Habitat Relationships		Comments	Life Requisite	Selection Rationale
		Conservation Focus	Habitat Attribute (Vegetative Structure)			
			> 50% optimum area providing nest/brood cover		Reproduction	
			> 0.25 km between nest/brood rearing habitat and winter habitat		Reproduction	
			> 75% cover deciduous shrubs and trees		Winter	
			> 10% optimum area providing winter habitat		Winter	
Sage grouse	Shrubsteppe	diverse herbaceous understory, sagebrush cover	sagebrush cover 10-30%	area sensitive; needs large blocks	Reproduction	shrubsteppe obligate; State threatened, Federal Candidate species
			forb cover > 10%		Food	
			open ground cover > 10%			
			non-native herbaceous cover < 10%			
Pygmy rabbit	Shrubsteppe	deep, rock-free soil	sagebrush cover 21-36%	area sensitive, needs large blocks	Reproduction	Shrubsteppe obligate; Federal, State endangered species
			shrub height 32”			

Focal Species	Focal Habitat Type	Key Habitat Relationships		Comments	Life Requisite	Selection Rationale
		Conservation Focus	Habitat Attribute (Vegetative Structure)			
Mule deer	Shrubsteppe	antelope bitterbrush	30-60% canopy cover of preferred shrubs < 5 ft.		Food	The mule deer is a management priority species and an indicator of healthy diverse shrub layer in east-slope shrubsteppe habitat.
			number of preferred shrub species > 3			
			mean height of shrubs > 3 ft.			
			30-70% canopy cover of all shrubs < 5 ft.			
Willow flycatcher	Eastside (Interior) Riparian Wetlands	shrub density	dense patches of native vegetation in the shrub layer > 35 $ft.^2$ in size and interspersed with openings of herbaceous vegetation	> 20 ac; frequent cowbird host; sites > 0.6 mi from urban/residential areas and > 3 mi from high-use cowbird areas	Reproduction	Indicator of healthy, diverse riparian wetland habitat
			shrub layer cover 40-80%		Reproduction	
			shrub layer height > 3 ft. high		Reproduction	
			tree cover < 30%		Reproduction	
Lewis' woodpecker	Eastside (Interior) Riparian Wetlands	large cottonwood trees/snags	> 0.8 trees/ac > 21" dbh	Dependent on insect food supply; competition from	Food	Indicator of healthy cottonwood

Focal Species	Focal Habitat Type	Key Habitat Relationships: Conservation Focus	Key Habitat Relationships: Habitat Attribute (Vegetative Structure)	Comments	Life Requisite	Selection Rationale
				starlings detrimental		stands with snags
			canopy cover 10-40%			
			shrub cover 30-80-%			
Red-eyed vireo	Eastside (Interior) Riparian Wetlands	canopy foliage and structure	canopy closure > 60%		Food, Reproduction	The red-eyed vireo is an obligate species in riverine cottonwood gallery forests and an indicator of healthy canopy cover.
			riparian zone of mature deciduous trees > 160 ft.		Food, Reproduction	
			> 10% of the shrub layer should be young cottonwoods		Food, Reproduction	
Yellow-breasted chat	Eastside (Interior) Riparian Wetlands	dense shrub layer	shrub layer 1-4 m tall	vulnerable to cowbird parasitism; grazing reduces understory structure	Food, Reproduction	The yellow-breasted chat is an indicator of healthy shrub dominated riparian habitat and is a management priority species in the Canadian Okanogan.
			30-80% shrub cover		Food,	

Focal Species	Focal Habitat Type	Key Habitat Relationships: Conservation Focus	Key Habitat Relationships: Habitat Attribute (Vegetative Structure)	Comments	Life Requisite	Selection Rationale
					Reproduction	
			scattered herbaceous openings		Food, Reproduction	
			tree cover < 20%		Food, Reproduction	
Beaver	Eastside (Interior) Riparian Wetlands	canopy closure	40-60% tree/shrub canopy closure		Food	The beaver is an indicator of healthy regenerating aspen stands and an important habitat manipulator.
			trees < 6" dbh; shrub height ≥ 6.6 ft.			
		permanent water	stream channel gradient ≤ 6% with little to no fluctuation		Water (cover for food and reproductive requirements)	
		shoreline development	woody vegetation ≤ 328 ft. from water		Food	
Red-winged blackbird	Herbaceous Wetlands	Open water with emergent wetlands				Wetland obligate species
Pygmy nuthatch	Ponderosa Pine	large trees	> 10/ac > 21" dbh with > 2 trees > 31" dbh	large snags for nesting; large trees for foraging	Food, Reproduction	The pygmy nuthatch is a species of management concern and is an obligate for healthy old-growth

Focal Species	Focal Habitat Type	Key Habitat Relationships		Comments	Life Requisite	Selection Rationale
		Conservation Focus	Habitat Attribute (Vegetative Structure)			
						Ponderosa pine forest with an abundant snag component.
			> 1.4 snags/ac > 8" dbh with > 50% > 25"			
Gray flycatcher	Ponderosa Pine	shrubsteppe/ pine interface; pine savannah w/ shrub-bunchgrass understory	Nest tree diameter 18" dbh		Reproduction	The gray flycatcher is an indicator of healthy fire-maintained regenerating ponderosa pine forest.
			Tree height 52'		Food	
White-headed woodpecker	Ponderosa Pine	large patches of old growth forest with large trees and snags	> 10 trees/ac > 21" dbh w/ > 2 trees > 31" dbh	large high-cut stumps; patch size smaller for old-growth forest; need > 350 ac or > 700 ac	Reproduction	The white-headed woodpecker is a species of management concern and it is an obligate species for large patches of healthy old-growth Ponderosa pine forest.
Flammulated owl	Ponderosa Pine	interspersion; grassy openings and dense thickets	> 10 snags / 40 ha > 30 cm dbh and 1.8m tall	thicket patches for roosting; grassy openings for foraging	Food	The flammulated is an indicator of a healthy landscape mosaic in Ponderosa pine and Ponderosa

Focal Species	Focal Habitat Type	Key Habitat Relationships		Comments	Life Requisite	Selection Rationale
		Conservation Focus	Habitat Attribute (Vegetative Structure)			
						pine/Douglas-fir forest and it is a Washington State priority species.

Limiting Factors

Beavers readily adapt to living in urban areas near humans and are limited primarily by the availability of permanent water with limited fluctuations and accessibility of food.

Riparian habitat along many waterways has been removed, thus, removing important habitat and food sources for beaver.

Beavers that create dams, that restrict fish passage, are removed in order to restore fish passage.

Current Distribution

The beaver is found throughout most of North America except in the Arctic tundra, peninsular Florida, and the Southwestern deserts (Figure 31) (Allen 1983; VanGelden 1982; Zeveloff 1988).

Source: Linzey and Brecht 2002.

Figure 31 North American distribution of beaver.

3.6.8 Pygmy Nuthatch

General Habitat Requirements

Among all breeding birds within Ponderosa pine forests, the density of Pygmy nuthatches is most strongly correlated with the abundance of Ponderosa pine trees (Balda 1969). In Colorado, 93% of breeding bird atlas observations occurred in coniferous forests, 70% of those in Ponderosa pines. Indeed the distribution of Pygmy nuthatches in Colorado coincides with that of Ponderosa pine woodlands in the state (Jones 1998).

Several studies identify the Pygmy nuthatch as the most, or one of the most abundant species in Ponderosa forests (e.g., Mt. Charleston, Nevada, Arizona’s mountains and plateaus, New

Mexico, Colorado statewide, and Baja California) (Reassumes 1941; Brandt 1951; Norris 1958; Stallcup 1968; Balda 1969; Farris 1985; Travis 1992; Kingery 1998), as well as in other yellow long-needled pines such as those of coastal California and Popocatépetl, Mexico (Norris 1958, Paynter 1962).

In California's mountains, the Pygmy nuthatch favors open park-like forests of Ponderosa and Jeffrey pines in the Sierra Nevada Mountains (Gaines 1988), but also ranges to 3,050 metres (10,007 feet) in open stands of large lodgepole pine in the White Mountains of California (Shuford and Metropulos 1996). In the Mogollon Rim region of central Arizona, the bird breeds and feeds in vast expanses of Ponderosa pine that extend throughout the Colorado plateau, and is also common in shallow snow-melt ravines that course through the pine forests. These snowmelt drainages contain white fir (*Abies concolor*), Douglas-fir, Arizona white pine (*Pinus strobiformis*), quaking aspen (*Populus tremuloides*), and an understory of maples (*Acer* spp.) (Kingery and Ghalambor 2001).

In New Mexico, it is most common in Ponderosa pine, including Ponderosa/oak and Ponderosa/Douglas-fir forests (Kingery and Ghalambor 2001). In Washington, it uses Douglas-fir zones rarely, and then only those in or near Ponderosa pines (Smith et al. 1997). In Summit County, Colorado, a small group of Pygmy nuthatches occupy a small section of lodgepole pine at the edge of an extensive lodgepole forest (Kingery and Ghalambor 2001).

In coastal California (Sonoma, Marin, Monterey, San Luis Obispo Counties) Pygmy nuthatches occur in the "coastal fog belt" (Burridge 1995) in Bishop pine (*Pinus muricata*), Coulter pine (*Pinus coulteri*), natural and planted groves of Monterey pine (*Pinus radiata*) (Roberson 1993, Shuford 1993), other pine plantations (Burridge 1995), and wherever Ponderosa pines grow (e.g., Santa Lucia Mountains, Monterey County) (Roberson 1993).

In Mexico, where it occurs in arid pine forests of the highlands, the Pygmy nutchatch follows pines to their upper limits at the tree line on Mount Popocatépetl (Paynter 1962) and Pico Orizaba (Cox 1895). Almost no other contemporary information is available on the habitat preferences of Pygmy nuthatches in Mexican mountain ranges (S. Howell, J. Nocedal, A. Sada, pers. comm.). They are known to favor pine and pine-oak woodlands; these pine species include Ponderosa-type pines: *Pinus engelmanii*, *P. arizonica*, *P. montezumae*, as well as non-Ponderosa-types: *Pinus teocote*, *P. hartwegii*, *P. leiophylla*, and *P. cooperi*. Associated Mexican tree species in Pygmy nuthatch habitat include oaks (*Quercus rugosa*, *Q. castanea*, *Q. durifolia*, and *Q. hartwegii*), madrones (*Arbutus xalpensis* and *A. glandulosa*), and alders (*Alnus firmifolia*; Nocedal 1984, 1994, A. Sada, pers. comm.). The species also occurs, in small numbers, in fir (*Abies religiosa*) forests (Nocedal 1984, 1994).

Limiting Factors

There is good evidence for at least two main limiting factors in Pygmy nuthatch populations: the availability of snags for nesting and roosting, and sufficient numbers of large cone-producing trees for food.

Pygmy nuthatches are dependent on snags for nesting and roosting, and reduced snag availability has been shown to have negative effects on populations. Because Pygmy nuthatches nest and roost in excavated tree cavities, the importance of snags is manifested during both the breeding and non-breeding season. During the breeding season, numerous studies have documented a decline in the number of breeding pairs and a reduction in population density on sites where timber harvesting reduced the number of available snags. During the non-breeding season, studies show that timber harvests, that remove the majority of snags, cause communally roosting groups to use atypical cavities with poorer thermal properties.

Pygmy nuthatches choosing roost sites during the non-breeding season use a different set of characteristics compared to nest sites. A considerable reduction in snag densities may affect overwinter survivorship, and possibly reproduction, by forcing Pygmy nuthatches to use cavities in snags they would normally avoid (Hay and Güntert 1983; Matthysen 1998). More research on the differences among snags is clearly needed in order to distinguish those factors that make some snags more desirable than others.

Pygmy nuthatch populations rely heavily on the availability of pine seeds and arthropods that live on pines. In comparison to other nuthatches and woodpeckers, Pygmy nuthatches forage more amongst the foliage of live trees rather than on the bark. The preferred foraging habitat for Pygmy nuthatches appears to contain a high canopy density, low canopy patchiness, and increased vertical vegetation density, a common feature of mature undisturbed forests.

Pygmy nuthatch populations are very sedentary. Young birds have been observed to only move 940 feet (287 metres) from their natal territories. Such limited dispersal reduces the number of individuals that emigrate and immigrate from local populations, which in turn, reduces gene flow and demographic stability. Thus, in contrast to the majority of North America's songbirds, movement and dispersal patterns in Pygmy nuthatch populations are limited to a relatively small geographic area. Pygmy nuthatches may, therefore, need a greater amount of connectivity between suitable habitat in comparison to other resident birds.

In a recent review of the effects of recreation on songbirds within Ponderosa pine forests, Marzluff (1997) hypothesized that nuthatches would experience moderate decreases in population abundance and productivity in response to impacts associated with established campsites (the Pygmy nuthatch was not specifically identified).

Impacts associated with camping that might negatively influence nuthatches include changes in vegetation, disturbance of breeding birds, and increases in the number of potential nest predators (Marzluff 1997). Other recreational activities associated with resorts and recreational residences may, however, moderately increase nuthatch population abundance and productivity (Marzluff 1997). This positive effect on nuthatch populations is likely to occur through food supplementation, such as bird feeders, that are frequently visited by Pygmy nuthatches.

Current Distribution

The Pygmy nuthatch is resident in Ponderosa and similar pines from south-central British Columbia and the mountains of the western United States to central Mexico. The patchy distribution of pines in western North America dictates the patchy distribution of the Pygmy nuthatch throughout its range. The reliance on pines distinguishes Pygmy nuthatches from other western nuthatches, such as the red-breasted and white-breasted nuthatches, which are associated with fir/spruce and deciduous forests respectively (Ghalambor and Martin 1999). The following is a review of the distribution of populations in the United States, Canada, and Mexico (based on Kingery and Ghalambor 2001).

The Pygmy nuthatch occurs in southern interior British Columbia, particularly in the Okanagan and Similkameen valleys and adjacent plateaus (Campbell et al. 1997), south into the Okanagan Highlands, and into the northeast Cascades of Washington. It is scattered along the eastern slope of the Cascades from central Washington (Jewett et al. 1953; Smith et al. 1997) into Oregon and in the Blue Mountains in southwest Washington (Garfield County only) (Smith et al. 1997), but widespread in Oregon along the west slope of the Cascades (Gabrielson and Jewett 1940; Jewett et al. 1953; Gilligan et al. 1994). It ranges south from the Cascades in Oregon into northern California, and south into the Sierra Nevadas and nearby mountains of Nevada (Brown 1978).

In the southern Sierra Nevadas, it is found on the east and west side of the range in the Mono Craters and Glass Mountain region (Gaines 1988, Shuford and Metropulos 1996) and in the White Mountains of Nevada and California (Norris 1958; Brown 1978; Shuford and Metropulos 1996). It is also found throughout the mountain ranges of southern California, including the Sierra Madres in Santa Barbara County, the Mt. Pinos area (Kern and Ventura Counties), the San Gabriel and San Bernardino Mountains in Los Angeles and San Bernardino Counties (Norris 1958; B. Carlson, K. Garrett, pers. comm.), the San Jacinto and Santa Rosa Mountains in Riverside County (Norris 1958; B. Carlson, pers. comm.), in the Laguna and Cuyamaca Mountains, as well as at Mt. Palomar and the Volcan and Hot Springs Mountains of San Diego County (San Diego County Breeding Bird Atlas preliminary data, B. Carlson, P. Unitt, pers. comm.). The range extends south into the Sierra Juarez and Sierra San Pedro Mártir Mountains in Baja California Norte, Mexico (Grinnell 1928; Norris 1958;).

In eastern Washington, the Pygmy nuthatch is common in the pine forests of Spokane County (Jewett et al. 1953; Smith et al. 1997) and adjacent Kootenai County, Idaho (Burleigh 1972). Only scattered records exist for the rest of Idaho’s mountains (Burleigh 1972; Stephens and Sturts 1991), but Pygmy nuthatches are well distributed in the Rocky Mountains of far western Montana (Montana Bird Distribution Committee 1996).

Population Trend Status

Survey-wide estimates of all BBS routes suggest Pygmy nuthatch populations are stable (Sauer et al. 2000); however, these estimates are based on small samples that do not provide a reliable population trend nor reliable trends for any states or physiographic regions, because of too few routes, too few birds, or high variability (Sauer et al. 2000). The lack of reliable data is most obvious in the Black Hills, where there are too few data to perform even the most basic trend analysis (Sauer et al. 2000).

Where long-term data are available for particular populations, natural fluctuations in population numbers have been documented. For example, a constant-effort nest-finding study in Arizona recorded a major population crash. On this site between 1991 and 1996, the number of nests found each year varied from 23 to 65 (mean = 50.2), whereas in the same site from 1997 to 1999, only two to five nests were found each year (Kingery and Ghalambor 2001). Likewise, Scott’s (1979) study also portrays a Pygmy nuthatch population swing, but no clear factor has been identified as being responsible for these rapid changes in population numbers. No definitive explanation currently exists for why some Pygmy nuthatch populations may be prone to large fluctuations, but it is suspected that an intolerance to cold winter temperatures and/or a poor cone crop may play a role.

3.6.9 Gray Flycatcher

General Habitat Requirements

Limiting Factors

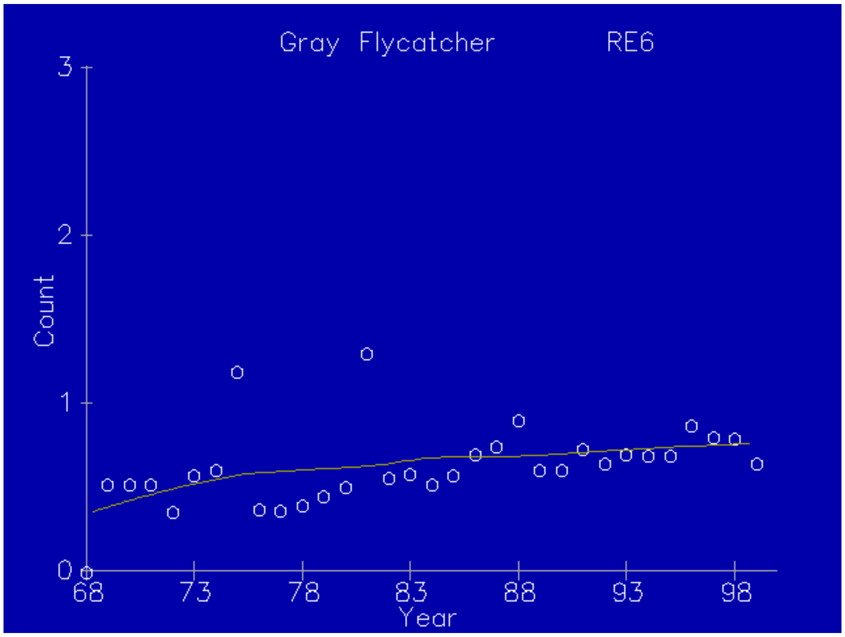
Gray flycatchers will be vulnerable to land clearing, but generally are found in very arid environments that are not usually converted to agriculture (USDA Forest Service 1994). Clearing of pinyon-juniper for mining of coal and oil shale deposits, or in favor of grassland for livestock grazing, or for widespread harvesting of pinyon-juniper could be detrimental (O'Meara et al. 1981, cited in Sterling 1999).

Current Distribution

Gray flycatchers are found in extreme southern British Columbia (Cannings 1992) and south-central Idaho, and south to southern California, southern Nevada, central Arizona, south-central New Mexico, and locally western Texas (Terres 1980, AOU 1983). Gray flycatchers during the non-breeding season occur in southern California, central Arizona, south to Baja California and south-central mainland of Mexico (Terres 1980).

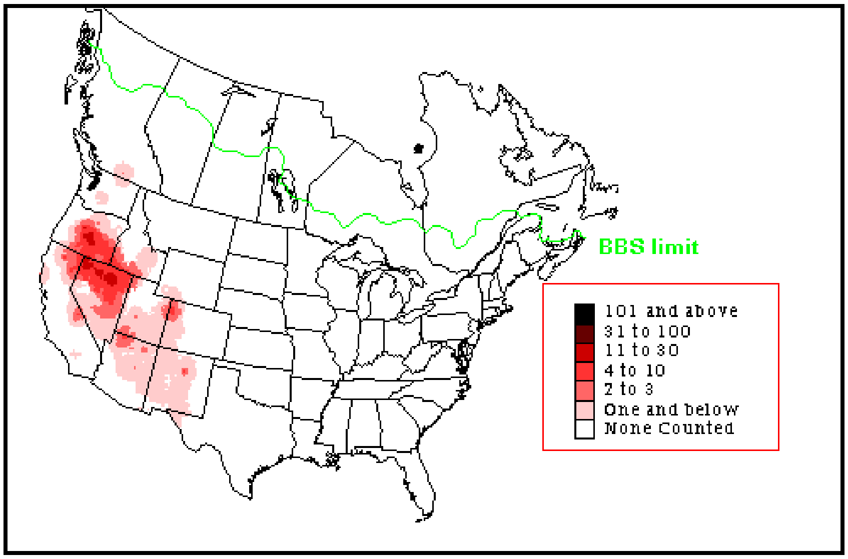
Population Trend Status

North American Breeding Bird Survey (BBS) shows a survey-wide significantly increasing trend of 10.2% average per year (n = 89) from 1966 to 1996, a nonsignificant decline of -1.0% average per year (n = 22) from 1966 to 1979, and a significant increase from 1980 to 1996 of 10.0% average per year (n = 84) (**Figure 32**). Data for Oregon reflect strong long-term increase of 7.9% average per year (n = 29) from 1966 to 1996. Sample sizes were too low for accurate trend estimates in other states (Sauer et al. 1997). Gray flycatcher breeding season abundance is illustrated in **Figure 33**.



Source: BBS data; Sauer et al. 1997

Figure 32 Gray flycatcher population trend data



(Sauer et al. 1997)

Figure 33 Gray flycatcher breeding season abundance (from BBS)

Christmas Bird Count (CBC) data for 1959 to 1988 show a significant survey-wide increase of 4.3% average per year, and a significant increase in Arizona (4.6% average per year, n = 28). Trend for California is apparently stable over the period (non-significant increase of 0.2% average per year, n = 21; Sauer et al. 1996).

The Gray flycatcher is reportedly declining as a wintering bird in southern California; extensions in Washington and California at western edges of breeding range were noted in the 1970s (USDA Forest Service 1994).

3.6.10 White-headed Woodpecker

General Habitat Requirements

White-headed woodpeckers prefer a conifer forest with a relatively open canopy (50 to 70% cover) and an availability of snags (i.e., a partially collapsed, dead tree) and stumps for nesting. The birds prefer to build nests in trees with large diameters, preference increasing with diameter. The understory vegetation is usually very sparse within the preferred habitat, and local populations are abundant in burned or cut forest where residual large-diameter live and dead trees are present. In general, open Ponderosa pine stands with canopy closures between 30% and 50% are preferred. The openness, however, is not as important as the presence of mature or veteran cone-producing pines within a stand (Milne and Hejl 1989).

The highest abundances of white-headed woodpeckers occur in old-growth stands, particularly ones with a mix of two or more pine species. The birds are uncommon or absent in monospecific Ponderosa pine forests and in stands dominated by small-coned or closed-cone conifers (e.g., lodgepole pine or knobcone pine).

Limiting Factors

Logging has removed, from throughout this species' range, much of the old growth cone-producing pines, which provide winter food and large snags for nesting. The impact from the decrease in old growth cone-producing pines is even more significant in areas where no alternate pine species exist for the white-headed woodpecker to utilize.

Fire suppression has altered the stand structure in many of the forests. Lack of fire has allowed dense stands of immature Ponderosa pine as well as the more shade tolerant Douglas-fir to establish. This has led to increased fuel loads, which has resulted in more severe stand-replacing fires where both the mature cone producing trees and the large suitable snags are destroyed. These dense stands of immature trees has also led to increased competition for nutrients as well as a slow change from a Ponderosa pine climax forest to a Douglas-fir dominated climax forest.

Predation does not appreciably affect the woodpecker population. Chipmunks are known to prey on the eggs and nestlings of white-headed woodpeckers. There is also limited predation by the great horned owl on adult white-headed woodpeckers.

Current Distribution

White-headed woodpeckers live in montane, coniferous forests from southern British Columbia in Canada, to eastern Washington, southern California and Nevada and northern Idaho in the United States).

Source: Sauer et al. 2003

Figure 34 Distribution of white-headed woodpeckers

Population Trend Status

The current distribution/year-round range of white-headed woodpeckers (Sauer et al. 2003) is shown in northern Idaho in the United States (**Figure 34**).

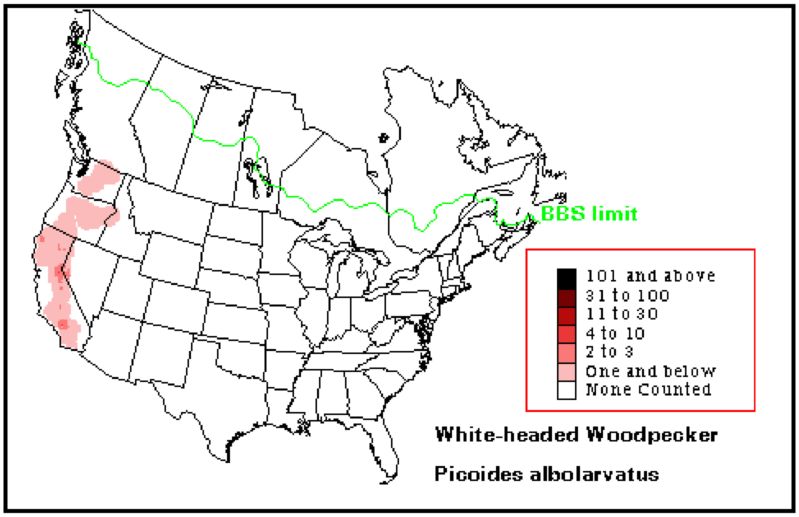
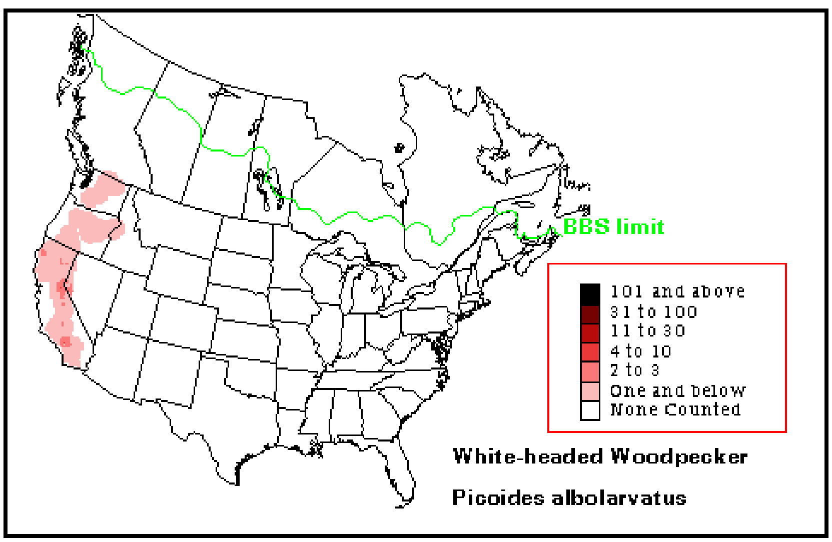


Figure 35 Distribution of white-headed woodpeckers

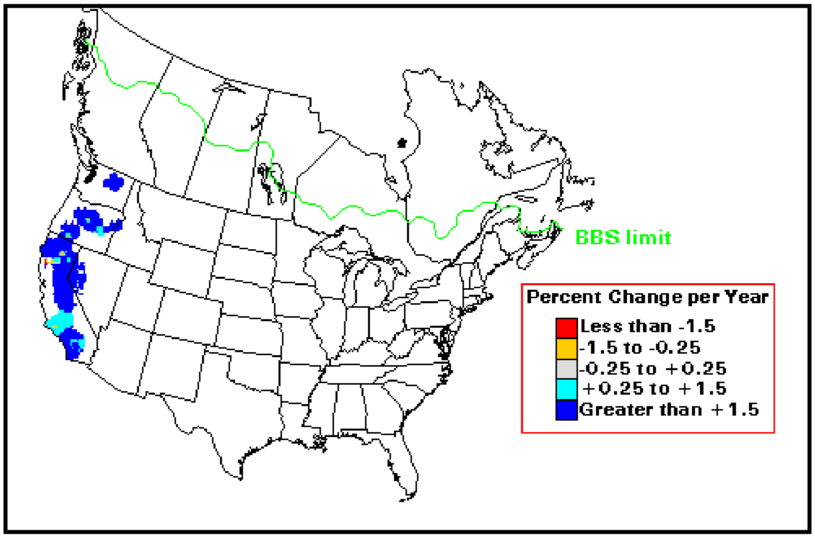
White-headed woodpecker abundance appears to decrease north of California. They are uncommon in Washington and Idaho, and rare in British Columbia; however, they are still common in most of their original range in the Sierra Nevada and mountains of southern California.

The species is of moderate conservation importance because of its relatively small and patchy year-round range (**Figure 36**) and its dependence on mature, montane coniferous forests in the West. Knowledge of this woodpecker's tolerance of forest fragmentation and silvicultural practices will be important in conserving future populations. BBS population trend data are illustrated in **Figure 37**.



Source: Sauer et al. 2003

Figure 36 Current distribution/year-round range of white-headed woodpeckers



Source: Sauer et al. 2003

Figure 37 White-headed woodpecker BBS population trend: 1966-1996

Structural Condition Associations

Structural conditions (IBIS 2003) associated with white-headed woodpeckers are summarized in **Table 32**. White-headed woodpeckers feed and reproduce (F/R) in, and are generally associated (A) with a multitude of structural conditions within the Ponderosa pine habitat type. Similarly, white-headed woodpeckers are present (P), but not dependent upon sapling/pole successional forests. According to IBIS (2003) data, white-headed woodpeckers are not closely associated (C) with any specific Ponderosa pine structural conditions.

Table 32 White-headed woodpecker structural conditions and association relationships (IBIS 2003)

Common Name	Focal Habitat	Structural Condition (SC)	SC Activity	SC Assoc.
White-headed Woodpecker	Ponderosa Pine	Giant Tree-Multi-Story	F/R-HE	A
		Grass/Forb-Closed	F/R-HE	A
		Grass/Forb-Open	F/R-HE	A
		Large Tree-Multi-Story-Closed	F/R-HE	A
		Large Tree-Multi-Story-Moderate	F/R-HE	A
		Large Tree-Multi-Story-Open	F/R-HE	A
		Large Tree-Single Story-Closed	F/R-HE	A
		Large Tree-Single Story-Moderate	F/R-HE	A
		Large Tree-Single Story-Open	F/R-HE	A
		Medium Tree-Multi-Story-Closed	F/R-HE	A
		Medium Tree-Multi-Story-Moderate	F/R-HE	A
		Medium Tree-Multi-Story-Open	F/R-HE	A

Common Name	Focal Habitat	Structural Condition (SC)	SC Activity	SC Assoc.
		Medium Tree-Single Story-Closed	F/R-HE	A
		Medium Tree-Single Story-Moderate	F/R-HE	A
		Medium Tree-Single Story-Open	F/R-HE	A
		Sapling/Pole-Closed	F/R-HE	P
		Sapling/Pole-Moderate	F/R-HE	P
		Sapling/Pole-Open	F/R-HE	P
		Shrub/Seedling-Closed	F/R-HE	A
		Shrub/Seedling-Open	F/R-HE	A
		Small Tree-Multi-Story-Closed	F/R-HE	A
		Small Tree-Multi-Story-Moderate	F/R-HE	A
		Small Tree-Multi-Story-Open	F/R-HE	A
		Small Tree-Single Story-Closed	F/R-HE	A
		Small Tree-Single Story-Moderate	F/R-HE	A
		Small Tree-Single Story-Open	F/R-HE	A

3.6.11 Flammulated Owl

General Habitat Requirements

The flammulated owl is a Washington State candidate species. Limited research on the flammulated owl indicates that its demography and life history, coupled with narrow habitat requirements, make it vulnerable to habitat changes. The flammulated owl occurs mostly in mid-level conifer forests that have a significant Ponderosa pine component (McCallum 1994b) between elevations of 1,200 to 5,500 feet in the north, and up to 9,000 feet in the southern part of its range in California (Winter 1974).

Flammulated owls are typically found in mature to old, open canopy yellow pine (Ponderosa pine and Jeffrey pine [*Pinus jeffreyi*]), Douglas-fir, and grand fir (Bull and Anderson 1978; Goggans 1986; Howie and Ritchie 1987; Reynolds and Linkhart 1992; Powers et al. 1996). It is a species dependent on large-diameter Ponderosa pine forests (Hillis et al. 2001) and are obligate, secondary cavity nesters (McCallum 1994b), requiring large snags in which to roost and nest.

Flammulated owls nest in habitat types with low to intermediate canopy closure (Zeiner et al. 1990). The owls selectively nest in dead Ponderosa pine snags, and prefer nest sites with fewer shrubs in front than behind the cavity entrance, possibly to avoid predation and obstacles to flight.

Limiting Factors

Logging disturbance and the loss of breeding habitat associated with it has a detrimental effect on the birds (USDA 1994a). The owls prefer late seral forests. The main threat to the species is the loss of nesting cavities as this species cannot create its own nest and relies on existing cavities. Management practices such as intensive forest management, forest stand improvement, and the felling of snags and injured or diseased trees (potential nest sites) for firewood effectively remove most of the cavities suitable for nesting (Reynolds et al. 1989). The owls will nest in selectively logged stands; however, as long as they contain residual trees (Reynolds et al. 1989).

Wildfire suppression has allowed many Ponderosa pine stands to proceed to the more shade-resistant fir forest types, that are is less suitable habitat for these species (Marshall 1957; Reynolds et al. 1989).

Roads and fuelbreaks, often placed on ridgetops, result in removal of snags for safety considerations (hazard tree removal); as well, removal of firewood can result in the loss of existing and recruitment nest trees.

Pesticides including aerial spraying of carbaryl insecticides to reduce populations of forest insect pests may affect the abundance of non-target insects important in the early spring diets of flammulated owls (Reynolds et al. 1989). Although flammulated owls rarely take rodents as prey, they could be at risk, like other raptors, of secondary poisoning by anticoagulant rodenticides. Possible harmful doses could cause hemmorhaging upon the ingestion of anticoagulants such as Difenacoum, Bromadiolone, or Brodifacoum (Mendenhall and Pank 1980).

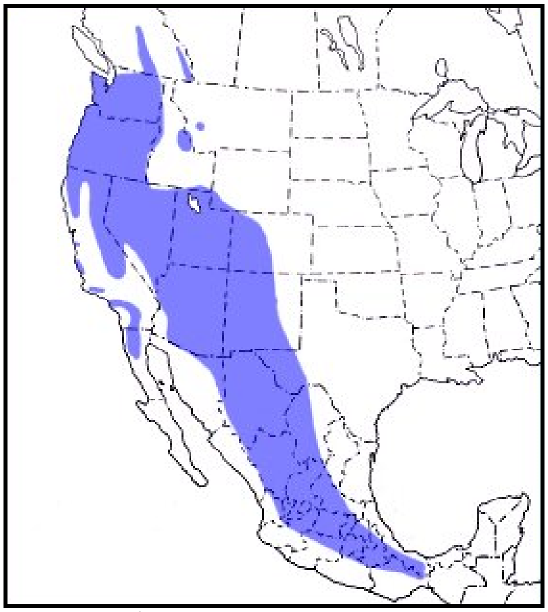
Predators/competitors include spotted owl and other larger owls, accipiters, long-tailed weasels (Zeiner et al. 1990), felids, and bears (McCallum 1994b). Nest predation has also been documented by the northern flying squirrel in the Pacific Northwest (McCallum 1994a). Saw-whet owls, screech owls, and American kestrels compete for nesting sites, but flammulated owls probably have more severe competition for nest cavities with non-raptors, such as woodpeckers, other passerines, and squirrels (Zeiner et al. 1990, McCallum 1994b).

Birds, from the size of bluebirds upward, are potential competitors. Owl nests containing bluebird eggs and flicker eggs suggest that flammulated owls evict some potential nest competitors (McCallum 1994b). Any management plan that supports pileated woodpecker and northern flicker populations will help maintain high numbers of cavities, thereby minimizing this competition (Zeiner et al. 1990). Flammulated owls may compete with western screech-owls and American kestrels for prey (Zeiner et al. 1990) as both species have a high insect component in their diets. Common poorwills, nighthawks, and bats may also compete for nocturnal insect prey especially in the early breeding season (April and May) when the diet of the owls is dominated by moths. (McCallum 1994b).

Exotic species impact flammulated owl populations. Flicker cavities are often co-opted by European starlings, reducing the availability of nest cavities for both flickers and owls (McCallum 1994a). Africanized honeybees will nest in tree cavities (Merrill and Visscher 1995), and may be a competitor where natural cavities are limiting, particularly in southern California where the bee has expanded its range north of Mexico.

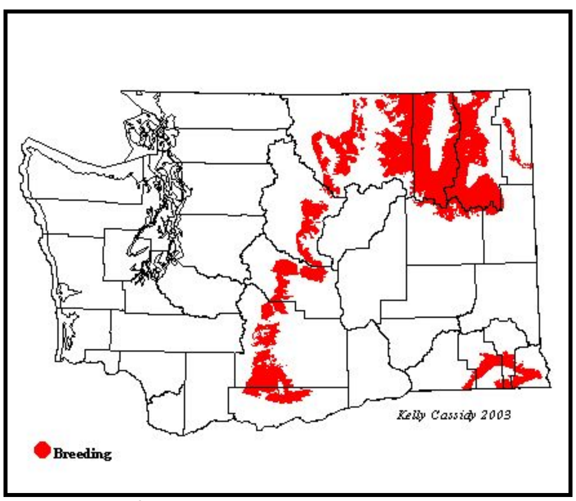
Current Distribution

Flammulated owl distribution is illustrated in **Figure 38** and **Figure 39**. Flammulated owls are uncommon breeders east of the Cascades in the Ponderosa pine belt from late May to August. There have been occasional records from western Washington, but they are essentially an east-side species. Locations where they may sometimes be found include Blewett Pass (straddling Chelan and Kittitas Counties), Colockum Pass area (Kittitas County), and Satus Pass (Klickitat County).



Source: Kaufman 1996

Figure 38 Flammulated owl distribution, North America



Source: Kaufman 1986

Figure 39 Flammulated owl distribution, Washington

Population Trend Status

Because old-growth Ponderosa pine is more rare in the northern Rocky Mountains than it was historically, and little is known about local flammulated owl distribution and habitat use, the USFS has listed the flammulated owl as a Sensitive species in the Northern Region (USDA 1994b). It is also listed as a Sensitive species by the USFS in the Rocky Mountain, Southwestern, and Intermountain Regions, and receives special management consideration in the states of Montana, Idaho, Oregon, and Washington (Verner 1994).

So little is known about flammulated owl populations that even large-scale changes in their abundance would probably go unnoticed (Winter 1974). Several studies have noted a decline in flammulated owl populations following timber harvesting (Marshall 1939; Howle and Ritcey 1987); however, more and more nest sightings occur each year, most likely due, however, to the increase in observation efforts.

Structural Condition Associations

Structural conditions (IBIS 2003) associated with flammulated owl are summarized in **Table 33**.

Table 33 Structural conditions associated with flammulated owls

Common Name	Focal Habitat	Structural Condition (SC)	SC Activity	SC Assoc.
Flammulated Owl	Ponderosa Pine	Giant Tree-Multi-Story	F/R-HE	C
		Large Tree-Multi-Story-Closed	F/R-HE	C
		Large Tree-Multi-Story-Moderate	F/R-HE	C
		Large Tree-Multi-Story-Open	F/R-HE	A
		Large Tree-Single Story-Closed	F/R-HE	P
		Large Tree-Single Story-Moderate	F/R-HE	P
		Medium Tree-Multi-Story-Closed	F/R-HE	C
		Medium Tree-Multi-Story-Moderate	F/R-HE	C
		Medium Tree-Multi-Story-Open	F/R-HE	A
		Medium Tree-Single Story-Closed	F/R-HE	P
		Medium Tree-Single Story-Moderate	F/R-HE	P
		Small Tree-Multi-Story-Closed	F/R-HE	A
		Small Tree-Multi-Story-Moderate	F/R-HE	A
		Small Tree-Multi-Story-Open	F/R-HE	P

Flammulated owls feed and reproduce (F/R) in and are closely associated (C) with medium to large, multi-story, moderate to closed canopy Ponderosa pine forest conditions. Similarly, flammulated owls are associated (A) with medium to large multi-story/open canopy forest, and will utilize dense stands of small trees. In contrast, flammulated owls are present (P), but not dependent upon open canopy forest (IBIS 2003). Of the three Ponderosa pine focal species, flammulated owls are the most structural-dependent species.

3.7 Wildlife Focal Habitats and Focal Species

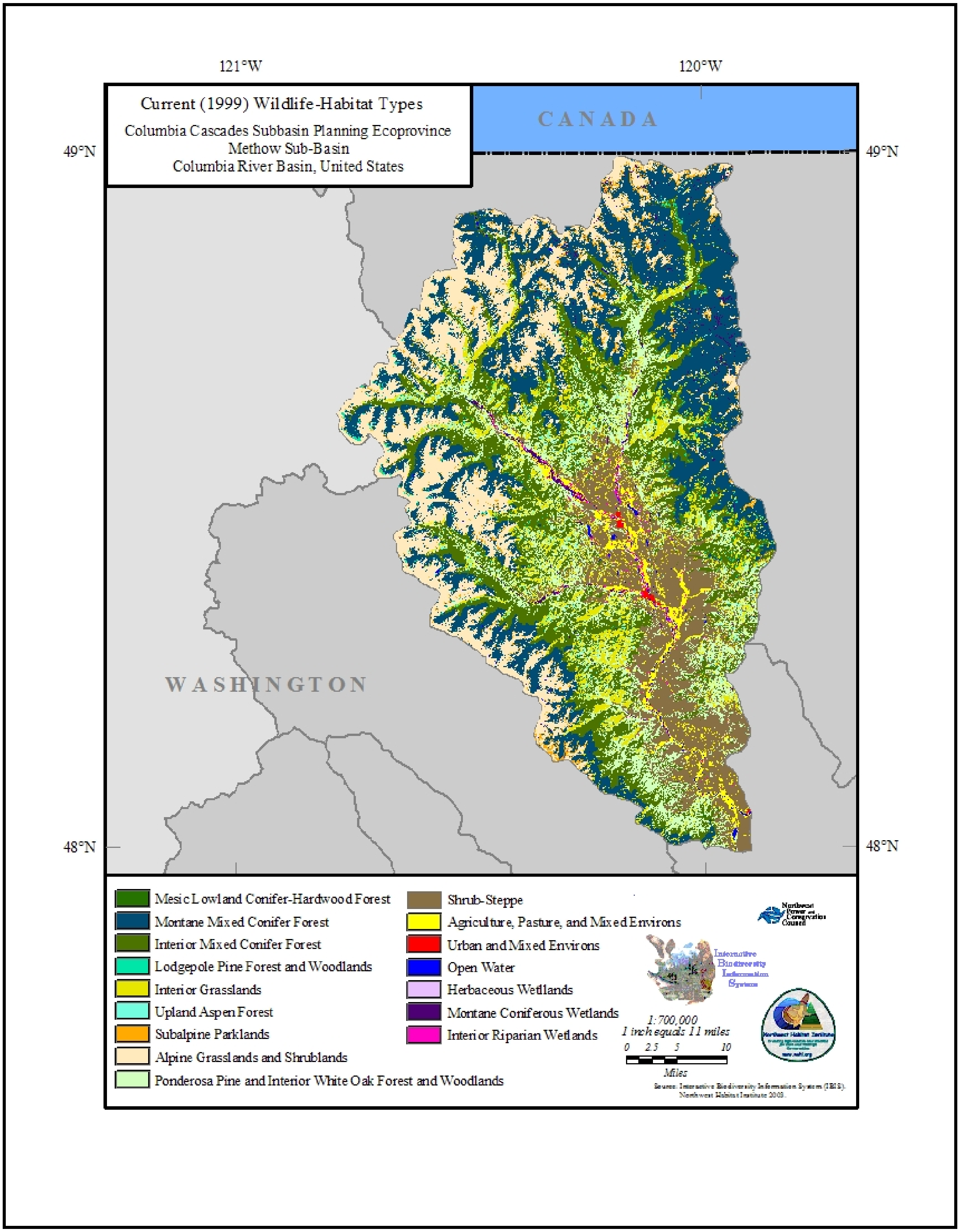
Since wildlife distribution is related more to habitat type than stream or creek reach, the following discussion of wildlife habitat is presented in terms of habitat assemblages rather than the subwatershed format used to describe fish habitat. Management across aquatic and terrestrial ecosystems, therefore, must explore the connection through habitat relationships and focal species relationships.

Ecoregion/subbasin planners assumed that by focusing resources primarily on selected habitat types, the needs of most listed and managed terrestrial species, dependent on those habitats, would be addressed during this planning period. While other listed and managed species occur within the subbasin (primarily forested habitat obligates), needs of those species are addressed primarily through the existing land management frameworks of the federal agencies within whose jurisdictions the overwhelming majority of forested habitats occur within the Okanogan subbasin (Okanogan/Wenatchee National Forest and Washington Department of Natural Resources).

Ecoprovince/subbasin planners then identified an assemblage of focal species for each focal habitat type. The focal species that compose the assemblage for each focal habitat type will serve as indicators of environmental health for species that use that habitat type. The planners combined life requisite habitat attributes for each species assemblage to form a recommended range of management conditions, that, when achieved, should result in functional habitats. The rationale for using focal species assemblages is to draw immediate attention to habitat features and conditions most in need of conservation or most important in a functioning ecosystem. The corollary is that factors that affect habitat quality and integrity, within the Ecoregion and subbasins, also impact wildlife species. As a result, identifying and addressing limiting factors that affect focal habitats should support the needs of obligate wildlife populations as well.

3.8 Wildlife Focal Habitats

The subbasin consists of 15 wildlife habitat types, which are illustrated in **Figure 40**. Detailed descriptions of these habitat types can be found in Appendix B of Ashley and Stovall (unpublished report, 2004). A comparison of the amount of current focal habitat types for each subbasin in the Columbia Cascade Ecoprovince is summarized in **Table 34** Additional information, including information about habitat requirements, limiting factors, distribution, and population trends, which will be useful to recovery project planners, is included in Ashley and Stovall (unpublished report, 2004).



Source: IBIS 2003

Figure 40 Habitat types in the Methow subbasin

Table 34 A comparison of the amount of current focal habitat types for each subbasin in the Columbia Cascade Ecoprovince, Washington (IBIS 2003)

Subbasin	Focal Habitat		
	Ponderosa Pine (acres)	Shrubsteppe (acres)	Riparian Wetlands (acres)
Entiat	55,807	32,986	94
Lake Chelan	45,480	45,018	5,079
Wenatchee	51,912	24,248	141
Methow	139,853	107,655	4,232
Okanogan	140,738	562,763	9,920
Upper Middle Mainstem Columbia River	50,843	753,073	3,898
Crab	4,660	991,397	12,227

Focal habitats selected for the subbasin include Ponderosa pine, shrubsteppe, and riparian wetlands. The planners also identified rugged lands as a habitat of concern. Neither the IBIS nor the Washington GAP Analysis data recognize the historic presence of riparian wetlands in the Methow subbasin.

The current extent of riparian wetlands, as reflected in these databases, is suspect at best; however, this habitat is a high priority habitat wherever it is found in the Ecoprovince. Agriculture, a habitat of concern, is not included or reported as a focal habitat type (but reflected in Appendix A).

Focal species and their association with focal habitat types are summarized in **Table 35**. The focal species will be used in other planning efforts in the subbasin and the Ecoregion, including Ecoregional Planning and Priority Habitat and Species planning.

Table 35 Wildlife Focal Species occurrence by habitat type in the Methow subbasin, Washington (IBIS 2003)

Common Name	Focal Habitat[1]	Status[2]		Native Species	PHS	Partners in Flight	Game Species
		Federal	State				
Sage thrasher	SS	n/a	C	Yes	Yes	Yes	No
*Brewer's sparrow		n/a	n/a	Yes	No	Yes	No
*Grasshopper sparrow		n/a	n/a	Yes	No	Yes	No
*Sharp-tailed grouse		SC	T	Yes	Yes	Yes	No
Sage grouse		C	T	Yes	Yes	No	No
Pygmy rabbit		E	E	Yes	Yes	No	No
*Mule deer		n/a	n/a	Yes	Yes	No	Yes
Willow flycatcher	RW	SC	n/a	Yes	No	Yes	No

Common Name	Focal Habitat[1]	Status[2]		Native Species	PHS	Partners in Flight	Game Species
		Federal	State				
Lewis woodpecker		n/a	C	Yes	Yes	Yes	No
*Red-eyed vireo		n/a	n/a	Yes	No	No	No
*Yellow-breasted chat		n/a	n/a	Yes	No	No	No
*American beaver		n/a	n/a	Yes	No	No	Yes
*Pygmy nuthatch	PP	n/a	n/a	Yes	No	No	No
*Gray flycatcher		n/a	n/a	Yes	No	No	No
*White-headed woodpecker		n/a	C	Yes	Yes	Yes	No
*Flammulated owl		n/a	C	Yes	Yes	Yes	No
Red-winged blackbird	HW	n/a	n/a	Yes	No	No	No

[1] SS = Shrubsteppe; RW = Riparian Wetlands; PP = Ponderosa pine; HW = Herbaceous Wetlands
[2] C = Candidate; SC = Species of Concern; T = Threatened; E = Endangered

* Identifies a focal species

3.9 Wildlife Focal Habitat Summaries

Focal wildlife habitat types are fully described in Ashley and Stovall (unpublished report, 2004). Only subbasin-specific focal habitat type anomalies and differences are described in this section.

3.9.1 Ponderosa Pine

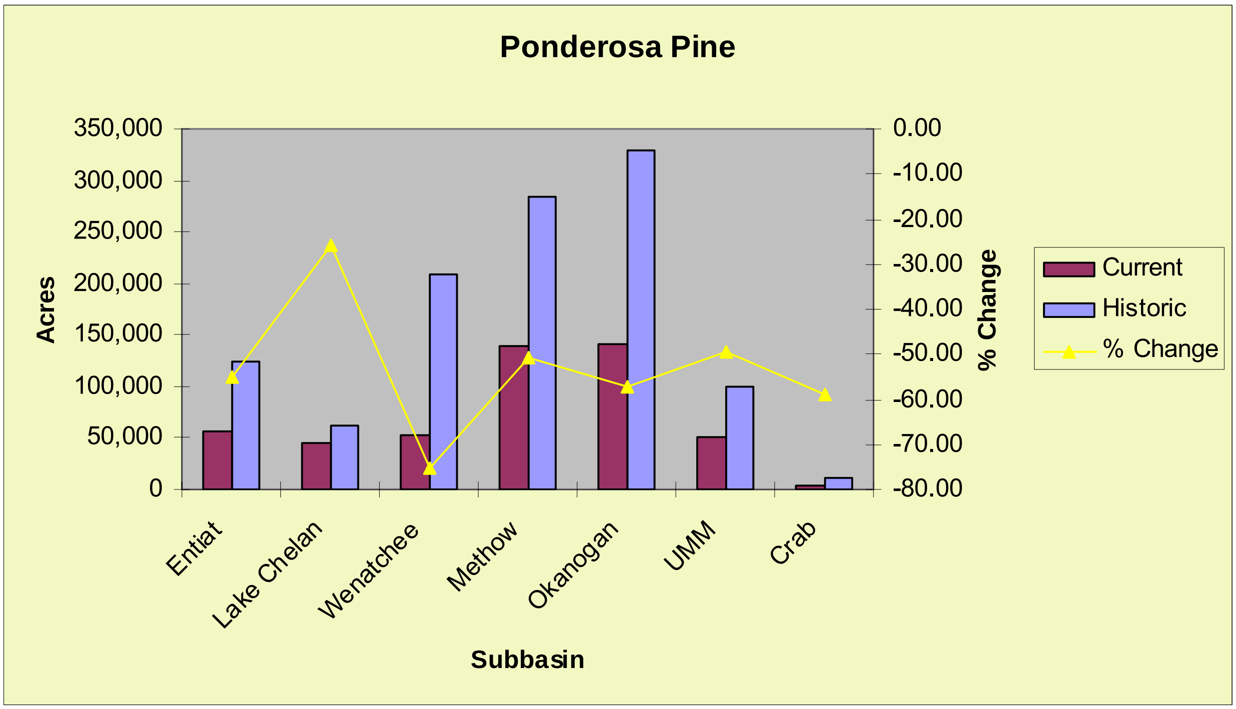
The Ponderosa pine habitat type is described in Ashley and Stovall (unpublished report, 2004). Historically in the subbasin, old-growth Ponderosa pine forests occupied large areas between the shrubsteppe zone and moister forest types at higher elevations. Large, widely spaced, fire-resistant trees and an understory of forbs, grasses, and shrubs characterized these forests. Periodic fires maintained this habitat type. With the settlement of the subbasin, most of the old pines were harvested for timber, and frequent fires have been suppressed. As a result, much of the original forest has been replaced by dense second growth of Douglas-fir and Ponderosa pine with little understory.

Extant Ponderosa pine habitat within the subbasin currently covers a wide range of seral conditions. Forest management and fire suppression have led to the replacement of old growth Ponderosa pine forests by younger forests with a greater proportion of Douglas-fir.

Currently, much of this habitat has a younger tree cohort of more shade-tolerant species that give the habitat a more closed, multi-layered canopy. For example, this habitat includes previously natural fire-maintained stands in which grand fir can eventually become the canopy dominant. Large late-seral Ponderosa pine and Douglas-fir are harvested in much of this habitat type. Under most management regimes, typical tree size decreases and tree density increases. In some areas, patchy tree establishment at forest-steppe ecotones has created new woodlands.

Introduced annuals, especially cheatgrass and invading shrubs under heavy grazing pressure, have replaced native herbaceous understory species. Four exotic knapweed species (*Centaurea* spp.) are spreading rapidly through the Ponderosa pine zone, and threatening to replace cheatgrass as the dominant increaser after grazing (Roche and Roche 1988). Dense cheatgrass stands eventually change the fire regime of these stands, often resulting in stand replacing, catastrophic fires. Bark beetles, primarily of the genus *Dendroctonus* and *Ips*, kill thousands of pines annually, and are the major mortality factor in commercial saw timber stands.

Current and historic acreages and percent change for the Ponderosa pine habitat type in the CCP are compared by subbasin in **Figure 41**. All subbasins in the Ecoprovince experienced a significant loss (25 to 75%) of Ponderosa pine habitat from historic (circa 1850) amounts (IBIS 2003).

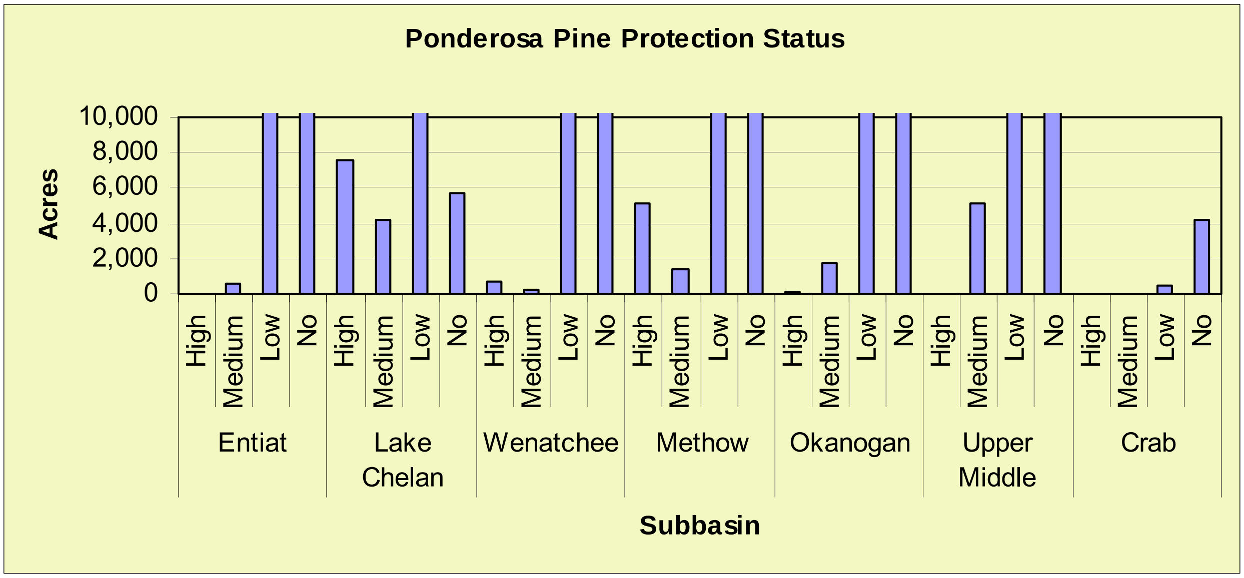


Source: IBIC 2003

Figure 41 A comparison of the Ponderosa pine habitat type in Ecoprovince subbasins

Protection Status

The protection status of Ponderosa pine habitat for the CCP subbasins is compared in **Figure 42**. The protection status of remaining Ponderosa pine habitat, in all subbasins, fall primarily within the “low” to “no protection” status categories. As a result, this habitat type will likely suffer further degradation, disturbance, and/or loss in all Ecoprovince subbasins. Protection status of Ponderosa pine habitat within the Methow subbasin is illustrated in **Table 36**.



Source: IBIS 2003

Figure 42 Protection status of Ponderosa pine in the Columbia Cascade Ecoprovince, Washington

Table 36 Ponderosa pine habitat GAP protection status in the Methow subbasin, Washington (IBIS 2003)

GAP Protection Status	Acres
High Protection	5,151
Medium Protection	1,381
Low Protection	119,451
No Protection	13,851

Factors Affecting Ponderosa Pine Habitat

Factors affecting Ponderosa pine habitat are explained in detail in Ashley and Stovall (unpublished report, 2004) and are summarized below:

- Timber harvesting, particularly at low elevations, has reduced the amount of old growth forest and associated large diameter trees and snags.
- Urban and residential development has contributed to loss and degradation of properly functioning ecosystems.
- Fire suppression/exclusion has contributed towards habitat degradation, particularly declines in characteristic herbaceous and shrub understory from increased density of small shade-tolerant trees. High risk of loss of remaining Ponderosa pine overstories from stand-replacing fires because of high fuel loads in densely stocked understories.
- In those minimal instances where overgrazing has occurred, this has resulted in lack of recruitment of sapling trees, particularly pines.
- Invasion of exotic plants has altered understory conditions and increased fuel loads.

- Fragmentation of remaining tracts has negatively impacted species with large area requirements.
- The timing (spring/summer versus fall) of restoration/silviculture practices such as mowing, thinning, and burning of understory removal may be especially detrimental to single-clutch species.

Recommended Future Condition

Recommended future conditions are described in Ashley and Stovall (unpublished report, 2004). Recommended conditions for Ponderosa pine habitat are summarized below:

- Condition 1a – mature Ponderosa pine forest: The white-headed woodpecker represents species that require/prefer large patches (greater than 350 acres) of open mature/old growth Ponderosa pine stands with canopy closures between 10 and 50%, and snags and stumps for nesting (nesting stumps and snags grater than 31 inches DBH). Abundant white-headed woodpecker populations can be present on burned or cut forest with residual large-diameter live and dead trees and understory vegetation that is usually very sparse. Openness, however, is not as important as the presence of mature or veteran cone-producing pines within a stand (Milne and Hejl 1989).
- Condition 1b – mature Ponderosa pine forest: The Pygmy nuthatch represents species that require heterogeneous stands of Ponderosa pine with a mixture of well-spaced, old pines and vigorous trees of intermediate age, and those species that depend on snags for nesting and roosting, high canopy density, and large-diameter (greater than 18 inches DBH) trees characteristic of mature undisturbed forests. Connectivity between suitable habitats is important for species, such as the Pygmy nuthatch, whose movement and dispersal patterns are limited to their natal territories.
- Condition 2 – multiple-canopy Ponderosa pine mosaic: Flammulated owls represent wildlife species that occupy Ponderosa pine sites that are comprised of multiple-canopy, mature Ponderosa pine stands or mixed Ponderosa pine/Douglas-fir forest interspersed with grassy openings and dense thickets. Flammulated owls nest in habitat types with low to intermediate canopy closure (Zeiner et al. 1990), two-layered canopies, tree density of 508 trees/acre (9-foot spacing), basal area of 250 feet^2/acre (McCallum 1994), and snags greater than 20 inches DBH, and three to 39 feet tall (Zeiner et al. 1990). Food requirements are met by the presence of at least one snag greater than 12 inches DBH/10 acres and eight trees/acre greater than 21 inches DBH.
- Condition 3 – Pine/shrubsteppe interface: Gray flycatchers represent wildlife species that occupy the pine/shrubsteppe interface (pine savannah) with a shrub/bunchgrass understory. Gray flycatchers require nest trees 18 inches DBH and a tree height of 52 feet for their reproductive life requisites.

3.9.2 Shrubsteppe

The shrubsteppe habitat type is described in Ashley and Stovall (unpublished report, 2004). Historically, sage-dominated steppe vegetation occurred throughout the majority of the lower elevations in the subbasin, and variations of shrubsteppe habitat once occupied most of the non-forested land in eastern Washington. The moister draws and permanent stream courses, imbedded in the shrubsteppe landscape, supported strands of riparian vegetation dominated by moisture-loving shrubs and small trees, including thick stands of water birch, a major component of the winter diet of sharp-tailed grouse. The drastic reduction of water birch in the subbasin by early settlers is likely a major factor in the decline of sharp-tailed grouse (NPPC 2002).

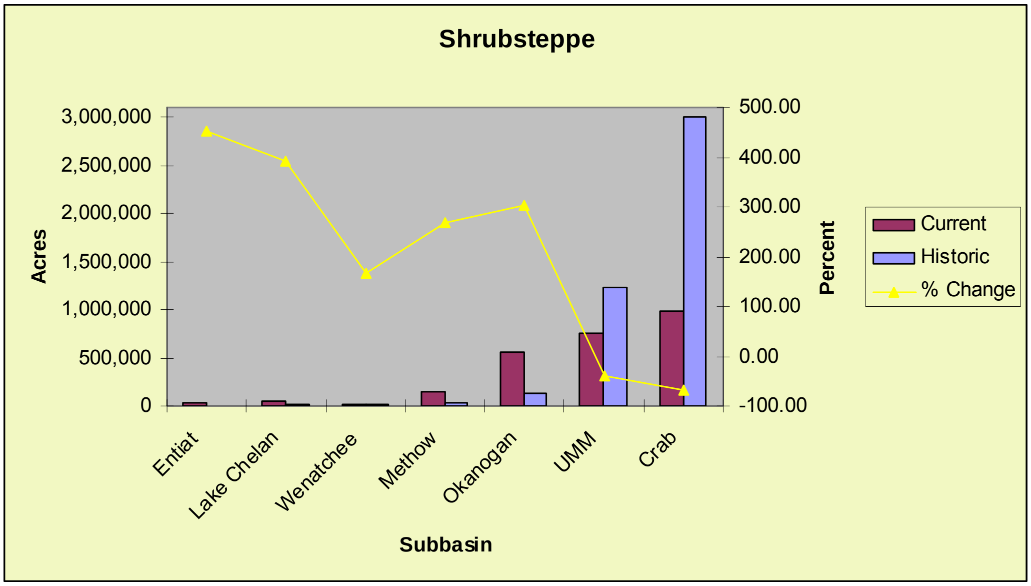
The greatest changes in shrubsteppe habitat from historic conditions are the reduction of bunchgrass cover in the understory and an increase in sagebrush cover. Soil compaction is also a significant factor in heavily grazed lands, affecting water percolation, runoff and soil nutrient content. A long history of grazing, fire, and invasion by exotic vegetation has altered the composition of the plant community within much of the extant shrubsteppe in this region (Quigley and Arbelbide 1997; Knick 1999), and it is difficult to find stands which are still in relatively natural condition.

Fire has relatively little effect on native vegetation in the three-tip sagebrush zone, since three-tip sagebrush and the dominant graminoids resprout after burning. Three-tip sagebrush does not appear to be much affected by grazing, but the perennial graminoids decrease and are eventually replaced by cheatgrass (*Bromus tectorum*), plantain (*Plantago* spp.), big bluegrass (*Poa secunda*), and/or gray rabbitbrush (*Chrysothamnus nauseosus*). In recent years, diffuse knapweed (*Centaurea diffusa*) has spread through this zone, and threatens to replace other exotics as the chief increaser after grazing (Roche and Roche 1998).

In areas of central arid steppe, with a history of heavy grazing and fire suppression, true shrublands are common and may even be the predominant cover on non-agricultural land. Most of the native grasses and forbs are poorly adapted to heavy grazing and trampling by livestock. Grazing eventually leads to replacement of the bunchgrasses with cheatgrass, Nuttall’s fescue (*Festuca microstachys*), eight-flowered fescue (*F. octofiora*), and Indian wheat (*Plantago patagonica*) (Harris and Chaney 1984). In recent years, several knapweeds (*Centaurea* spp.), have become increasingly widespread. Russian star thistle (*Centaurea repens*) is particularly widespread, especially along and near major watercourses (Roche and Roche 1988 in Cassidy 1997).

Sizable areas of healthy shrubsteppe still remain. These areas occur primarily on public lands and the few remaining large private ranches in the Methow valley. Much of the deeper soil shrubsteppe habitat on flat bench lands has been converted to agriculture or developed as home sites. As agriculture increasingly gives way to subdivision and housing developments in the valley, private land parcels containing healthy shrubsteppe habitat may be lost (NPPC 2002). Currently, the largest block of undeveloped shrubsteppe in private ownership is located north of Twisp just south of WDFW land in the vicinity of the last known active sharp-tailed grouse lek in the subbasin.

Current and historic acreages and percent change for the shrubsteppe habitat type are compared by subbasin in **Figure 43**. The Upper Middle Mainstem Columbia River and Crab subbasins have experienced considerable losses (39% and 67%, respectively), while the remaining subbasins show increases in shrubsteppe habitat ranging from 165 to 462% over historic (circa 1850) amounts (IBIS 2003).

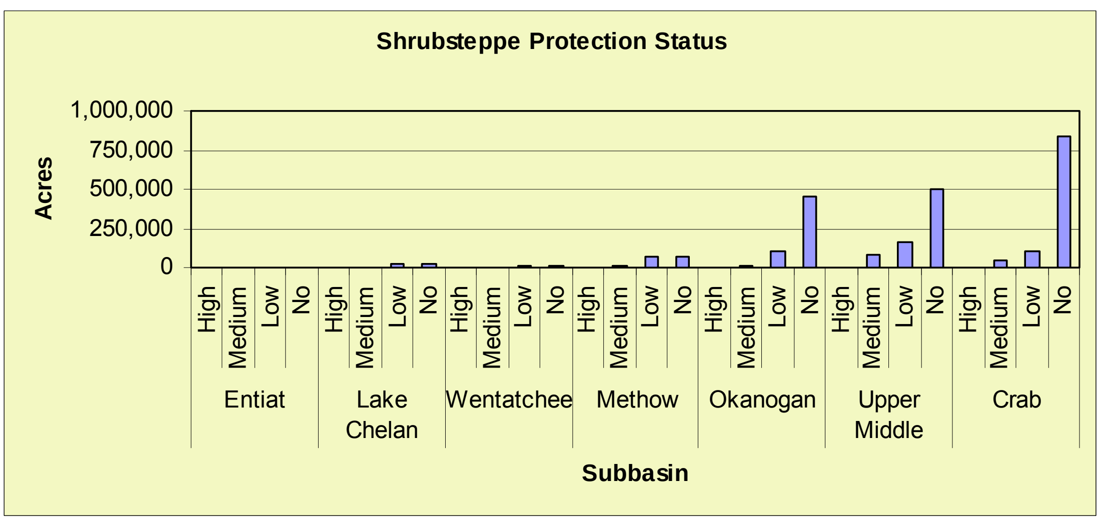


Source: IBIS 2003

Figure 43 A comparison of the shrubsteppe habitat type in Ecoprovince subbasins

Protection Status

The protection status of shrubsteppe habitat for Ecoprovince subbasins is compared in Figure 44. The protection status of remaining shrubsteppe habitats, in all subbasins, fall primarily within the "low" to "no' protection status categories. As a result, this habitat type will likely suffer further degradation, disturbance, and/or loss in all Ecoprovince subbasins. Protection status of shrubsteppe habitat within the Methow subbasin is summarized in **Table 37**.



Source: IBIS 2003

Figure 44 GAP protection status of shrubsteppe habitat in the Columbia Cascade Ecoprovince, Washington

Table 37 Shrubsteppe habitat GAP protection status in the Methow subbasin, Washington (IBIS 2003)

GAP Protection Status	Acres
High Protection	42
Medium Protection	8,274
Low Protection	65,670
No Protection	73,647

Factors Affecting Shrubsteppe Habitat

Factors affecting shrubsteppe habitat are explained in detail in Ashley and Stovall (unpublished report, 2004) and are summarized below:

- Permanent habitat conversions of shrubsteppe/grassland habitats (e.g., approximately 60% of shrubsteppe in Washington [Dobler et al. 1996]) to other uses (e.g., agriculture, urbanization).
- Fragmentation of remaining tracts of moderate to good quality shrubsteppe habitat.
- Degradation of habitat from intensive grazing and invasion of exotic plant species, particularly of annual grasses, such as cheatgrass, and woody vegetation, such as Russian olive.
- Degradation and loss of properly functioning shrubsteppe/grassland ecosystems resulting from the encroachment of urban and residential development and conversion to agriculture. Most of the remaining shrubsteppe in Washington is in private ownership (57%).
- Loss of big sagebrush communities to brush control (may not be detrimental relative to interior grassland habitats).
- Conversion of CRP lands back to cropland.
- Loss and reduction of cryptogamic crusts, which help maintain the ecological integrity of shrubsteppe/grassland communities.
- Fire management, either suppression or over-use.
- Invasion and seeding of crested wheatgrass and other introduced plant species that reduces wildlife habitat quality and/or availability.

Recommended Future Condition

Recommended future conditions are described in Ashley and Stovall (unpublished report, 2004). Recommended conditions for shrubsteppe habitat are summarized as follows:

3.9.3 Sagebrush-dominated Shrubsteppe

Condition 1 – Diverse shrubsteppe habitat: Mule deer were selected to represent species that require and prefer diverse, dense (30 to 60% shrub cover less than five feet tall) shrubsteppe habitats (Ashley and Berger 1999) comprised of bitterbrush, big sagebrush, rabbitbrush, and other shrub species (Leckenby 1969; Kufeld et al. 1973; Sheehy 1975; Jackson 1990), with a palatable herbaceous understory exceeding 30% cover (Ashley and Berger 1999).

Condition 2 – Sagebrush dominated shrubsteppe habitat: Brewer's sparrow was selected to represent wildlife species that require sagebrush-dominated sites. Brewer's sparrow prefers a patchy distribution of sagebrush clumps, 10-30% cover (Altman and Holmes 2000), lower sagebrush height (between 20 and 28 inches) (Wiens and Rotenberry 1981), 10 to 20% native grass cover (Dobler 1994), less than 10% non-native herbaceous cover, and bare ground greater than 20% (Altman and Holmes 2000). It should be noted, however, that Johnsgard and Rickard (1957) reported that shrublands comprised of snowberry, hawthorne, chokecherry, serviceberry, bitterbrush, and rabbitbrush were also used by Brewer's sparrows for nesting in southeast Washington. Specific, quantifiable habitat attribute information for this mixed shrub landscape could not be found.

3.9.4 Steppe/Grassland-dominated Shrubsteppe:

Condition 1 – Shrubsteppe habitat with multi-structured deciduous trees and shrubs: Sharp-tailed grouse was selected to represent species that require multi-structured fruit/bud/catkin-producing deciduous trees and shrubs dispersed throughout the landscape (10 to 40% of the total area). Other habitat conditions include:

- Native bunchgrass greater than 40% cover
- Native forbs at least 30% cover
- Visual obstruction readings (VOR) of at least 6 inches
- At least 75% cover deciduous shrubs and trees
- Exotic vegetation/noxious weeds less than 5% cover

Condition 2 – Shrubsteppe habitat with native bunch grasses: Grasshopper sparrow was selected to represent species that require healthy steppe habitat dominated by native bunch grasses. Grasshopper sparrow require native bunchgrass cover greater than 15% and comprising greater than 60% of the total grass cover.

3.9.5 Eastside (Interior) Riparian Wetlands

The eastside (interior) riparian wetlands habitat type refers only to riverine and adjacent wetland habitats in both the Ecoprovince and individual subbasins. Historic (circa 1850) and, to a lesser degree, current data concerning the extent and distribution of riparian wetland habitat are a significant data gap at both the Ecoprovince and subbasin level.

The lack of data for this habitat type is a major challenge as Ecoprovince and subbasin planners attempt to quantify habitat changes from historic conditions, and develop strategies that address limiting factors and management goals and objectives.

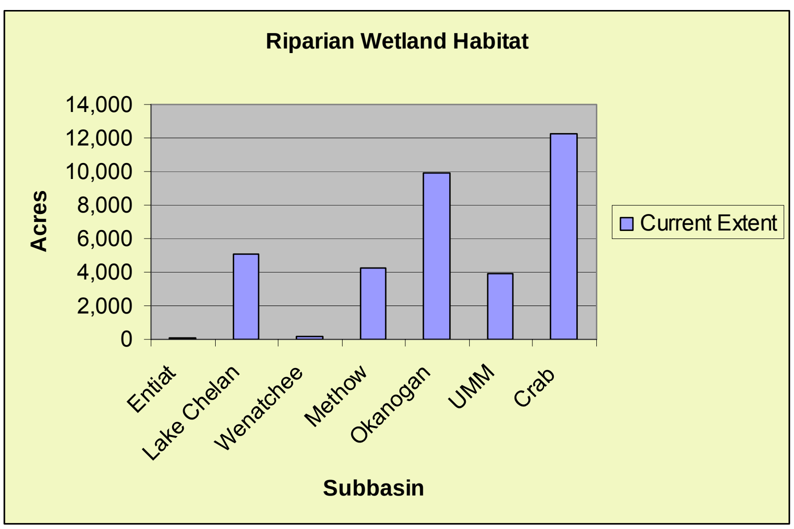
Because of the lack of historic riparian wetland data, the IBIS database cannot be relied upon for comparisons, in the Ecoprovince and individual subbasins, between the historic and current extent of riparian wetlands. According to the IBIS database (2003), there are an estimated 3,898 acres of riparian wetland habitat currently in the subbasin. Although there are no historic data, the actual number of acres or absolute magnitude of the change is less important than recognizing that the loss of riparian habitat and lack of permanent protection continues to place this habitat type at further risk.

Historically, riparian wetland habitat was characterized by a mosaic of plant communities occurring at irregular intervals along streams, and dominated singularly, or in some combination, by grass-forbs, shrub thickets, and mature forests with tall deciduous trees. Beaver activity and natural flooding are two ecological processes that affected the quality and distribution of riparian wetlands.

Today, agricultural conversion, altered stream channel morphology, and water withdrawal have played significant roles in changing the character of streams and associated riparian areas. The subbasin, however, is still host to some of eastern Washington's best remaining tracts of cottonwood gallery forests, found in the wide floodplain portions of the Methow Valley and its major tributaries.

Significant riparian habitat remains along the Methow River between Winthrop and Lost River. Additional stands are located along the Twisp and Chewuch rivers, and more fragmented pockets can be found along the Methow between Winthrop and Carlton. Large areas once dominated by cottonwoods, which contribute considerable structure to riparian habitats, are being lost. Because of its proximity to roads and other developed areas, much of the remaining riparian/floodplain habitat may be at risk of conversion to housing development.

The current extent of riparian wetland habitat throughout the Columbia Cascade Ecoprovince is illustrated in **Figure 45**.

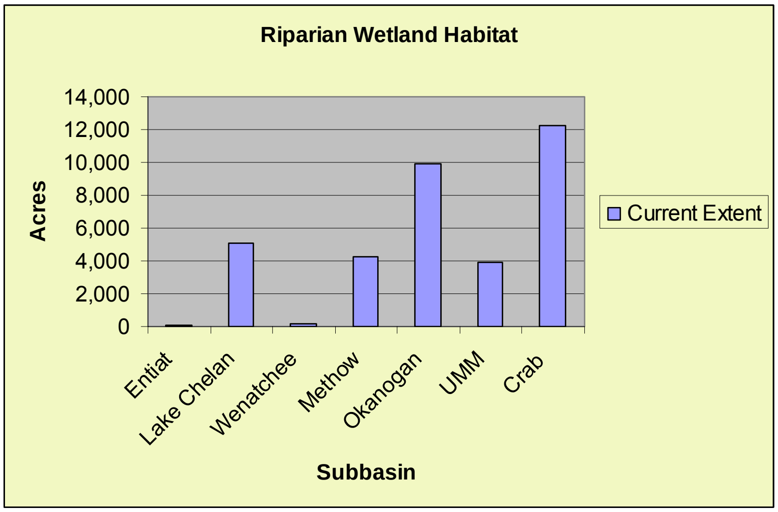


Source: IBIS 2003

Figure 45 Current extent of riparian wetland habitat in the Columbia Cascade Ecoprovince, Washington

Protection Status

The protection status of riparian habitat is compared by subbasin in **Figure 46**. Riparian habitats are provided high protection status predominantly in the Lake Chelan subbasin. The vast majority of Ecoprovince riparian habitat is designated "low" or "no" protection status, and is at risk for further degradation and/or conversion to other uses. The GAP protection status of riparian wetland habitat in the Methow subbasin is depicted in **Table 38**.



Source: IBIS 2003

Figure 46 Protection status of riparian wetlands in the Columbia Cascade Ecoprovince, Washington

Table 38 Eastside (interior) riparian wetlands GAP protection status in the Methow subbasin, Washington (IBIS 2003)

GAP Protection Status	Acres
High Protection	0
Medium Protection	168
Low Protection	434
No Protection	3,632

Factors Affecting Eastside (Interior) Riparian Wetland Habitat

Factors affecting grassland habitat are described in Ashley and Stovall (unpublished report, 2004) and summarized below:

- Loss of habitat because of numerous factors including riverine recreational developments, inundation from impoundments, cutting and spraying of riparian vegetation for eased access to water courses, gravel mining, etc.;
- Habitat alteration from: a) hydrological diversions and control of natural flooding regimes (e.g., dams) resulting in reduced stream flows and reduction of overall area of riparian habitat, loss of vertical stratification in riparian vegetation, and lack of recruitment of young cottonwoods, ash, willows, etc., and; b) stream bank stabilization which narrows stream channel, reduces the flood zone, and reduces extent of riparian vegetation;
- Habitat degradation from livestock overgrazing which can widen channels, raise water temperatures, and reduce understory cover;

- Habitat degradation from conversion of native riparian shrub and herbaceous vegetation to invasive exotics such as reed canary grass, purple loosestrife, perennial pepperweed, salt cedar, indigo bush, and Russian olive;
- Fragmentation and loss of large tracts necessary for area-sensitive species such as yellow-billed cuckoo;
- High energetic costs associated with high rates of competitive interactions with European starlings for cavities may reduce reproductive success of cavity-nesting species such as Lewis' woodpecker, downy woodpecker, and tree swallow, even when outcome of the competition is successful for these species, and;
- Recreational disturbances (e.g., ORVs), particularly during nesting season, and, in particular, in high-use recreation areas.

Recommended Future Condition

Recommended future conditions are described in detail in section 4.1.7.3.3 in Ashley and Stovall (unpublished report, 2004). Recommended conditions for riparian wetland habitat are summarized in the following paragraphs.

Condition 1a – Cottonwood gallery forests with healthy canopy cover: Red-eyed vireo was selected to represent species that require greater than 60% canopy closure. For their food and reproductive requirements, red-eyed vireo require mature deciduous trees greater than 160 feet tall; greater than 10% of the shrub layer should be young cottonwoods.

Condition 1b – Deciduous riparian zone with high canopy closure: Beaver was selected to represent species that require 40-60% tree/shrub canopy closure and shrub height greater than 6.6 feet. Beavers also require trees less than 6 inches DBH.

Condition 2 – Riparian habitat with a dense shrub layer: Yellow-breasted chat was selected to represent species that require riparian habitat with a shrub layer one to four metres (three to 13 feet) tall, 30-80% shrub cover, scattered herbaceous openings, and less than 20% tree cover.

The change in extent of the riparian wetland habitat type from circa 1850 to 1999 is not included because of inaccurate IBIS (2003) data/GIS products.

3.9.6 Agriculture (Habitat of Concern)

Agricultural habitat varies substantially in composition among the cover types it includes. Cultivated cropland includes at least 50 species of annual and perennial plants, and hundreds of varieties, ranging from vegetables such as carrots, onions, and peas, to annual grains such as wheat, oats, barley, and rye. Row crops of vegetables and herbs are characterized by bare soil, plants, and plant debris along bottomland areas of streams and rivers and areas having sufficient water for irrigation. Annual grains, such as barley, oats, and wheat are typically produced in almost continuous stands of vegetation on upland and rolling hill terrain without irrigation.

Improved pastures are used to produce perennial herbaceous plants for grass seed and hay. Alfalfa and several species of fescue and bluegrass, orchardgrass (*Dactylis glomerata*), and timothy (*Phleum pratensis*) are commonly seeded in improved pastures. Grass seed fields are single-species stands, whereas pastures maintained for haying are typically composed of several species.

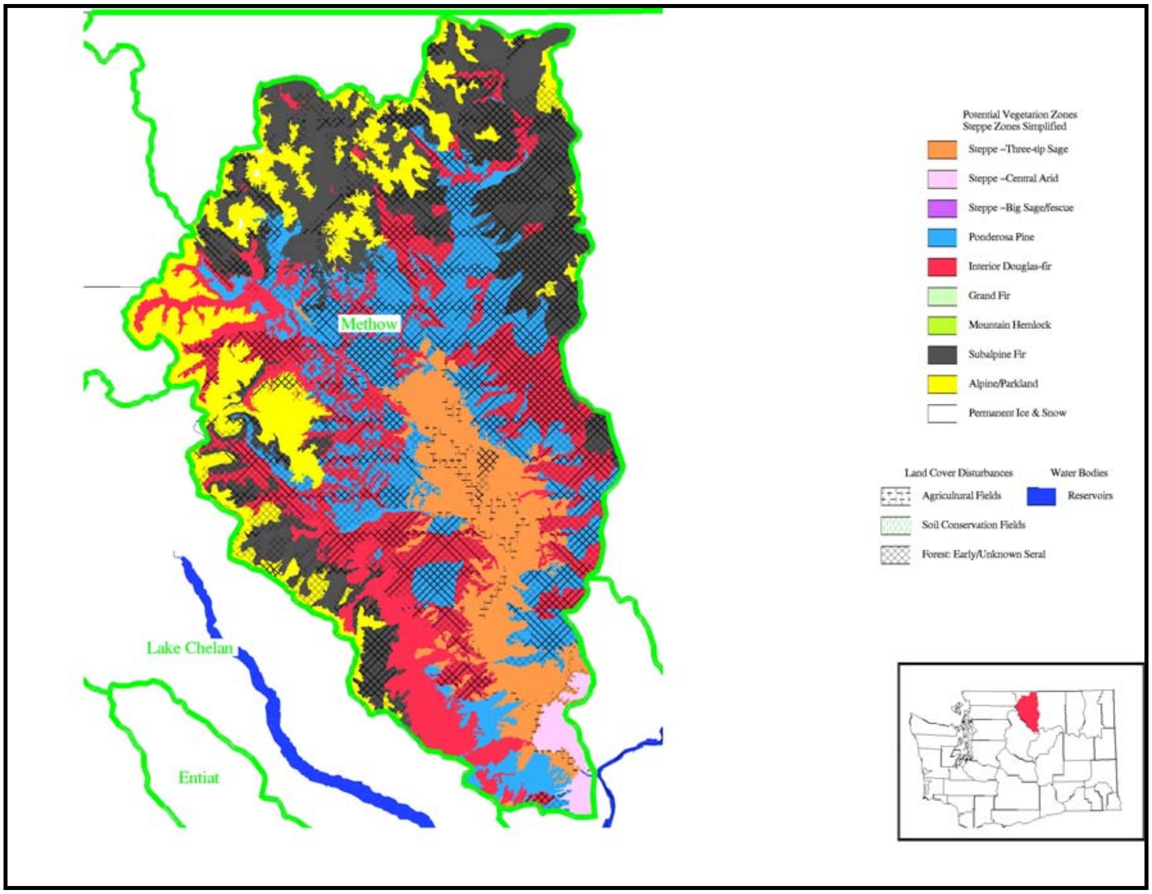
Unimproved pastures are predominantly grassland sites and often abandoned fields that have little or no active management such as irrigation, fertilization, or herbicide applications. These sites may or may not be grazed by livestock. Unimproved pastures include rangelands planted to exotic grasses that are found on private land, state wildlife areas, federal wildlife refuges, and CRP sites. Grasses commonly planted on CRP sites include crested wheatgrass (*Agropyron cristatum*), tall fescue (*F. arundinacea*), perennial bromes (*Bromus* spp.), and wheatgrasses.

Intensively grazed rangelands have been seeded to intermediate wheatgrass (*Elytrigia intermedia*), crested wheatgrass to boost forage production, or are dominated by increaser exotics such as Kentucky wheatgrass or tall oatgrass (*Arrhenatherum elatius*). Other unimproved pastures have been cleared and intensively farmed in the past, but are allowed to convert to other vegetation.

These sites may be composed of uncut hay, litter from previous seasons, standing dead grass and herbaceous material, invasive exotic plants including tansy ragwort (*Senecio jacobea*), thistle (*Cirsium* spp.), Himalaya blackberry (*Rubus discolor*), and Scot's broom (*Cytisus scoparius*) with patches of native black hawthorn, snowberry, spirea (*Spirea* spp.), poison oak (*Toxicodendron diversilobum*), and various tree species, depending on seed source and environment.

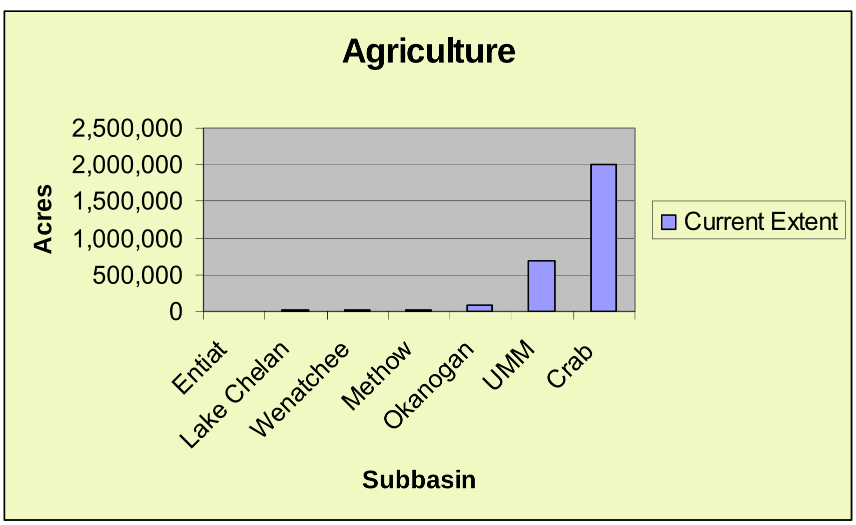
Because agriculture is not a focal wildlife habitat type, and there is little opportunity to effect change in agricultural land use at the landscape-scale, Ecoprovince and subbasin planners did not conduct a full-scale analysis of agricultural conditions. Agricultural lands converted to CRP, however, can significantly contribute toward benefits to wildlife habitat and other species that utilize agricultural lands.

Agricultural extent in the Methow subbasin is illustrated in Figure 47.



Source: Cassidy 1997

Figure 47 Agricultural extent in the Methow subbasin, Washington

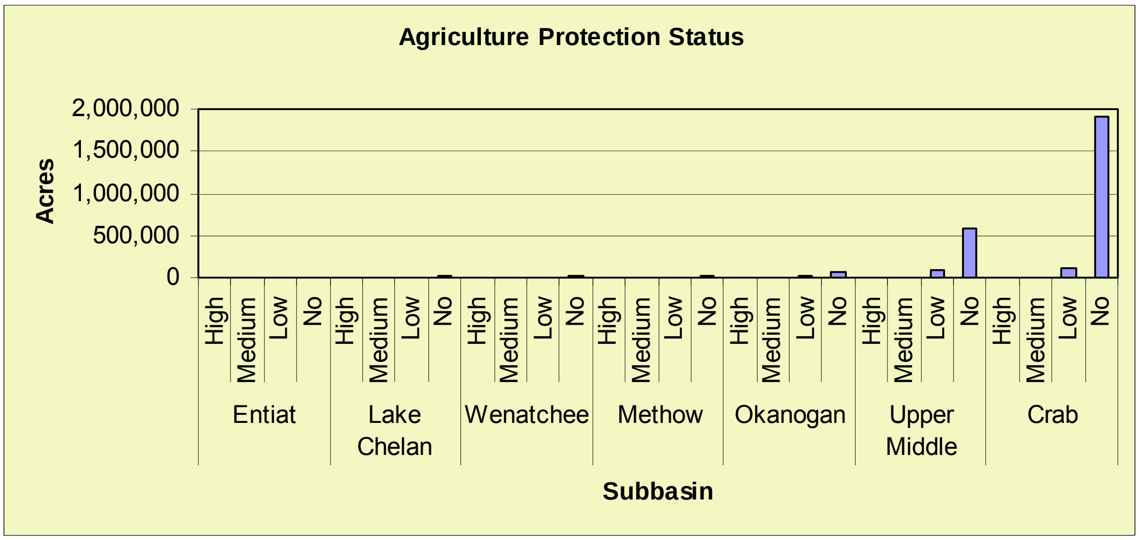


Source: IBIS 2003

Figure 48 Current extent of agriculture in the Columbia Cascade Ecoprovince, Washington

Protection Status

The protection status of agricultural habitat is compared by subbasin in **Figure 49**. IBIS (2003) data clearly indicate that nearly all of this cover type has been provided protection status across the Ecoprovince. Small amounts of agricultural lands, however, are given "low" and "medium" protection status. "Low" and "medium" protection is limited to lands enrolled in conservation easements, or to those that are under other development restrictions, such as County planning ordinances. The GAP protection status of agricultural habitat in the subbasin is illustrated in Table 39



Source: IBIS 2003

Figure 49 Protection status of agriculture in the Columbia Cascade Ecoprovince, Washington

Table 39. Agriculture GAP protection status/acres in the Methow subbasin, Washington (IBIS 2003).

GAP Protection Status	Acres
High Protection	412
Medium Protection	710
Low Protection	8,004
No Protection	22,873

3.10 Environmental Conditions

3.10.1 Changes in Wildlife Habitats

Dramatic changes in wildlife habitat have occurred throughout the subbasin since pre-European settlement (circa 1850). IBIS data limitations for describing historic and current habitat conditions at the subbasin level are described in Section 1.1 (Ashley and Stovall, unpublished report, 2004). Because of the limitations and inaccuracies associated with the IBIS mapping, the IBIS historic versus current characterizations of habitats is not used for subbasin level analyses.

Summary of Factors Affecting Focal Wildlife Habitats and Species

The presence, distribution, and abundance of wildlife species in the Methow subbasin have been affected by habitat losses due primarily to:

- agricultural development;
- timber management;
- livestock grazing;
- mining; and
- commercial and residential development

Agricultural Development

Agricultural development in the Methow subbasin has altered or destroyed vast amounts of native shrubsteppe habitat and fragmented riparian/floodplain habitat. Agricultural operations have increased sediment loads and introduced herbicides and pesticides into streams.

Conversion to agriculture has decreased the overall quantity of habitat for many native species, but the loss of specific communities may be particularly critical for habitat specialists. However, conversion of land to agriculture has practically diminished, and there has been an actual decrease in agricultural lands in the Methow subbasin within the last 30 years.

Timber Management

Timber management activities, including extensive timber harvest in sections of the Methow subbasin, have negatively impacted wildlife habitat, particularly in the Chewuch River and Beaver Creek drainages (NPPC 2002).

Historic timber harvest activities and related road building have contributed to erosion and sediment loading, loss of shading for creeks and streams, loss of recruitment material for LWD, and overall decrease in nutrients. Construction of logging roads also resulted in the construction of numerous culverts in the subbasin. However, timber harvest activities have dramatically decreased in the last 20 years in the Methow subbasin, to being very limited.

Livestock Grazing

Livestock grazing has negatively impacted wildlife habitat in the subbasin, particularly in the Chewuch River and Beaver Creek drainages. Mismanaged grazing has contributed to increased soil erosion and displaced native plant communities. In the 1950s and 1960s, approximately 12,000 mother cows were producing in the subbasin. Currently, there are approximately 100 mother cows in the subbasin.

Mining

Mining activity in the Methow Subbasin is currently minimal; however, abandoned mine sites pierce the valley hillsides and historically have contributed sediment, which, in some cases, is relatively toxic loads to rivers and creeks (WCC, 2000).

Mining degrades aquatic habitats, used by bull trout, by altering water chemistry (e.g., pH), by altering stream morphology and flow, and by altering the substrate composition and benthic insect community composition where in-channel mining activity occurs, causing sediment, fuel, and heavy metals to enter streams (Martin and Platts 1981; Spence et al. 1996; Thomas 1985).

Commercial and Residential Development

While urban areas comprise only a small percentage of the land base within the subbasin (0.1%), their habitat impacts are significant. Residential growth within the subbasin is largely occurring along creeks and rivers. Channelization and development along water courses has been altered, and in some cases, replaced riparian and wetland habitats.

Environmental / Ecological Relationships

Expansion of residential areas affects drainage, and homes built along streams have affected both water quality and the ability of the floodplain to function normally. Residential development has resulted in the loss of large areas of all focal habitat types. Disturbance by humans in the form of highway traffic, noise and light pollution, and various recreational activities have the potential to displace wildlife and force them out of their native areas, or force them to use less desirable habitat.

The conversion of forested uplands and riparian habitat to residential use has negatively affected wildlife habitat connectivity and composition. Road construction and dispersed residential development have impeded stream access and changed vegetative communities, resulting in the reduction of wildlife range and quality.

Human activities have increased the number of fire starts, but historic fire control policies have kept the size of fires small, resulting in a buildup of fuel in the forested uplands of the subbasin. This absence of fire has resulted in changes to the composition of the forest and plant communities and the related capacity to store and transport water.

3.10.2 Changes in Fish habitats

Diking, conversion of riparian areas to agriculture and residential uses, and LWD removal along the mainstem Methow River, have resulted in loss of side-channel access, riparian vegetation, and overall habitat complexity. Much of the habitat within this area, however, has not been adequately inventoried or assessed and data gaps exist regarding the extent of habitat alterations.

As noted in Section 2.1, Washington Water Power Company’s dam near the mouth of the Methow River significantly altered salmonid production in the early decades of the 20th century. The dam is thought to have had significant effects on production of coho, Chinook, and steelhead.

When the dam was removed in 1930, coho salmon, once the most abundant salmonid in the Methow subbasin (Craig and Suomela 1941) were extirpated, Chinook were nearly extirpated, and steelhead persisted as resident rainbow trout (Mullan et al. 1992b).

Much of the watershed remains undeveloped, and large tracts of high quality fish habitat remain, particularly within the middle and upper elevations. These areas are contained in lands held largely in public ownership, and include several thousand acres managed as wilderness/roadless condition by the Okanogan National Forest. Within these management boundaries, plant communities and succession are shaped largely through such natural processes as fire, avalanches, storms, and temperature ranges.

Current Reference Condition

Within these management boundaries, plant communities and succession are shaped largely through such natural processes as fire, avalanches, storms and temperature ranges. Early successional habitats are underrepresented, however, due largely to historic emphasis on fire suppression.

Outside of these protected areas, little habitat has been lost to development at middle and upper elevations, but acreage within the lower elevations has been altered and/or degraded through road building, grazing, and timber harvest. The most significant changes in wildlife habitat have occurred in the dry forest, riparian/floodplain, and shrubsteppe habitats at lower elevations.

Native habitats have been lost or altered by commercial and residential development, conversion to agricultural use, grazing, timber harvest and road building. Fire suppression and noxious weed invasion have also altered the landscape and native plant communities considerably.

There are 29 fish and wildlife species listed as Endangered, Threatened, or as Species of Concern within the Methow subbasin. The watershed contains 14 Priority Habitats as identified by WDFW.

The riparian and wetlands of the Methow subbasin support the greatest wildlife diversity and abundance, but occupy the lowest percentage of acreage within the watershed. It has been widely quoted that in semi-arid environments like the Methow, riparian habitats typically occupy less than 10% of the land area, but are used by more than 90% of the wildlife species for some or all of their life history requirements.

The Methow subbasin is host to some of Eastern Washington’s best remaining tracts of cottonwood gallery forests, found in the wide floodplain portions of the Methow River valley and its major tributaries. Almost all of this habitat type is in private ownership and much has been converted to residential development or agriculture; significant forest parcels remain along the Methow River between Winthrop and Lost River.

Additional significant stands are located along the Twisp and Chewuch rivers, and more fragmented pockets can be found along the Methow between Winthrop and Carlton. Below Carlton, a higher stream gradient and a more constrained channel preclude the development of large patches of this habitat type (J. Foster, WDFW, pers. comm.). Because of its proximity to roads and other developed areas, much of the remaining riparian/floodplain habitat may be at risk of conversion to housing development.

Protection status

Much of the land within the subbasin is set aside as protected, particularly in the upper elevations. Protected areas (**Figure 50**) include two wilderness areas: the Pasayten Wilderness Area and the Lake Chelan-Sawtooth Wilderness Area. The WDFW also manages the Methow Valley Wildlife Area.

The subbasin contains the largest amount (27% or 317,865 acres) of permanently protected lands than any other subbasin in the Ecoprovince. The Pasayten Wilderness Area and the Lake Chelan-

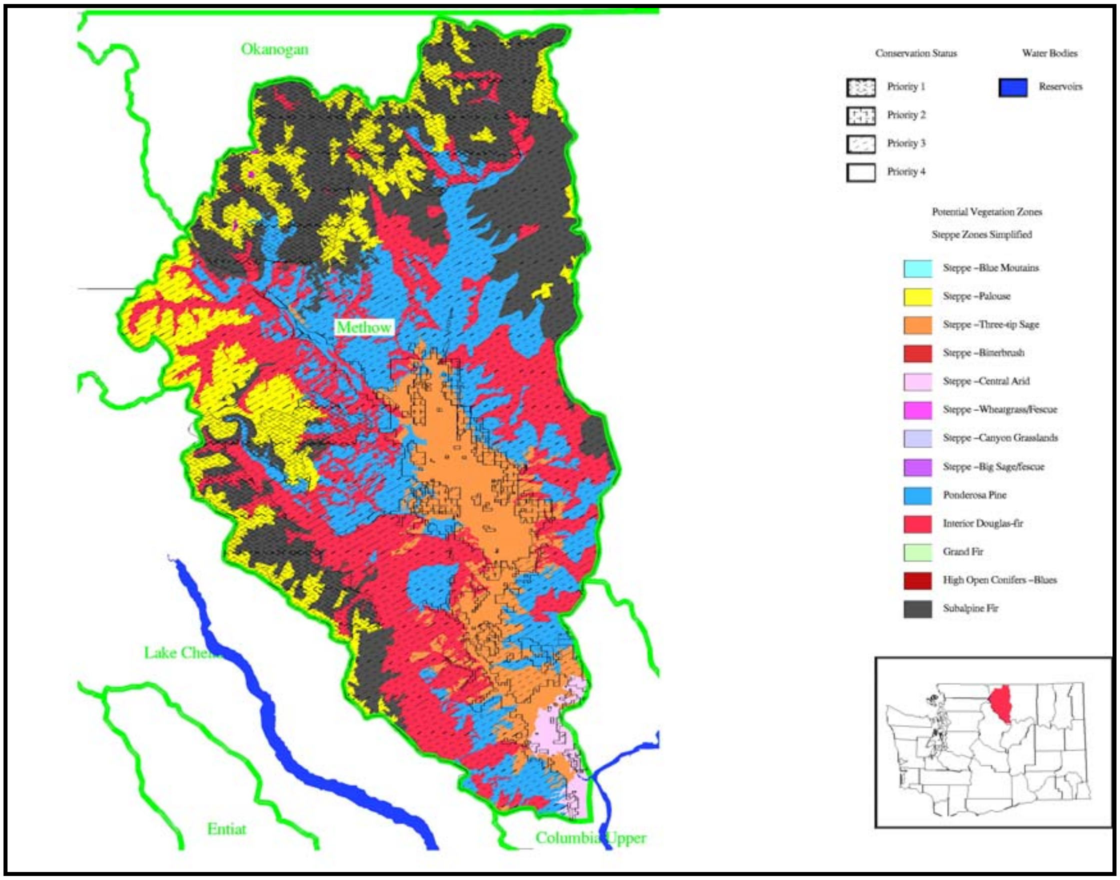
Sawtooth Wilderness Area have permanent protection from conversion of natural land cover and a mandated management plan in operation to maintain a natural state within which disturbance events of natural type are allowed to proceed without interference.

Approximately 1.2% (14,078 acres) of the subbasin has permanent protection from conversion of natural land cover and a mandated management plan in operation to maintain a primarily natural state ("medium" protection status). The majority of lands in the subbasin (60% or 706,058 acres) has permanent protection from conversion of natural land cover for the majority of the area, but is subjected to uses of either a broad, low intensity type or localized, intense type ("low" protection status).

The NPPC designated a number of river reaches throughout the Columbia Basin as protected areas. Those protected river reaches total approximately 178.8 miles within the Methow subbasin, and include portions of Bear Creek, Buttermilk Creek, Chewuch River, Early Winters Creek, Lost River, Methow River, South Creek, War Creek, and the Twisp River (StreamNet 2001).

Approximately 80% of the Upper Methow subwatershed is managed by the U.S. Forest Service (USFS) as Congressionally Withdrawn (Wilderness), Late-Successional Reserve, or Riparian Reserve (USFS 1998d). These designations provide a high level of protection of aquatic areas and the surrounding uplands.

The Lost River subwatershed contains 102,100 acres (95% of the subwatershed) that is protected within the Pasayten Wilderness.



Source: Cassidy 1997

Figure 50 Protection status and vegetation zones of the Methow subbasin

The Early Winters Creek subwatershed contains approximately 51,548 acres (approximately 99% of the subwatershed) that are managed by the USFS. The majority of that land is designated as a Scenic Highway Corridor along state Route Highway 20, with the remainder designated as a Late Successional Reserve.

In the Chewuch River subwatershed, 108,000 acres (34% of the subwatershed) are protected within the Pasayten Wilderness. Other lands within the subwatershed include 5,000 acres (1.5%) that are managed by WDFW.

The Twisp River subwatershed, including the headwaters and much of the uplands, contains approximately 72,000 acres (approximately 50% of the subwatershed) that fall within the Lake Chelan-Sawtooth Wilderness area. Additional federally managed land within the Twisp subwatershed is managed as Late Successional Reserves or Matrix (USFS 1995c). Lower elevation Forest Service land above the confluence with Buttermilk Creek has been allocated as Late Successional Reserves.

The majority of the Lower Methow River is federally owned and managed by the National Forest Service as the Okanogan National Forest, with a small portion of upper Libby Creek lying within the Lake Chelan-Sawtooth Wilderness.

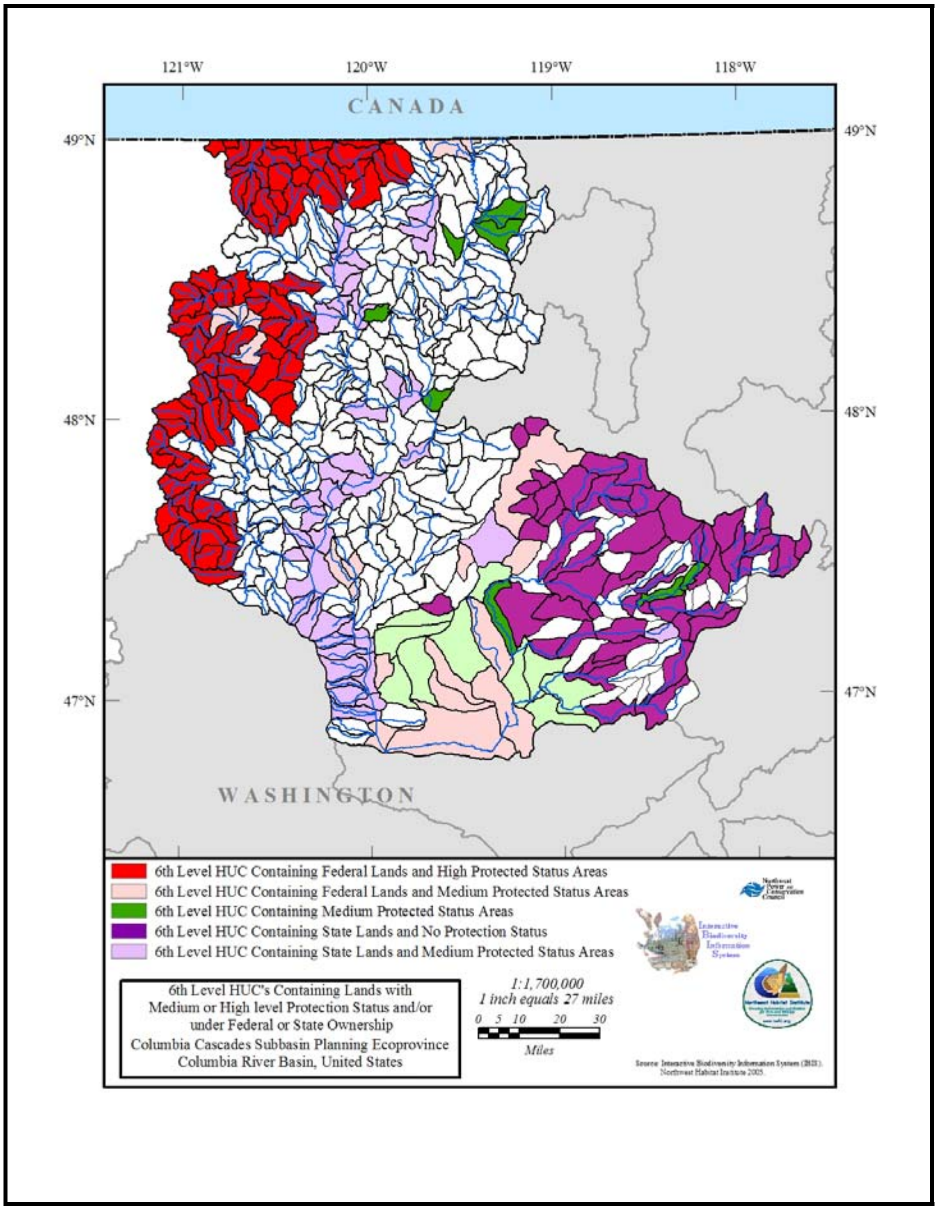
The Methow Conservancy works to provide successful, voluntary, private land conservation easements. To date, the Methow Conservancy protects a total of 3,774 acres.

Approximately 11% (129,794 acres) of the lands within the subbasin lack irrevocable easements or mandates to prevent conversion of natural habitat types to anthropogenic habitat types (“no” protection). Lands owned by WDFW fall within the “medium” and “low” protection status categories.

GAP protection status acreage for each Ecoprovince subbasin is compared in **Figure 51**. As illustrated, the Upper Middle Mainstem Columbia River subbasin and the Crab subbasin are the only subbasins in the Ecoprovince without “high” protection status lands (status 1). “Medium,” “low,” and “no” protection status lands (status 2, 3, and 4 respectively) show similar trends as those found in other Ecoprovince subbasins.

Additional habitat protection, primarily on privately owned lands, is provided through the Conservation Reserve Program (CRP) and the Conservation Reserve Enhancement Program (CREP). The CRP is intended to reduce soil erosion on upland habitats through establishment of perennial vegetation on former agriculture lands. Similarly, CREP conservation practices reduce stream sedimentation and provide protection for riparian/riverine habitats using buffer strips comprised of herbaceous and woody vegetation.

Both programs provide short-term (CRP-10 years; CREP-15 years), high protection of habitats enrolled in either program. The U.S. Congress authorizes program funding/renewal, while the USDA determines program criteria. Program enrollment eligibility and sign-up is decentralized to state and local NRCS offices (R. Hamilton, FSA, pers. comm., 2003).



Source: IBIS 2003

Figure 51 GAP protection status for all Ecoprovince/subbasin habitat types

Ecological Features

Vegetation

The following landscape-level vegetation information is derived from the Washington GAP Analysis Project (Cassidy 1997) and IBIS data (2003).

Cassidy (1997) identified six historic (potential) vegetation zones that occur within the subbasin in Section 2.2 (**Table 14**). The three-tip sage, central arid steppe, and Ponderosa pine vegetation zones are described in detail in Ashley and Stovall (unpublished report, 2004). These vegetation zones constitute focal habitat types. Douglas-fir, subalpine fir, and alpine parkland are not focal habitat types, but these vegetation zones occur throughout the subbasin.

Vegetation zone status has been summarized in **Figure 50**. An estimated 1.5% of central arid steppe and 5.2% of three-tip sage has been lost to agriculture. Similarly, 1.1% of the Ponderosa pine vegetation zone has been converted to agriculture. Historic and current extent of GAP vegetation zones in the Methow subbasin is illustrated in **Figure 52** and **Figure 53**.

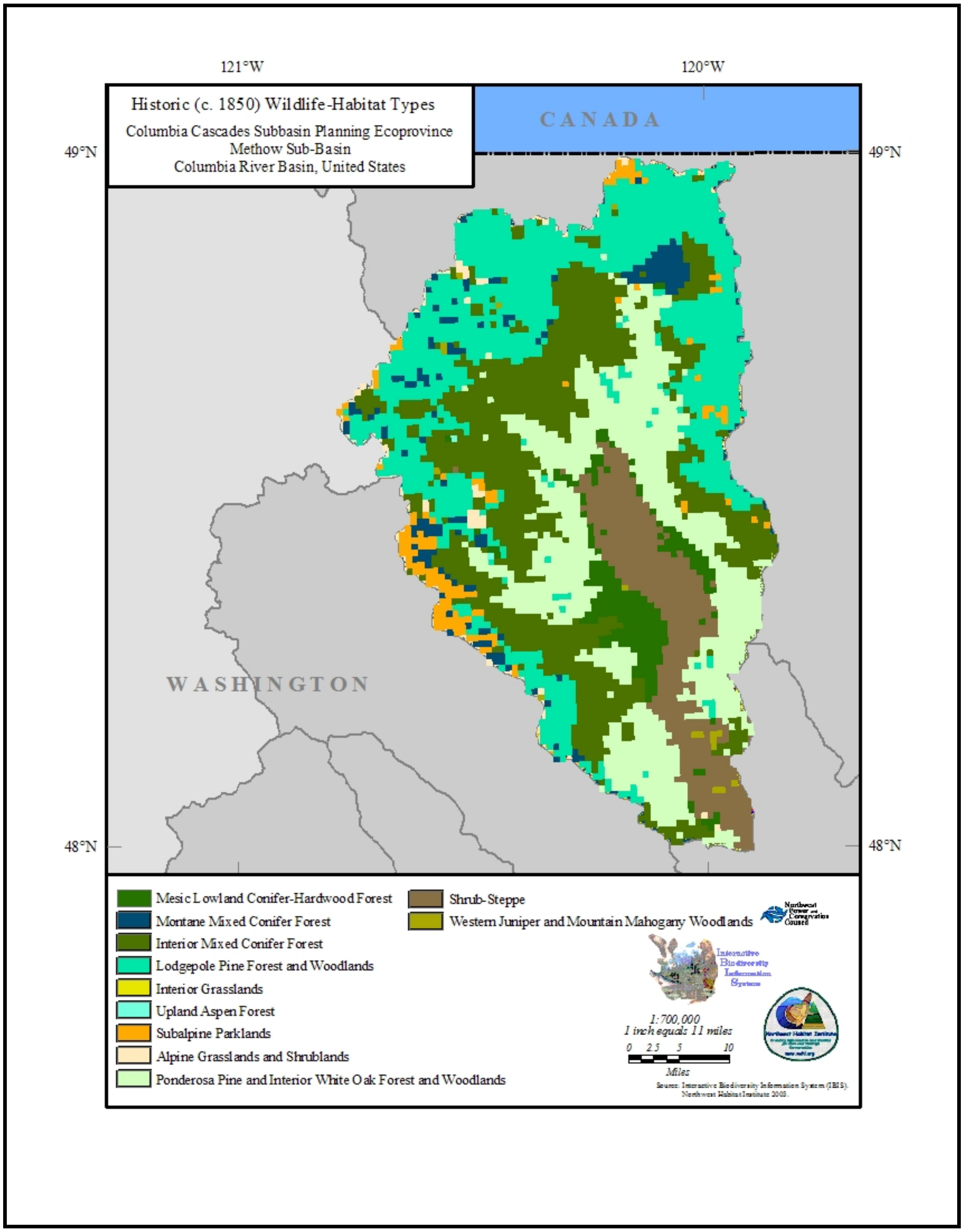


Figure 52 Historic wildlife habitat types of the Methow subbasin, Washington (IBIS 2003)

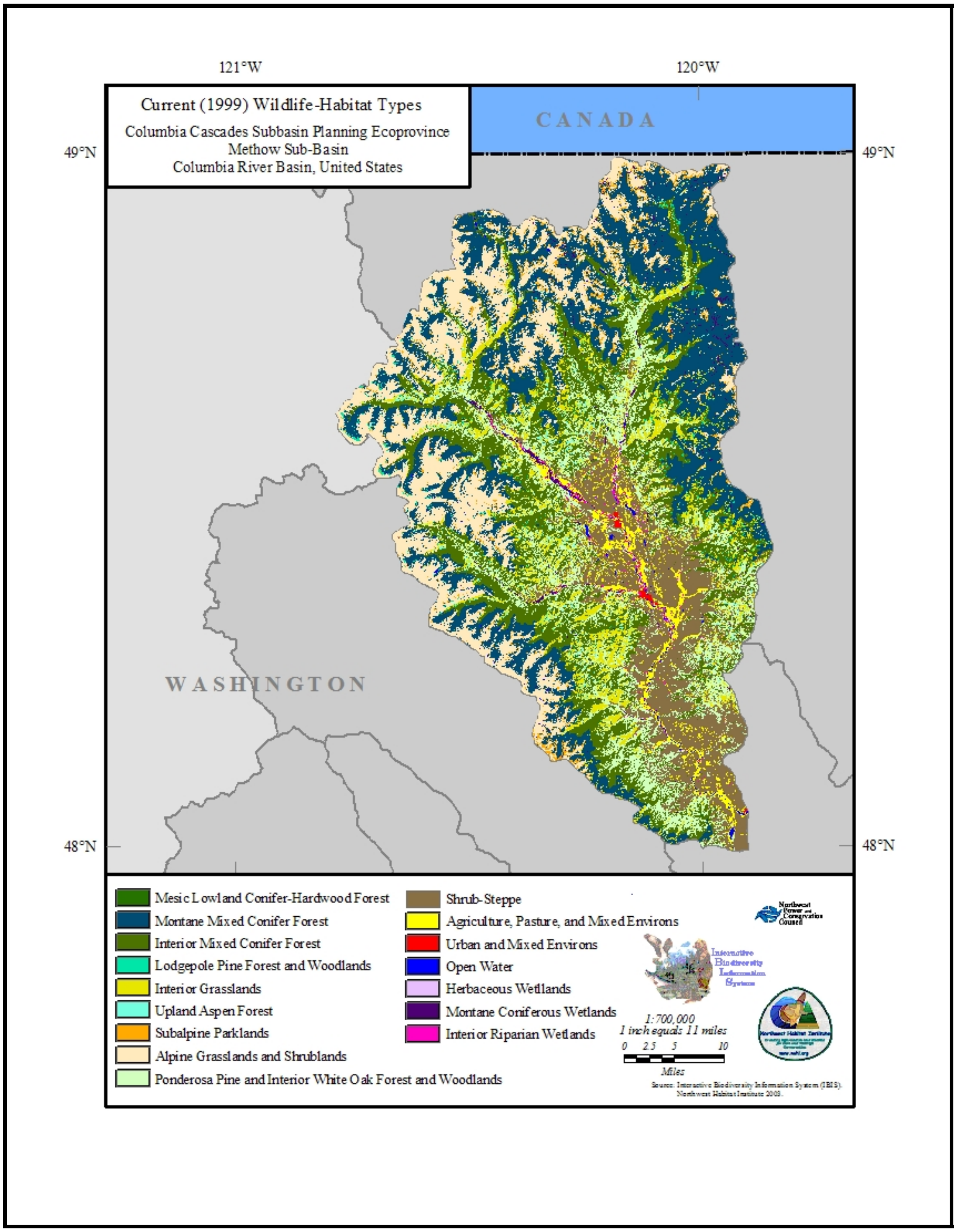


Figure 53 Current wildlife habitat types of the Methow subbasin, Washington (IBIS 2003)

Rare Plant Communities

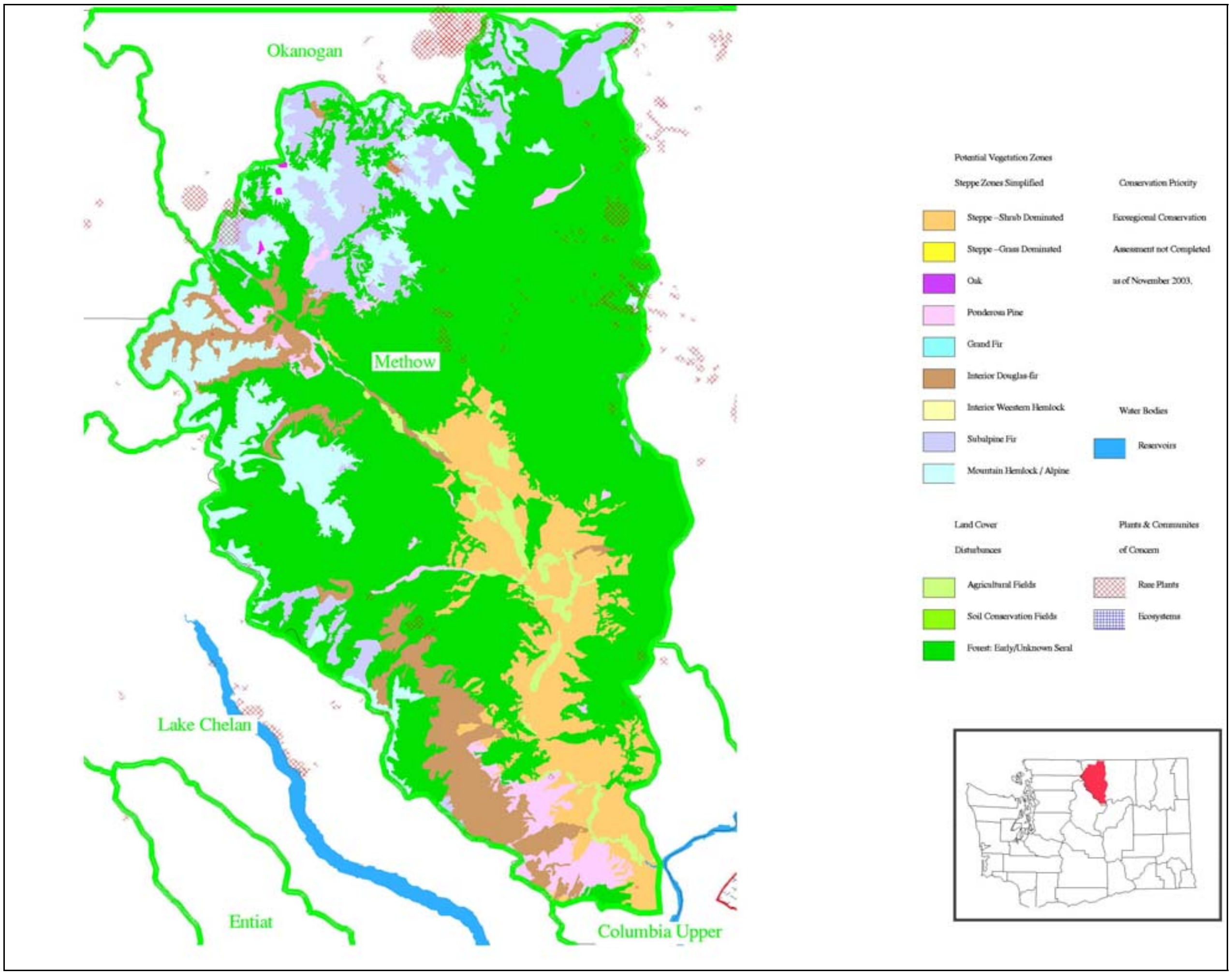
The subbasin contains 50 rare plant communities (Appendix C). Approximately 28% of the rare plant communities are associated with shrubsteppe habitat, 16% with riparian or wetland habitats, and 56% with upland forest habitat. Rare/high-quality plant occurrences and communities are illustrated in **Figure 54**.

Introduced wildlife

A list of 17 species of introduced or exotic wildlife species has been developed by the WDFW (**Table 40**).

Table 40 Introduced/exotic wildlife present in the Methow subbasin (IBIS 2003)

Common Name	Scientific Name
Bullfrog	*Rana catesbeiana*
Chukar	*Alectoris chukar*
Gray Partridge	*Perdix perdix*
Ring-necked Pheasant	*Phasianus colchicus*
Wild Turkey	*Meleagris gallopavo*
California Quail	*Callipepla californica*
Rock Dove	*Columba livia*
European Starling	*Sturnus vulgaris*
House Sparrow	*Passer domesticus*
Virginia Opossum	*Didelphis virginiana*
Eastern Cottontail	*Sylvilagus floridanus*
Cascade Golden-mantled Ground Squirrel	*Spermophilus saturatus*
Eastern Gray Squirrel	*Sciurus carolinensis*
Black Rat	*Rattus rattus*
Norway Rat	*Rattus norvegicus*
House Mouse	*Mus musculus*
Nutria	*Myocastor coypus*



(Cassidy 1997; WNHP 2003)

Figure 54 Rare plant occurrence and high-quality plant communities in the Methow subbasin, Washington

Noxious Weeds

Changes in biodiversity have been closely associated with changes in land use. Grazing, agriculture, and accidents have introduced a variety of exotic plants, many of which are vigorous enough to earn the title "noxious weed." Twenty-six species of noxious weeds occur in the Subbasin (**Table 40**).

Noxious weeds alter riparian vegetative cover by reducing the complexity of vegetative layering and diversity, on which indigenous aquatic and semi-aquatic species rely (USDA 2000). Infestations on stream banks may lead to increased sediment delivery when weeds replace native, fibrous-rooted plants with tap-rooted weeds, such as knapweed. The weeds use available water, but do not provide enough ground cover to prevent erosion. (USDA 2000).

Herbicide treatment of weeds also impacts streams if the herbicide reaches the channel. Herbicides may enter surface or shallow groundwater when sprayed directly on running or standing water, through drift or soil erosion, or in the case of an accidental spill.

Herbicides may indirectly affect surface waters by reducing the riparian zone vegetation, leading to increased water temperatures (USDA 2000). Herbicides may contaminate water through accidental spills, direct application to water bodies, surface runoff, or movement through the soil (USDA 2000).

Table 41 Exotic terrestrial plant/noxious weeds in the Methow subbasin and their origin (Callihan and Miller 1994)

Common Name	Scientific Name	Origin
Babysbreath	*Gypsophila paniculata*	
Canadian thistle	*Cirsium arvense*	Eurasia
Cheat grass	*Bromus tectorum*	
Cocklebur	*Xanthium spionosum*	
Dalmatian toadflax	*Linaria dalmatica*	Mediterranean
Diffuse knapweed	*Centaurea diffusa*	Eurasia
Hounds tongue	*Cunoglossum officinale*	
Japanese knotweed	*Polygonum cuspidatum*	
Kochia	*Kochia scoparia*	
Leafy spurge	*Euphorbia esula*	Eurasia
Longspine sandbur	*Cenchrus longispinus*	
Meadow hawkweed	*Hieracium caespitosum*	Europe
Mullein	*Verbascum thapsus*	
Musk thistle	*Carduus nutans*	Eurasia
Orange hawkweed	*Hieracium aurantiacum*	Europe
Oxeye daisy	*Leucanthemum vulgare*	
Perennial sowthistle	*Sonchus arvengis*	
Plumeless thistle	*Carduus acanthoides*	
Puncturevine	*Tribulus terrestris*	Europe
Purple loosestrife	*Lythrum salicaria*	Europe
Russian knapweed	*Centaurea repens*	Southern Russia and Asia
Russian thistle	*Salsola iberica sennen*	
Scotch cottonthistle	*Onopordum acanthium*	Europe
Scotchbroom	*Cytisus scoparius*	Europe
Spotted knapweed	*Centaurea maculosa*	Europe
Spurge flax	*Thymelaea passerina*	
St. Johnswort	*Hypericum perforatum*	
Sulfur cinquefoil	*Potentilla recta*	
Tansy ragwort	*Senecio jacobaea*	Eurasia
Whitetop	*Cardaria draba*	Europe
Wild Four o'clock	*Mirabilis nyctaginea*	
Yellow star thistle	*Centaurea solstitialis*	Mediterranean and Asia
Yellow toadflax	*Linaria vulgaris*	Europe

3.11 Ecological Relationships

The biotic communities of aquatic systems in the Upper Columbia Basin are highly complex. Within communities, assemblages and species have varying levels of interaction with one another. Direct interactions may occur in the form of predator-prey, competitor, and disease- or parasite-host relationships. In addition, many indirect interactions may occur between species.

These interactions continually change in response to shifting environmental and biotic conditions. Human activities that change the environment, the frequency and intensity of disturbance, or species composition can shift the competitive balance among species, alter predatory interactions, and change disease susceptibility. All of these changes may result in community reorganization.

3.12 Community Structure

Few studies have examined the fish species assemblages within the Upper Columbia Basin. Most information available is from past surveys (e.g., Dell et al. 1975; Dobler et al. 1978; McGee et al. 1983; Burley and Poe 1994; Hillman 2000; Duke Engineering 2001), dam passage studies (e.g., Mullan et al. 1986; Tonseth and Petersen 1999; Chelan PUD unpublished data), and northern pikeminnow studies (e.g., Burley and Poe 1994; West 2000).

The available information indicates that about 41 species of fish occur within the Upper Columbia Basin (from the mouth of the Yakama River upstream to Chief Joseph Dam) (**Table 12**). This is an underestimate because several species of cottids (sculpins) live there. Of the fishes in the basin, 15 are cold-water species, 18 are cool-water species, and eight are warm-water species. Most of the cold-water species are native to the area; only four were introduced: brown trout (*Salmo trutta*), brook trout (*Salvelinus fontinalis*), lake whitefish (*Coregonus clupeaformis*), and Atlantic salmon (*S. salar*). Four of the 18 cool-water species are exotics: pumpkinseed (*Lepomis gibbosus*), walleye (*Stizostedion vitreum*), yellow perch (*Perca flavescens*), and smallmouth bass (*Micropterus dolomieu*), while all warm-water species are exotics.

Table 42 Fish Species of the Upper Columbia River Basin (Pevan 2004)

Common name	Species	Native (N) or Exotic (E)	Feeding location in water column				Primary prey			
			Surf	Mid	Bot	Plant	Detrit	Mic	Mac	Fish
Cold-water species:										
White sturgeon	*Acipenser transmontanus*	N			x	x	x	x	X	x
Chinook salmon (juv)	*Oncorhynchus tshawytscha*	N	X	x	x				X	
Coho salmon (juv)	*Oncorhynchus kisutch*	N	X	x	x				X	

Common name	Species	Native (N) or Exotic (E)	Feeding location in water column				Primary prey			
			Surf	Mid	Bot	Plant	Detrit	Mic	Mac	Fish
Sockeye/kokanee (juv)	*Oncorhynchus nerka*	N	x	X	x			x	X	
Steelhead/ rainbow	*Oncorhynchus mykiss*	N	x	X	x				X	x
Interior redband trout	*Oncorhynchus mykiss gairdneri*	N	x	X	x				X	x
cutthroat trout	*Oncorhynchus clarki*	N	X	x	x				X	x
Westslope cutthroat trout	*Oncorhynchus clarki lewisi*	N	X	x	x				X	x
Brown trout	*Salmo trutta*	E	x	X	x				X	x
Atlantic salmon	*Salmo salar*	E	x	X	x				X	x
Bull trout	*Salvelinus confluentus*	N	x	x	X				X	x
Brook trout	*Salvelinus fontinalis*	E	x	X	x				X	x
Mountain whitefish	*Prosopium williamsoni*	N	x	x	X				X	
Lake whitefish	*Coregonus clupeaformis*	E		x	X				X	x
Longnose sucker	*Catostomus catostomus*	N			X	x	x	x	x	
Sculpins	*Cottus spp.*	N			X				X	x
Cool-water species:										

Common name	Species	Native (N) or Exotic (E)	Feeding location in water column				Primary prey			
			Surf	Mid	Bot	Plant	Detrit	Mic	Mac	Fish
Longnose dace	*Rhinichtys cataractae*	N			X				X	
Peamouth	*Mylocheilus caurinus*	N			X				X	x
Chiselmouth	*Acrocheilus alutaceus*	N			X	X			x	
Northern pikeminnow	*Ptychocheilus oregonensis*	N	x	x	X				X	x
Redside shiner	*Richardsonius balteatus*	N	x	X	x				X	
Sand roller	*Percopsis transmontana*	N			X				X	
Bridgelip sucker	*Catostomus columbianus*	N			X	X			x	
Mountain sucker	*Catostomus platyrhynchus*	N			X	X	x	x	x	
Largescale sucker	*Catostomus macrocheilus*	N			X	X	x	x	x	
Pacific lamprey (juv)	*Lampetra tridentata*	N			X	x		X	x	
Western brook lamprey (juv)	*Lampetra richardsonii*	N			X	x	X	x		
Threespine stickleback	*Gasterosteus aculeatus*	N	x	X	x				X	
Pumpkinseed	*Lepomis gibbosus*	E		X	x				X	x

Common name	Species	Native (N) or Exotic (E)	Feeding location in water column				Primary prey			
			Surf	Mid	Bot	Plant	Detrit	Mic	Mac	Fish
Walleye	*Stizostedion vitreum*	E		x	X				x	X
Yellow perch	*Perca flavescens*	E	x	X	x				X	x
Smallmouth bass	*Micropterus dolomieu*	E	x	x	X				X	x
Sculpin	*Cottus spp.*	N			X				X	x
Warm-water species:										
Channel catfish	*Ictalurus punctatus*	E			X				X	x
Black bullhead	*Ameiurus melas*	E			X	x			X	
Brown bullhead	*Ameiurus nebulosus*	E			X	x	x	x	X	x
Tench	*Tinca tinca*	E			X	x			X	
Common carp	*Cyprinus carpio*	E			X	x	x	x	X	
Bluegill	*Lepomis macrochirus*	E	x	X	x				X	x
Black crappie	*Pomoxis nigromaculatus*	E	x	X	x				X	x

Common name	Species	Native (N) or Exotic (E)	Feeding location in water column				Primary prey			
			Surf	Mid	Bot	Plant	Detrit	Mic	Mac	Fish
Largemouth bass	*Micropterus salmoides*	E	x	X	x				x	X

Anadromous species within the upper basin include spring and summer/fall Chinook salmon (*Oncorhynchus tshawytscha*), coho salmon (*O. kisutch*), sockeye salmon (*O. nerka*), steelhead (*O. mykiss*), and Pacific lamprey (*Lampetra tridentata*). Atlantic salmon (*Salmo salar*) and exotic species are also anadromous, but their status in the basin is largely unknown. White sturgeon (*Acipenser transmontanus*), which may have been anadromous historically, are present as a resident population.

Fish community interactions, interspecies competition, and species that are likely to interact with Chinook and steelhead, in particular, have been described for the Upper Columbia Basin in order to provide some management context for multi-species and ecosystem management objectives.

3.13 Competition

Competition among organisms occurs when two or more individuals use the same resources and when availability of those resources is limited (Pianka 2000). That is, for competition to occur, demand for food or space must be greater than supply (implies high recruitment or that the habitat is fully seeded), and environmental stresses few and predictable. Two types of competition are generally recognized: a) interference competition, where one organism directly prevents another from using a resource through aggressive behavior, and b) exploitation competition, where one species affects another by using a resource more efficiently. Salmonids likely compete for food and space both within species (intraspecific), and between species (well-coordinated). Well-coordinated interactions are more likely to occur between native and exotic species, rather than between species that coevolved together.

Although coevolved sympatric species should segregate (i.e., partition resources in space or time or both), native species may still interact along the margins of their spatial and temporal distributions. An example of this may occur between Chinook salmon and steelhead. This interaction was studied in the Wenatchee Basin by Hillman et al. (1989a, 1989b), and found to be relatively unimportant in limiting the production of the species. Interaction between the species was minimized because of disparate times of spawning, which tended to segregate the two species.

Currently, there is no evidence that the focal species interact with bull trout or westslope cutthroat trout. Indeed, Martin et al. (1992) indicated that juvenile bull trout and Chinook have different habitat preferences and, thus, do not interact competitively.

Significant interaction between redside shiners, Chinook, and steelhead may occur as a result of changes or modifications in water quality (e.g., temperature). In both field and laboratory studies, Hillman (1991) found that redside shiners displaced Chinook salmon from rearing areas at temperatures greater than 64°F (18°C). In fact, at these warmer temperatures, shiners negatively affected the distribution, behavior, and production of Chinook salmon. Reeves et al. (1987) documented similar results with redside shiners and juvenile steelhead. Thus, if water temperatures increase within the basin, one can expect increased interactions between shiners and Chinook and steelhead.

Exotic species may be more likely to interact with Chinook and steelhead because exotics have not had time to segregate spatially or temporally in their resource use. For example, there is a possibility that brook trout interact with Chinook and steelhead in the upper basin. Welsh (1994), however, found no evidence that brook trout displaced Chinook salmon. On the other hand, Cunjak and Green (1986) found that brook trout were superior competitors to rainbow/steelhead at colder temperatures (48°F or 9°C), while rainbow/steelhead were superior at warmer temperatures (61°F or 16°C).

A potentially important source of exploitative competition occurring outside the geographic boundary of the ESUs may be between the exotic American shad (*Alosa sapidissima*) and juvenile Chinook and steelhead. Palmisano et al. (1993a, 1993b) concluded that increased numbers of shad likely compete with juvenile salmon and steelhead.

Although coho salmon were native to the upper basin, they have been absent for many decades. Recently, there have been efforts to re-establish them in the upper basin (Murdoch et al. 2002). There is the potential that reintroduced coho will interact negatively with Chinook and steelhead; however, studies conducted in the Wenatchee Basin indicate that there is little to no interaction between the species (Spaulding et al. 1989; Murdoch et al. 2002).

3.14 Predation

Fish, mammals, and birds are the primary natural predators of Chinook and steelhead in the Upper Columbia Basin. Although the behavior of Chinook and steelhead precludes any single predator from focusing exclusively on them, predation by certain species can, nonetheless, be seasonally and locally important. Recent changes in predator and prey populations along with major changes in the environment, both related and unrelated to development in the Upper Columbia basin, have reshaped the role of predation (Mullan et al. 1986; Li et al. 1987).

About half of the resident species in the upper basin are piscivorous (eat fish). Ten cold-water species, seven cool-water species, and five warm-water species are known to eat fish. About 59% of these piscivores are exotics. Although 59% of the piscivores are exotics, these exotics constitute a small fraction of the total fish biomass within the project area (S. Hays, Chelan PUD, pers. comm.).

Before the introduction of exotics, northern pikeminnow (*Ptychocheilus oregonensis*), sculpin (*Cottus* spp.), white sturgeon, bull trout (*Salvelinus confluentus*), rainbow trout (*O. mykiss*), cutthroat trout (*O. clarki*), and burbot (*Lota lota*) were the primary piscivores in the region (Li et al. 1987; Poe et al. 1994). Presently, burbot are rare in the upper basin (Dell et al. 1975; Burley and Poe 1994), and probably have little effect on the abundance of juvenile Chinook and steelhead in the region. The status of white sturgeon in the upper basin is mostly unknown, although their numbers appear to be quite low (DeVore et al. 2000).

Introduced species such as walleye, smallmouth bass, and channel catfish (*Ictalurus punctatus*) are important predators of Chinook and steelhead in the Columbia River (Poe et al. 1994). Channel catfish are rare (Dell et al. 1975; Burley and Poe 1994) and likely have little to no effect on abundance of Chinook and steelhead. Other piscivores, such as largemouth bass (*M. salmoides*), black crappie (*Pomoxis nigromaculatus*), bluegill (*Lepomis macrochirus*), brown bullhead (*Ameiurus nebulosus*), yellow perch, and pumpkinseed are either rare or not known to prey heavily on juvenile anadromous fish (Dell et al. 1975; Burley and Poe 1994).

Although several fish species can consume Chinook and steelhead in the upper basin, northern pikeminnow, walleyes, and smallmouth bass have the potential for significantly affecting the abundance of juvenile anadromous fish (Gray and Rondorf 1986; Bennett 1991; Poe et al. 1994; Burley and Poe 1994). These are large, opportunistic predators that feed on a variety of prey and switch their feeding patterns when spatially or temporally segregated from a commonly consumed prey.

Most adult salmonids within the upper basin are opportunistic feeders and are, therefore, capable of preying on juvenile Chinook and steelhead. Those likely to have some effect on the survival of Chinook and steelhead include adult bull trout, rainbow/steelhead trout, cutthroat trout, brook trout, and brown trout. Of these, bull trout and rainbow trout are probably the most important. These species occur together with Chinook and steelhead in most tributaries, hence the probability for interaction is high. The presence of both fluvial and adfluvial stocks of bull trout in the region further increases the likelihood for interaction there.

Predation by piscivorous birds on juvenile anadromous fish may represent a large source of mortality. Fish-eating birds that occur in the upper basin include great blue herons (*Ardea herodias*), gulls (*Larus* spp.), osprey (*Pandion haliaetus*), common mergansers (*Mergus merganser*), American dippers (*Cinclus mexicanus*), cormorants (*Phalacrocorax* spp.), Caspian terns (*Sterna caspia*), belted kingfishers (*Ceryle alcyon*), common loons (*Gavia immer*), western grebes (*Aechmophorus occidentalis*), black-crowned night herons (*Nycticorax nycticorax*), and bald eagles (*Haliaeetus leucocephalus*) (T. West, Chelan PUD, pers. comm.).

These birds have high metabolic rates and require large quantities of food relative to their body size. In the Columbia River estuary, avian predators consumed an estimated 16.7 million smolts (range, 10 to 28.3 million smolts), or 18% (range, 11 to 30%) of the smolts reaching the estuary in 1998 (Collis et al. 2000). Caspian terns consumed primarily salmonids (74% of diet mass), followed by double-crested cormorants (*P. auritus*) (21% of diet mass) and gulls (8% of diet mass). The NMFS (2000) identified these species as the most important avian predators in the Columbia River basin.

Mammals may be an important agent of mortality to Chinook and steelhead in the upper basin. Predators such as river otters (*Lutra canadensis*), raccoons (*Procyon lotor*), mink (*Mustela vison*), and black bears (*Ursus americanus*) are common in the upper basin. These animals, especially river otters, are capable of removing large numbers of salmon and trout (Dolloff 1993). Black bears consume large numbers of salmon, but generally scavenge post-spawned salmon.

Pinnipeds, including harbor seals (*Phoca vitulina*), California sea lions (*Zalophus californianus*), and Stellar sea lions (*Eumetopia jubatus*) are the primary marine mammals preying on Chinook and steelhead originating from the Upper Columbia basin (Spence et al. 1996). Pacific striped dolphin (*Lagenorhynchus obliquidens*) and killer whale (*Orcinus orca*) may also prey on adult Chinook and steelhead. Seal and sea lion predation is primarily in saltwater and estuarine environments though they are know to travel well into freshwater after migrating fish. All of these predators are opportunists, searching out locations where juveniles and adults are most vulnerable.

3.15 Disease and Parasitism

Chinook and steelhead can be infected by a variety of bacterial, viral, fungal, and microparasitic pathogens. Numerous diseases may result from pathogens that occur naturally in the wild or that may be transmitted to wild fish via infected hatchery fish. Among these are bacterial diseases, including bacterial kidney disease (BKD), columnaris, furunculosis, redmouth disease, and coldwater disease; virally induced diseases, including infectious hepatopoietic necrosis (IHN), infectious pancreatic necrosis (IPNV), and erythrocytic inclusion body syndrome (EIBS); protozoan-caused diseases, including ceratomyxosis and dermocystidium; and fungal infections, such as saprolegnia (Bevan et al. 1994).

Chinook in the Columbia River have a high incidence of BKD (Chapman et al. 1995). Incidence appears higher in spring Chinook (Fryer 1984), and can be a major problem in hatchery-reared Chinook in the upper Columbia region (Chapman et al. 1995). Viral infections such as IPNV have been detected in hatchery steelhead in the upper Columbia region (Chapman et al. 1994). Other epizootics, including *Ceratomyxa shasta* and tuberculosis, are endemic to the Columbia River basin, but it is unknown if these affect the production of Chinook and steelhead in the upper Columbia region.

Generally, one thinks of epizootics killing fish outright. However, sublethal chronic infections can impair the performance of Chinook and steelhead in the wild, thereby contributing secondarily to mortality or reduced reproductive success. Fish weakened by disease are more sensitive to other environmental stresses. Additionally, they may become more vulnerable to predation (Hoffman and Bauer 1971), or less able to compete with other species. For example, both Hillman (1991) and Reeves et al. (1987) found that water temperature affected interactions between redside shiners and the focal species. Both researchers noted that outcomes of interactions were, in part, related to infection with *F. columnaris*. In their studies, most Chinook and steelhead were infected at warmer temperatures, whereas shiners showed a higher incidence of infection at cooler temperatures.

3.16 Competition

As noted in the Ecological Interactions section, competition among organisms occurs when two or more individuals use the same resources, and when availability of those resources is limited (Pianka 2000). Although competition is difficult to demonstrate, a few studies conducted within the Upper Columbia Basin indicate that competition may affect the production of Chinook salmon and steelhead in the basin.

3.16.1 Chinook/Steelhead

It is possible that competition may occur between juvenile Chinook and steelhead along the margins of their spatial and temporal distributions. Hillman et al. (1989a, 1989b) investigated the interaction between these species in the Wenatchee River between 1986 and 1989. They reported that Chinook and steelhead used dissimilar daytime and nighttime habitat throughout the year.

During the daytime in summer and autumn, juvenile Chinook selected deeper and faster water than steelhead. Chinook readily selected stations associated with brush and woody debris for cover, while steelhead primarily occupied stations near cobble and boulder cover. During winter days, Chinook and steelhead used similar habitat, but Hillman et al. (1989a) did not find them together. At night, during both summer and winter, Hillman et al. (1989b) found that both species occupied similar water velocities, but subyearling Chinook selected deeper water than steelhead.

Within smaller streams, Hillman and Miller (2002) found that Chinook were more often associated with pools and woody debris during the summer, while steelhead occurred more frequently in riffle habitat. Hillman et al. (1989a, 1989b) concluded that interaction between the two species would not strongly negatively affect production of either species, because disparate times of spawning tended to segregate the two species. This conclusion is consistent with the work of Everest and Chapman (1972) in Idaho streams.

3.16.2 Redside shiners

Under appropriate conditions, well-coordinated interaction may also occur between redside shiners and juvenile Chinook and steelhead. Hillman (1991) studied the influence of water temperature on the spatial interaction between juvenile Chinook and redside shiners in the field and laboratory. In the Wenatchee River during summer, Hillman (1991) noted that Chinook and shiners clustered together, and that shiners were aggressive toward salmon. He reported that the shiners used the more energetically profitable positions, and that they remained closer than Chinook to instream and overhead cover.

In laboratory channels, shiners affected the distribution, activity, and production of Chinook in warm (64-70°F or 18-21°C) water, but not in cold (54-60°F or 12-15°C) water (Hillman 1991). In contrast, Chinook influenced the distribution, activity, and production of shiners in cold water, but not in warm water. Reeves et al. (1987) documented similar results when they studied the interactions between redside shiners and juvenile steelhead. Although Hillman (1991) conducted his fieldwork in the lower Wenatchee River, shiners are also present in the Entiat, Methow, and Okanogan rivers, and are abundant in the mainstem Columbia River. At warmer temperatures, shiners likely negatively affect the production of Chinook salmon and steelhead in the upper basin.

3.16.3 Coho salmon

It is possible that the re-introduction of coho salmon into the Upper Columbia Basin may negatively affect the production of Chinook and steelhead. One of the first studies in the upper basin that addressed effects of coho on Chinook and steelhead production was conducted by Spauling et al. (1989) in the Wenatchee River.

This work demonstrated that the introduction of coho into sites with naturally produced Chinook and steelhead did not affect Chinook or steelhead abundance or growth. However, because Chinook and coho used similar habitat, the introduction of coho caused Chinook to change habitat. After removing coho from the sites, Chinook moved back into the habitat they used prior to the introduction of coho.

Steelhead, on the other hand, remained spatially segregated from Chinook and coho throughout the study. More recent studies conducted by Murdoch et al. (2002) found that juvenile coho, Chinook, and steelhead used different microhabitats in Nason Creek, and at the densities tested, coho did not appear to displace juvenile Chinook or steelhead from preferred microhabitats.

In addition, Mullan et al. (1992) studied the growth and survival of juvenile coho, chinook, and steelhead in Icicle Creek and concluded that little interaction was apparent among age-0 chinook salmon, coho salmon, and steelhead, and that the introduced coho did not negatively affect the abundance or growth of chinook and steelhead.

These studies indicate that the re-introduction of coho should have little to no effect on the production of Chinook and steelhead.

3.16.4 Various salmonids

Most adult salmonids within the upper basin are capable of preying on juvenile Chinook and steelhead. Those likely to have some effect on the survival of Chinook and steelhead include adult bull trout, rainbow/steelhead trout, cutthroat trout, brook trout, and brown trout. Because brown trout are rare in the region, they probably have little effect on the survival of Chinook and steelhead.

The other salmonids often occur in the same areas as Chinook and steelhead, and are known to be important predators of Chinook and steelhead (Mullan et al. 1992). Of these, bull trout and rainbow trout are probably the most important. These species occur together in most tributaries; hence, the probability for interaction is high. The presence of both fluvial and adfluvial stocks of bull trout in the region further increases the likelihood for interaction there.

Bull trout are opportunistic feeders and will eat just about anything including squirrels, birds, ducklings, snakes, mice, frogs, fish, and insects (Elliott and Peck 1980; Goetz 1989); although, adult migrant bull trout primarily eat fish. Because adult migrant bull trout occur throughout the upper basin, including the mainstem Columbia River (Stevenson et al. 2003), they likely prey on juvenile Chinook and steelhead.

In the upper Wenatchee basin, Hillman and Miller (2002) noted that juvenile Chinook and steelhead were rare in areas where adult bull trout were present. Like northern pikeminnow, adult bull trout frequent the tailrace areas of upper Columbia dams. These areas provide concentrated prey items that include juvenile Chinook and steelhead.

It is likely that adult bull trout prey heavily on migrant salmon and steelhead in these areas. Indeed, Stevenson et al. (2003) found bull trout staging near the Wells Hatchery outfall, apparently seeking opportunistic feeding opportunities. As the number of bull trout increase in the upper basin, the interaction between them and Chinook and steelhead will increase.

Rainbow/steelhead trout feed on Chinook fry in the upper basin. In the Wenatchee River, for example, Hillman et al. (1989a) observed both wild and hatchery rainbow/steelhead feeding on Chinook fry. Predation was most intense during dawn and dusk. At these times, rainbow/ steelhead occupied stations immediately adjacent to aggregations of Chinook. Hillman et al. (1989a) noted that within the prey cluster, the largest, light-colored Chinook were closest to shelter and seldom eaten. Small, darker-colored Chinook were farther from escape cover and usually eaten by predators. Hillman et al. (1989a; 1989b) suggest that predator-mediated interaction for shelter was strong and contributed to the rapid decline in Chinook numbers in May. Although this work was done in the Wenatchee River, the results probably hold for other tributaries where the two species occur together.

Although adult salmonids prey on juvenile Chinook and steelhead in the upper basin, the predation rate is unknown. Because of the abundance of both bull trout and rainbow/steelhead trout in the upper basin, it is reasonable to assume that large numbers of fry are consumed by these fish.

3.16.5 American shad

A potentially important source of exploitative competition occurring outside the geographic boundary of the ESUs may be between the exotic American shad and juvenile Chinook and steelhead. Changes in stream flow in the Columbia River system have resulted in increased plankton production that has apparently increased the success of introduced shad.

Shad prey on the most abundant foods (Walburg 1956; Levesque and Reed 1972). Shad in the Columbia River estuary consume amphipods, calanoid copepods (*Neomysis mercedis*), cladocerans (*Daphnia* spp.), and insects (Durkin et al. 1979). Juvenile salmonids eat the same foods (McCabe et al. 1983). Palmisano et al. (1993a, 1993b) concluded that increased numbers of shad likely compete with juvenile salmon and steelhead.

Predation

Fish, mammals, and birds are the primary natural predators of Chinook and steelhead in the Upper Columbia basin. Although the behavior of Chinook and steelhead precludes any single predator from focusing exclusively on them, predation by certain species can nonetheless be seasonally and locally important. Below is a discussion on the importance of specific predators on the production of Chinook and steelhead in the Upper Columbia basin.

3.16.6 Smallmouth bass

Smallmouth bass were introduced into the Columbia River before 1900 (Poe et al. 1994). Given their behavioral characteristics, it is assumed that they could significantly affect the abundance of juvenile Chinook and steelhead. In spring and early summer, they inhabit rocky shoreline areas that are also used by juvenile salmonids (Scott and Crossman 1973; Wydoski and Whitney 1979).

Studies in Columbia basin reservoirs and Lake Sammamish, Washington, showed that smallmouth bass were highly predacious on outmigrating juvenile salmonids (Gray et al. 1984; Gray and Rondorf 1986). In contrast, studies by Bennett et al. (1983) and Zimmerman (1999) found that even though salmonids were present in Snake and Columbia River reservoirs, they were less important in the diets of smallmouth bass than other fish.

Smallmouth bass commonly consumed sculpins, minnows, suckers, and troutperches in impounded and unimpounded reaches of the lower Columbia and lower Snake rivers during the outmigration of juvenile anadromous salmonids (Zimmerman 1999).

Sampling in the Upper Columbia Basin indicates that smallmouth bass are relatively rare (Dell et al. 1975; Burley and Poe 1994). Burley and Poe (1994) described studies that assessed the relative abundance of northern pikeminnow, walleye, and smallmouth bass in the Rocky Reach project area. Smallmouth bass constituted only 5% of the catch; northern pikeminnow and walleye made up 91% and 4% of the respective catch. Most (63%) smallmouth bass resided in the tailrace.

Very few (3%) were captured mid-reservoir. Mullan (1980), Mullan et al. (1986), and Bennett (1991) suggested that few smallmouth bass occur within the Upper Columbia because of low ambient water temperatures. Optimum growth temperatures for smallmouth bass range from 79-84°F (26-29°C) (Armour 1993a).

Because Upper Columbia reservoirs function as a cold-tailwater to the reservoir of Grand Coulee Dam, optimal temperatures for bass occur primarily in warm backwaters (Mullan et al. 1986; Bennett 1991). The typical low water temperatures in the project area result in late spawning times, slow fry and fingerling growth, and small body size of smallmouth bass entering the first winter. This contributes to high overwinter mortality of juvenile smallmouth bass (Bennett 1991).

One could theorize that if sustained removals of northern pikeminnow significantly reduce mortality of juvenile salmonids in the project area, predation by smallmouth bass may be enhanced because of increased availability of juvenile salmonid prey. Studies in the lower Columbia and Snake rivers found that smallmouth bass did not respond to sustained removals of northern pikeminnow (Ward and Zimmerman 1999). Smallmouth bass density, year-class strength, consumption of juvenile salmonids, survival, growth, and relative weight did not increase concurrent with removals of northern pikeminnow. Likewise, it is unlikely that smallmouth bass will respond to sustained removals of northern pikeminnow in the Upper Columbia basin.

Because smallmouth bass are not abundant in the upper Columbia, they probably have a minor influence on the survival of juvenile Chinook and steelhead. Of the anadromous fish in the project area, subyearling summer/fall Chinook may be consumed more readily because their habitats overlap seasonally with smallmouth bass, and because the subyearlings are ideal forage size for adult smallmouth bass (Poe et al. 1994).

3.16.7 Walleye

According to Li et al. (1987), walleye recently invaded the Columbia River from the reservoir of Grand Coulee Dam, where they are now very abundant. This fish is a large, schooling predator, unlike the native fauna, and its affect on juvenile Chinook and steelhead could be significant because of the potential for depensatory predatory-prey interactions.

Gray et al. (1984) found a high frequency of occurrence (42%) of juvenile salmonids in the stomachs of walleyes collected in the John Day tailrace during spring. In John Day Reservoir, however, Maule (1982) reported that walleyes ate few juvenile salmonids, and suggested that the probable reason was the spatial and temporal segregation of the species when walleyes were feeding most actively. Perhaps the reason that walleyes eat more juvenile salmonids in the tailrace is because the dam creates habitat that increases potential for spatial overlap, and, therefore, predation between the species. This is supported by the high occurrence of juvenile salmonids in walleye stomachs collected between 1800 and 2400 hours (Gray et al. 1984), when the greatest fraction of smolts move through the powerhouse at John Day Dam (Sims et al. 1981), and when walleyes feed most heavily (Maule 1982).

Work by Zimmerman (1999) in impounded and unimpounded reaches of the lower Columbia River indicated that walleyes, like smallmouth bass, more commonly consumed sculpins, suckers, minnows, and troutperches during the outmigration of juvenile salmonids. This comports with the observations of Vigg et al. (1991), who estimated that nonsalmonid consumption rates of walleye were similar to those of smallmouth bass, and exceeded those of northern pikeminnow in John Day Reservoir.

Walleyes are relatively rare in the upper Columbia (Dell et al. 1975; Burley and Poe 1994). Burley and Poe (1994) reported that walleyes made up only 4% of the catch of the major predators in the Rocky Reach project area; the other two major predators, northern pikeminnow and smallmouth bass, made up 91% and 5% of the respective catch.

Most of the walleyes were captured in the tailrace. Few were captured in the forebay or mid-reservoir. The abundance of walleye appears to be limited by poor recruitment and low turbidity (Bennett 1991). Bennett (1991) reported that the most significant factor limiting abundance of walleyes is the short reservoir retention times (5.5-0.7 days), especially at the time of larvae abundance. High mortality and low food abundance for larvae probably limit recruitment of walleyes in reservoirs. In addition, low water turbidity likely affects the temporal and spatial distribution of feeding and reproduction of walleyes.

Walleyes attain maximum population sizes in shallow, large, turbid waters (Scott and Crossman 1973). They prefer turbid water because their eyes are sensitive to bright light. In clear waters, walleyes retain contact with the substrate during the day (Ryder 1977) and increase activity as light conditions decrease in the evening. Peak periods of activity in clear waters are dusk and dawn (Kelso 1976).

Mullan et al. (1992) believed that low water temperatures may limit recruitment of walleyes in the upper Columbia. Optimal water temperatures for embryo incubation range from 9-15°C (48-59°F) (Armour 1993b). Optimal growth temperatures for juveniles and adults range from 22-28°C (72-82°F) and 20-28°C (68-82°F), respectively (Armour 1993b). These thermal requirements suggest that water temperatures in the project area may not increase sufficiently fast or high enough for successful incubation, hatching, and rearing (Mullan et al. 1986; Bennett 1991). Successful incubation, hatching, and rearing may occur in backwater areas.

Because walleyes are not abundant in the upper Columbia, they probably do not significantly reduce the abundance of juvenile Chinook or steelhead in the area. Walleye predation on juvenile salmonids is probably greatest on subyearling summer/fall Chinook. Gray et al. (1984) found that about 80% of the juveniles identified in walleye stomachs were subyearlings, probably a result of their smaller size. Subyearling Chinook spend more time in shallower water than yearling spring Chinook, also increasing the likelihood of encountering walleyes.

3.16.8 Northern pikeminnow

The northern pikeminnow is a native cyprinid widely distributed throughout the Columbia River system (Mullan et al. 1986). It is the dominant predator of juvenile salmonids in the system, and predation by this species is clearly important compared to other sources of mortality (Poe et al. 1991; Rieman et al. 1991; Vigg et al. 1991; Ward and Zimmerman 1999; Zimmerman 1999).

Petersen (1994) estimated the annual loss of juvenile salmonids to predation by northern pikeminnow in John Day Reservoir to be 1.4 million, or approximately 7.3% of all juvenile salmonids entering the reservoir. Predation varies throughout the system and is often highest near dams (Ward et al. 1995). Although the work by Gadomski and Hall-Griswold (1992) suggests that northern pikeminnow prefer dead juvenile Chinook to live ones, Petersen (1994) found that 78% of juvenile salmonids eaten by northern pikeminnow near a dam were consumed while alive.

Ward et al. (1995) estimated that 48% of predation occurs in mid-reservoir areas away from dams, where juvenile salmonids are presumably alive and uninjured when consumed. Of the estimated 200 million juvenile salmonids that emigrate annually through the Columbia River system, about 16.4 million (8%) are consumed by northern pikeminnow (Beamesderfer et al. 1996).

Northern pikeminnow are abundant in the Upper Columbia Basin (Dell et al. 1975; Mullan 1980; Mullan et al. 1986; Bennett 1991; Burley and Poe 1994), and large numbers pass through the fishways at dams. Of the three major predators in the Rocky Reach project area (northern pikeminnow, smallmouth bass, and walleye), northern pikeminnow made up 91% of the catch (Burley and Poe 1994). These fish were most abundant in the mid-reservoir (45% of the total catch of northern pikeminnow), with the remaining catch of northern pikeminnow split equally between the forebay and tailrace.

At other dams in the Upper Columbia basin, Burley and Poe (1994) found larger numbers of northern pikeminnow in the tailrace areas. Northern pikeminnow in the Rocky Reach project area averaged 296 millimetres (12 inches) fork length (range, 115-515 millimetres [4.5-20 inches]) (Burley and Poe 1994). Vigg et al. (1991) reported that juvenile salmonids are the major dietary component of northern pikeminnow larger than 250-mm (10 inches) fork length; therefore, one would assume that northern pikeminnow could significantly affect the abundance of juvenile Chinook and steelhead in the upper basin.

Burley and Poe (1994) summarize studies that assessed the significance of northern pikeminnow predation in the Upper Columbia region. They reported that northern pikeminnow in the Rocky Reach project area consumed primarily fish during the spring and summer; crustaceans, molluscs, insects, and plants were also consumed. Typically, the highest percentage of gut contents consisting of fish occurred in pikeminnows feeding in the tailrace and forebay areas. Juvenile salmonids were a significant component of northern pikeminnow diets, especially in tailrace areas.

The concern that northern pikeminnow could significantly affect the abundance of Chinook and steelhead in the upper basin, resulted in the initiation of a pikeminnow population reduction program. Since its initiation (1994), the program has removed well over 75,000 northern pikeminnow from Rocky Reach and Rock Island project areas (West 2000). At Rocky Reach, the program removed 44,743 (average, 6,400 per year; range, 2,482-9,633) pikeminnow. The number of northern pikeminnow ascending fish ladders at both dams has declined and catch rates have decreased (West 2000).

It is reasonable to assume that the reduction in numbers of northern pikeminnow has increased survival of juvenile Chinook and steelhead in the upper basin. In the lower Columbia and Snake rivers, potential predation on juvenile salmonids by northern pikeminnow decreased 25% after a pikeminnow removal program was implemented there (Friesen and Ward 1999). Friesen and

Ward (1999) estimated a reduction in potential predation of 3.8 million juvenile salmon (representing 1.9% of the total population).

Knutsen and Ward (1999) found no evidence that the surviving pikeminnow compensated for removals. That is, estimates of relative weight, growth, and fecundity of pikeminnow were similar to estimates made before pikeminnow removals. Zimmerman and Ward (1999) concluded that consumption of juvenile salmonids by surviving pikeminnow has not increased in response to pikeminnow removal. It is likely that similar results occur within the Upper Columbia basin.

Northern pikeminnow are abundant in the Upper Columbia basin, and have the potential to significantly affect the abundance of juvenile Chinook and steelhead. They consume large numbers of juvenile salmonids, primarily those concentrated in the tailrace and forebay areas during the spring outmigration. They also consume large numbers of juvenile salmonids (probably summer/fall Chinook) during summer.

Currently, the factor limiting the abundance of northern pikeminnow in the upper basin is the sustained population reduction program. The program has removed large numbers of northern pikeminnow from the project area. As a result, dam passage counts of pikeminnow have decreased. This has likely resulted in increased survival of juvenile anadromous fish in the project area.

3.16.9 Sculpins

Sculpins are native and relatively common in the upper basin (Dell et al. 1975; Mullan 1980; Burley and Poe 1994). Although sculpins are not considered a major predator of outmigrating anadromous fish, they do prey on small Chinook and steelhead (Hunter 1959; Patten 1962, 1971a, 1971b; Hillman 1989).

In the Wenatchee River, Hillman (1989) noted that large concentrations (20 fish/m$^{2)}$ of juvenile Chinook and steelhead occupied inshore, shallow, quiet-water positions on the streambed during the night. Hillman (1989) found that many sculpins moved into these areas at night and preyed heavily on Chinook and steelhead fry. Predation on fry appeared to be limited to sculpins larger than 85 millimetres (3.3 inches) and ceased when prey reached a size larger than 55 millimetres (2 inches). The number of fry eaten per night appeared to be related to sculpin size, with the largest sculpins consuming the most fry per individual.

Because sculpins are abundant in Upper Columbia River tributaries, they are likely an important agent of mortality of Chinook, and of steelhead eggs and fry. As Chinook and steelhead fry grow, they are released from this source of mortality. It is unknown what fraction of the Chinook and steelhead population is removed by sculpins.

3.16.10 White sturgeon

White sturgeon, a native species, are not abundant in the upper basin (Mullan 1980; Mullan et al. 1986; Gray and Rondorf 1986; DeVore et al. 2000). According to Mullan (1980), sturgeon were perhaps the most important predator on young and adult salmon, as well as other fishes. This is not the case now because of greatly reduced sturgeon abundance.

Using setlines and gill nets, DeVore et al. (2000) found few sturgeon in the Upper Columbia River. In Rock Island Reservoir, a total of 95 overnight setlines captured only four sturgeon. The researchers did not sample in Rocky Reach Reservoir and used only setlines in Rock Island Reservoir. Sturgeon in Rock Island Reservoir ranged in lengths from 144-192 centimetres (57-76 inches) and in weight from 31-57 kilograms (68-126 pounds). The researchers aged two fish, one at 17 years and the other at 30 years.

White sturgeon are occasionally captured during the northern pikeminnow reduction program. For example, anglers collected two sturgeon in 1998, one at Rocky Reach Dam and another at Rock Island Dam (West 1999). Angling in 1999 captured three sturgeon at Rock Island Dam (West 2000). No sturgeon were captured at Rocky Reach Dam in 1999. All sturgeon captured during the northern pikeminnow control program were 91 centimetres (36 inches) or larger (T. West, Chelan PUD, pers. comm.).

White sturgeon are opportunistic bottom feeders, as indicated by morphological adaptations that include ventral barbels and a ventral, protrusible, sucker-like mouth (Wydoski and Whitney 1979; Ford et al. 1995). Juveniles predominantly eat chironomids and to a lesser degree, zooplankton, molluscs, and immature mayflies, caddisflies, and stoneflies (Scott and Crossman 1973). In the lower Columbia River, juveniles primarily ate the tube-dwelling amphipod *Corophium salmonis* (McCabe et al. 1993).

Individuals larger than 48 centimetres (18 inches) in length eat primarily fish (Scott and Crossman 1973; Ford et al. 1995). In the Kootenai River, white sturgeon larger than 80 centimetres (32 inches) fed on fish (whitefish, suckers, and other unidentified fish), aquatic insects, snails, clams, leeches, and chironomids (Partridge 1983).

DeVore et al. (2000) concluded that the white sturgeon in the Upper Columbia region are recruitment-limited because spawning habitat appears to be absent and no juveniles were found. Spawning coincides with peak flows during spring and early summer. Mature adults typically spawn in swift water (mean water column velocity, 0.8-2.8 m/s) over large substrate (cobble, boulder, or bedrock) (Parsley et al. 1993; Ford et al. 1995). In the upper basin, these conditions likely exist just downstream from Wells Dam and Rocky Reach Dam. It is unknown if white sturgeon spawn in these areas.

Because white sturgeon are rare in the upper basin, they probably do not significantly affect the abundance of juvenile Chinook or steelhead. Small Chinook that rear in the Columbia River may be vulnerable to predation by white sturgeon. Theoretically, this would occur primarily at night when Chinook and steelhead are stationed on the streambed.

3.16.11 Birds

Predation by piscivorous birds on juvenile anadromous fish may represent a large source of mortality. Birds have high metabolic rates and require large quantities of food relative to their body size.

In the Columbia River estuary, avian predators consumed an estimated 16.7 million smolts (range, 10-28.3 million smolts), or 18% (range, 11-30%) of the smolts reaching the estuary in 1998 (Collis et al. 2000). Caspian terns consumed primarily salmonids (74% of diet mass), followed by double-crested cormorants (21% of diet mass) and gulls (8% of diet mass). The NMFS (2000) identified these species as the most important avian predators in the Columbia River basin.

Currently, there is little information on the effects of bird predation on the abundance of juvenile Chinook and steelhead in the upper basin. Fish-eating birds that occur in the region include great blue herons, gulls, osprey, common mergansers, American dippers, cormorants, Caspian terns, belted kingfishers, common loons, western grebes, black-crowned night herons, and bald eagles (T. West, Chelan PUD, pers. comm.).

According to Wood (1987a, 1987b), the common merganser limited salmon production in nursery areas in British Columbia. Wood found that during smolt migrations, mergansers foraged almost exclusively on juvenile salmonids (Wood 1987a). Maximum mortality rate declined as fish abundance increased (i.e., depensatory mortality), and did not exceed 10% for any salmonid species. Wood (1987b) also estimated that young mergansers consumed almost one-half pound of subyearling Chinook per day. A brood of ten ducklings, therefore, could consume between four and five pounds of fish daily during the summer.

The loss of juvenile Chinook and steelhead to gulls is potentially significant. Ruggerone (1986) studied the consumption of migrating juvenile salmon and steelhead below Wanapum Dam, and found that the foraging success of gulls averaged 65% during bright light conditions, and 51% during the evening. The number of salmonids consumed ranged from 50 to 562 fish/hour. Ruggerone (1986) estimated that the number of salmonids consumed by gulls foraging downstream from the turbines during 25 days of peak salmonid migration was about 111,750 to 119,250 fish, or 2% of the estimated spring migration. Ruggerone (1986) noted that gulls consumed some salmonids that had been killed when passing through the turbines.

Cormorants may take large numbers of juvenile Chinook and steelhead in the upper basin. Roby et al. (1998) estimated that cormorants in the estuary consumed from 2.6 to 5.4 million smolts in 1997, roughly 24% of their diet, most being hatchery fish. Although Caspian terns are not common in the upper basin, there is evidence that they consume fish from the area. Bickford (Douglas PUD, pers. comm.) found both PIT-tags and radio tags at a Caspian Tern nesting area near Moses Lake. Tag codes indicated that consumed fish were from the Upper Columbia region.

Although there are no estimates of the losses associated with bird predation in the Upper Columbia basin, it appears that bird predation can significantly affect the survival of juvenile Chinook and steelhead. Accordingly, the PUDs have implemented bird harassment measures, and in some cases, placed piano wire across tailraces. The degree to which these measures have reduced predation on juvenile anadromous fish is unknown at this time, but they have reduced bird predation on fish in the region (T. West, Chelan PUD, pers. comm.).

3.16.12 Mammals

No one has studied the effects of mammals on numbers of Chinook and steelhead in the upper Columbia basin. Observations by BioAnalysts (unpublished data) indicate that river otters occur throughout the region. BioAnalysts (unpublished data) found evidence of otters fishing the Wenatchee, Chiwawa, Entiat, and Methow rivers, and Icicle Creek.

Otters typically fished in pools with LWD. According to Hillman and Miller (2002), juvenile Chinook are most abundant in these habitat types; thus, the probability for an encounter is high.

Dolloff (1993) examined over 8,000 otoliths in scats of two river otters during spring 1985 and found that at least 3,300 juvenile salmonids were eaten the otters in the Kadashan River system, Alaska. He notes that the true number of fish eaten was much higher, as it is unlikely that searchers found all the scats deposited by the otters.

Other predators, such as raccoon and mink also occur in tributaries throughout the Upper Columbia basin. Their effects on numbers of Chinook and steelhead are unknown.

Black bears are relatively common in the upper Columbia basin, and frequent streams used by spawning salmon during autumn. Studies have shown that salmon are one of the most important meat sources of bears, and that the availability of salmon greatly influences habitat quality for bears at both the individual level and the population level (Hilderbrand et al. 1999; Reimchen 2000).

Observations by crews conducting Chinook spawning surveys in the upper basin indicate that bears eat Chinook, but it is unknown if the bears remove pre-spawned fish, or are simply scavenging post-spawned fish. Regardless, there is no information on the role that bears play in limiting survival and production of Chinook and steelhead in the upper basin.

Pinnipeds, including harbor seals, California sea lions, and Stellar sea lions are the primary marine mammals preying on Chinook and steelhead originating from the Upper Columbia basin (Spence et al. 1996). Pacific striped dolphin and killer whale may also prey on adult Chinook and steelhead. Seal and sea lion predation is primarily in saltwater and estuarine environments, though they are know to travel well into freshwater after migrating fish. All of these predators are opportunists, searching out locations where juveniles and adults are most vulnerable.

Although there are no estimates of the losses associated with mammal predation in the upper Columbia basin, it appears that mammals can significantly affect the survival of Chinook and steelhead, especially in the estuary and near-shore ocean environments.

3.17 Habitat Conditions and Limiting Factors to Fish Production

Both naturally occurring and human-induced habitat conditions affect fish spawning, rearing and passage within the Methow subbasin. While the Methow region has accommodated human habitation for close to 7,500 years, substantial changes to overall habitat conditions caused by human activities have taken place in the mid and lower reaches of the basin during the last century.

Three habitat factors identified as limiting to salmon, steelhead, and bull trout in the WSCC analysis, require additional research (Williams 2000). Those factors are:

- the extent to which irrigation diversion affects natural runoff patterns, water temperature, chemical enrichment, and fish production;
- the role that LWD played historically within the Methow in producing fish; and,
- the affect of man’s placement of 35 miles of riprap on fish production

Natural factors

Naturally occurring habitat conditions can cause both benefit and harm to fish species. Some tributaries within the Methow subbasin experience naturally occurring seasonal low flows and occasional instances of dewatering. Some creeks and streams throughout the subbasin, such as the mainstem Methow upstream of Weeman Bridge, are subject to naturally occurring seasonal dewatering. In the upper elevations of the watershed, avalanches, landslides, flooding and creek icing can both negatively and positively affect salmonid habitat.

Throughout the subbasin, naturally occurring influences, such as: fire, which can contribute to erosion and sediment delivery; high stream flow events, which potentially alter stream channels and structure, and; low stream flow, which can limit fish passage and strand LWD, play a role in altering and defining habitat. Although the short-term effects of naturally occurring habitat changes like fire, avalanches, and flooding tend to be detrimental to fish and wildlife, in the long run these changes are often beneficial.

Landslides and avalanches in the upper reaches of the drainage periodically alter habitat conditions, sometimes destroying, and at other times creating, rearing and spawning habitat. Harsh winter temperatures in the Methow subbasin also play a role in limiting productive fish habitat. Additionally, fire events have altered habitat in many portions of the watershed.

Harsh winter temperatures also contribute to seasonal limitations in water quantity. Water quality, primarily in terms of temperature, is to a lesser degree, also a limiting factor in the subbasin. In general, stream temperatures within the basin are conducive to fish health; although, elevated temperatures have been noted in select reaches, and in winter, freezing creeks pose a limiting factor in some reaches.

The reduction in the number of beaver historically found within the watershed has potentially detracted from overall spawning and rearing habitat by eliminating pools, LWD recruitment, and decreasing water and nutrient storage capacity, previously facilitated by beaver activity. The overall decrease in nutrients, caused by lack of large numbers of salmon carcasses throughout the watershed, has potentially contributed to reductions of both fish and wildlife abundance.

Anthropogenic Effects

Over the course of the last century, a number of human-induced physical changes have redefined the quality and quantity of aquatic and terrestrial habitat found in the mid-upper and lower reaches of the Methow subbasin. Most significant among these changes is habitat fragmentation compounded by degradation in overall habitat quality, the result of historic and current agricultural practices, timber management, mismanaged grazing, mining, and commercial and residential development activities.

Combinations of these activities have contributed to:

- alteration, reduction, and elimination of riparian habitat;
- alteration and elimination of floodplains;
- degradation of instream habitat through sediment loading, elimination of LWD, and loss of stream bank integrity;
- construction of artificial barriers to fish passage, such as push up dams, diversions, and ill-functioning fish screens and culverts;
- increased road densities and related erosion, as well as loss of canopy cover; and

- changes to overall vegetative composition and forage availability in both riparian and upland areas.

Irrigation and low flows

Irrigated agriculture took root in the Methow Valley around 1887. By the 1890s farmers were regularly diverting water from the Methow River and other tributaries to grow crops in the valley. Irrigated land has always comprised a relatively small percentage of the basin's total acreage (currently about 1.7%)(Mullan et al. 1992b).

In some areas of the Methow basin, irrigation water is still delivered via unlined open ditches. Whether irrigation ditches and diversions contribute to stream dewatering or groundwater recharge, is a matter of great concern and speculation in the Methow subbasin, but the exact nature of that relationship is not fully understood. Substantial future growth in agricultural activity within the Methow Valley is not anticipated; nevertheless, ongoing small-scale conversions of riparian habitat to residential, pasture and agricultural uses are likely to continue.

Seasonal, naturally occurring, and human-influenced low stream flows and occasional dewatering can alter fish passage to upstream spawning and rearing habitat in the Methow. Low flows also affect water quality by contributing to higher stream temperatures in summer months, particularly prevalent in the lower reaches of the Methow. In addition, low stream flows tend to concentrate any toxic materials or other contaminants entrained in the stream flow.

The effects of the myriad of small irrigation diversion and hydropower projects throughout the range of bull trout are likely of even greater significance than the large hydropower and flood control projects. Many of these are located further up in watersheds, and either physically block fish passage by means of a structure (i.e., a dam), or effectively block passage by periodically dewatering a downstream reach (e.g., diversion of flows through a penstock to a powerhouse; diversion of flows for the purposes of irrigation). Even if diversions are not so severe as to dewater downstream reaches, reduced flows can result in structural and thermal passage barriers. Other effects include water quality degradation resulting from irrigation return flows and runoff from fields and entrainment of bull trout into canals and fields (MBTSG 1998). Some irrigation diversion structures are reconstituted annually with a bulldozer as “push up” berms and not only affect passage, but also significantly degrade the stream channel. The prevalence of these structures throughout the range of bull trout has resulted in the isolation of bull trout populations in the upper watersheds in many areas.

Bull trout may enter unscreened irrigation diversions and become stranded. Diversion dams without proper passage facilities prevent bull trout from migrating, and may isolate groups of fish (Dorratcaque 1986; Light et al. 1996). Other effects on aquatic habitat include stream channelization and LWD removal (Spence et al. 1996).

Sediments

High road densities, poor road placement and land management practices have contributed to persistent sediment delivery to streams in the middle reaches of the Chewuch River subwatershed (USFS 1994). The lower reaches of the watershed host the greatest concentrations of human activity, and are the site of the much of the basin's recent habitat changes. For example, diking, channelization, and conversion of riparian areas to agricultural and residential uses have occurred throughout much of the lower reaches of the Methow mainstem, Twisp, Chewuch and Lost Rivers, and the middle and lower Methow River subwatersheds.

Migratory fish and many wildlife species depend on intact, complex, and functioning habitat extending over broad geographic ranges to support healthy populations. Resident, non-migratory populations of fish and wildlife also indirectly depend on basin-wide habitat connectivity since migratory species make essential contributions to overall ecosystem balance, such as providing essential nutrients and maintaining predator/prey balances (NPPC 1996). Overarching habitat loss, brought about as a result of human settlement activities within the Columbia River Basin since the early 1800s, and the development of hydropower facilities along the Columbia River, have irrevocably altered both terrestrial and aquatic habitat in the Methow subbasin.

Forest Practices

The middle reaches of the Methow subbasin, particularly areas within the Chewuch River and Goat Creek drainages, exhibit significant habitat degradation as a result of past logging activities. Forest management activities, including timber extraction and road construction, affect stream habitats by altering recruitment of LWD, erosion and sedimentation rates, runoff patterns, the magnitude of peak and low flows, water temperature, and annual water yield (Cacek 1989; Furniss et al. 1991; Wissmar et al. 1994; Spence et al. 1996; Spencer and Schelske 1998; Swanson et al. 1998).

Activities that promote excessive substrate movement reduce bull trout production by increasing egg and juvenile mortality and reducing or eliminating habitat (e.g., pools filled with substrate), important to later life history stages (Fraley and Shepard 1989; Brown 1992). The length and timing of bull trout egg incubation and juvenile development (typically more than 200 days during winter and spring), and the strong association of juvenile fish with stream substrate make bull trout vulnerable to changes in peak flows and timing that affect channels and substrate (Goetz 1989; Pratt 1992; McPhail and Baxter 1996; MBTSG 1998).

Roads

Roads for logging access and log skidding can, and have locally introduced fine sediments to spring Chinook and summer steelhead habitat. Riparian communities have, at times, been disrupted, reducing shade and availability of LWD. Timber removal alters hydrology of tributaries until regrowth occurs.

Roads constructed for forest management are a prevalent feature on managed forested and rangeland landscapes in the Methow. Roads have the potential to adversely affect several habitat features, (e.g., water temperature, substrate composition and stability, sediment delivery, habitat complexity, and connectivity) (Baxter et al. 1999; Trombulak and Frissell 2000). Roads may also isolate streams from riparian areas, causing a loss in floodplain and riparian function. The aquatic assessment portion of the Interior Columbia Basin Ecosystem Management Project provided a detailed analysis of the relationship between road densities and bull trout status and distribution (Quigley and Arbelbide 1997). The assessment found that bull trout are less likely to use streams in highly roaded areas for spawning and rearing, and do not typically occur where average road densities exceed 1.1 kilometres per square kilometre (1.7 miles per mile^2).

Roads degrade bull trout habitats by: creating flow constraints in ephemeral, intermittent, and perennial channels; increasing erosion and sedimentation; impacting groundwater-streamwater interactions, important to bull trout spawning and rearing habitat; creating passage barriers; channelizing stream reaches, and; reducing riparian vegetation (Furniss et al. 1991; Ketcheson and Megahan 1996; Trombulak and Frissell 2000). Effects are not limited to direct effects on occupied bull trout habitat, but can indirectly affect occupied habitat by altering natural functions in smaller upstream tributaries. For example, Wipfli and Gregovich (2002) identified small headwater tributaries not occupied by salmonids as important sources of aquatic invertebrate production for areas occupied by salmonids downstream. Roads also provide access to many activities, including undesired activities such as illegal fishing and fish stocking, and accidental discharges into streams.

Roads may affect aquatic habitats considerable distances away. For example, increases in sedimentation, debris flows, and peak flows affect streams longitudinally so that the area occupied by a road can be small compared to the entire downstream area subjected to its effects (Jones et al. 2000; Trombulak and Frissell 2000). Upstream from road crossings, large areas of suitable habitats may become inaccessible to bull trout because of fish passage barriers (e.g., culverts).

Livestock Grazing

Improperly managed livestock grazing degrades salmonid habitats, including bull trout habitat, by removing riparian vegetation, destabilizing streambanks, widening stream channels, promoting incised channels, lowering water tables, reducing pool frequency, increasing soil erosion, and altering water quality (Howell and Buchanan 1992; Mullan et al. 1992; Overton et al. 1993; Platts et al. 1993; Uberuaga 1993; Henjum et al. 1994; MBTSG 1995a,b,c; USDA and USDI 1996, 1997). These effects reduce overhead cover, increase summer water temperatures, and promote formation of anchor ice (ice attached to the bottom of an otherwise unfrozen stream, often covering stones, etc.) in winter, and increase sediment in spawning and rearing habitats.

Negative effects of livestock grazing may be minimized if grazing is managed appropriately for conditions at a specific site. Practices generally compatible with the preservation and restoration of fish habitats include fences to exclude livestock from riparian areas, rotation schemes, maintenance of fences, relocation of water, placement of salting facilities away from riparian areas, and use of herders.

Agricultural Practices

Agricultural practices, such as cultivation, irrigation diversions, and chemical application contribute to non-point source pollution in some areas within the range of bull trout (IDHW 1991; WDE 1992; MDHES 1994). Impacts resulting from these practices are as follows: release of sediment, nutrients, pesticides, and herbicides into streams; increased water temperature; reduced riparian vegetation, and; altered hydrologic regimes, typically by reducing flows in spring and summer.

Mining

Mining degrades aquatic habitats used by bull trout by altering water chemistry (e.g., pH), by altering stream morphology and flow, and by altering the substrate composition and benthic insect community composition where in-channel mining activity occurs, causing sediment, fuel, and heavy metals to enter streams (Martin and Platts 1981; Spence et al. 1996; Thomas 1985). The types of mining that occur within the range of bull trout include extraction of hard rock minerals, coal, gas, oil, and sand and gravel. Past and present mining activities have adversely affected bull trout and bull trout habitats in Washington (Johnson and Schmidt 1988; Moore et al. 1991; WDW 1992; Platts et al. 1993; MBTSG 1995a, c, 1996b, c; McNeill et al. 1997; Ramsey 1997).

Development and Urbanization

Residential and commercial development has altered habitat in the subbasin. Approximately 874 building permits were issued in the Methow watershed between 1984 and 1994 (Methow Valley Water Pilot Planning Project Planning Committee 1994). During that time, the majority of development activity occurred in the middle and lower reaches of the watershed.

Residential development has altered stream and riparian habitats through contaminant inputs, stormwater runoff, changes in flow regimes, streambank modification and destabilization, increased nutrient loads, and increased water temperatures (MBTSG 1995b). Indirectly, urbanization within floodplains alters groundwater recharge by rapidly routing water into streams through drains rather than through more gradual subsurface flow (Booth 1991).

Urbanization negatively affects the lower reaches of many of the large rivers and their associated side channels and wetlands. Activities such as: dredging; removing LWD (e.g., snags, logjams, driftwood); installing revetments, bulkheads, and dikes, and; filling side channels have led to the reduction, simplification, and degradation of habitats (Thom et al. 1994; Spence et al. 1996;). Pollutants associated with urban environments such as heavy metals, pesticides, fertilizers, bacteria, and organics (oil, grease) have contributed to the degradation of water quality in streams, lakes, and estuaries (NRC 1996; Spence et al. 1996).

3.17.1 Summary of Limiting Factors

Following is a summary of limiting factors in the Methow subbasin based primarily on the WSCC Limiting Factors Analysis (WSCC 2000) and on the RTT draft report to the UCSRB (RTT 2001).

Habitat Fragmentation compounded by degradation in overall habitat quality

- Alteration and reduction of riparian habitat (fish & wildlife)
- Habitat connectivity (fish & wildlife)
- Instream and floodplain habitat degradation (fish and wildlife)
- Artificial and natural fish passage barriers (fish)
- Land management practices (fish & wildlife)
- Noxious Weeds

Water Quantity and Quality

- Low flows and dewatering (fish & wildlife)
- Temperatures (fish)
- Sediment load (fish)

- Freezing creeks and streams (fish & wildlife)

Additional Key Factors

- Severely reduced numbers of returning naturally produced adults (fish & wildlife)
- Decrease in nutrients (i.e., salmon carcasses [fish & wildlife])
- Presence of brook trout in many Methow subbasin streams and creeks (bull trout)
- Data and knowledge gaps (fish & wildlife)

3.18 The Form and Function of Ecosystem Change

Alteration and Reduction of Riparian Habitat

Loss of riparian areas in the Methow basin because of logging, agriculture, and residential development affects streambanks, water quality, water quantity, and overall habitat complexity; the loss leads to increased erosion, which in turn, increases sedimentation. Riparian habitat losses also contribute to higher water temperatures in summer months and lower temperature in winter months.

Riparian zones play many essential roles in maintaining ecosystem health and integrity. They provide connectivity between aquatic and upland habitats, moderate stream temperature through shading, maintain water quality by performing filtering and bank stabilizing functions, and supply in-stream nutrients through insect and vegetative contributions (Platts 1991; Johnson and Carothers 1982; Mitsch and Gosselink 1986; Lee et al. 1987).

Additionally, riparian zones act to “meter” water delivery by holding water in plant root wads and soils, and gradually releasing that moisture as humidity and groundwater (Knutson and Naef 1997). Riparian zones also assist in recruitment of LWD, the loss of which reduces instream pools and channel complexity. In addition to the role riparian zones play in moderating and improving overall habitat conditions, many species of fish and wildlife depend directly on riparian zones to provide cover and forage (Federal Caucus 2000).

Instream and Floodplain Habitat Degradation

Loss of instream habitat complexity limits spawning and rearing habitat for fish, and in egregious cases, limits passage. Large woody debris plays an important role in maintaining varied and functional instream habitat. Logging and destruction of riparian habitat decrease available LWD recruitment materials, particularly in the lower Methow subbasin.

Reduced riparian cover, conversion of riparian zones to agricultural and residential uses, road construction, road failures, accelerated scour at culverts, and logging all contribute sediment materials to streams. Increased sedimentation alters stream channel characteristics and reduces spawning gravels and egg/alevin survival.

Floodplains help to moderate river flows by dissipating flow velocity and providing storage capacity for excess flows. Loss of floodplain wetland habitat in the developed reaches of the Methow and tributaries further reduces the already limited overwintering habitat for salmonids, eliminates forage and cover for wildlife, and reduces recharge potential of shallow groundwater in dry seasons.

Loss of floodplain wetland also contributes to higher stream velocities with associated bank erosion and sediment delivery.

Artificial and Naturally Occurring Barriers

Dikes and dams constructed for irrigation purposes can reduce fish passage to spawning and rearing grounds, block passage to floodplain habitat, prevent development of stream side channels, limit spawning gravel recruitment, and can confine the stream channel which in turn concentrates stream flows and facilitates scouring of stream beds.

Unscreened irrigation diversions can divert fish from the main river or creek flow, leaving them stranded when the irrigation flow is cut off. Maintenance of irrigation diversions can damage streambeds and banks. Inadequate or inappropriate screens, associated with diversion, can entrap fish or simply not function properly, allowing fish to pass into irrigation diversions.

Culverts can prevent access to spawning and rearing grounds by concentrating flow to the extent that they become impassable, and by concentrating debris. The high velocities of water moving through culverts also sometimes downcut the streambed to such an extent that upstream fish passage eventually becomes impossible.

While all of these man-made diversions play a role in reducing passage within the Methow subbasin, even before human settlement, waterfalls and high gradient steams characterized by high velocity spring run-off prevented and reduced passage to many reaches of the Methow subbasin.

Land Management Practices

Timber management activities, including extensive timber harvest in sections of the Methow subbasin and livestock grazing, have negatively impacted both fish and wildlife habitat in mid and lower reaches of the watershed, particularly in the Chewuch River and Beaver Creek drainages. Both logging and grazing contribute to fragmentation of habitat, soil erosion, sediment delivery to creeks and streams, channel simplification from loss of LWD recruitment within the riparian zone, and changes to upland and riparian vegetative communities, including displacement of native plant communities with exotic species.

Timber harvest changes upland vegetative cover and influences snow accumulation and melt rates. Road building associated with timber harvest further exacerbates erosion, habitat fragmentation, and contributes barriers to fish passage through construction of culverts. Uncontrolled livestock grazing compacts soil, contributes to stream bank destabilization, affects compositions of riparian plant communities, and slows recovery of damaged riparian habitat.

Conversion of forestland and riparian habitat to residential and agricultural uses also negatively affects habitat connectivity and composition. Human developments often constrain wildlife range and quality through construction of roads, dispersed residential developments, impediments to stream access, and changes to vegetative communities. Human activities have increased the number of fire starts, but historic fire control policies have kept the size of fires small, resulting in a buildup of fuel in the forested uplands of the subbasin. This absence of fire has resulted in changes in the composition of the forest and plant communities, and in the related capacity to store and transport water. Areas of the Methow subbasin burn periodically because of lightning and human causes, and will continue to do so.

Policy, Social, and Cultural Effects

Humans and salmon colonized and expanded their range in the Columbia River Basin after the most recent Ice Age (10,000-15,000 years BP). American Indians developed a culture that relied extensively upon anadromous fish for sustenance in some portions of the area (Craig and Hacker 1940). Their catches must have increased as their populations rose and techniques of fishing developed. Particularly at partial obstacles for passage, Indians captured large numbers of fish for both sustenance and trade.

Native Americans had access to an abundant fish resource comprised of spring, summer, and fall runs of Chinook salmon, coho and sockeye salmon, and steelhead, as well as Pacific lamprey and white sturgeon. Estimates of pre-development (late 1700s) abundance of Columbia River salmon and steelhead ranged from about 8 million (Chapman 1986) to 14 million (NPPC 1986) fish. Estimates of pre-development salmon and steelhead numbers were based on maximum catches in the latter part of the 1800s and assumed catch rates by all fishing gear. Inherent in such calculations is the assumption that fish populations in the late 1800s represented a reasonable expression of average effects of cyclic variation in freshwater and ocean habitat conditions. No one, currently, has determined validity of that assumption. It is, however, quite certain that salmon and steelhead have declined to a small fraction of their former abundance (Figure 3-2 in NRC 1996). Peak catches in the 1800s by all fishers may have included 3-4 million salmon and steelhead (Chapman 1986). Total run size for all salmon and steelhead recently has ranged from 1 to 2 million fish. About three-quarters of recent spring Chinook and summer steelhead runs have consisted of fish cultured to smolt size in hatcheries.

While actual numbers of adult spring Chinook salmon and steelhead produced by the upper Columbia River basin in the pre-development period are not available, one can attempt to estimate them, albeit roughly. From Fulton (1968, his Table 2), one can total formerly-used spring Chinook salmon habitat throughout the Columbia River basin as 10,002 kilometres (6215 miles), and upper Columbia habitat (upstream from the Yakima River) as 899 kilometres (559 miles), or about 9% of the total. Chapman (1986) estimated that about 500,000 spring Chinook returned to the Columbia River in the latter portion of the 1800s. Nine percent of that total would be about 45,000 spring Chinook salmon attributable to the upper Columbia River.

Anadromous fish of the upper Columbia area must have fluctuated because of variable environmental conditions. Certain combinations of freshwater and ocean habitat conditions appear to have caused very low salmon returns in some years, well before non-Indians degraded habitat or began fishing intensively (Mullan et al. 1986).

Numbers of spring Chinook that escaped to the Columbia River at Priest Rapids Dam in the most recent decade have averaged about 15,800 (adults plus jacks). This escapement would convert to approximately 21,000 fish downstream from Bonneville Dam (adjusting for 4% loss of adults for each dam between the estuary and counting station at Priest Rapids Dam, and a fishing rate of about 5%, mostly upstream from Bonneville Dam). Hatcheries had contributed about 75-80% of these fish. Thus naturally produced spring Chinook salmon abundance in the upper Columbia area can be estimated to have declined to about 5,000 fish, a decrease of 89% . Estimation of the percentage decline in wild summer steelhead produced in the upper Columbia River would indicate a similar major decline. Salmon and steelhead genetic diversity has also declined as a result of artificial propagation and widespread stock transfers.

Both spring Chinook and summer steelhead in the upper Columbia River have been listed under provisions of the Endangered Species Act (ESA) of 1972. Factors that depressed numbers of wild spring Chinook and steelhead sufficiently enough to lead to ESA listing include range extirpation, fishing, artificial propagation, and habitat degradation caused by dams, irrigation, channelization, overgrazing, and public policy. Lackey (2001) wrote:

> The depressed abundance of wild stocks was caused by a well known, but poorly understood combination of factors, including: unfavorable ocean or climatic conditions; excessive commercial, recreational, and subsistence fishing; various farming and ranching practices; dams built for electricity generation, flood control, and irrigation, as well as many other purposes; water diversions for agricultural, municipal, or commercial requirements; hatchery production to supplement diminished runs or produce salmon for the retail market; degraded spawning and rearing habitat; predation by marine mammals, birds, and other fish species; competition, especially with exotic fish species; diseases and parasites; and many others.

Lackey (2001) also wrote that "technocrats," who represent various organizations, have developed estimates of the proportions of wild fish declines attributable to one or more of the above-mentioned factors for decline. He pointed out that models that resulted in that work usually ended up supporting the favoured policy position of the supporting organization.

Fishing

Pre-development harvests and effects

Until 7,000 to 10,000 B.P., glacial ice blocked upper reaches of many rivers of the Pacific Northwest (Lackey 1999). Improved ecological conditions for salmon likely developed about 4,000 years ago, and aboriginal fishermen benefited. Lackey (1999) speculated that salmon populations reached their highest levels within the last few centuries.

It seems quite unlikely that aboriginal fishing was responsible for run declines in the Columbia River (Craig and Hacker 1940; Chapman 1986; Lackey 1999). Their artisanal fishing methods (Craig and Hacker 1940) were incapable of harvesting upper Columbia River spring Chinook and summer steelhead at rates that approached or exceeded optima for maximum sustained yield (probably 68% and 69% for spring Chinook and steelhead, respectively, as estimated in Chapman (1986)).

Indian populations declined sharply about 100-500 years ago, attacked by smallpox, measles, sexually-transmitted diseases, cholera, and other pathogens imported from Europe. Fishing rates likely declined in concert.

The year 1957 marked a major change in Native American fisheries. The Dalles Dam, completed in that year, and flooded the most important traditional and important Indian fishing dipnetting site in the Columbia River, at Celilo Falls. Catch rates in 1957 in Zone 6 dropped dramatically, and did not increase until the early 1960s, once Indians shifted to set gillnets.

Fisheries of the late 1800s

The population of humans in the Columbia River basin developed rapidly, beginning in the mid-1800s, with extensive immigration from the eastern U.S. Efficient fishing techniques, and preservation methods such as canning, set the stage for overexploitation of Columbia River salmon stocks. The onslaught of techniques included gillnets, traps, horse-pulled beach seines, purse seines, and fish wheels.

Intense fishing first targeted the abundant late-spring and summer components of what was a bell-shaped abundance function for Chinook salmon. Spring Chinook entered first and in relatively small numbers (Chapman 1986). The late-spring and summer runs formed the central bulk of the abundance timing function, then finally, fall Chinook arrived in lesser numbers. Thompson (1951) showed that fishing had all but extirpated the central bulk of the return distribution by 1919. As that fishery disappeared, industry shifted to sockeye, steelhead, coho, and fall Chinook. These shifts partially masked the decline of over-fished run components.

Although governmental agencies existed, with nominal responsibility, for fishery management (e.g., U.S. Bureau of Fisheries, Oregon Fish Commission), demand for fish and gear competition, chiefly among commercial fishermen, brooked little interference with seasons and fishing intensity. Washington passed its first gear restriction in 1866, some six years after commercial fishing became an important Columbia River industry. Oregon's first restriction came in 1878. Not until 1899, did Oregon and Washington begin to jointly manage Columbia River fisheries.

There can be little doubt that the relentless fishing intensity in most of the latter half of the 1800s and early 1900s substantially exceeded optimum rates. Chapman (1986) assumed that extant rates were 80-85% on spring and summer Chinook, 88% on fall Chinook, and 85% on steelhead.

The 1900s - decades of change

In 1909, Oregon and Washington instituted joint consistent fishing seasons. From about 1910 to 1912, as reasonably dependable internal combustion engines became available, troll fishing for salmon developed, enabling offshore fishing on Columbia River stocks mixed with fish from other rivers. Some inflation of early Columbia River landing statistics likely occurred as a result of troll-caught salmon sales inside the Columbia River mouth.

Industrial fishing practices

An intense industrial fishery in the lower Columbia River, employing traps, beach seines, gillnets, and fishwheels, developed in the latter half of the 1800s. In the early 1900s, troll fisheries developed to catch salmon even before they reached the Columbia River. The late-spring and early-summer Chinook salmon returns, which constituted the heart of the Columbia River runs, were decimated by the early 1900s (Thompson 1951).

In 1917, purse seines were prohibited in the Columbia River. These regulations, as several others later, likely resulted in part from gear wars, rather than from conservation. Whip seines became illegal in 1923, and fish wheels in Oregon were prohibited in 1927. Fish wheels in Washington remained legal until 1935. Washington prohibited drag seines, traps, and set nets in 1935, while Oregon waited until 1949 to take similar steps.

Washington law prohibited commercial take or sale of steelhead from the Columbia River after 1934, while Oregon continued to permit take and sale of steelhead by non-Indians until 1975.

Meanwhile, upriver dams began to deny salmon access to habitat. Swan Falls Dam on the Snake River was the first mainstem obstacle (1910). On the Columbia River mainstem, Rock Island

Dam was completed in 1933, Bonneville Dam in 1938. These facilities provided the first consistent numerical assessments of fish passage (only harvest data were available formerly). Grand Coulee Dam denied fish access to salmon and steelhead that formerly used Canadian tributaries and the Spokane and San Poil rivers. Small irrigation dams also chipped away at fish habitat, beginning in the 1800s.

Commercial fishing, and most Native American subsistence fishing in the latter half of the 1900s, was confined to gillnets. Downstream from Bonneville Dam, in zones 1-5, only drift nets were employed. In Zone 6, set gillnets were used. Gillnets do not facilitate release of gilled fish alive; hence, the principal means for protecting weak stocks of salmon and steelhead are area and time closures. Large mesh sizes in the 1990s afforded some protection for upper Columbia A-group steelhead (most upper Columbia summer steelhead are in this group of smaller steelhead); although, some larger steelhead, that spent two years at sea, were taken during late summer during the fall Chinook season.

As upriver spring Chinook populations declined sharply in the last quarter of the 1900s, managers reduced commercial fishing seasons in zones 1-5, and tribes reduced harvest rates in Zone 6. Hatchery-produced salmon and steelhead increasingly dominated runs.

Effects of harvest on wild/natural spring Chinook and steelhead of the upper Columbia River are very difficult to control in mixed-stock fisheries of zones 1-5 (Columbia River mouth to Bonneville Dam) and Zone 6 (upstream from Bonneville Dam, concentrated in Bonneville, The Dalles, and John Day pools). Gillnets are the most utilized fishing technique, indiscriminate in selecting one stock or another, or hatchery fish over wild ones. Mixed-stock fisheries are particularly detrimental to naturally small populations or those depressed (Spence et al. 1996; NRC 1996).

Only through virtual elimination of fishing on weak stocks can managers achieve protection for them. Fisheries in zones 1-6 have been curtailed sharply to protect ESA-listed stocks, chiefly destined for the Snake and upper Columbia rivers. This has led to excess escapements of spring Chinook of hatchery origin, leading to public policy conflicts with respect to management use of the excess returns when the fish arrive at the hatchery.

Near elimination of harvest on weak stocks can be accomplished by fishery closures, restrictions on area and times of fishing, and limitations on gillnet mesh sizes, sometimes combined with net modifications (e.g., trammel nets that entangle rather than gill fish).

Sport and Native American subsistence catches have been confined largely to areas short distances downstream from hatcheries where managers expect sufficient returns (e.g., on Icicle Creek downstream from Leavenworth National Fish Hatchery).

Columbia River fishery management in the last third of the 1900s was based in large measure on the concept of maximum sustained yield (MSY) (NRC 1996). At least two important issues make that concept obsolete for future management. The first is that stock-recruit models, from which MSY was determined, are based on historical adult and progeny adult information obtained under past environmental conditions. Those conditions changed, or re-set, as successive mainstem dams came on line, especially after the early 1950s. They may also change markedly over time with cyclicity of the ocean environment. Furthermore, MSY management does not acknowledge value of “excess” escapement as: a) a means of augmenting nutrient levels by bringing marine nutrients to the infertile streams of the upper Columbia River; or b) important in fostering competition for mates and spawning sites. The MSY paradigm now does not well serve managers, especially regarding upriver anadromous stocks.

Although the long-term effects of non-native species introductions often remain unpredictable because of the intricate nature of aquatic food webs and ecosystems, experience has demonstrated the establishment of certain non-native species will usually have predictable negative effects, resulting in serious population declines of bull trout.

Current fisheries

Extremely restrictive fisheries are allowed in the lower Columbia River for spring Chinook and steelhead in order to protect listed fish (including upper Columbia River spring Chinook and steelhead). For example, a federally-established limit of 2% incidental kill of wild spring Chinook and wild steelhead was set in 2004 for non-tribal fisheries; of that allowance, a maximum kill of 1.2% was set for the recreational fishery and 0.8% for the commercial fishery in zones 1-5. These conservative impacts were emplaced in spite of an expected spring Chinook run to the Columbia River of 500,000 fish, the second largest run since 1938, when Bonneville Dam counts began. Tribal gillnet fisheries in Zone 6 are likely to harvest an additional 8 to 10%.

Current restrictions also require sport anglers, between the Rocky Point/Tongue Point line in the estuary upstream to the I-5 bridge, to maintain caught fish that have intact adipose fins in the water as they remove the hook. Commercial fishers must use a combination of tangle net (4.25 inch mesh) and large mesh sizes (9-9.75 inches), not longer than 150 fathoms. Recovery boxes on board must be used for any wild fish captured, and on-board observers determine the number of wild fish caught and released.

ESA-listed upriver stocks, including those in the upper Columbia, prevent directed fisheries, even though substantial numbers of hatchery-produced spring Chinook can be taken. Upriver summer steelhead may not be harvested in the commercial fishery of zones 1-5.

A set-gillnet fishery for spring Chinook and steelhead, classed as “ceremonial and subsistence” is prosecuted by Indians in Zone 6. Steelhead captured by Indians in Zone 6 can be sold or used as “ceremonial and subsistence” harvest. Mean catch rates in the last half of the 1990s equalled about 10%.

Fishing in the future

Schaller et al. (1999) estimated spawner numbers required for full seeding of spawning areas used by wild Columbia River spring Chinook salmon as 4,808 for the Wenatchee River, 496 for the Entiat River, and 1,379 fish for the Methow River, for a total of 6,683. Other estimates have placed the spawner requirement higher.

Mainstem multipurpose dam projects in the Columbia River kill upper Columbia River spring Chinook and steelhead smolts at cumulative rates that may approach 45-50%. Adult inter-dam loss at 4% per project accumulates to 25% (Wenatchee River fish), and more for fish destined for tributaries upstream from Rocky Reach and Wells dams. Under these pressures from dam-related mortality, wild fish cannot sustain a directed fishery prosecuted with gillnets, and their escapements, even at full seeding, are insufficient to return one progeny spawner for each parent spawner.

Four solutions are theoretically feasible. The first, the approach now employed, is to severely restrict harvest, and to supplement wild fish with hatchery programs aimed at maintaining and fostering genetic adaptiveness peculiar to each upper Columbia River spawning/rearing area. The long-term utility and appropriateness of this approach has yet to be demonstrated.

A second approach is to shift mainstem fisheries to live-catch methods that permit identification and release of wild fish unharmed (NRC 1996). Although live-catch systems would permit substantially greater harvest of hatchery fish, political resistance to this option is strong. Tribal interests regard such proposals as interference with treaty rights.

The third is to confine fisheries aimed at hatchery fish to terminal areas (e.g., Icicle Creek spring Chinook, supported by Leavenworth National Fish Hatchery and by some natural spawners not listed under the ESA, are harvestable in Icicle Creek downstream from the hatchery). Fish quality for spring Chinook destined to spawn in terminal areas of the upper Columbia River declines as fish progress upstream. Quality in the terminal areas cannot compete with quality of pen-reared, or ocean- or estuary-caught salmon. Pen-reared salmon have made up over 50% of marketed salmon in recent years.

The fourth is to stop all fishing other than terminal harvests. NRC (1996) discussed this option, but noted that it is fraught with treaty and international, political, and legal issues.

Effects of fishing on population characteristics

High fishing rates in the 1800s virtually extirpated some late-spring and summer stocks of Chinook salmon. Past effects of fishing on now-listed spring Chinook and steelhead of the upper Columbia River are unknown. Attempts to sustain fishing by use of hatchery fish influenced genetic composition of at least summer steelhead, as progeny of adults trapped at Priest Rapids and Wells dams were, for several generations, liberated as smolts in the major tributaries of the upper Columbia River without regard to fostering local adaptations. NRC (1996) noted: "The continual erosion of the locally adapted groups that are the basis of salmon reproduction constitutes the pivotal threat to salmon conservation today."

Nelson and Soule (1987) and Thorpe (1993) reviewed effects of fishing on genetic makeup of salmon populations. Intense fishing probably altered genetics of pink salmon in the north Pacific, for example, with the result that adult size declined. Historically, intense gillnetting in the Columbia River may have increased the proportion of smaller fish in escapements, with potential increases in jack fractions and reduced fecundity of females. Three-ocean spring Chinook adults may have been selected against at earlier high fishing rates. At current low fishing rates, genetic selection against large spring Chinook and steelhead by gillnets likely does not occur (Chapman et al. 1995).

Despite the implementation of restrictive fishing regulations and strong educational efforts, both legal and illegal angling have direct impacts on bull trout populations. In streams open to general fishing, without legal harvest of bull trout, bull trout adults and juveniles are vulnerable to incidental catch, poaching, or disturbance. Incidental hooking mortality varies from less than 5% to 24% for salmonids caught on artificial lures, and between 16% and 58% for bait-caught salmonids (Taylor and White 1992; Pauley and Thomas 1993; Lee and Bergersen 1996; Schill 1996; Schill and Scarpella 1997). Although salmonid eggs in the early developmental stages are resistant to crushing (Hayes 1949), eggs and alevins in redds are vulnerable to wading-related mortality. Wading can cause mortality of up to 46% from a single wading event (Roberts and White 1992). In addition, harvest of bull trout may occur within their range because of misidentification. Schmetterling and Long (1999) found that only 44% of anglers correctly identified bull trout, and anglers frequently confused related species.

Illegal harvest is a significant common theme across the range of bull trout. Bull trout are also susceptible to incidental mortality associated with gill-net fisheries that target salmon and steelhead. Many of the life history attributes of bull trout increase their susceptibility to interception in gillnet fisheries. The highly migratory behavior of bull trout, coupled with their ability to repeat spawn, and their longevity, increase the number of possible encounters with nets located at river mouths.

Because they are a predator on other, more highly prized fish, intentional fisheries management efforts in the past have also negatively affected bull trout. Bull trout were sometimes targeted for elimination in many parts of their range through bounties, liberal daily bag limits on recreational angling, or the removal of limits entirely (Bond 1992; Brown 1992; Colpitts 1997; Stuart et al. 1997). Additionally, streams and reservoirs were sometimes treated with toxicants to remove undesirable species (usually targeting native suckers and minnows) in preparation for introduction of native and non-native sport fishes; (MBTSG 1996b).

Effects of fishing on salmonid populations

As these run components rapidly declined, fishing shifted earlier, later, and to other species, changes that, for a time, numerically masked the precipitous decline in the sought-after late-spring and early-summer fish.

By the early 1930s, mean escapement of spring Chinook into the upper Columbia River upstream from Rock Island Dam had declined to fewer than 3,000 fish. That escapement would represent perhaps 12,000 fish arriving in the lower Columbia River, inasmuch as fishing rates exceeded 75% in that period. Only Rock Island Dam (1933) lay athwart the Columbia River. Mean returns of summer steelhead to the upper Columbia River were lower than 4,000 fish in the first part of the 1930s. Harvest rates of 70%, and probably higher, were common before the 1940s. If one assumes a 70% rate, returns of upper Columbia summer steelhead to the estuary may have amounted to about 13,000 fish.

By the 1930s and 1940s, restrictions on fishing time and gear had increased. For example, purse seines were outlawed in 1917, whip seines in 1923, fish wheels in 1927 (in Oregon), seine, and traps east of Cascade Locks in Oregon in 1927, and drag seines, traps, and set nets in 1935 (Washington). Seasons were gradually shortened. Catch rates almost certainly were much higher than those appropriate for MSY or populations for several decades before then.

It is important to remember that fishing intensity, unless pursued to stock extinctions, can be relaxed by management action. If habitat remains intact, stocks can rebound. Presently, fishing rates have been reduced well below 10% for spring Chinook and below 13% for summer steelhead, yet wild and natural components of the respective runs in the upper Columbia River have not responded markedly. Currently, factors other than fishing depress these fish of the upper Columbia River.

Mainstem Columbia River Dams

Spring Chinook and steelhead production areas in the pre-development period included the Wenatchee, Entiat, Methow, Okanagan, and limited portions of the Spokane, San Poil, Colville, Kettle, Pend O'pers. comm., and Kootenay rivers . The Grand Coulee Dam project and Chief Joseph Dam eliminated access to the Columbia River upstream. The Grand Coulee Fish Maintenance Project (GCFMP), designed to transfer populations formerly produced upstream, into remaining habitat downstream from Grand Coulee, trapped fish at Rock Island from 1939 to 1943. Managers placed some adults in tributaries (e.g., Nason Creek) to spawn naturally, and artificially propagated others. Spring Chinook from outside the upper Columbia were introduced. The extreme changes in population structures permanently transfigured populations of spring Chinook and steelhead of the upper Columbia River (Chapman et al. 1995).

The era of mainstem multi-purpose dams downstream from the Grand Coulee project began with Rock Island Dam in 1933 and culminated with completion of Wells Dam. Seven mainstem dams lie between the Wenatchee River and the sea, eight downstream from the Entiat River, and nine between the Methow/Okanagan systems and the estuary. Dam-related losses are substantial. For example, adult salmon and steelhead mortality in the reaches between projects has been estimated as 4% or more in some years (Chapman et al. 1994 and 1995), and juvenile losses at each project can amount to about 10%. Some of the losses result from physical effects of adult and smolt passage. Others are derived from altered limnological conditions that increase predation by fish and birds, or that cause gas-bubble trauma. The cumulative loss rates also explain why so much mitigative effort has been allocated to project-related mortality rates.

Dams for storage, like Grand Coulee, and mainstem multipurpose dams, have had other effects on ecology of salmon and steelhead. Estuarine limnology has shifted from a basis of macrodetritus and benthos to a microdetrital, planktonic, trophic structure that favors non-salmonids. Spring freshet flows and turbidity have declined in the river and estuary, and the Columbia River plume has been reduced seasonally (Ebbesmeyer and Tangborn 1993, Chapman et al. 1994 and 1995, NRC 1996) with potential, but largely unknown effects on survival of salmon and steelhead in the estuary and nearshore ocean.

Tributary Habitat Degradation

Residential development is rapidly increasing within portions of the range of bull trout, including the Methow subbasin.

Perhaps the most important habitat influence on wild spring Chinook and steelhead in the upper Columbia River, including the Methow subbasin, involves water diversion, withdrawal, and application to crops. The Columbia Basin Project, operated by the U.S. Bureau of Reclamation, constitutes the largest single water diversion and application system in the area. In the Wenatchee, Okanagan, and Entiat River basins, water diversion for orchards is important. In the Methow River system, crops and pasturage divert tributary and mainstem water.

For wild spring Chinook and summer steelhead, diversions on tributaries of the Methow river must be considered a factor for decline. Instream flows have been depleted downstream from irrigation diversion dams, reducing instream habitat and improving predator access to rearing juvenile fish. Diversions were unscreened for many decades, permitting downstream migrants to pass into, and perish, in fields and orchards. Today some fish diversion screens are less than

100% effective. Diversion dams were built in some cases without adequate provision for adult passage.

Cattle pastures adjacent to tributaries can, and have, denuded riparian vegetation, and permitted nutrients from fecal material, and fine sediment, to enter salmon and steelhead habitat. Overgrazing by sheep and cattle has locally increased runoff of fine sediments and increased stream flow peaks (Mullan et al. 1992).

Channelization reduces instream habitat by straightening meanders, increasing water velocity, and eliminating or reducing riparian cover and input of LWD. It can, and has, occurred associated with roads and railroad grades, residential encroachment, and protection of agricultural land. Diking and channel-bank riprap prevents stream lateral movements across alluvial floodplains, particularly in the Methow and Okanagan drainages.

Note: Of the foregoing habitat factors, diversions and associated diversion dams probably constitute the most important factors for decline.

Hatcheries

NRC (1996) and Flagg et al. (2001) discussed at length the risks and problems associated with use of hatcheries to compensate for, or supplement, fish produced in the wild. NRC (1996) noted demographic risk, pointing out that large-scale releases of hatchery fish exacerbate mixed-stock harvest problems. Wild fish cannot sustain harvest rates that would be appropriate for hatchery fish. Demand is essentially unlimited for salmon and steelhead, and advocacy groups for various fisheries often clamour to have access to ever-more harvestable fish from hatcheries.

Solutions to the mixed-stock fishing problem are elusive. Gillnets, for example, have only limited potential for releasing wild spring Chinook and steelhead unharmed. Terminal fisheries, particularly for spring Chinook after they enter waters that contain only hatchery fish, are impractical for commercial fisheries because fish quality there has declined greatly. Steelhead are somewhat easier to manage in sport fisheries, where fish known to be of wild origin (identifiable by an intact adipose fin) can be released with minimal mortality, and hatchery fish (with adipose intact) kept.

Genetic and evolutionary risks for hatchery fish and interacting populations include inbreeding depression, loss of population identity and within-population diversity, and domestication selection (NRC 1996). Recognition of these possible factors has increased in recent decades. Unfortunately, measures used in the GCFMP and steelhead management in the upper Columbia (until recently) almost certainly realized some of the listed risks, and contributed to decreased genetic diversity of wild fish. Steelhead adults were collected at Priest Rapids, and later at Wells Dam, their progeny reared in hatcheries and released as smolts to the various tributaries, without regard to fostering local adaptation in tributaries.

Foraging, social behaviour, time of spawning, and predator avoidance can differ for fish reared in the hatchery and in the wild (Flagg et al. 2001). While resulting differences may primarily reduce survival of hatchery-produced salmon and steelhead, negative effects may carry into the wild population where adults of hatchery origin spawn with wild fish. Effects of disease on released hatchery fish and on wild fish are poorly understood, but likely to be negative (Flagg et al. 2001, tables 10-11 summarize these).

Also poorly understood, are ecological effects of hatchery programs. NRC (1996) noted that 5.5 billion salmon smolts of all species are released to the wild each year around the Pacific rim, with potential trophic effects that may lead to altered body size and survival of wild fish. Emphasis on hatchery fish denies marine nutrients to infertile rearing streams used by relatively few wild spring Chinook salmon and steelhead.

Intentional and unintentional introductions of non-native aquatic species have contributed to declines in bull trout abundance, local extirpations, and hybridization (Bond 1992; Howell and Buchanan 1992; Leary et al. 1993; Donald and Alger 1993; Pratt and Huston 1993; MBTSG 1995b,d, 1996g, h; Platts et al. 1995; J. Palmisano and V.Kaczynski, Northwest Forest Resource Council, in litt. 1997). The historical record documents many cases of both authorized and unauthorized introductions of non-native species by government agencies, as well as by private parties, across the range of bull trout.

Public policy

The Marine Mammals Protection Act of 1976 afforded seals and sea lions complete protection from killing by humans. These animals increased sharply in abundance thereafter (Fresh 1996). NRC (1996) discussed the potential for effects on salmon and steelhead. They concluded that such predation was “probably not a major factor in the current decline of salmon in general.” Chapman et al. (1994 and 1995) suggested a need for adaptive management, including population control through selective harvest and/or sterilization of live-captured seals on haul-out beaches. They pointed out that, although pinnipeds and salmon coexisted long before man interfered ecologically, contrary views hold that it is unrealistic for man to manage and prey upon salmon without managing one of their principal predators.

The Corps of Engineers dredges shipping channels in the lower Columbia River and has created artificial islands with the spoils. Caspian terns have exponentially increased in the Columbia River estuary after dredge spoils created near-ideal nesting sites within the boundaries of a U.S. Fish and Wildlife Service refuge. Many PIT tags have been found on artificial island sites, demonstrating that terns may be very important predators on smolts that must pass through the estuary to reach the sea.

Public policy clearly has more ubiquitous influences, both direct and indirect, than the foregoing examples (NRC 1996). Mainstem dams are a direct outgrowth of public policy, constructed by the federal government (Chief Joseph, Grand Coulee, and four mainstem Columbia River dams downstream from the Snake River) or by public utilities licensed by the Federal Energy Regulatory Commission (Wells, Rocky Reach, Rock Island, Wanapum, and Priest Rapids dams).

Human population growth in the Pacific Northwest, often fostered by local government boosters, places more pressure every year on salmon and steelhead. Lackey (1999, 2001) eloquently described the ramifications for salmon of human population growth, and of public policies and decisions. He noted that the Pacific Northwest has a population increase rate that rivals many developing third-world nations. Public policies affect water diversions, instream flows, water temperature, dam operations, manufacturing, urban development, national defence, fishing, hatchery outputs, and transportation of people and goods. All of these factors, and more, some of greater influence than others, have depressed salmon and steelhead abundance and potential for restoration of depressed fish populations.

Marsh (1994) may have inadvertently captured an essence of the effects of public policy on salmon when he wrote:

> the process is seriously, significantly, flawed because it is too heavily geared towards a status quo that has allowed all forms of river activity to proceed in a deficit situation – that is, relatively small steps, minor improvements and adjustments – when the situation literally cries out for a major overhaul.

He was referring to salmon restoration and management. But the underlying question was identified by Lackey's papers: Given human population growth and perceived needs, is Pacific Northwest society prepared to make the sacrifices necessary to restore wild listed spring Chinook and steelhead in the upper Columbia River (and elsewhere in the Columbia River basin)? The answer to date appears to be "no."

3.19 Synthesis Of Previous Efforts to Determine Important Factors For Decline of Methow Subbasin and Upper River Columbia Fish Populations

A number of key documents and reports have addressed factors affecting the decline of wild bull trout, spring Chinook and steelhead in the upper Columbia. Often the assessments take the form of limiting factor analyses, and are reported as such. There is not always clear agreement regarding the importance of various factors. Here we summarize and compare some of the central findings and conclusions offered in a number of key reports.

Chapman et al. (1995) reviewed the status of the spring Chinook salmon ESU of the upper Columbia Basin, including populations in the Wenatchee, Entiat, Methow, and Okanogan rivers. Their key findings and conclusions regarding factors affecting the decline of these wild populations are:

- The extensive development of mainstem dams and upstream storage reservoirs reduced productivity by 43% from the 1950s through the 1980s.
- Spawning and rearing habitat has not suffered functional degradation in most areas. However, water withdrawal for irrigation is a serious concern in several key tributaries, particularly in the Methow River Basin.
- There is no evidence to indicate that inter-specific competition from exotic or native fish species reduced the productivity of this ESU.
- Inriver harvest rates have been minimal since 1974, but in decades before that, harvest rates ranged from 40-85%. Marine harvest impacts are low, less that 1% for the years 1978-1993.

Their report emphasized hydro-passage effects as the primary factor limiting the productivity of this ESU. Risks associated with hatchery programs, and modest degradation in tributary habitat conditions were discussed, but they were not identified as critical factors responsible for the decline in the ESU. In-river harvest pressures were substantial before 1974, but subsequent to that year, harvest rates had been minimal or negligible with the imposition of harvest restrictions.

Chapman et al. (1994) wrote a similar status report for steelhead populations comprising the listed upper Columbia ESU. In their assessment, the following factors were identified as the chief causes of the decline of wild steelhead:

- Over fishing prior to the 1950s;
- Elimination of access to productive habitat above Grand Coulee Dam with dam emplacement, and;
- Mainstem dams, that have been the major cause for the depressed runs in recent decades.
- Additionally, they suspect two other human activities probably contributed to the decline of wild steelhead:
- Hatchery practices that mixed fish from a variety of sources to seed tributaries, and;
- Mortality (direct and incidental) associated with sport fishing for hatchery-released and resident trout.

They did not identify tributary habitat conditions as being important factors in the population decline. In fact, they characterize most spawning and rearing areas as being in fair to good condition; however, they noted that irrigation withdrawals in late summer in the Methow, Wenatchee, and Okanogan rivers posed a risk.

Specific land and water management activities that depress bull trout populations and degrade habitat include dams and other diversion structures, forest management practices, livestock grazing, agriculture, agricultural diversions, road construction and maintenance, mining, and urban and rural development (Beschta et al. 1987; Chamberlin et al. 1991; Furniss et al. 1991; Meehan 1991; Nehlsen et al. 1991; Sedell and Everest 1991; Craig and Wissmar 1993; Frissell 1993; Henjum et al. 1994; McIntosh et al. 1994; Wissmar et al. 1994; USDA and USDI 1995, 1996, 1997; Light et al. 1996; MBTSG 1995a e, 1996a f).

Mullan et al. (1992) focused on conditions and processes (including both hatchery influences and habitat factors) within three major watersheds: the Wenatchee, Entiat, and Methow rivers. In general, they concluded that the carrying capacity of those rivers is similar to what it was historically. On page 28, they conclude that natural production of Chinook salmon and steelhead smolts now may be similar to historical production. Overall, human activities have not badly degraded the tributary habitat, although some localized problem areas were identified. Even so, they note that coho are now extinct in this area. Furthermore, they point to mainstem dams and reservoirs as critical factors impacting stocks emanating from this basin, noting that 62-71% of smolts die while passing through the hydrosystem.

More recently a series of draft subbasin summaries have been published that address limiting factors in the subbasins of the upper Columbia. Electronic copies of these are on the NPCC website. The summaries are supported by a series of limiting factor analyses that were conducted for individual subbasins. Their characterization of tributary habitat conditions as limiting factors contrast with the portrayal by Mullan et al. (1992) and Chapman et al. (1994, 1995). In general, the limiting factors analyses describe a network of tributaries that has been degraded by assorted human activities, and where ecological processes have been compromised, the implication being that some of these areas may well be important in limiting the productivity of anadromous fish in the basin.

3.19.1 Mortalities Inside Methow subbasin

The Limiting Factors Analysis (Andonaegui 2000) points to poor salmonid productivity as a consequence of habitat fragmentation and loss of ecological function in important areas within the subbasin.

3.19.2 Mortality Outside the ESU

Mortality Assumptions

Decadal-scale, climate-driven fluctuations in marine conditions are a dominant factor influencing salmonid survival in marine waters. This factor appears to account for the greatest amount of change in survival from smolt through return as adults documented over the decades.

NOAA Fisheries (Williams et al. 2003-draft) recently characterized the importance of marine-based processes on the abundance of Columbia River salmon as follows:

Increasing evidence points to dramatic changes in the marine ecosystem of the northern Pacific Ocean resulting from shifts in climate over the past 2000 years (Finney et al. 2002, Moore et al. 2002). Throughout this region, changes in ocean-climate conditions have influenced zooplankton, benthic invertebrate, seabird, and fish populations (McGowan et al. 1998). In particular, analyses of data from the last 100 years demonstrate a strong relationship between ocean conditions and the production of Pacific salmon (*Oncorhynchus* spp.) across a range of spatial and temporal scales (Mantua et al. 1997, Beamish et al. 1999). The varied response of salmon to these past environmental changes likely reflects their complex life history and the wide diversity of freshwater and marine habitats that they occupy (Hilborn et al. 2003).

Recent evidence links Chinook salmon from the Columbia River basin to cyclic changes in ocean-climate conditions. Modeling exercises, directed at explaining the negative effects of various anthropogenic activities on the productivity of Snake River spring-summer (SRSS) Chinook salmon, identified the estuary and ocean environments as important sources of unexplained variation in stock performance (Kareiva et al. 2000, Wilson 2003). Using catch records from commercial fisheries, Botsford and Lawrence (2002) found reasonable correlations between the inferred survival of Columbia River Chinook salmon and physical attributes of the ocean, such as sea-surface temperature and coastal upwelling. Building upon these previous studies, Scheuerell and Williams (in review) found that they could actually forecast changes in the smolt-to-adult survival of SRSS Chinook from changes in coastal ocean upwelling over the past 37 years, including the rapid decline in the 1960-70s and the increase in the late 1990s.

All of these analyses highlight the important effects of the ocean in determining smolt-to-adult survival, and further support Pearcy's, {1992 #307} assertion that the primary influence of the ocean on salmon survival occurs early within the first year that juveniles occupy coastal waters.

Smolt and adult mortality associated with passage through the hydrosystem is still problematic, but efforts are underway to improve passages conditions and evaluate progress.

System survival studies conducted during the 1980s revealed that the survival of spring-migrating smolts was poor. Skalski and Giorgi (1999) summarized results from seven studies from that decade, conducted by either the Public Utility Districts or the Fish Passage Center. Four studies used yearling spring Chinook, and three used steelhead. The average annual per-project survival across all studies was 86.2% (range = 83.4 to 88.7%).

This equates to only 47.6% survival for smolts passing through five hydroelectric projects, from Wells Dam to Priest Rapids Dam. Today the HCP for Douglas and Chelan County PUDs specifies a smolt survival goal of 93% per project for all species of smolts. If this goal can be realized through passage improvements currently being implemented or explored at all five dams, then the smolt survival through that system would equate to 69.6%. If these passage survival goals can be achieved, they would provide a substantive contribution to the recovery of ESA-listed spring Chinook and steelhead ESUs in the Upper Columbia.

The existence and magnitude of delayed effects associated with passage through the hydrosystem remains unresolved, and constitutes a critical uncertainty in the context of ESU recovery.

It has been hypothesized that cumulative effects may be incurred as smolts migrate through the hydrosystem, effects that are not expressed until smolts enter saltwater. Such a scenario has proved difficult to test and verify. NOAA Fisheries established the Plan for Analyzing and Testing Hypotheses (PATH) in 1995. For five years, this issue was one of many key ones that were investigated. Consensus was never reached. Subsequent to PATH, a number of papers were published, some supporting and some contesting the hypothesis. The debate still continues today, and is a prominent topic treated in a recent draft technical memorandum published by NOAA Fisheries (Williams et al. 2003-draft).

The condition of smolts migrating from a watershed can influence survival in subsequent life stages; thus, improving habitat conditions may realize benefits beyond those reflected in egg-to-smolt survival.

Total Mortality Outside The Subbasin

The most comprehensive and instructive index of ESU survival beyond the watershed is smolt-to-adult return rate (SAR). It is a common survival index used to characterize the performance of salmonid populations throughout the Pacific Northwest. This survival index reflects all sources of mortality affecting migrating smolts through returning adults. These include effects associated with:

- Hydrosystem operations;
- Migration conditions in the mainstem, including both natural and anthropogenic causes (e.g., actions associated urbanization and industrialization);
- Fish condition that can vary annually by hatchery or rearing stream;
- Marine/estuarine conditions and processes influenced by natural and anthropogenic factors;
- Harvest in marine and riverine waters, and;
- Predation.

SARs can be calculated in different ways. Juvenile salmonids implanted with either PIT tags or CWT can be used to estimate SAR if returning adults can be sampled at strategic locations. Alternatively, the survival index can be calculated by estimating smolt abundance passing some site (a dam or the mouth of a tributary), then subsequently estimating adult returns to that location for a specific brood year. Often, SARs are expressed in terms of return rates to the mouth of the Columbia River. This calculation requires additional information such as estimates of inriver harvest and adult passage mortality.

3.20 Upper Columbia Smolt-to-Adult Survival

3.20.1 Spring Chinook

Historical estimates of SAR for naturally produced spring Chinook in the upper Columbia River have been reported by Mullan et al. (1992) and Raymond (1988). Mullan et al. estimated the smolt-to-adult return rate for the collective populations produced in the Wenatchee, Entiat, and Methow rivers for the years 1967 to 1987. Over that period, SAR ranged from 2.0 to 10.1%. They noted that the estimates reflect corrections for adult passage mortality, as well as for marine and inriver harvest.

Raymond (1988) estimated the percent of returning adults to the uppermost dam on the upper Columbia River for the years 1962 through 1984. Values for wild spring Chinook ranged from 0.7 to 4.9% over those years. One reason Raymond's values are generally lower than those reported by Mullan et al. (1992) may be that his estimates are not adjusted for adult passage mortality and marine harvest, whereas Mullan et al.'s (1992) were. Also, the reference locations for calculating SARs differed, with Raymond focusing on the upper dam, and the other investigators referencing the spawning grounds. This raises an important point; when comparing SAR values among investigators, the locations where smolts and adults are enumerated must be known.

SAR estimates for the most recent decade have not been calculated and published by any other investigators; thus, the historical estimates provide the only guidance on this matter.

3.20.2 Steelhead

Raymond (1988) estimated smolt-to-adult return percentages for the combined wild and hatchery steelhead population, 1962-1984 (**Figure 55**). Adult return rates to the upper dam ranged from a low of 0.2% for the smolt migration of 1977, to a high of 6.4% for the 1982 smolt migration. Mullan et al. (1992) reported SARs for only one stock (Well Hatchery steelhead), for the years 1982 to 1987.

The percent return to the mouth of the Columbia River averaged 6.38%, ranging from 1.32% to 14.28%. Survival back to Wells Dam averaged 3.01%,and ranged from 0.72% to 7.31%. These estimates aligned closely with Raymond's estimates for the overlapping years 1982 to 1984. Chapman et al. (1994) compiled data from three hatcheries in the upper Columbia (Chelan, Entiat, and Leavenworth) for the years 1961 to 1991. Smolt-to-adult survival averaged 1.7%, with a range from 0.16% to 7.54%.

The reference point for smolt abundance is the upper dam on the Columbia and estimated return of adults to that location. Years refer to smolt migration years

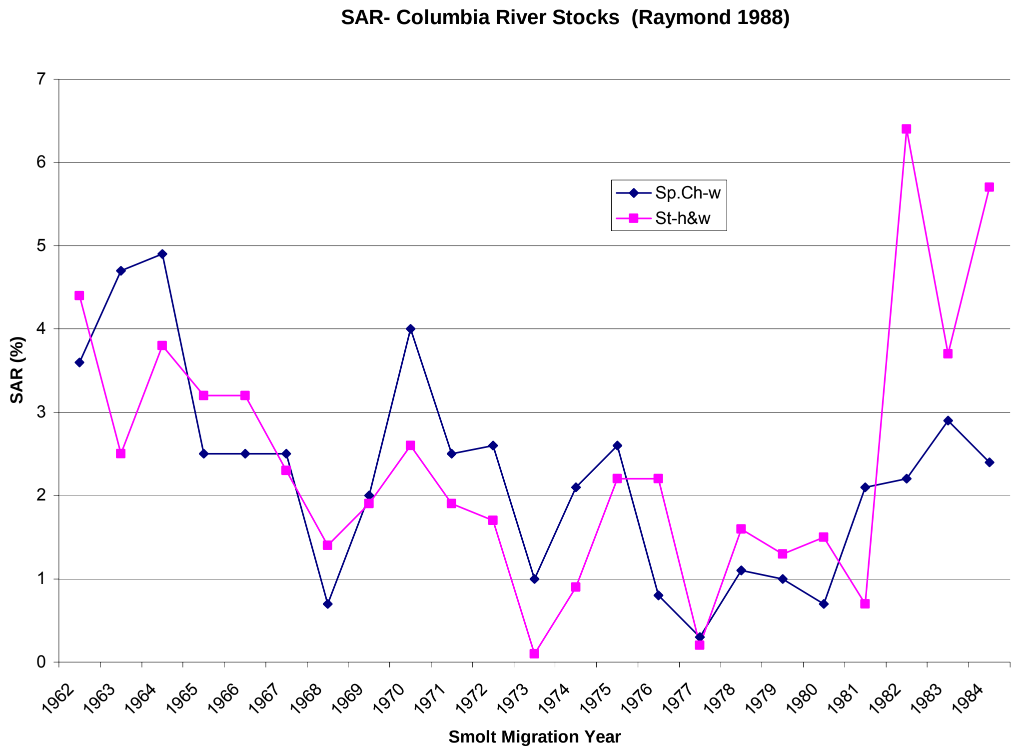


Figure 55 Survival from smolt to returning adult for upper Columbia wild spring Chinook and steelhead stocks as estimated by Raymond (1988)

Selecting Values for SAR to Use in ESU-Level Habitat Effectiveness Evaluations

Clearly SAR estimates for both spring Chinook and steelhead vary greatly across years. Over the decades, changes spanning at least an order of magnitude were commonly observed; thus, no single survival index value is satisfactory for accurately representing the performance of an ESU beyond the watershed. But accuracy may not be a central requirement for selecting a standard SAR that can be applied universally in habitat evaluations that use models like Ecosystem Diagnosis and Treatment (EDT). In years when smolt-to-returning-adult survival is low, survival from pre-spawner through parr in the tributaries carries more weight in terms of overall lifecycle survival. Conversely, when SARs are high, the contribution of survival during the subbasin residence stages contributes less proportionately to overall gravel-to-gravel survival.

What is the importance in establishing the magnitude of survival expressed outside the boundaries of a subbasin? When resource managers wish to compare the effectiveness of tributary habitat actions among subbasins, or across ESUs, then effects beyond the bounds of the subbasin or watershed become an issue. For example, if analysts in subbasin A assume a high SAR index, and they use adult abundance as a performance measure in modeling analyses, then the contribution from tributary-resident life stages is diluted. In contrast, if analysts in subbasin B assume a low SAR index, then the contribution of tributary survival is magnified in importance. One could imagine that funding agencies may prefer to invest in habitat projects where the “bang for the buck” might be greatest. This will be difficult to determine unless a standard out-of-subbasin survival index is adopted by all parties.

Is it practical to ignore effects outside the subbasin, and not incorporate them in quantitative analyses? Not if performance measures like productivity and adult abundance are of interest; these are sensitive to hydro, marine, and harvest effects. A SAR-like component, therefore, should be incorporated into whatever analytical model is employed; however, it may not be practical to run a series of model analyses over a range of SARs to reflect the sensitivity of every watershed population to variable marine or hydrosystem conditions. This is another reason why it is advantageous if a standard SAR value and approach can be selected for application when analyzing various populations emanating from different subbasins.

Out-of-subbasin Survival Effects in EDT Analyses

Ecosystem Diagnosis and Treatment (EDT) evaluates habitat across the life history of a focal fish species. For anadromous species, this evaluation addresses conditions within a subbasin, as well as outside of it, for example, in the mainstem Columbia River, estuary and ocean. Conditions outside the subbasin are often referred to as “out-of-subbasin effects” or OOSE. While EDT includes out-of-subbasin effects, the focus of an EDT evaluation is on the potential of a habitat condition within a subbasin; however, it is of interest to understand how survival conditions outside the subbasin might affect protection and restoration priorities within the subbasin.

Estimating out-of-subbasin effects in the upper Columbia will require a separate and dedicated effort under the Management Plan in years two and three. Because only general guidance for estimating the overall effects outside the subbasins exists at the time of this plan, we chose to identify the methods for this estimation protocol, and recommend that a dedicated effort be done as part of the overall Management Strategy. Estimates of SAR for each population are available at this time; however, estimates of mortality associated with specific locations or causal mechanisms in the mainstem or ocean are not.

Once the picture is complete, and under this proposed OOSE approach, a hypothetical generic situation forms the blueprint. There is one SAR reference value selected for each species. The maximum value of this SAR index value stock is realized for a generic stock of smolts entering the mainstem from tributaries downstream from Bonneville Dam. In modeling analyses, that generic stock of smolts is moved upstream to subbasins that enter the mainstem above an increasing number of dams. This effort will require additional modeling that was not made available to subbasin planners during the assessment phase.

Out-of-subbasin Effects (OOSE) Approach

Since subbasins enter the mainstem Columbia at differing distances from the point of ocean entry, each subbasin population will incur different levels of hydrosystem-related mortality. Mobrand Biometrics, in conjunction with the NPCC, has devised an approach to generically treat all populations entering the mainstem. They refer to the composite mortality through the hydrosystem and marine waters, including harvest removals, as “out-of-subbasin-effects” (OOSE). We propose adopting that approach at this time.

Under this OOSE approach, a hypothetical generic situation forms the blueprint. There is one SAR reference value selected for each species. The maximum value of this SAR index value stock is realized for a generic stock of smolts entering the mainstem from tributaries downstream from Bonneville Dam. In modeling analyses, that generic stock of smolts is moved upstream to subbasins that enter the mainstem above an increasing number of dams.

The SAR index value is then reduced by incremental amounts to reflect the number of dams the generic stock now has to pass enroute to the mouth of the Columbia River. The values initially selected as the SAR index do not need to represent the "truth," nor do values representing dam passage survival, but they should fall within an accepted range of observed values. The purpose is to prescribe a standard OOSE that can be applied to all ESUs or populations entering the mainstem at different locations.

Out-of-subbasin survival effects in EDT analyses are described in 2.6 Synthesis and Interpretation.

3.21 Synthesis and Interpretation of Assessment for Fish Ecosytems

The review of limiting factors for focal species of fish was carried out using an extensive and powerful tool called EDT (Ecosystem Diagnosis and Treatment). The major results of EDT are captured under the plan sections entitled Major Findings and Assessment Unit Summaries. In brief, they show that in the Methow Basin habitat losses have chiefly resulted from artificial and natural fish passage barriers, alteration and reduction of riparian habitat, loss of habitat connectivity, instream and floodplain habitat degradation, low flows and dewatering, and extreme water temperatures. Added to these limiting factors within the Methow are out-of-basin problems including fish passage over mainstem dams and harvest.

Thus, the ecosystem diagnosis method used was intended primarily to address the question: *Is there potential to improve anadromous salmonid population status through improvements to habitat conditions in tributary environments?*

Said in a form of a **central subbasin hypothesis** (for fish and adaptable for wildlife): *Improvements in habitat conditions will have a positive effect on habitat productivity and thus, improve fish population status through increased abundance, diversity, and spatial structure.*

Ecosystem Diagnosis and Treatment

Reach analysis tables (EDT consumer reports tables) were used to determine primary and secondary limiting factors within each Assessment Unit (AU). The Subasin Core Team factored in the results of assessments on focal species, and across all reaches in each AU. In general, a survival factor was considered a primary limiting factor if there were high or extreme impacts on key life stages.

Exceptions included some reaches where sediment load or temperature only had a high impact to spawning or egg incubation. Additionally, a survival factor was considered to be a primary limiting factor if there were small to moderate impacts across most (9-12) life stages, thereby producing a cumulative impact that could be just as severe as high and extreme impacts on fewer life stages. Secondary limiting factors, generally, had small to moderate impacts on several (5-8) life stages.

An exception occurred with the survival factor “food”; when there were small to moderate impacts on two or three juvenile life stages in most of the reaches of a particular AU, then we considered “food” to be a secondary limiting factor.

In most reaches and AUs, the break between primary and secondary limiting factors was fairly obvious. In some cases, where EDT results were not as obvious, other information, such as the Limiting Factors Reports, the RTT Biological Assessment, professional opinion, and local knowledge were factored into the decision.

Out-of-subbasin Survival Effects in EDT Analyses

Ecosystem Diagnosis and Treatment (EDT) evaluates habitat across the life history of a focal fish species. For anadromous species, this evaluation addresses conditions within a subbasin, as well as outside of it, for example, in the mainstem Columbia River, estuary and ocean. Conditions outside the subbasin are often referred to as “out-of-subbasin effects” or OOSE. While EDT includes out-of-subbasin effects, the focus of an EDT evaluation is on the potential of a habitat condition within a subbasin; however, it is of interest to understand how survival conditions outside the subbasin might affect protection and restoration priorities within the subbasin.

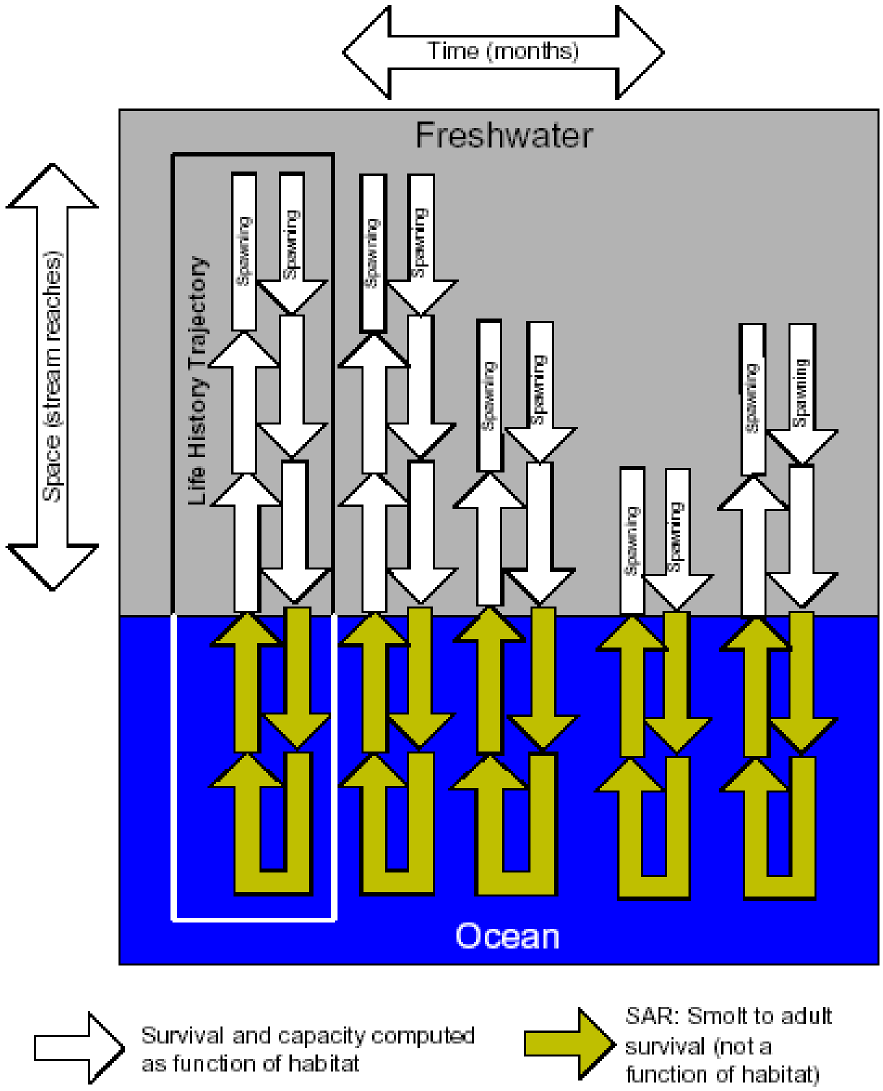
In contrast to the situation within a subbasin, in EDT, OOSE survival is not calculated from habitat information; instead, a set of survival multipliers are used to achieve reported smolt-to-adult survival rates (SAR). These multipliers result in a SAR value for the focal population, which is reported in the standard EDT output summary.

The SAR, as reported in the EDT output, represents the survival from a juvenile leaving the subbasin to an adult returning. Since EDT accounts for age at emigration and at maturation, the survival value will vary depending on the age composition of a population. However, since age-composition for a given population is stable, a single SAR value can be used for each population. For some populations in some watersheds, significant numbers of juveniles that emigrate from the subbasin are not smolts. In these cases, the SAR reported by EDT may be an underestimate.

SAR has been estimated from empirical data, for some species, in a limited number of subbasins (NOAA 2004). From these estimates, it is clear that the SAR is highly variable from year to year, and from subbasin to subbasin, and that spatial or temporal trends in SAR are difficult to discern. The variability in SAR indicates that the survival rate of smolts leaving a subbasin is highly dependent on conditions both inside and outside the subbasins.

Life History Trajectories in Ecosystem Diagnosis and Treatment

To understand how the SAR affects results in EDT, it is necessary to explain the concept of life history trajectories. A life history trajectory is the unbroken sequence of life stages and habitat segments that a fish moves through while completing its full life cycle. Trajectories start and end with spawning at a particular spot (i.e., a stream reach), and at a particular time within a year (**Figure 56**). At each trajectory segment (defined by a life stage, a location, and a time), the survival conditions are computed from habitat characteristics as they affect the life stage.

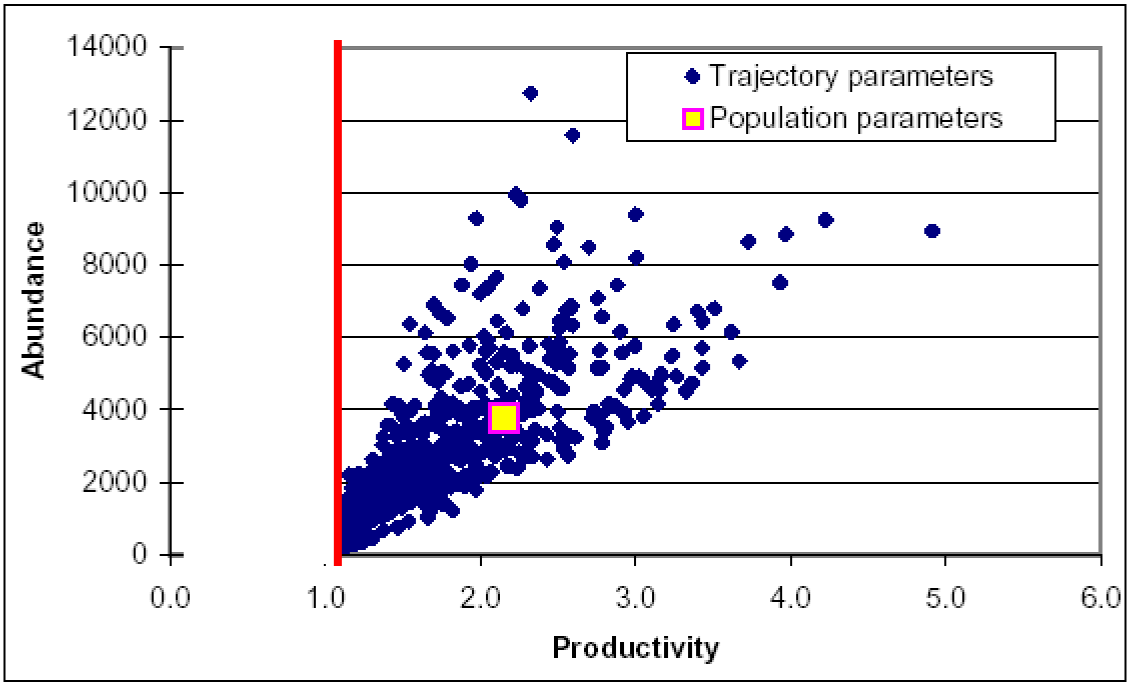


Source: Mobrand Biometrics 2004

Figure 56 Life History Trajectory Concept in EDT

Trajectory segments outside the subbasin are greatly simplified by applying constant, population-specific survival factors. EDT then computes the cumulative survival of all segments along each trajectory. EDT samples the environment by starting trajectories in a regular pattern along the stream course, and at regular time intervals during the spawning season (**Figure 56**). In a typical stream, EDT generates hundreds of life history trajectories to sample and characterize the habitat conditions within a stream. EDT finally estimates survival parameters for the focal population from this collection of trajectories (**Figure 57**); thus, the SAR computation is embedded in the trajectory calculations.

To capture the seasonal variations of hydroelectric operations and conditions in the estuary and ocean, survival conditions outside the subbasin are shaped by month within a year.

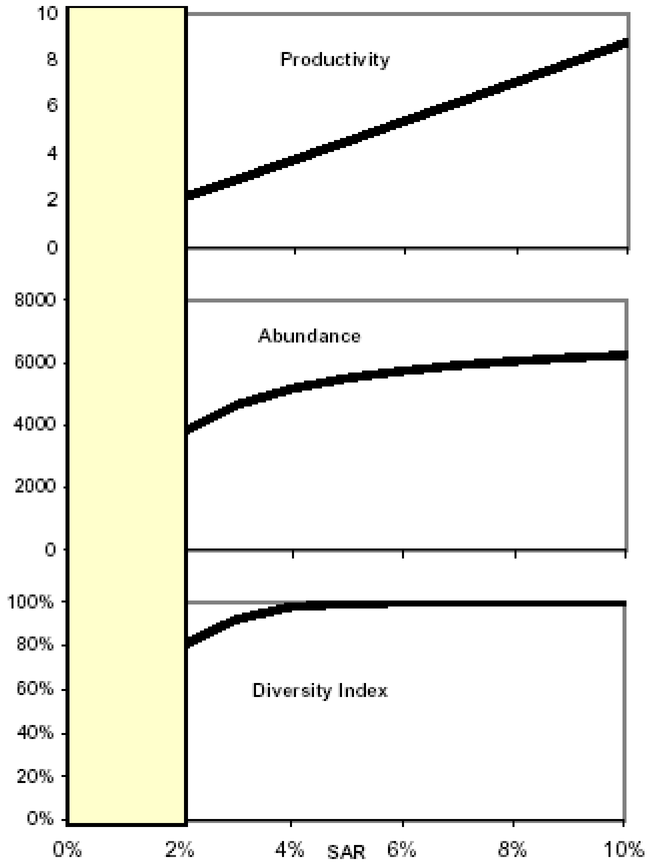


Source: Mobrand Biometrics inc., 2004

Figure 57 Hypothetical population depicting individual trajectories, population abundance and productivity parameters EDT derives from the trajectories

Effects of OOSE on Population Parameters

A hypothetical example might help illustrate how the survival outside the subbasin, the SAR, affects the EDT estimates of the population parameters of the focal population. There is a near-linear relationship between productivity and the SAR, as might be expected (**Figure 58**). The deviation from linearity is because of the fact that the SAR affects the population productivity parameter through the individual trajectories described above. For small SAR's (< 2% in the example), both equilibrium abundance and the diversity index are very sensitive to changes in SAR (**Figure 58**). One of the consequences of this is that errors in the estimate of SAR in this range will have a significant effect on the abundance and diversity estimates. Also implied is that overall improvements in productivity (e.g. through habitat restoration) will stabilize the population, making it less vulnerable to changes in SAR.



Source: Mobrand Biometrics 2004

Figure 58 Effects of SAR on EDT estimates of population productivity, abundance and diversity

Qualitative Habitat Analysis (QHA)

Qualitative Habitat Analysis (QHA), as modified from its original intent to meet the specific needs of the Methow subbasin planning process regarding bull trout and westslope cutthroat trout, has been a useful tool to organize and summarize a large amount of information into a useable format.

The QHA relies on the expert knowledge of natural resource professionals, with experience in a local area, to describe bull trout and westslope cutthroat trout use in the target stream. From this assessment, planners are able to develop hypotheses about the population and environmental relationships of the bull trout and westslope cutthroat trout. The ultimate result is an indication of the relative importance for restoration and/or protection management strategies at the sub-watershed scale addressing specific habitat attributes.

The primary strength of the QHA is its ability to conveniently store and summarize a substantial amount of information relating focal species to their habitats. Consequently, planners chose to view the assessment as a tool for examining four fundamental questions:

1. Where have significant bull trout and westslope cutthroat trout use changes occurred since the historic reference condition?

7. What changes are thought to have most significantly affected the distribution and abundance of bull trout and westslope cutthroat trout (sub-populations within the watersheds)?
8. Where are the greatest opportunities to protect and/or enhance habitat attributes that will potentially provide the greatest benefits to fish populations within the subbasin?

Current and historic focal species distribution was described by ranking focal species use for each of the stream reaches. The QHA values were compared to existing literature to ensure consistency and credibility as well as the EDT habitat analysis.

The technical sub-committee used the subbasin vision, goals and biological objectives as a backdrop for describing a desired future condition. The technical team evaluated where the most affective application of various actions might occur, and described the extent to which specific attributes may need to change in order to achieve stated goals and objectives.

Each of these reference conditions was evaluated and compared. Findings from this evaluation are found in the Assessment / Synthesis sub-chapter within this document.

The QHA was used in the Methow subbasin planning process for two fundamental reasons; a) the tool is a straight forward means to summarize a substantial amount of information, associated with bull trout and westslope cutthroat trout, in an accessible manner, and; b) rules of bull trout and westslope cutthroat trout have not been developed for the EDT model. The subbasin planners have developed various approaches to communicate the findings of the QHA to the general public and scientific community as a basis for the development of management strategy recommendations. Regardless of the shortcomings of the QHA, the methodology was successful in its intent in describing the fundamental changes in bull trout and westslope cutthroat trout use that have occurred in the Methow subbasin, and has served as a catalyst for describing future management direction.

Technologies Employed

Scaled Versus Unscaled EDT Output

We analyzed two sets of EDT model output: scaled and unscaled. The unscaled output estimated the total potential for increase or decrease (because of restoration or protection actions) within an assessment unit (AU), regardless of its length relative to other AUs.

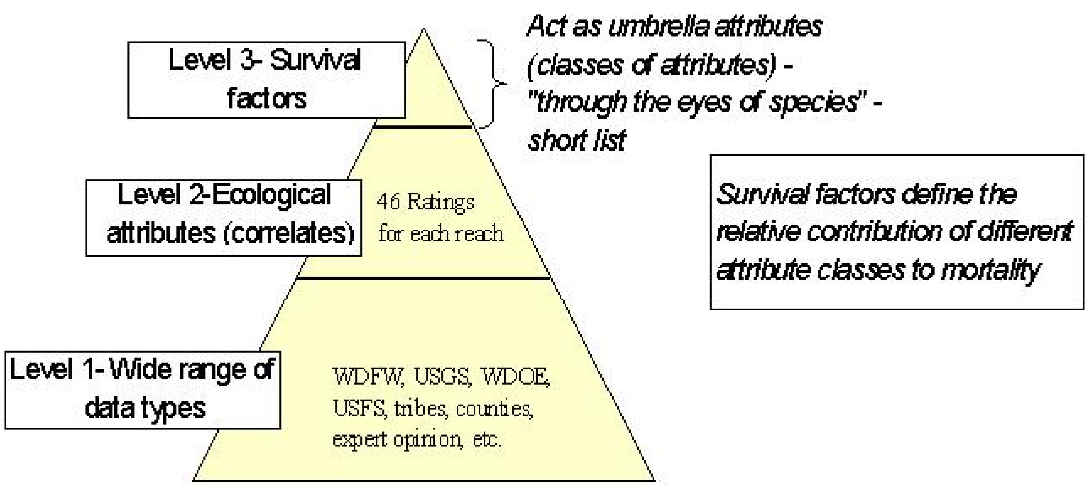
Unscaled output allowed us to evaluate in-basin versus OOSE, and showed the critical areas for restoration and protection, regardless of size or efficiency. The scaled output calculated the potential benefit on a per kilometre basis, which gave us “bang for the buck,” or the most efficient areas to work in to benefit focal species.

Scaled output can de-emphasize large, important areas, and there may be some segments within these larger AUs where it is just as efficient to perform restoration or protection actions. Therefore, it is important to consider both scaled and unscaled output, as well as site-specific nuances within the AUs when evaluating the final list of priority AUs.

Methow Subbasin Habitat Assessment Methods

The Methow subbasin habitat was assessed using the Ecosystem Diagnosis and Treatment (EDT) method; EDT is an analytical model relating habitat features and biological performance to support conservation and recovery planning for salmonids (Lichatowich et al. 1995; Lestelle et al. 1996; Mobrand et al. 1997; Mobrand et al. 1998). It acts as an analytical framework that brings together information from empirical observation, local experts, and other models and analyses.

The Information Structure and associated data categories are defined at three levels of organization. Together, these can be thought of as an information pyramid in which each level builds on information from the lower level (**Figure 59**). As we move up through the three levels, we take an increasingly organism-centered view of the ecosystem. Levels 1 and 2, together, characterize the environment, or ecosystem, as it can be described by different types of data. This provides the characterization of the environment needed to analyze biological performance for a species. The Level 3 category is a characterization of that same environment from a different perspective: “through the eyes of the focal species" (Mobrand et al. 1997). This category describes biological performance in relation to the state of the ecosystem described by the Level 2 ecological attributes.



Source: Mobrand Biometrics 2004

Figure 59 Data/information pyramid—information derived from supporting levels

The organization and flow of information begins with a wide range of environmental data (Level 1 data) that describe a watershed, including all of the various types of empirically-based data available. These data include reports and unpublished data. Level 1 data exist in a variety of forms and pedigrees. The Level 1 information is then summarized or synthesized into a standardized set of attributes (Level 2 ecological attributes) that refine the basic description of the watershed. The Level 2 attributes are descriptors that specify physical and biological characteristics about the environment relevant to the derivation of the survival and habitat capacity factors for the specific species in Level 3. Definitions for Level 2 and Level 3 attributes can be found at www.edthome.org, together with a matrix showing associations between the two levels and various life stages.

The Level 2 attributes represent conclusions that characterize conditions in the watershed at specific locations, during a particular time of year (season or month), and for an associated management scenario; hence, an attribute value is an assumed conclusion by site, time of year, and scenario. These assumptions become operating hypotheses for these attributes under specific scenarios. Where Level 1 data are sufficient, these Level 2 conclusions can be derived through simple rules; however, in many cases, experts are needed to provide knowledge about geographic areas and attributes where Level 1 data are incomplete. Regardless of the means whereby Level 2 information is obtained, the characterization it provides can be ground-truthed and monitored over time through an adaptive process.

To perform the assessment we first structured the entirety of the relevant geographic areas, including marine waters, into distinct habitat reaches. The Methow drainage was subdivided into 148 stream segments within the estimated historic range of steelhead by an assembled technical workgroup (**Table 43**). We identified reaches on the basis of similarity of habitat features, drainage connectivity, and land use patterns. Such a detailed reach structure, however, is counterproductive for displaying results and implementing a management plan. Therefore the reaches were regrouped into 13 larger geographic areas or AUs (**Table 43**). A set of standard habitat attributes and reach breaks, developed by MBI, were used for the mainstem Columbia and Snake Rivers, estuarine, nearshore, and deep water marine areas. We then assembled baseline information on habitat and human-use factors and fish life history patterns for the watersheds of interest. This task required that all reaches be completely characterized by rating the relevant environmental attributes.

Table 43 Stream reaches and assessment units (AUs) defined in the Methow River for Ecosystem Diagnosis and Treatment (EDT) modeling

Reach Codes	Location/Description	Assessment Unit
Met1-Met7	Methow River mainstem; mouth to RM 33 (1 mile below Beaver Ck)	Lower Methow
Met8-Met17	Methow River mainstem; RM 33 to Weeman Bridge (RM 60).	Middle Methow
Met18-Met23	Methow River mainstem; Weeman Bridge to Robinson Ck (RM 74)	Upper-Middle Methow
Met24-Met26a; EarlyW1-3, Cedar1; Lost1-3, Eureka1	Methow River mainstem; Robinson Ck to falls above Brush Ck.; Early Winters Ck (RM 0-8.2), Cedar Ck (RM 0-2.3), Lost R (RM 0-7.5), Eureka Ck (0-0.3)	Upper Methow
Gold1-4, GoldSF1-3, Foggy1, GoldNF1; Libby1-5, Smith1-2	Gold Creek; mouth to North Fork (RM 5.5), South Fork of Gold Creek to falls (RM 0-7.3); Foggy Dew Cr to falls (RM 0-2.7), North Fork Gold Ck to Crater Ck (RM 0-1.3); Libby Ck, mouth to confluence of N and S forks (RM 0-7.4), Smith Canyon Ck (RM 0-2.9)	Gold Ck/Libby Ck
Beav1-5, Fraz1-2;Bear1-2	Beaver Ck, mouth to South Fork Confluence (RM 0-10), Frazer Ck to Jack Ck (RM 0-4.7); Bear Ck, mouth to RM 6	Beaver Ck/Bear Ck

Reach Codes	Location/Description	Assessment Unit
Twisp1-11; LBridge1-4, Cany1-2, Butter 1-2, EF Butter1, WF Butter1	Twisp River, from mouth to Eagle Ck (RM 0-17.3); Little Bridge Creek to West Fork (RM 0-7.7), Canyon Ck to RM 2.2, Buttermilk Ck to E and W forks (0-2.8), West Fork Buttermilk Ck (RM 0-2.2), East Fork Buttermilk Ck (RM 0-2.6)	Lower Twisp
Twisp12-17a; Eagle1-2, War1-2, Reynolds1-2, South Cr1, North 1	Twisp River from Eagle Ck to falls at RM 31; Eagle Ck to falls at RM 0.5, War Ck to falls at RM 1.4, Reynolds Ck to falls at RM 0.7, South Ck to falls at 0.6, North Ck to falls at 0.5	Upper Twisp
Chew 1-6; Pearrygin Lake Ck1; Cub1; Boulder1;	Chewuch River, mouth to Eightmile Ck (RM 11.3); Pearrygin Lake Ck (RM 0-0.2), Cub Ck to falls at RM 0.41, Boulder Ck to falls at RM 1	Lower Chewuch
Chew7-16; Eight1-3, Twenty1, Dodd1, Lake1-4, Farewell1, Andrews1, Sheep1, Thirty1	Chewuch River, from Eightmile Ck to Chewack Falls (RM 11-35); Eightmile Ck (RM 0-14.2), Twentymile Ck (RM 0-0.5), Dodd Ck (RM 0-0.7), Lake Ck (RM 0-9.5), Farewell Ck (RM 0-0.4), Andrews Ck (RM 0-0.3), Sheep Ck (RM 0-0.4), Thirtymile Ck (RM 0-0.3)	Upper Chewuch
Wolf1-3; Hancock1	Wolf Creek, mouth to North Fork (RM 0-6.2); Hancock Ck, mouth to springs (RM 0-0.81)	Wolf/Hancock Ck
Goat1-6; LBoulder1-2	Goat Ck, mouth to Montana Ck (RM 0-3.2); Little Boulder Ck, mouth to Left Fork (RM 0-1.1)	Goat/L.Boulder Ck

3.22 Methow Subbasin EDT Results

Species Prioritization

Reach analysis tables (EDT consumer reports tables) were used to determine primary and secondary limiting factors within each Assessment Unit (AU). The Subbasin Core Team factored in the results of assessments on focal species and across all reaches in each AU. In general, a survival factor was considered a primary limiting factor if there were high or extreme impacts on key life stages. Exceptions included some reaches where sediment load or temperature only had a high impact to spawning or egg incubation. Additionally, a survival factor was considered a primary limiting factor if there were small to moderate impacts across most (9-12) life stages, thereby producing a cumulative impact that could be just as severe as high and extreme impacts on fewer life stages.

Secondary limiting factors generally had small to moderate impacts on several (5-8) life stages. An exception occurred with the survival factor “food”; when there were small to moderate impacts on two or three juvenile life stages in most of the reaches of a particular AU, then we considered it a secondary limiting factor. In most reaches and AU, the break between primary and secondary limiting factors was fairly obvious. In some cases where EDT results were not as obvious, other information, such as the Limiting Factors Reports, RTT Biological Assessment, professional opinion, and local knowledge were factored into the decision.

Species Findings – Ecosystem Diagnosis and Treatment (EDT)

Intraspecific priorities were generated using the Ecosystem Diagnosis and Treatment (EDT) model scaled (percent potential benefit / kilometre) for anadromous fish and qualitative habitat assessment method for resident fish. Well-coordinated (integrated) priorities were generated by giving preference to Endangered fish first, then Threatened, then all focal species. Categories (A,B,C) represents groups of AUs with the highest, intermediate, and lowest potential for benefit to focal species. Throughout the Methow subbasin, habitat diversity (floodplain connection, off-channel habitat, LWD, riparian vegetation) was the greatest limiting factor to anadromous fish (**Table 45**).

Other critical limiting factors included key habitat quantity (primarily a function of fewer quality pools for rearing and holding and fewer pool tailouts for spawning), sediment load (turbidity, embeddedness, and percent fines), obstructions, and channel stability. Common secondary limiting factors included flow (reduced base flow, increased peak flow), food (reduced salmon carcasses and benthic productivity), and temperature (high summer temperatures) (**Table 45**).