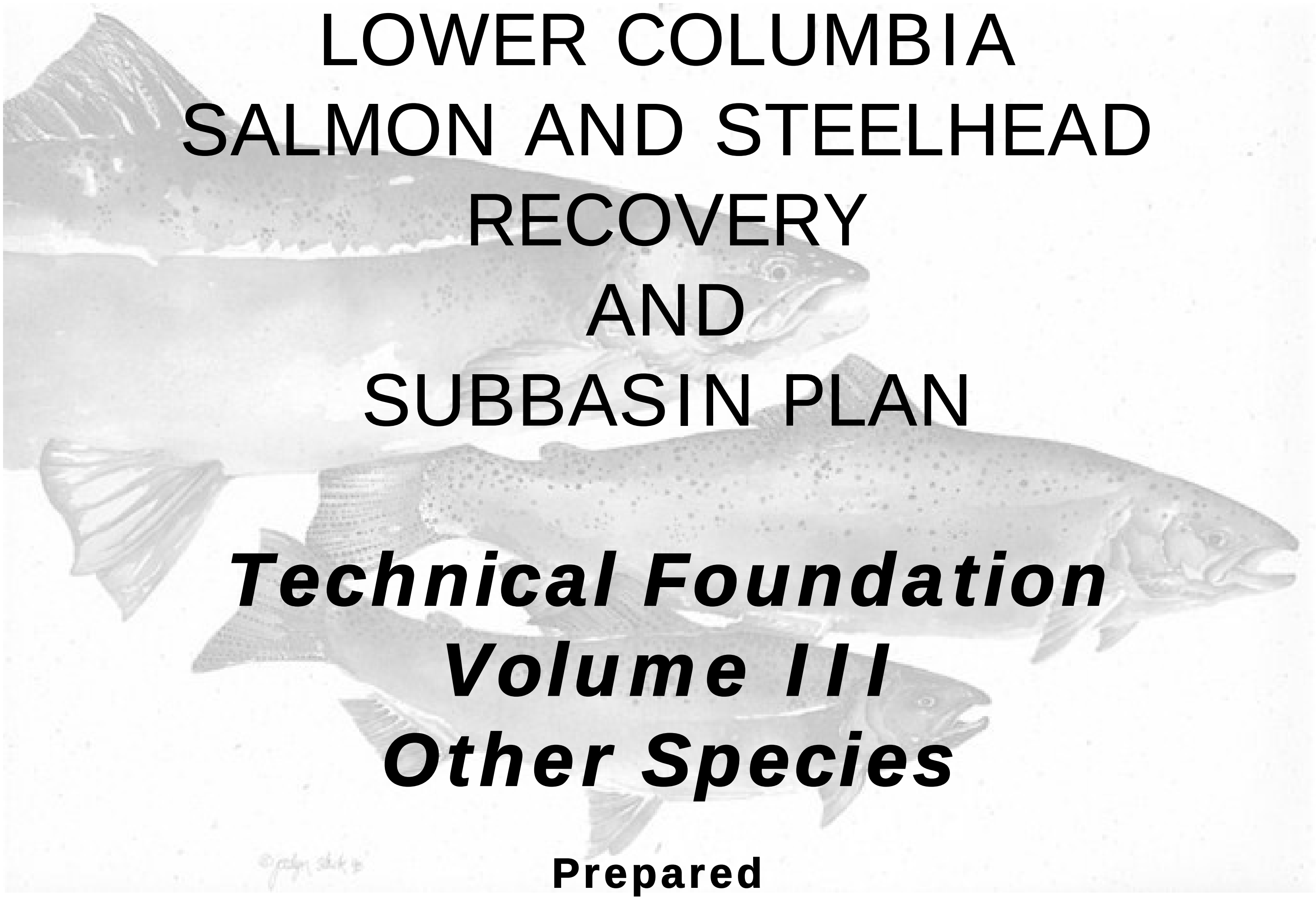

LOWER COLUMBIA
SALMON AND STEELHEAD
RECOVERY
AND
SUBBASIN PLAN

Technical Foundation
Volume III
Other Species

Prepared
For
Northwest Power
And
Conservation Council

MAY 28, 2004
DRAFT

Prepared By:
Lower Columbia Fish Recovery Board

Preface

This is number three of six volumes of a Technical Foundation for Recovery and Subbasin Planning prepared under direction of the *Washington Lower Columbia River Fish Recovery Board*. This information provides a basis for an integrated Salmon Recovery and Subbasin Plan prepared by the *Fish Recovery Board*. The Technical Foundation is an encyclopedia of information relating to focal and other species addressed by the plan, environmental conditions, ecological relationships, limiting factors, existing programs, and economic considerations. The Technical Foundation summarizes existing information and new assessments completed as part of the planning process. A separate Executive Summary document provides an overview of the entire Technical Foundation.

Technical Foundation volumes include:

Vol. I	Focal Fish Species	*Species overviews, limiting factors, recovery standards, and status assessments for lower Columbia River chinook salmon, coho salmon, chum salmon, steelhead, bull trout, and cutthroat trout*
Vol. II	Subbasins	*Fish populations and habitat conditions in each of 11 Washington lower Columbia River subbasins*
Vol. III	Other Species	*Descriptions, status, and limiting factors of other fish and wildlife species of interest to recovery and subbasin planning*
Vol. IV	Existing Programs	*Descriptions of Federal, State, Local, Tribal, and non governmental programs and projects that affect or are affected by recovery and subbasin planning*
Vol. V	Economic Assessment	*Potential costs and economic considerations for recovery and subbasin planning*
Vol. VI	Appendices	*Methods and detailed discussions of assessments completed as part of this planning process*

This work was funded by the *State of Washington* and the *Northwest Power and Conservation Council*. The Technical Foundation was completed primarily by the *Washington Lower Columbia Fish Recovery Board*, *Washington Department of Fish and Wildlife*, *S.P. Cramer and Associates*, and *The White Company*. This second draft of the Technical Foundation incorporates suggestions and revisions provided by a wide array of agency and public reviewers of an initial draft distributed in 2003. Additional opportunities for review and revision of the current draft will occur as part of ongoing recovery and subbasin planning processes

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White Sturgeon

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1.0 White Sturgeon (*Acipenser transmontanus*)

Distribution of the world's Acipenseriformes has been classified according to nine biogeographic provinces identified by Bemis and Kynard (1997). All extant taxa within Acipenseriformes exist in biogeographic provinces within the Holoarctic region (Bemis *et al.* 1997a). All known fossil Acipenseriformes were also found exclusively in north temperate localities (Grande and Bemis 1991; Jin 1995; Bemis *et al.* 1997b; Grande and Bemis 1997; in Bemis and Kynard 1997). With the exception of the Pearl River in China, all spawning within Acipenseriformes occurs in rivers located entirely within the north temperate zone of Asia, Europe, and North America (Bemis and Kynard 1997). The absence of Acipenseriformes from waters south of the north temperate zone is likely explained by geographic distribution of empirical thermal maxima (~68°F [20°C]) for successful maturation and early development of many sturgeon species (Artyukhin 1988; Conte *et al.* 1988; Detlaff *et al.* 1993; Anders and Beckman 1995; Bemis and Kynard 1997).

Sturgeons *(Acipensaeridae)* have become the subject of intense worldwide conservation efforts in response to increasing numbers of imperiled and extirpated sturgeon populations (Rochard 1990; Birstein 1993; Waldman 1995; Bemis *et al.* 1997; Birstein *et al.* 1997, 1997a, 1997c; Secor *et al.* 2002). In North America, sturgeons were second only to marine *Sebastes* (Scorpaenidae) in terms of numbers of threatened and endangered species (Musick *et al.* 2000; Secor *et al.* 2002). Being predominantly associated with the world's large river systems, sturgeons have collectively exhibited declining population trajectories due to severe large-scale habitat alterations and the effects of harnessing the world's large rivers for human purposes. Overharvest and habitat loss, degradation, and alteration are the causes most commonly cited of sturgeon population declines (Birstein 1993; Bemis and Kynard 1997; Waldman 1995; *et al.* 1997c; Anders *et al.* 2002; Secor *et al.* 2002).

Sturgeons are evolutionarily unique, ancient fish in need of modern protection (Birstein 1993; Waldman 1995; Anders 2000). Ironically, the very life history traits and behaviors responsible for sturgeons' long successful evolutionary history now serve as obstacles to their conservation, management, and recovery (Secor *et al.* 2002). Sturgeons worldwide share a predominantly threatened status (Birstein 1993; Findeis 1997; Birstein *et al.* 1997), as do many populations of North American taxa (Birstein 1993; Waldman 1995; Beamesderfer and Farr 1997; Birstein *et al.* 1997c; Secor *et al.* 2002). Despite valuable recently published collections of peer-reviewed research on various aspects of sturgeons (Birstein *et al.* 1997c; Bruch *et al.* 2001; Van Winkle *et al.* 2002), much remains unknown or poorly understood about many sturgeon taxa, including the white sturgeon, *(Acipenser transmontanus).*

White sturgeon are endemic to the Pacific coast of North America and its major river systems west of the Rocky Mountain Continental Divide, from central California to the Gulf of Alaska and the Aleutian Islands (Scott and Crossman 1973) (Figure 1-1). Sturgeon have been reported in other rivers of Yukon and Alaska, including the Taku, Skeena, Nass, and Yukon Rivers (Perrin *et al.* 1999). However, Lane (1991) suggested that these observations in the extreme northern extent of the species' range may have been of green sturgeon (*A. medirostris*) rather than white sturgeon.

Although white sturgeon occupy marine and estuarine habitats, marine residence is not required (Perrin *et al.* 1999). Thus, white sturgeon can be referred to as facultatively anadromous where dams have not blocked or restricted their access to marine and estuarine habitats. Although white sturgeon are found along the Pacific Coast from central California to the Gulf of Alaska, spawning populations have been confirmed in only three large river drainages: Columbia, Sacramento-San Joaquin, and Fraser (PSMFC 1992) as illustrated by the following map. Individuals have been observed as far south as Ensenada, Mexico, but did not appear to represent spawning populations (Moyle 1976).

Unlike salmonid fishes, white sturgeon do not require specific physiological changes (e.g. smoltification) prior to entering salt water, and can freely migrate between fresh and salt water environments or remain in estuarine habitat for prolonged periods (DeVore *et al.* 1999). Empirical tag-recapture data have confirmed their ability to migrate in excess of 1,550 miles (2,500 km) within, between, and among major river systems of western North America (DeVore *et al.* 1999; ODFW 1996). However, all sturgeons spawn exclusively in fresh water (Bemis and Kynard 1997).

The following paragraph from Parsley *et al.* (2002) summarizes challenges to restoring natural recruitment of white sturgeon populations in altered large river systems:

> *Recovery or maintenance of sturgeon populations through natural production in perturbed rivers requires adequate knowledge of the abiotic and biotic factors that influence spawning and cause mortality of embryonic, larval, and juvenile life stages. Although year-class strength of white sturgeon is determined within 2-3 months after spawning, little is known about specific causes of mortality to early life stages during this period. Initial spawning success is critical in the development of a strong year-class, and maximized recruitment may be dependent upon water temperature and the availability of optimal in-river habitat. Analyses have shown that increased river discharge combined with suitable water temperatures during spawning, egg incubation, yolk-sac larvae dispersal, and first exogenous feeding result in greater recruitment. However, little is known about the importance of other variables, such as food availability or losses due to predation that influence year class strength*

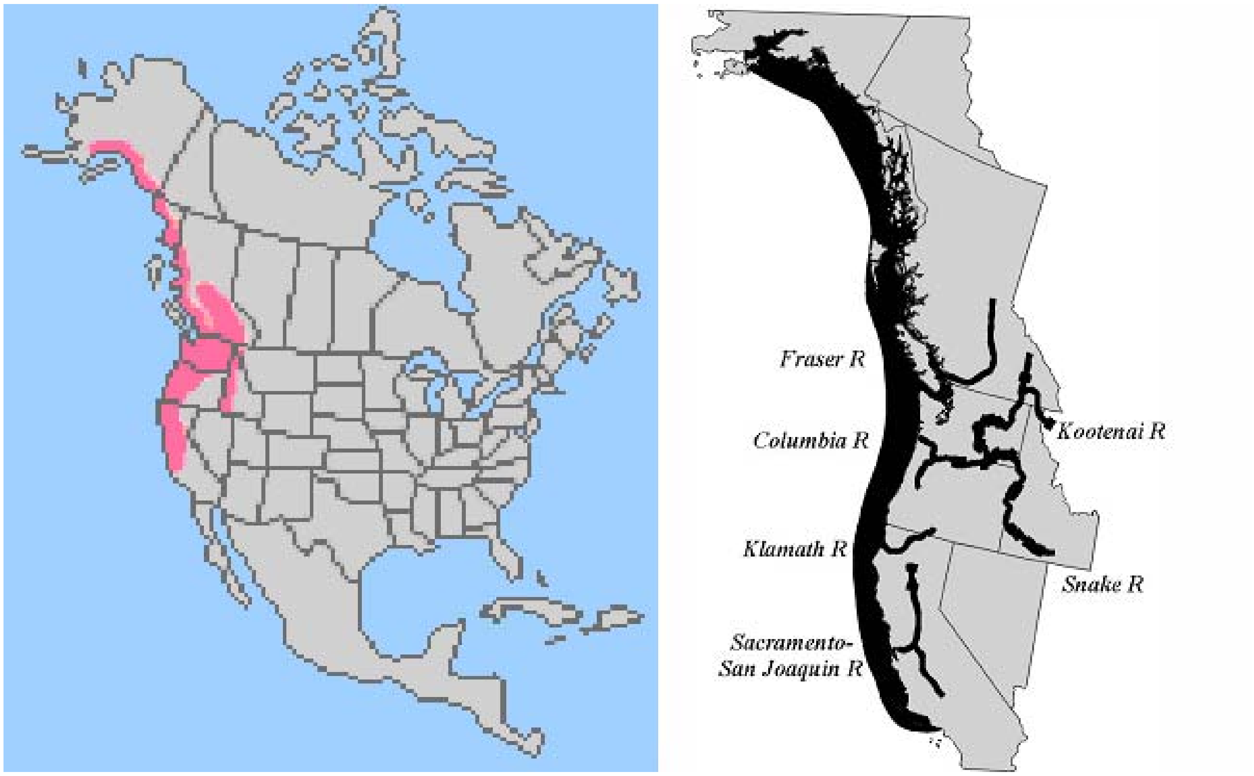

Figure 1-1. Range of white sturgeon (*Acipenser transmontanus*) along the Pacific Coast of North America. White sturgeon inhabit large Pacific coastal river systems and adjacent marine environments.

The lower Columbia River downstream from Bonneville Dam has the most productive white sturgeon population in the species' range (DeVore *et al.* 1995). This high productivity supports healthy sport and commercial fisheries with mean annual harvests since 1992 ranging from 40,000 to 55,000 (Figure 1-2). The sturgeon fishery ranks as the largest sport fishery in the Columbia Basin in terms of effort, with 10-year annual angler trips exceeding 175,000; in some years, angler trips exceed 200,000. Factors most responsible for the favorable production potential of the population are access to marine areas, abundant food resources, and consistently favorable hydrologic conditions during the spawning timeframe, which enhances recruitment (Parsley and Beckman 1994; DeVore *et al.* 1995; Counihan *et al.* in press). This high productivity can be sustained in the long term only with careful scientifically-based management.

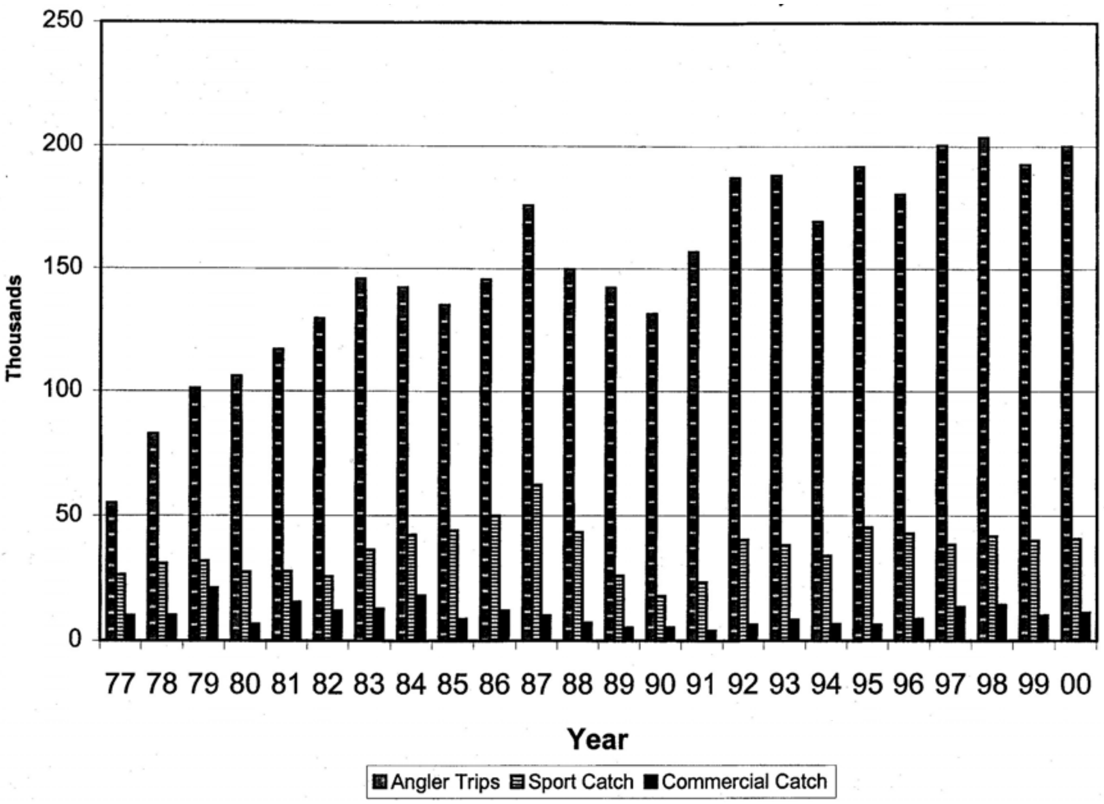

Figure 1-2. Sturgeon effort and catch on the lower Columbia River, 1977-2000

The longevity, slow growth, and delayed maturation of sturgeon make them vulnerable to overexploitation (Rieman and Beamesderfer 1990; Rochard *et al.* 1990; Birstein 1993). Excessive harvest during the 19th century resulted in the collapse of Columbia River sturgeon stocks. Intensive sturgeon fishing on the Columbia River began in 1889 and peaked in 1892 with about 2,500,000 kg (5.5 million pounds) of sturgeon landed. The stock was depleted by 1899 after a 10-year period of unregulated exploitation (Craig and Hacker 1940). Season, gear, and minimum size restrictions failed to bring about an increase in sturgeon production as evidenced by poor yields during the first half of this century.

The lower Columbia River sturgeon population rebounded after a maximum size regulation designed to protect sexually mature sturgeon was enacted in 1950. Annual harvests doubled by the 1970s and doubled again by the 1980s. Increased interest in the recreational sturgeon fishery was due to decreased salmon fishing opportunities, increased stock size, and greater appreciation of sturgeon as gourmet fare. In 1987, 72,100 white sturgeon were harvested in the lower Columbia River—a recent year record. Research indicates that the harvest rate of 30% of the 3-6 foot population, estimated to have occurred during 1985–87, was twice what the population could sustain in the long term.

Management actions to reduce the annual harvest rate in lower Columbia River sturgeon fisheries to a long-term sustainable level were decided on between 1988–97, and a management accord was struck between ODFW and WDFW to manage fisheries to assume adequate recruitment to the broodstock population. The legal size slot for lower Columbia River white sturgeon eventually was reduced to 42-60 in for sport fisheries and 48-60 in for commercial fisheries. The daily bag limit was reduced to one fish and the annual possession limit to ten fish. Maximum harvest guidelines and allocations also were placed on lower Columbia River sport and commercial fisheries.

White sturgeon historically had access from the ocean all the way to the Columbia's Canadian headwaters and Shoshone Falls in the upper Snake River. Mainstem dams have now fragmented sturgeon habitat into short riverine sections connected by long impoundments. White sturgeon in the Columbia and Snake Rivers have been isolated into at least 30 separate reaches, functionally extirpated from eight reaches, and are likely to become extirpated in another eight without intervention. Remaining subpopulations are restricted primarily to reaches with significant riverine habitat; subpopulations in marginal habitat areas have been lost, or consist of a few remnant individuals. A significant white sturgeon population remains in Bonneville Reservoir between Bonneville and The Dalles Dams, although this impounded population is substantially less productive than the anadromous population in the free-flowing river downstream from Bonneville Dam.

1.1 Life History & Requirements

In addition to pre-spawning recruitment failure mechanisms (e.g. stock limitation) a variety of early life (post-spawn) mortality factors may affect white sturgeon egg, larval, fingerling, and YOY as well as additional density-dependent and density-independent factors. These factors are illustrated in the following charts (Figure 1-3 and Figure 1-4).

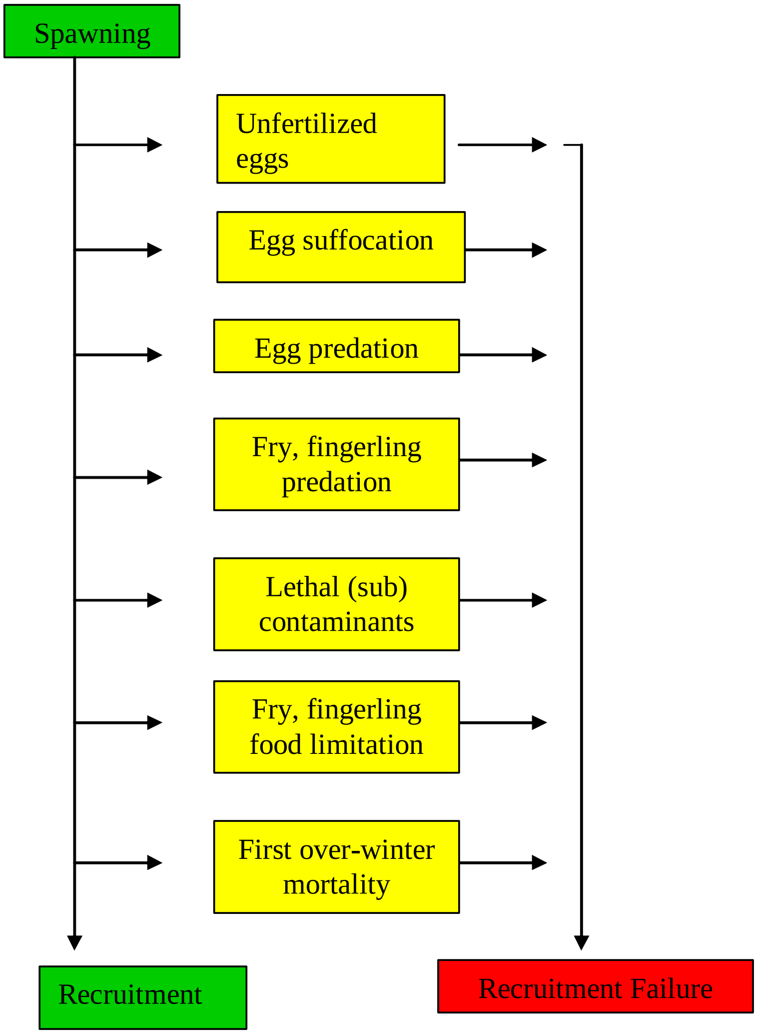

Figure 1-3. Potential early life mortality factors affecting white sturgeon (from Anders et al. 2002).

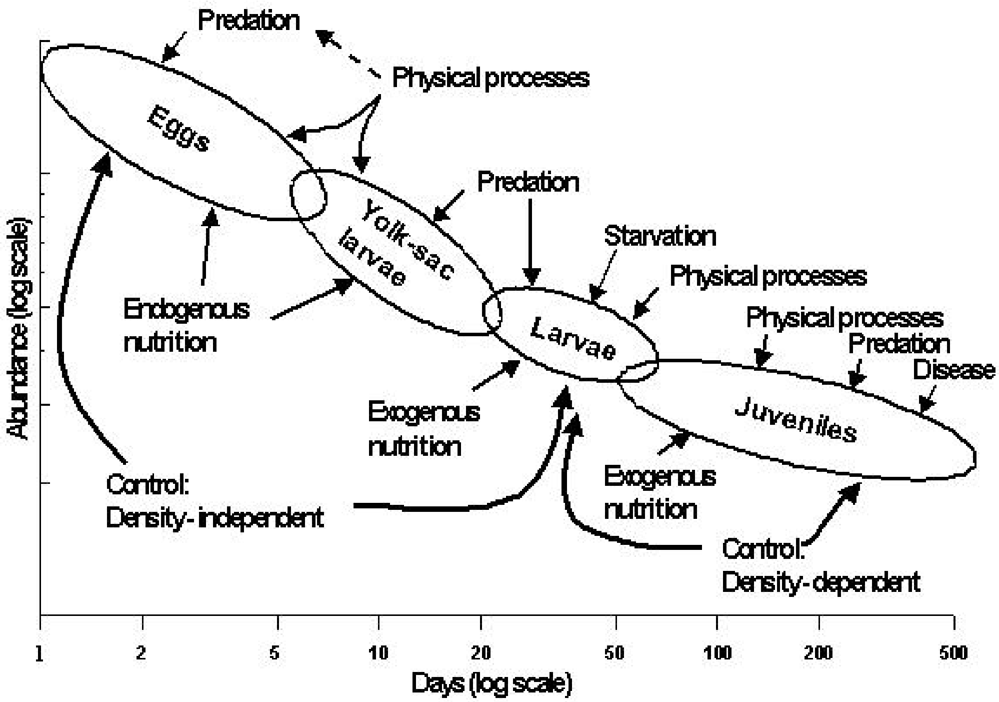

Figure 1-4. Conceptualization of the recruitment process adapted from Houde (1987) for white sturgeon showing probable sources of death, nutrition, and mechanisms likely controlling early life stages. Negative slopes in abundance throughout each life stage are an approximation not a measure of the degree of expected mortality (from Parsley et al. 2002).

1.1.1 Spawn timing and conditions

Timing of white sturgeon spawning is largely a function of water temperature, which varies annually, but is linked to time of year and associated photoperiods. White sturgeon in the Columbia Basin generally spawn from April through July, when water temperatures across the basin range from 46°F (8°C) in upper basin areas to near 68°F (20°C) in lower river areas. Table 1-1 illustrates physical habitat conditions (Parsley *et al.* 2002; Paragamian *et al.* 2001; RL&L 1994, 1996; Golder Associates 2003; IPC 2003). In the lower Columbia River, annual white sturgeon spawning appears to be triggered consistently when water temperature reaches 50°F (10°C) (M. Parsley, US Geological Survey, G. McCabe, NMFS (retired), personal communication). Spawning in the four impoundments farthest downstream occurs exclusively in tailrace areas immediately downstream from hydropower dams when water temperatures reach 54°F (12°C) (Parsley *et al.* 1993). Because water temperatures generally reach spawning temperatures first in downstream areas of the Columbia Basin, annual spawning is usually initiated downstream from Bonneville Dam when water temperatures reach 50°F (10°C), followed by spawning activity in each adjacent upstream tailrace when lower impoundment water temperatures reach and exceed 54°F (12°C). Most spawning occurs in the four farthest downriver Columbia River impounded areas at 57°F (14°C) (Parsley *et al.* 1993; Anders and Beckman 1995) with an optimum range generally cited as 54-57°F (12-14°C) for those areas. Paragamian *et al.* (2001) reported that Kootenai River white sturgeon spawned when water temperatures were between 47 and 54°F (8.5-12^{o}C)

Table 1-1. Physical habitat conditions at sites where white sturgeon (*Acipenser transmontanus*) eggs were collected in the Columbia, Fraser, and Sacramento River basins (from Anders 2002, Chapter 1).

Location		Water Temperature *	Mean Column Velocities (m/s)**	Velocity Near Substrate	Substrate Type	References
Lower Columbia River	1987–91	10-18	1.0-2.8	0.06-2.4	Boulder	Parsley *et al.* 1993
Columbia River Impoundments	1987–91	12-18	0.81-2.10	0.52-1.62	Cobble	Parsley *et al.* 1993
Kootenai River	1994	7.8-11.2	0.03-0.27	—	Fine sediment and sand	Anders 1994
	1995	8.4-12.9	0.68	0.93	Fine sediment and sand	Anders and Westerhof 1996
	1991–98	8.5-12.0	0.19-0.83	—	Fine sediment and sand	Paragamian *et al.* 2001
Columbia River, BC	1993	15.5-17.0	—	—	Clean small boulder, large cobble	Hildebrand and McKenzie 1994
	1995	15.5-21.6	0.5-1.8	—	Bedrock, boulder, cobble	RL&L 1996
Fraser River, BC	1998	15.1	—	—	Bedrock	RL &L 1998; Perrin *et al.* 1999
Sacramento River, CA	1970	14-22	—	—	Gravel	Stevens and Miller 1970
	1973	—	—	—	Mud and sand	Kohlhorst 1976

*°C; **m/s=meters per second

Empirical data generally support negative correlations between elevation, latitude, and mean spawning temperatures (i.e. farthest upstream and farthest north populations spawn at coldest mean temperatures). Wang *et al.* (1985) reported that white sturgeon embryos died when exposed to 164°F (8°C), whereas 68°F (20°C) water temperature was lethal to all exposed larvae. Anders and Beckman (1995) reported 98% mortality (129 of 132) of all eggs collected from The Dalles Pool during 1987 at water temperature of 64°F (18°C) and warmer. However, these authors also documented egg mortality in water 55 to 63°F (13-17°C), indicating additional egg mortality factors.

White sturgeon generally spawn in high velocity areas associated with gravel and larger substrates (Wydowski and Whitney 1979; Simpson and Wallace 1981; RL&L 1994, 1996; Perrin *et al.* 1999; Parsley *et al.* 2002; Paragamian *et al.* 2001; Golder Associates 2003, IPC 2003). Hard-bottom, high-velocity, structured habitats with adequate interstitial space are critical as spawning and incubation substrate and predation refuge areas for broadcast-spawning white sturgeon (Parsley *et al.* 1993; Perrin *et al.* 1999; Parsley *et al.* 2002; Secor *et al.* 2002).

The following three paragraphs on spawn timing and associated conditions are from Parsley *et al.* 2002:

Many physical factors and environmental conditions influence spawning. The amount and quality of spawning habitat available to individual populations of white sturgeon differs among reaches because of channel morphology and among years because of variation in river discharge (Parsley and Beckman 1994). White sturgeon spawning in the Columbia and Snake rivers generally occurs in areas with high water velocities, coarse substrates and water depths of 3 m or more (Parsley et al. 1993; R. L. & L. Environmental Services Ltd. 1994; Parsley and Kappenman 2000). Spawning in high water velocities would separate and disperse the adhesive eggs, and the coarse substrates would provide a good surface for the adhesive eggs to attach.

Impoundments have reduced the hydraulic slope of the river over vast reaches and inundated many rapids and falls that historically may have provided spawning habitat for white sturgeon. Because of differences in channel morphology and a greater hydraulic slope, the free-flowing reaches provide more spawning habitat than the impoundments at reduced discharges (Parsley and Beckman 1994) causing variability in spawning habitat quantity and quality among areas within years.

Inter-annual variation in spawning timing is also caused by the thermal regime of rivers within the Columbia River Basin. Developments within the Columbia River Basin for hydroelectric power generation and operations at dams that are used to store water for flood control and power generation have resulted in temperature variations from the historic thermal regime. The timing and duration of the spawning season for white sturgeon in any given year vary with water temperature. White sturgeon spawning in the Columbia River Basin generally occurs when water temperatures are between 10-18°C (Parsley et al. 1993; R. L. & L. Environmental Services; Ltd. 1994) with the peak of spawning occurring when temperatures are generally between 13 and 15°C. These temperatures can occur for variable periods during the months of April, May, June, or July. Kootenai River white sturgeon also spawn during May and June but at water temperatures that are much cooler. Typically, spawning by white sturgeon in the Kootenai River begins when temperatures are 8-9°C and ceases when temperatures approach 12°C (Paragamian et al. 1995; Paragamian et al. 1997). Though the primary force behind the thermal regime is regional climatic conditions, the hydropower system is often manipulated to provide cooler water temperatures during the summer to benefit outmigrating juvenile anadromous salmonids. These manipulations can lower river water temperature by several degrees and often occur during times when white sturgeon are spawning. It is unknown, but probable, that these temperature variations disrupt spawning activities by white sturgeon.

1.1.2 Incubation

Recruitment failure in sturgeon populations frequently results from loss and degradation of spawning, incubation and early rearing habitats (Beamesderfer and Farr 1977; Bemis and Kynard 1997; Jager *et al.* 2001; Paragamian *et al.* 2001; Anders *et al.* 2002; Parsley *et al.* 2002). As mentioned above, hard-bottom, high-velocity, structured habitats with adequate interstitial space are critical for broadcast-spawning white sturgeon (Parsley *et al.* 1993; Perrin *et al.* 1999; Parsley *et al.* 2002; Secor *et al.* 2002). Furthermore, hypoxia (oxygen limitation) may have disproportionately negative effects on sturgeons, relative to other fauna, due to their limited capacity to osmoregulate at low dissolved oxygen concentrations (Klyashtorin 1976; Secor and Gunderson 1998 as cited in Secor *et al.* 2002). Although hypoxic effects may be particularly important during the first year of life due to increased sensitivity and reduced ability of sturgeons—especially incubating embryos—to escape anoxia environments (Secor and

Niklitschek 2001), specific oxygen and gas exchange requirements for incubating white sturgeon embryos are currently unknown.

In addition to potential threats of suffocation, hypoxia, and reduced gas exchange, demersal white sturgeon embryos are vulnerable to fish predation (Anders 1994, 1996; Miller and Beckman 1996; Parsley *et al.* 2002). During 1994 and 1995, 632 stomach content samples from predatory fishes collected from the Kootenai River (northern pikeminnow *Ptychocheilus oregonensis*, peamouth chub *Mylocheilus caurinus* and suckers *Catastomus spp*.) were analyzed (Anders 1996). Of 428 naturally-spawned white sturgeon eggs collected from the Kootenai River during 1994 and 1995, 12.2% (52) were collected from 623 predatory fish stomach samples. Although a low percentage of the total catch, predation likely was underestimated due to sampling and observational constraints. Miller and Beckman (1996) reported the occurrence of one to 70 white sturgeon eggs in guts of four omnivorous fishes in the Columbia River. These authors noted that a single largescale sucker (*Catastomus macrocheilus*) consumed 70 white sturgeon eggs.

1.1.3 Emergence

Emergence is typically a term associated with post-hatching salmon ecology and reproductive biology, and is not directly associated with white sturgeon embryos, which typically hatch in less than two weeks at a mean incubation temperature of 50°F (10°C) (Wang *et al.* 1985). Important details concerning subsequent larval white sturgeon behavior and emergence from interstitial spaces within river substrates are provided in the following section.

1.1.4 Larvae

Brannon *et al.* (1985) conducted laboratory studies to characterize distribution behaviors of Columbia River white sturgeon larvae and fry. These authors concluded:

> *substrate composition in a river may influence both the emergence and settling response of white sturgeon larvae and could affect whether they remain in an area once they become bottom oriented. Upon hatching, larvae enter the water column and are subject to the influences of current. Larvae then seek the substrate for places that provide cover. Larvae remained in the substrate until yolk is absorbed and feeding initiated. Larvae were noted to enter just about every conceivable space where they could hide their head. Beneath rocks, gravel interstices, amongst plants, and under detrital material were the places harboring the larvae during the hiding phase.*

Larval white sturgeon were observed in aquaria to burrow into fine sediments, resulting in mortality by suffocation in some observed cases (E. Brannon, University of Idaho, personal communication). If these behaviors represent those in the wild, empirical observation indicates the importance of suitable larval rearing habitats, including interstitial space among substrate particles of appropriate sizes.

The following three paragraphs regarding larval white sturgeon ecology are from Parsley *et al.* (2002):

> *In addition to successful hatching of embryos, onset of exogenous feeding constitutes a critical period of potentially high mortality. However, virtually no empirical data could be found addressing prey selection and food availability for naturally produced white sturgeon at onset of exogenous feeding. The larval stage for white sturgeon generally lasts approximately 25 to 30 days. Following yolk sac absorption, larvae end their hiding phase and move out onto the substrate to begin feeding. Mortality*

> *of larval fishes is often greatest during the period of transition from endogenous to exogenous feeding (Hjort 1926). It is not known if white sturgeon larvae experience high mortality rates at this juncture in natural populations, but it is probable that some of the variation in year-class strength observed in white sturgeon populations is due to mortality during the larval stage.*
>
> *Starvation is one biotic factor thought to regulate juvenile fish abundance in some freshwater and marine fish populations (Rice et al. 1987; Sinclair 1988). It is unknown if or when irreversible starvation (May 1974) occurs for larval white sturgeon deprived of food. Muir et al. (2000) found no evidence of larval starvation in the Columbia River downstream from Bonneville Dam and in the two lowermost impoundments. In a laboratory study, if food was not present, white sturgeon larvae re-entered the water column, presumably to be displaced farther downriver to a food source (Brannon et al. 1985a). White sturgeon larvae collected in the Columbia River fed primarily on amphipods of the genus* Corophium *(Muir et al. 2000), a food that historically was found in the Columbia River estuary but not upriver in free-flowing environments. Other food items consumed that would have been historically available to larvae upstream of the upper extent of* Corophium *included copepods, Ceratopogonidae larvae and Diptera pupae and larvae.*
>
> *Another source of mortality that can have significant affects on year-class strength is predation. Predation on white sturgeon larvae has been noted in laboratory experiments (Brannon et al. 1986) but has not been investigated under natural conditions. Larvae develop sharp scutes as they grow, possibly reducing their vulnerability to predation. Potential predators collected in association with larvae included bridgelip sucker* Catostomus columbianus, *largescale suckers, bullheads Ameiurus spp., common carp, peamouth* Mylocheilus caurinus, *chiselmouth* Acrocheilus alutaceus, *northern pikeminnow, prickly sculpin, larger white sturgeon, and starry flounder* Platichthys stellatus.

1.1.5 Juvenile

The following three paragraphs regarding juvenile white sturgeon ecology were taken directly from Parsley *et al.* (2002):

> *White sturgeon larvae metamorphose into juveniles within 3–4 months after egg fertilization. Predation, starvation, disease, parasitism, and physical processes caused by direct and indirect human actions reduce juvenile white sturgeon numbers. For many fish species, relative year-class strength is set prior to this life stage (Bradford 1992). Losses of juvenile white sturgeon to predation are probably slight because of the protective scutes, benthic habits, and fast growth. Only one juvenile white sturgeon was consumed by a channel catfish Ictalurus punctatus during a study of the gut contents of more than 4,780 northern pikeminnow, 1,050 walleye Stizostedion vitreum, 4,800 smallmouth bass Micropterus dolomieui, and 650 channel catfish (US Geological Survey, unpublished data). Other previously listed predators on young white sturgeon were not examined in that study.*
>
> *Juvenile white sturgeon feed primarily on benthic invertebrates (McCabe et al. 1993; Muir et al. 2000). Studies investigating productivity of benthic invertebrates that juvenile white sturgeon prey on between free flowing and impounded areas are lacking. Generally, growth rates, mean length at age, and condition factors of juvenile white sturgeon (1-8 years of age) were greater for those captured in the impounded areas than*

of those collected in the free-flowing reach (Miller and Beckman 1992), suggesting that food resources for juvenile white sturgeon were more limiting in the free-flowing reach than in the impounded areas at existing white sturgeon densities.

Losses of fish to disease and parasites in the wild are difficult to quantify. Hatchery reared white sturgeon are susceptible to many of the same diseases and parasites common to other fishes reared in culture facilities (LaPatra et al. 1995; Conte et al. 1988) and the white sturgeon iridovirus can cause significant mortality in cultured fish (LaPatra et al. 1994). This size-specific and stress-mediated virus has been found in white sturgeon throughout the Columbia River Basin (LaPatra et al. 1994). Fish weakened by disease or parasites could be more vulnerable to predation (Mesa et al. 1998) but this has not been investigated in white sturgeon. The nematode parasite Cystoopsis acipenseri is common to smaller white sturgeon and creates blister-like cysts located just under the skin of affected fish (McCabe 1993). The degree of infestation of white sturgeon by the nematode parasite varied spatially and temporally in the lower Columbia River and was greater in smaller white sturgeon (McCabe 1993). However, it is unknown if infestation increases mortality.

Human actions sometimes cause mortality of juvenile white sturgeon. Suction dredging in deep areas (66-85 ft [20-26 m]) in the lower Columbia River is known to seriously injure and kill juvenile white sturgeon (Buell 1992), and there is speculation that the dredging operations may attract feeding white sturgeon, compounding the losses. Lost and abandoned gill nets from commercial and subsistence fisheries can kill substantial numbers of juvenile and adult white sturgeon in impounded areas (M. Parsley, USGS Cook, Washington, Blaine Parker, Columbia River Inter-Tribal Fish Commission, personal communication), and large numbers of fish are occasionally killed during maintenance activities at the dams (J. DeVore, WDFW, personal communication). Mortality among sublegal-sized fish caused by hooking by anglers probably accounts for a minor loss of juvenile white sturgeon, but has not been investigated.

Juvenile white sturgeon recruited to the population after age 1 generally exhibit very high survival (e.g. 90%, Paragamian *et al.* in review). Thus, if habitat suitability and food availability are suitable and not limiting, it would appear that the juvenile white sturgeon life stage does not appear to likely produce population bottlenecks. However, based on results of elasticity analysis of life history attributes of three sturgeon species (Gross *et al.* In Press), the potential to increase population growth rate (λ) remained high for YOY and juvenile age classes. Simulated changes in fecundity had relatively little effect on the potential for increased population growth. Although YOY survival elasticity was equal to that of other juvenile ages, the overall opportunity for affecting λ was strongest at the YOY stage due to its exceptional potential to increase survival.

Regarding juvenile food habits, Scott and Crossman (1973) reported that age 0 white sturgeon diets consisted primarily of *Chironomid* larvae. Amphipods (*Corophium spp.)* accounted for 98% of diet items from 149 age 0 white sturgeon (0.78-10.5 in. TL [20-267 mm]) collected from Bonneville and The Dalles pools in the Columbia River from 1988 through 1991 (Sprague *et al.* 1993). Wydowski and Whitney (1979) reported that the stomachs of small white sturgeon in California contained primarily Mysis shrimp (*M. relicta*) and amphipods. Age 0 lake sturgeon *(Acipenser fulvescens)* in the Lake Winnebago system in Wisconsin were observed in close contact with the substrate, oriented upstream, and apparently feeding on drifting benthic organisms (Kempinger 1996). Kempinger (1996) also reported that species of Baetidae nymphs and dipteran larvae were the two principal organisms consumed by lake sturgeons during their first summer of life.

1.1.6 Adult

General life history characteristics of sturgeons were recently summarized by Bemis and Kynard (1997) and Kynard (1997). Sturgeons exhibit several life history forms, including:

- Diadromy—migrate between fresh and salt water
- Anadromy—spawn in fresh water, spend non-reproductive periods in marine environment
- Amphidromy—bi-directional, non-reproductive migration between fresh and salt water
- Potadromy—all feeding and reproductive migrations within a freshwater river system

Facultative potadromy, which occurs when dams prohibit expression of historically anadromous or amphidromous life history strategies is poorly understood, but accounts for most white sturgeon in impounded reaches of the Columbia River system in the US and Canada (Kynard 1997). Where not damlocked, and based on observed life history white sturgeon appear to be best described as facultatively anadromous. Regardless of life history strategies expressed, all sturgeons spawn exclusively in large freshwater river systems, often following upstream migrations of considerable distance (Bemis and Kynard 1997).

Like other sturgeons, white sturgeon are characterized by delayed onset of first reproduction. First maturation generally occurs from 10–20 years of age for males, and from 15 to 30 for females (Scott and Crossman 1973; Semakula and Larkin 1968; Conte *et al.* 1988; Paragamian *et al.* in review). This trait, coupled with empirically confirmed migratory and dispersal ability, are theorized to contribute to gene flow in white sturgeon (Brown *et al.* 1992, 1993). Furthermore, individual longevity ($\leq$ 82 years of age, Simpson and Wallace 1982) infrequently exceeding 100 years (Smith *et al.* 2001) also may contribute to observed migration, dispersal, and gene flow (Brown *et al.* 1993, 1996).

White sturgeon are iteroparous, communal spawners, which broadcast gametes into the water column where fertilization occurs before the demersal, adhesive embryos settle to the substrate (Wang *et al.* 1985; Conte *et al.* 1998; Paragamian *et al.* 2001, and references therein). In demographically viable white sturgeon populations, iteroparity provides the opportunity for within-year reproduction by numerous generations of fish. Reproductive periodicities vary between sexes; males may reproduce every 2-4 years, while females may reproduce no more frequently than at 5-year intervals (Conte *et al.* 1988; Chapman *et al.* 1996, Anders *et al.* 2002; Paragamian *et al.* in review). Simpson and Wallace (1982) reported 4–11 year spawning periodicity for white sturgeon, but made no mention of gender. Little is known regarding reproductive senescence in *A. transmontanus*. One perspective suggests that natural selection would not favor the persistence of this life history trait because longevity beyond reproductive age would serve no advantageous purpose to the population (E. Brannon, University of Idaho, personal communication). Mature adults are thought to spawn numerous times over a 30–40 year period, and possibly longer (S. Doroshov, University of California, Davis, personal communication). If an individual female initially reproduced at age 25 and successfully spawned in subsequent 5-year intervals until age 65, theoretically it could contribute gametes to subsequent generations up to nine times. Finally, communal spawning, along with the above reproductive mechanisms, likely contributes to increased gene flow and maintenance of genetic diversity in white sturgeon relative to that of paired, semelparous fishes (e.g. Salmonidae), especially in the absence of confirmed homing fidelity.

1.1.7 Movements in Fresh Water

White sturgeon movements within fresh water remain somewhat unclear (reviewed in Perrin *et al.* 1999). Semakula and Larkin (1968) reported migrations in the Columbia River, and

Pycha (1956) reported that large fish moved upstream during winter and spring and downstream during summer in the Sacramento-San Joaquin system. Other vague movements, such as from deeper water in winter to shallower water in summer, also have been described (Migdalski 1962; Anders 1991). In the Columbia River, white sturgeon were reported to exist in groups representing distinct geographic cohorts concentrated downstream from impoundments (North *et al.* 1993). Tagged white sturgeon generally moved more within than between reservoirs (Warren and Beckman 1993; ODFW 1996; DeVore *et al.* 1999). However, Galbreath (1985) did not detect clear migration patterns, and suggested that movement appeared to be random. Conversely, DeVore and Grimes (1993) reported that adults migrated upstream during the fall, downstream during spring, and congregated at the Columbia River estuary during summer, presumably in relation to food availability, with such movements exceeding 62 miles (100 km). Alternatively, in the Columbia River in British Columbia, most adult white sturgeon tagged with radio transmitters moved < 3.1 miles (5 km), and only 2% were recaptured > 6.2 miles (10 km) from the point of original tagging (RL&L 1994, 1996). These findings were suggested to have resulted from the close proximity of suitable habitat for feeding, overwintering, and spawning (RL&L 1994).

Review of empirical data on movement also suggested that Columbia River white sturgeon are not completely isolated within individual reservoirs, and where not damlocked (Kynard 1997), can migrate great distances (Warren and Beckman 1993; ODFW 1996; DeVore *et al.* 1999; Gallion and Parsley 200; Paragamian *et al.* 2001). From 1987–94, 9,323 white sturgeon were tagged in the four Columbia River reservoirs that are farthest downstream; of these fish, 1,162 (13%) were recaptured (ODFW 1996). During these years, 661 (57%) of these 1,162 fish were recaptured in the reservoir of original capture and tagging, 68 (6%) were recaptured in downstream reservoirs, or downstream from Bonneville Dam (thus given unimpounded access to the Columbia River estuary and Pacific Ocean), and 2 (0.3%) were recovered in an adjacent upstream reservoir (ODFW 1996).

Regarding movement and migration in unimpounded habitat, 471 white sturgeon were originally tagged in the unimpounded lower Columbia River (downstream from Bonneville Dam) and recaptured in 23 separate locations outside the Columbia River Basin from 1976–97 (DeVore *et al.* 1999). These fish were recaptured in the Fraser River (~310 miles [500 km] to the north in southwestern British Columbia), in the Sacramento River (~560 miles [900 km] to the south in central California), and in 21 additional intermediate locations along the Pacific Coast of Oregon and Washington (DeVore *et al.* 1999). Summarized empirical data from white sturgeon recaptured in the unimpounded lower Columbia River but tagged elsewhere were unavailable. Gallion and Parsley (2001) reported migrations among subadult and adult sturgeon up to 149 miles (240 km) in free-flowing areas of the mid-Columbia River.

Movement of adult white sturgeon in the Kootenai River system, Idaho, and British Columbia, was categorized into five movement patterns, based on ultrasonic telemetry data (Anders 1991). This research revealed seasonal movement and migration patterns between different habitats by over half of the tagged fish; the remaining 47% did not migrate, and their movements were generally < 19 miles (30 km), appearing random (Anders 1991). Although this categorization according to general movement patterns was presented for illustrative purposes, over a 14-month period, white sturgeon exhibited a wide array or a continuum of movement, from very short to considerable (>62 miles [100 km]). Hourly locations determined by ultrasonic telemetry for 24 consecutive hours of 14 adult white sturgeon in the Kootenai system revealed no distinct daily movement or activity patterns; however, most individuals moved slightly during the 24-hour investigations. More recently, Paragamian and Kruse (2001) reported consistent upstream migrations in the Kootenai system during spring and fall from downstream Kootenay Lake and the lower Kootenai River to reported prespawning staging areas in Idaho. These authors also reported that female white sturgeon demonstrated more consistent behavior, appeared more sensitive to changing environmental conditions, and were more useful in predicting the probability of migration to the spawning reach than males.

Before impoundment, white sturgeon were reported to range freely in large systems like the Columbia River (Bajkov 1951), undertaking extensive migrations among habitats to presumably take advantage of scattered and seasonally available resources (Beamesderfer *et al.* 1995). Dam construction and operations reduced access of these fish to different habitats, reduced seasonal hydrographic and thermographic variation, and reduced habitat diversity and heterogeneity (Beamesderfer *et al.* 1995).

1.1.8 Ocean Migration

Although ocean migrations of white sturgeon measured in thousands of kilometers are supported by mark-recapture data (Devore *et al.* 1999) and by widespread distribution of genetic signatures and empirical gene flow measures (Anders *et al.* 2002 [Chapter 2]; Anders and Powell 2002 [Chapter 3]), surprisingly little is known about specific ocean migratory behavior of white sturgeon. Of thousands of white sturgeon originally tagged in the unimpounded lower Columbia River (downstream from Bonneville Dam), just 471 were recaptured in 23 separate locations outside the Columbia River Basin from 1976–97 (DeVore *et al.* 1999). Likewise, statistical comparisons of mitochondria DNA (mtDNA) haplotype frequency distributions revealed a mean distance of over 620 miles (1,000 km) for significant differences in white sturgeon haplotype frequency distributions (Anders and Powell 2002), although not all comparisons involved intermediate marine environments.

Based on mark and recapture data from thousands of white sturgeon tagged in the lower Columbia River (downstream from Bonneville Dam) and recapture of 471 fish during a 21-year period, marine movements and migration of white sturgeon are not uncommon. These fish were recaptured up and down the Oregon and Washington coasts (Table 1-2). These recoveries occurred when few research program dollars were available to sample areas outside the Columbia Basin. No programs have been in place to specifically document migrations and movements of white sturgeon in marine environments. However, with recently increasing interest in green sturgeon—primarily a marine, estuarine, and lower river species—and with increased interest in developing a better understanding of white sturgeon movements and migration in the ocean, increasing effort may be directed at sampling near-shore marine habitats to better understand the species’ use of the marine environment (T. Rien ODFW, personal communication).

Table 1-2. Columbia River white sturgeon out-of-system tag recoveries by recovery area, 1976–97 tag groups.

Recovery	Tag Group																					
Area	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	Totals
Sacramento	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1
Oregon coast	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1	0	0	0	0	1	1	0	4
Coos Bay	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	4	0	0	5
Umpqua	0	0	0	0	0	0	0	0	0	0	1	0	0	2	0	0	0	5	0	8	0	1	17
Siuslaw	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1	0	2
Yaquina	1	0	1	0	0	0	0	0	0	0	0	0	0	2	0	0	1	2	0	2	0	0	9
Siletz	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1
Tillamook	0	0	0	0	0	0	0	0	0	1	3	3	1	5	2	0	0	9	1	6	1	12	44
Nehalem	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	1	1	0	0	0	0	4
Washington coast	0	1	0	1	0	0	1	1	0	3	0	0	0	1	0	0	0	1	0	0	0	0	9
Willapa Bay	2	1	0	3	3	4	4	3	0	4	8	14	8	7	4	10	4	14	1	15	12	6	127
Grays Harbor	1	1	1	1	4	9	4	3	3	4	14	24	11	18	8	17	0	24	5	30	6	3	191
Quinault	0	0	0	0	0	0	0	0	1	2	3	0	3	3	1	0	0	0	0	2	0	0	15
Hoh	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	1	0	0	0	0	3
Neah Bay	0	0	0	0	0	0	0	0	1	1	11	4	2	0	0	2	0	0	0	0	0	1	22
SJDF	0	0	0	0	0	0	0	0	0	0	1	0	0	1	0	0	1	2	0	1	0	0	6
Hood Canal	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	1	0	0	3
Puget Sound	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Nisqually	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1
Duwamish	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1
Snohomish	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0	0	2
Stillaguamish	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	2
Fraser	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1
Totals	4	3	2	5	8	14	9	7	5	15	41	45	26	41	17	31	8	64	7	73	22	24	471

1.2 Population Identification & Distribution

Currently, reference to white sturgeon populations is simply a convenient way to describe groups of fish in particular geographic areas, due largely to artificially-imposed migration barriers. Unlike semelparous, paired-spawning salmonids that exhibit strong homing fidelity and distinct population structure (Brannon *et al.* 2002), white sturgeon population structure appears to exist at very large geographic scales, measured in hundreds to thousands of kilometers (Anders and Powell 2002). These authors coined the term "expansive gene flow model" to describe observed population structure of white sturgeon in western North America. This model was strongly supported by empirical white sturgeon genetic, dispersal, life history, and reproductive data. Anders *et al.* (2002) and Anders and Powell (2002) rejected a null hypothesis that all white sturgeon in the Columbia River Basin represented one gene pool based on analysis of geographic heterogeneity of two mtDNA marker systems. Past white sturgeon stock delineation research using protein allozymes (Setter and Brannon 1992; Bartley *et al.* 1985) and mtDNA sequencing (Brown *et al.* 1992) from a relatively limited number of samples and sampling locations in the Columbia, Fraser, and Sacramento river basins also identified statistically significant differences in white sturgeon allele and haplotype frequencies across geography.

Extensive development of hydropower dams throughout the Columbia River Basin during the past century has severely fragmented large free-flowing river habitats (National Research Council 1996) occupied by white sturgeon. These changes likely have resulted in major alterations to historical gene flow patterns in the Columbia Basin and among large Pacific Coast river systems in North America (Anders and Powell 2002). Thus, future revelation of historical population structure and delineation will become increasingly difficult as historical gene flow signal is lost, and current migrations and gene flow patterns are limited by hydropower development, genetic drift, and recruitment failure in resulting small populations.

1.2.1 Genetic Differences

Small but significant differences in genetic frequencies and diversity are apparent among white sturgeon populations in the Sacramento, Columbia, and Fraser systems based on electrophoretic and mtDNA analysis (Bartley *et al.* 1985; Brown *et al.* 1992; Anders and Powell 2002). White sturgeon populations along the Pacific coast of North America are closely related. Anders and Powell (2002) observed 26 unique mtDNA sequences (haplotypes) in samples from 13 locations in the Columbia, Snake, Kootenai, Fraser, Nechako, and Sacramento Rivers. The two most common haplotypes were represented by 64% of the 260 fish sequenced and were observed at 100% and 85% of the sample sites (Anders and Powell 2002). Similar overlap among populations was reported by Bartley *et al.* (1985) based on electrophoretic analysis of allele frequencies, Brown *et al.* (1992) based on mtDNA, and McKay *et al.* (2002) based on mtDNA. Expansive haplotype distribution indicates little genetic divergence and significant gene flow throughout a major portion of the species' range (Anders and Powell 2002). However, there is little evidence to support high levels of contemporary gene flow, especially in post-impoundment systems (Anders, personal communication). This conclusion is consistent with observed recaptures of small numbers of tagged Columbia River sturgeon in the Sacramento and Fraser Rivers (DeVore *et al.* 1999).

White sturgeon genetic studies have consistently documented decreasing diversity with distance upstream (Bartley *et al.* 1985; Brannon *et al.* 1987; Brown *et al.* 1992; Anders and Powell 2002). Total number of haplotypes were negatively correlated with inland distance from the Pacific Ocean in all river systems studied (Anders and Powell 2002). Genetic differences were most pronounced in the Kootenai River white sturgeon population where heterozygosity was the lowest observed in the Kootenai River (Bartley *et al.* 1985; Brannon *et al.* 1987; Setter and Brannon 1990; Anders and Powell 2002). Kootenai River white sturgeon are believed to be a post-glacially isolated population of ancestral Columbia River stock (Duke *et al.* 1999; USFWS 1999). This population was listed in 1994 as endangered under the ESA (USFWS 1994).

Sturgeon populations impounded in the lower Columbia River mainstem between Bonneville Dam and the Snake River were created by dam construction and do not represent unique genetic units.

1.2.2 Life History Differences

General life history characteristics of Acipenserids were recently summarized by Bemis and Kynard (1997) and Kynard (1997). As discussed above, sturgeons exhibit several life history forms (diadromy, anadromy, amphidromy, and potadromy). Based on observed life history expressions, white sturgeon where not damlocked appear to be best described as facultatively anadromous when dams prohibit expression of historically anadromous or amphidromous life history strategies (Kynard 1997). Regardless of life history strategies expressed, all sturgeons spawn exclusively in large freshwater river systems often following upstream migrations of considerable distance (Bemis and Kynard 1997).

Unlike some Pacific salmonids, expression of specific life history forms do not appear correlated with specific white sturgeon populations. A possible exception may have been a longitudinal gradient (upstream-downstream) in expression of anadromy before such movements were prohibited or severely reduced by hydropower development (NRC 1996). While differences in expression of life history traits are well documented for Pacific salmonids, if such differences exist in within sturgeon they have been severely dampened by extensive hydropower development in the Columbia basin (NRC 1996), and they likely exist at much broader geographic scales, relative to salmonids. The extent of naturally expressed anadromy by white sturgeon in the Columbia Basin is unknown. However, based on gene flow, genetic distance, and geographic distribution of genetic signal (Anders and Powell 2002), the prevalence of expressed anadromy by white sturgeon was likely negatively correlated with inland (upstream) distance.

1.3 Status & Abundance Trends

1.3.1 Below Bonneville

1.3.1.1 Abundance

The current white sturgeon population in the lower Columbia River is considered to be healthy with more than 1 million fish exceeding 2 feet.

The lower Columbia River downstream from Bonneville Dam white sturgeon population is the most productive in the species' range (DeVore *et al.* 1995). This high productivity supports significant sport and commercial fisheries with annual harvest ranging from 40,000 to 55,000 during 1992–2000. The sturgeon fishery ranks as the largest sport fishery in the Columbia Basin in terms of effort with a 10–year annual average of over 175,000 angler trips. Factors most responsible for the favorable production potential of the population are access to marine areas, abundant food resources, and consistently favorable hydrologic conditions during the spawning timeframe, which enhances recruitment (Parsley and Beckman 1994; DeVore *et al.* 1995; Counihan *et al.* in press). This high productivity can be sustained in the long term only with careful, scientifically based management.

WDFW and ODFW cooperatively monitor the lower Columbia River white sturgeon population status through a study supported by federal sport fishing restoration and State of Oregon monies. The objectives of the study are to:

- evaluate natural production,
- estimate population abundance and appropriate fishery exploitation rates, and
- monitor white sturgeon fisheries.

ODFW traditionally captures sturgeon for tagging during the months of May, June, July, and August. They collect recovery data through November of the year following tagging. This allows for mixing of tagged fish within the population (a key assumption for mark and recapture models) and it covers four key sport and commercial fishing periods when most tagged fish are recovered: the initial summer sport fishery, fall and winter commercial fisheries, and the following year's summer sport fishery. Abundance estimates are made about 1-1/2 years after the fish are initially tagged.

The abundance trend has shown a significant increase in the 3-6 foot population since 1989 after size limit and harvest regulatory actions were implemented by Oregon and Washington (Figure 1-5). Oregon and Washington biologists believe the lower Columbia white sturgeon broodstock population is healthy as indicated by significant and consistent production of juvenile fish. Abundance estimates of 36-40 inch (age 9-10) white sturgeon have ranged from 66,400 in 1986 to 256,000 in 1993. These estimates display significant annual recruitment to the legal size age class. The key to maintaining this current high level of productivity is assuring adequate escapement of legal sized fish (42-60 inches) through the fisheries to reach broodstock age and protecting the fish which have reached spawning age (25 years).

Table 1-3. River abundance estimates.

River	River Reach	Abundance Estimates	Year(s)	Reference	River reach length (km)	Fish/km
Columbia	Lower Columbia River	174,900-445,000	1987–97	DeVore *et al.* 1999b	235	744-1,893
	Bonneville Pool	17,900-48,700	1989–99	Kern *et al.* 2001	73	245-667
	The Dalles Pool	4,500-46,800	1987–97	Burner *et al.* 2000	39	115-1,200
	John Day Pool	2,200-24,100	1990–96	Burner *et al.* 2000	123	18-196
	McNary Pool[a]	4,600	1995	Burner *et al.* 2000	185	25
	Upper Columbia, British Columbia	1,427	2001	RL&L Env. Serv. Ltd.	NA	NA
Snake	Ice Harbor Pool	4,560	1996	DeVore *et al.* 1998	51	89
	Lower Monumental	3,891	1997	DeVore *et al.* 1999a	43	91
	Little Goose Pool	4,860	1997	DeVore *et al.* 1999a	60	81
	Lower Granite Reservoir	1,372[b]	1990–91	LePla 1994	66	21
		1,524[c]	1990–91	LePla 1994	66	23
		1,804[b]	1992	Bennett *et al.* 1993	66	27
	Clearwater River-Hells Canyon Dam	3,955[b]	1982–84	Lukens 1985	174	23
	Salmon River-Hells Canyon Dam	1,312[b]	1997–2000	LePla *et al.* 2001	80	16
		1,600[c]	1997–2000	LePla *et al.* 2001	80	20
	Lower Granite-Hells Canyon dams	8,000-12,000	1972–75	Coon *et al.* 1977	224	36-54
	Lower Granite-Hells Canyon reach	3,625	1997–2000	LePla *et al.* 2001	209	18
	Lower Granite Dam-Salmon River	2,544[b]	1997–99	Tuell and Everett 2001	129	20
		1,823[c]	1997–1999	Tuell and Everett 2001	129	14
		3,625	1997–2000	LePla *et al.* 2001	224	16
	Hells Canyon Pool	[d]	1998	LePla *et al.* 2001	42	d
	Oxbow Reservoir	[d]	1998	LePla *et al.* 2001	21	d
	Brownlee-Swan Falls	155	1996–97	LePla *et al.* 2001	277	<1
	Swan Falls Reach	726	1996–97	LePla and Chandler 1997	58	13
	C.J. Strike Reach	2,622	1994–1996	LePla and Chandler 1995	106	25
Kootenai	Kootenai	1,469	1996	Duke *et al.* 1999		

a: Includes Columbia and Snake rivers

b: Schnabel estimator

c: Jolly-Seber estimator

d: Too few were captured to generate estimate

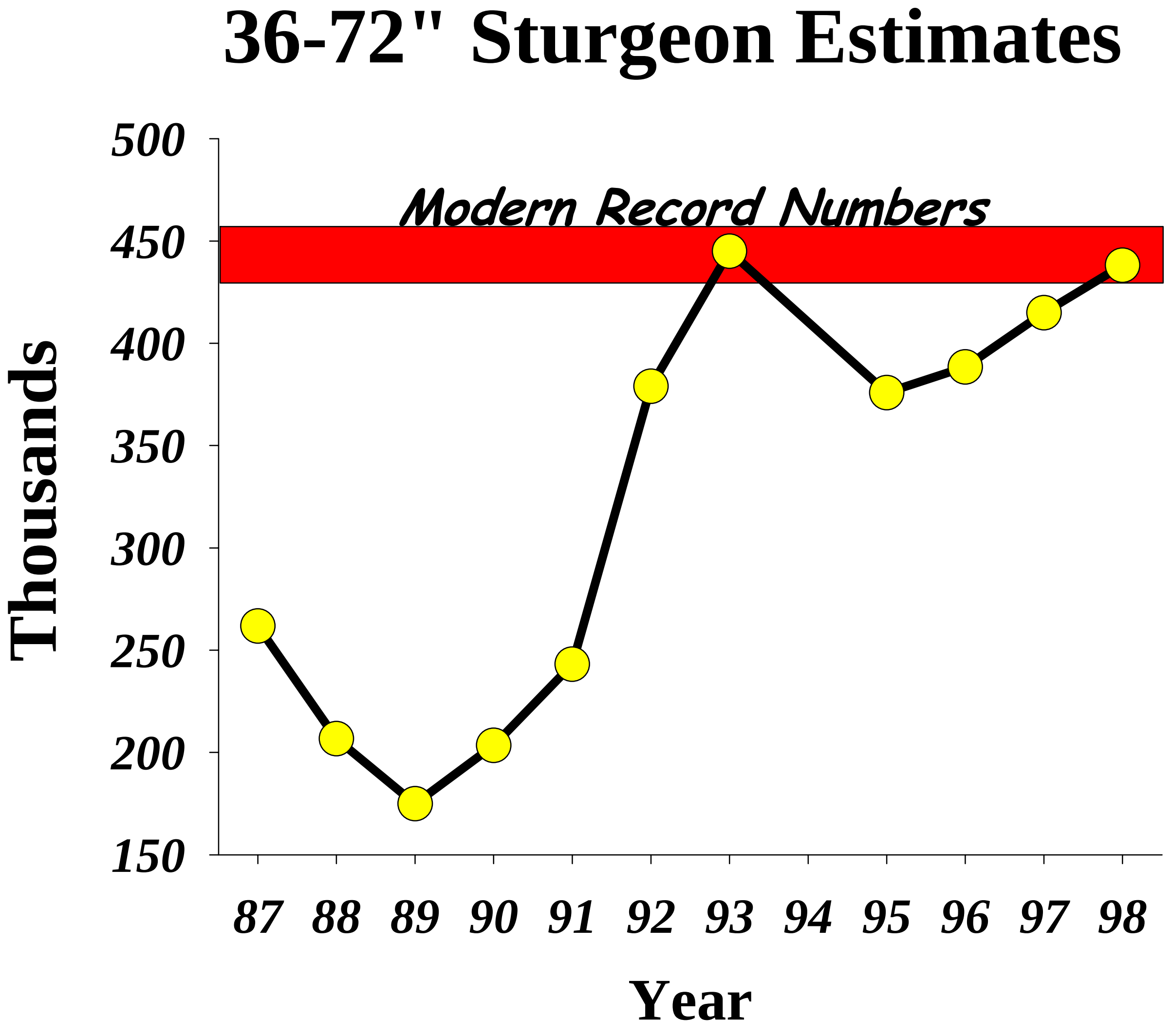

Figure 1-5. Number of 36-72 inch white sturgeon.

Biologists expected an increase in legal-sized white sturgeon abundance following harvest management actions taken in 1997. However, the most recent assessment of data indicates the legal-sized population at best is remaining static at 1996 levels or is more likely declining. The accuracy of the population estimates have been compromised by inconsistencies in sampling approach and an unforeseen mass emigration of sturgeon in 1996 and a reduction in individual growth rates.

Recoveries of tagged white sturgeon from areas outside the Columbia River peaked in 1996. This was believed due to a mass emigration of white sturgeon from the Columbia River that was ostensibly associated with El Niño ocean conditions that devastated eulachon (Columbia River smelt) survival (DeVore *et al.* 1999). Declines in recent years in white sturgeon abundance in the Columbia River were partially attributed to this emigration. Recoveries from outside the Columbia River have since diminished each year, indicating that most of the fish that emigrated had returned by 1997–98. However, population estimates of 42-60 inch fish based on tag recoveries continue to demonstrate a declining trend through 2001 (Table 1-5 in harvest section). Conversely, the sport fishery catch per effort remained steady at 0.19 to 0.21 legal white sturgeon per trip during 1997–2000.

1.3.1.2 Productivity

The white sturgeon population in the Columbia River downstream of Bonneville is among the most productive sturgeon populations in North America. Abundance and biomass have been estimated at 36.1 fish/ac (14.6 fish/ha) and 88 lbs/ac (87.5 kg/ha), respectively (DeVore *et al.* 1995). Current sturgeon biomass in the unimpounded lower mainstem appears similar to levels during pristine conditions before significant exploitation in the late 1800s (Beamesderfer and Farr 1997). Productivity is sufficient to sustain large commercial and sport fisheries. Habitat conditions are suitable for consistent annual recruitment. Large volumes of suitable rearing habitat exist. Large food resources are provided by anadromous fish species including smelt, shad, salmon, and lamprey. Sturgeon range freely throughout the lower river mainstem, estuary, and ocean to take advantage of dynamic seasonal patterns of food availability. Individual growth, condition, and maturation are among the greatest observed for white sturgeon anywhere they occur.

1.3.1.3 Supplementation

No white sturgeon supplementation or conservation hatchery programs exist in the lower Columbia River downstream from Bonneville Dam. Pelfry's, a small-scale commercial sturgeon culture facility downstream from Bonneville Dam, maintains an agreement with ODFW to spawn a small number of wild lower Columbia River broodstock (2-3 per year) for economic/ commercial production purposes. In return, an agreement between Pelfry's and ODFW ensures that a percentage of produced progeny (e.g. 1,000 progeny/female spawned) is returned to the Columbia to compensate for potential lost production associated with removing that fish from the wild population.

1.3.1.4 Harvest

Sturgeon abundance in the lower Columbia River collapsed at the end of the 19th century due to overharvest (Rieman and Beamesderfer 1990). Harvest in the lower Columbia River (US) was so severe that over 30 years elapsed before commercial harvest again became economically feasible (Rieman and Beamesderfer 1990) (Figure 1-6.). The population began to rebound after 1950 when maximum size limits were adopted to protect broodstock-size sturgeon. Since 1950, the population has increased significantly. The lower Columbia River white sturgeon population is currently healthy, with an abundance of more than 1 million fish 2 feet or longer, dominated (>95%) by immature fish (DeVore *et al.* 1999).

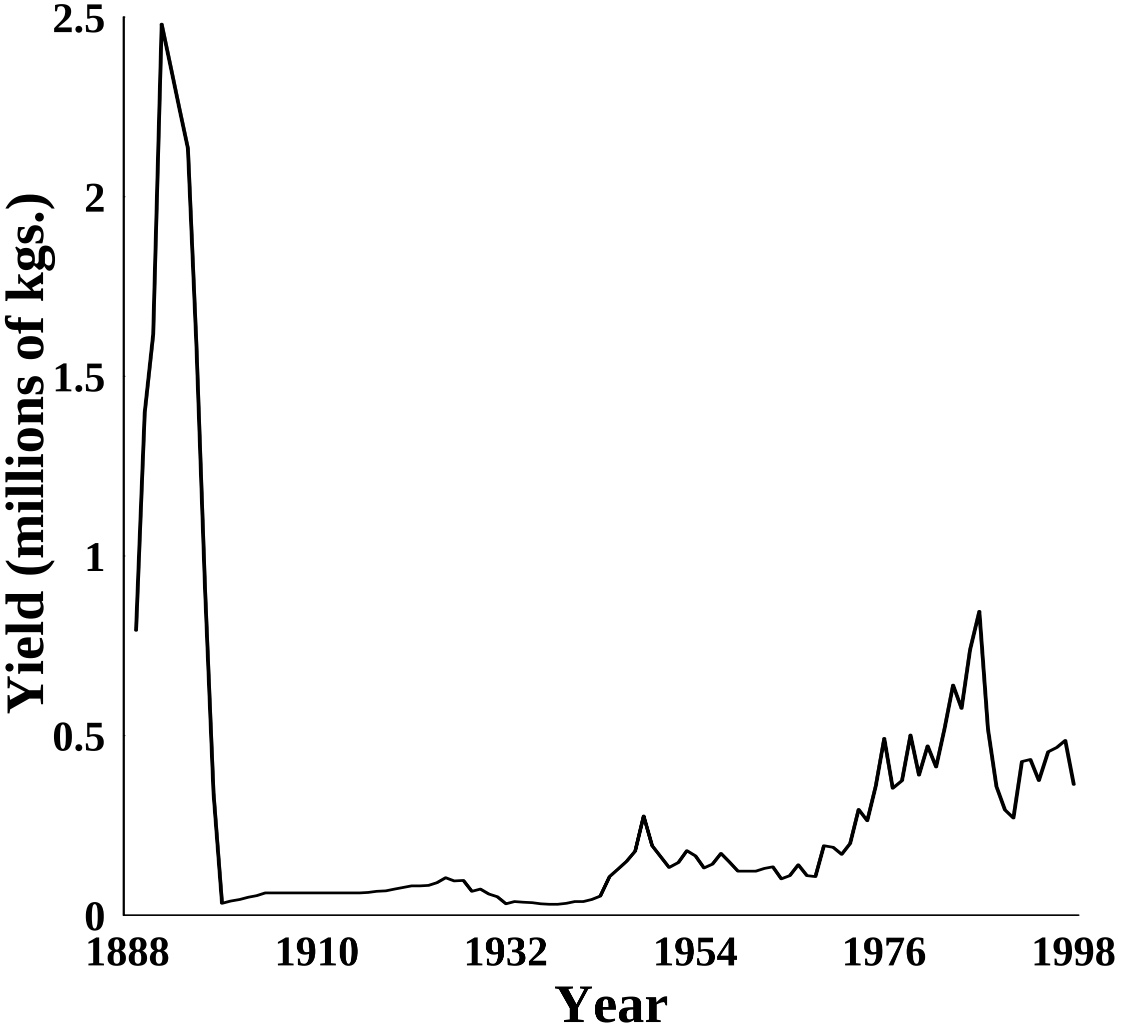

Figure 1-6. Historical yield of Columbia River white sturgeon.

During the 1980s, coincident with reductions in salmon harvest opportunities, sport effort for white sturgeon increased, along with harvest, compared to harvest levels during the 1970s. This increased harvest drove the legal-sized population down to levels of concern. Oregon and Washington responded with several regulation changes (e.g. size and bag limit changes, elimination of commercial target fisheries to control harvest growth). After the 1980s, harvest management increasingly included the setting of empirical abundance-based harvest levels.

Table 1-4 provides a 100-year summary (1899–1999) of white sturgeon management regulations in the lower Columbia River (from DeVore *et al.* 1999).

Table 1-4. Summary of historical changes in Columbia River sport and commercial fishery regulations for sturgeon (DeVore et al. 1999).

	Sport			Commercial	
Year	Daily Bag Limit	Size Limits (in)	Other Rules	Size Limits (in)	Gear & Other Restrictions
1899	None	None	None	48 min	Chinese gang lines prohibited (snagging setlines)
1899–1908	None	None	None	48 min	Sturgeon sales closed
1909	None	None	None	48 min	Sturgeon sales allowed during salmon seasons
1938	None	None	None	48 min	Beacon Rock-Bonneville Dam sanctuary established
1940	Only 3<48 in	None	None		
1942	3<4' & 2≥48 in	None	None		
1950	3<4' & 2≥48 in	30 min/72 max	None	48 min/72 max	
1951	3 fish	30 min/72 max	None		
1957	3 fish	30 min/72 max	Cannot remove head or tail in field		
1958	3 fish	36 min/72 max			
1968				48 min/72 max	Zone 6 became exclusive treaty Indian fishery
1975–82				48 min/72 max	Setline seasons allowed outside of salmon seasons
1983–85				48 min/72 max	Setline seasons phased out
1983–88				48 min/72 max	Target sturgeon gill net seasons (in lieu of setlines)
1986	2 fish	36 min/72 max	OR—sturgeon tag with 30 annual limit		
1989	2 fish	36 min/72 max 40 min/72 max (effective April 1)	WA—no gaffing, sturgeon tag with 15 annual limit	48 min/72 max	Target sturgeon gillnet seasons eliminated
1990	2 fish	40 min/72 max	Single-point barbless hooks OR—annual limit 15 and no gaffing	48 min/72 max	9 1/4 in max. mesh restriction in late fall salmon seasons
1991	1<48" &1≥48"	40 min/72 max		48 min/72 max	WA—adopted 2 lbs lead/fathom of leadline rule
1992	1<48" &1≥48"	40 min/72 max	WA—60 in max (effective Apr. 16, 1992–Apr 15, 1993), Beacon Rock–Bonn. Dam sanctuary (Apr 16–June 15, 1992)	48 min/72 max	WA—adopted 60 in max. length for fall seasons
1993				48 min/66 max	9 1/4 in max. mesh adopted as permanent rule. Sturgeon sales closed during last 2 weeks of fall salmon season (6,000 catch expectation for 1993 reached)
1994	1<48" &1≥48"	42 min/66 max	Annual limit 10	48 min/66 max	Catch ceiling of 6,000 for year. Sturgeon sales closed after first day of fall salmon season

	Sport			Commercial	
Year	**Daily Bag Limit**	**Size Limits (in)**	**Other Rules**	**Size Limits (in)**	**Gear & Other Restrictions**
1995	1<48" &1≥48"	42 min/66 max	Closed to retention Sept. 1– Dec. 31	48 min/66 max	Annual catch ceiling of 8,000 during salmon seasons of which not more that 6,800 (85%) may be taken in fall fisheries
1996	1 fish as of April 1	42 min/66 max	Beacon Rock-Bonneville Dam sanctuary (closed to boat angling May and June)		
1997–98	1 fish	42 min/60 max	53,840 harvest guideline	48 min/60 max (whites) 48 min/66 max (greens)	Annual harvest guideline of 13,460 whites
1999	1 fish	42 min/60 max	50,000 harvest guideline	48 min/60 max (whites) 48 min/66 max (greens)	Annual harvest guideline of 10,000 whites

Joint Oregon and Washington tagging and recovery programs to estimate annual abundance began in 1989 and these empirical abundance estimates have been used to base harvest management decisions during the past 21 years. Since 1989, fisheries have been managed for an optimum sustained yield (OSY), which requires harvest management plans to allow enough escapement through the legal size slot for optimum levels of sturgeon to recruit to the broodstock population on a sustainable basis. Management measures employed to achieve OSY built the sturgeon legal-sized population back to healthy levels by 1995.

In 1996, Oregon and Washington adopted a 1997–99 Management Accord, which specified white sturgeon harvest management objectives for both states, including total allowable harvest and allocation between sport and commercial fisheries (Table 1-5). The following management objectives were adopted for lower Columbia River white sturgeon:

- Provide adequate recruitment to the broodstock population,
- Manage fisheries for optimal sustainable yield (OSY),
- Maintain an OSY harvest rate determined for the legal-sized population in sport and commercial fisheries,
- Maintain concurrent Washington and Oregon regulations in the Columbia River,
- Provide for year-round sport fishing opportunity,
- Maintain sport and commercial shares in the fishery, and
- Consider emergency regulatory action if harvest is projected to compromise management objectives.

These management objectives were designed to build Columbia River sturgeon populations to carrying capacity for the habitats in which they reside. They also reflect a desire to manage for healthy, stable fisheries that provide a long-term, sustainable yield. Optimal sustainable yield, as defined for lower Columbia River sturgeon management, is a level of harvest that allows enough survival of juvenile fish through the fisheries to insure adequate recruitment into the protected broodstock population (DeVore *et al.* 1995).

The 1997–99 sport and commercial harvest sharing agreement (80% sport, 20% commercial) was renewed by Oregon and Washington for 2000–02. The state commissions also adopted a total allowable annual harvest reduced from 67,300 (1997–99) to 50,000 (2000–02). This harvest reduction was in response to empirical population measures indicating that the growth in the population necessary to achieve OSY was not being met at the 1997–99 harvest level.

Table 1-5. Annual sport and commercial catches of white sturgeon and comparisons to catch guidelines, 1993–2002.

	Sport		Commercial	
Year	Catch	Guidelines	Catch	Guidelines
1993	37,900		8,100	6,000
1994	33,500		6,400	6,000
1995	45,100		6,200	8,000
1996	42,800		8,400	8,000
1997	38,200	53,840	12,800	13,460
1998	41,600	53,840	13,900	13,460
1999	39,800	40,000	9,500	10,000
2000	40,500	40,000	10,870	10,000
2001	40,200	39,500	9,430	9,100
2002[1]	37,500	38,500	9,760	9,800

[1] Preliminary. Sport catch includes projection for November 23 through December 31, 2002

In October 2002, Oregon and Washington concluded that although lower Columbia white sturgeon had rebuilt to healthy levels as a result of harvest management actions taken in the past decade, the legal size white sturgeon (42-60 in) had not increased as intended over the past 6 years, as illustrated in Table 1-6. Positive growth in the legal-sized population is important to provide adequate recruitment into the broodstock population (sturgeon 6 ft and larger).

Table 1-6. Estimated abundance of harvestable white sturgeon in the lower Columbia River.

	Total Length Interval		
Year	42-48 inch	48-60 inch	42-60 inch
1989	32,500	16,800	49,300
1990	26,100	12,000	38,100
1991	32,900	11,700	44,600
1992	59,900	8,700	68,600
1993	85,000	14,200	99,200
1994	N/A	N/A	N/A
1995	143,200	59,000	202,200
1996	131,700	33,500	165,200
1997	123,700	33,400	157,100
1998	161,600	24,700	186,300
1999	116,800	17,600	134,400
2000	119,200	17,000	136,200
2001	100,200	22,400	122,600

Consequently, Oregon and Washington reduced annual harvest from 50,000 to 40,000 per year during 2003–05. Commercial and sport shares remain at 80% sport and 20% commercial but the earlier objective of a year-round sport fishery was no longer possible with sport harvest reduced to 32,000 fish annually (Figure 1-7).

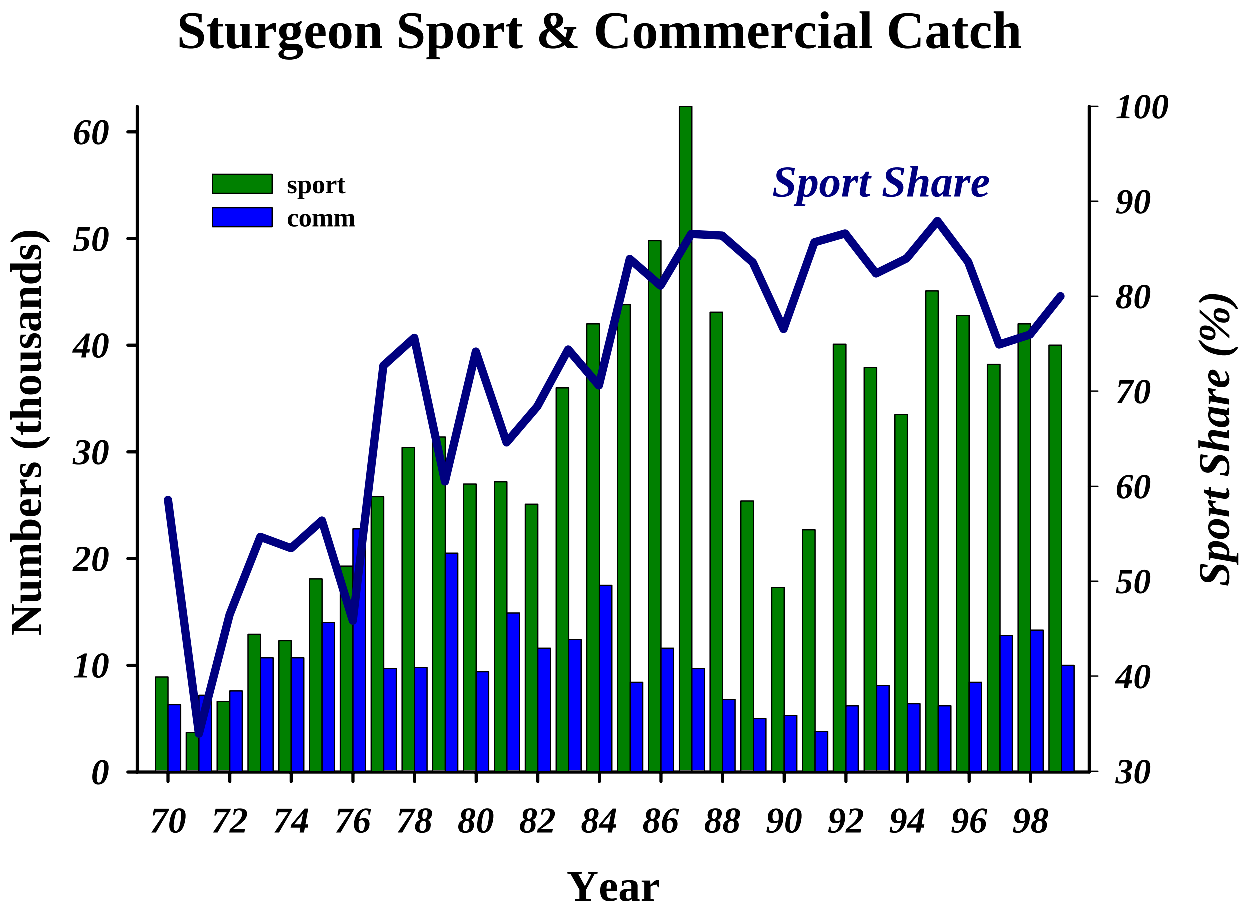

Figure 1-7. Sturgeon sport and commercial catches.

The 3-year Oregon/Washington management agreements provide stability within the annual (in-season) management processes and coincide with the ability to assess population trends needed for recommended management adjustments.

In addition to adjusting total harvest guidelines, Oregon and Washington have established a boat fishery sanctuary in the area between Beacon Rock and Bonneville Dam during peak spawning months (May–July) to reduce handle of broodstock fish by the sport fishery. Washington and Oregon biologists are studying the effect of the sport fishery handle on spawning broodstock and may consider further precautionary regulations in the future.

1.3.2 Above Bonneville

1.3.2.1 Abundance

Sturgeon abundance in the three reservoirs immediately upstream from Bonneville Dam is evaluated every 3–5 years using mark-recapture methodology. The Bonneville Reservoir population of 3-6 foot sturgeon was estimated at 48,600 in Bonneville (1999). Population size in 2002 was projected to be 59,300, based on observed growth and mortality rates. Revised population estimates are being developed based on analysis of 2002 mark-recapture data.

1.3.2.2 Productivity

Productivity of the impounded sturgeon populations upstream from Bonneville Dam is much less than that of the population in the free-flowing river between Bonneville Dam and the ocean. Reduced productivity likely results from reduced access to diverse anadromous, estuarine, and ocean food resources as well as sporadic conditions for recruitment. Sturgeon population productivity between Bonneville and McNary Dams (measured in harvestable lb/ac/year) has been estimated at only 15% that of the unimpounded population downstream from Bonneville Dam (Beamesderfer *et al.* 1995). In general, impounded populations grow slower, mature later, and maintain lower condition factors than the unimpounded population. Growth, condition, and maturation of Bonneville Reservoir sturgeon is among the poorest observed in the lower Columbia River basin but it is unclear whether this results from habitat conditions, competition, or contaminants.

Spawning habitat availability is a key determinant in the productivity of impounded sturgeon populations. Impounded reservoirs and river segments no longer provide suitable spawning conditions under all flow conditions. Spawning habitat is generally limited to the high-energy zones of dam tailraces but tailrace morphometry results in variable spawning habitat suitability and varying sensitivity to flow. The tailrace below The Dalles Dam at the head of Bonneville Reservoir appears to provide suitable spawning habitat under most flow conditions, hence, consistent sturgeon recruitment occurs to the Bonneville Reservoir population. In contrast, recruitment below John Day and McNary dams is sporadic, because suitable habitat provided only in years of high spring runoff.

1.3.2.3 Supplementation

No white sturgeon supplementation currently occurs in Bonneville Pool. An ongoing trap and transplant program from downstream of Bonneville Dam to The Dalles and John Day reservoirs is intended to compensate for migration blockage and sporadic recruitment. However, recruitment in Bonneville Reservoir is apparently consistent to seed the available rearing habitat.

1.3.2.4 Harvest

The productivity of impounded sturgeon populations between Bonneville and McNary Dams is sufficient to provide for limited harvest in Treaty Indian commercial, Treaty Indian subsistence, and non-Indian sport fisheries. The Bonneville Reservoir sturgeon population accounts for a substantial share of the harvest. Since 1991, annual harvests in combined fisheries generally have ranged from 2,000 to 3,000 sturgeon per year.

Sturgeon catch guidelines and sport/treaty commercial allocations have been reviewed annually since 1987 by the Sturgeon Management Task Force (SMTF), made up of representatives from state fish management agencies and the Columbia River treaty Indian tribes.

Guidelines are based on desired harvest rates and current stock assessments. In March 1997, the SMTF agreed to pool-specific management with catch guidelines based on OSY designed to allow for survival of adequate numbers of juvenile sturgeon through existing fisheries to increase harvestable and broodstock numbers. Commercial and sport retention is closed for the year when fishery-specific guidelines are reached.

Allocation is approximately 50:50 between sport and tribal fisheries, although reservoir-specific guidelines are shaped to meet fishery demands. For instance, the sport fishery is allowed a greater share of the Bonneville Reservoir catch, while the treaty Indian fishery is allowed a greater share of the catch in The Dalles and John Day reservoirs. Treaty Indian fishers may continue to take sturgeon for subsistence purposes after commercial seasons have been completed. Subsistence catch is estimated through a monitoring program conducted by the Yakama Indian Nation and annually averages less than 300 sturgeon. Subsistence catch is not included in the aforementioned catch guidelines. Sport anglers may continue to fish for sturgeon and release them unharmed when catch guidelines are reached and retention is prohibited.

Current sturgeon size limits are 48-60 inches for all treaty Indian fisheries, 48-60 inches for sport fisheries in The Dalles and John Day reservoirs, and 42-60 inches in the Bonneville Reservoir sport fishery. (Catches are shown in Table 1-7.) The Bonneville Reservoir size regulation matches that in the lower Columbia and is consistent with lower growth rates in Bonneville than in the upstream reservoirs.

Table 1-7. Sturgeon catches in Zone 6 reservoirs above Bonneville Dam, 1991–2001.

	Non-Indian Sport				Treaty Indian Commercial				
Year	Bonneville	The Dalles	John Day	Total	Bonneville	The Dalles	John Day	Total	Subsistence
1991	2,270	200	150	2,620	1,000	460	40	1,500	NA
1992	1,720	140	150	2,010	1,150	430	20	1,600	210
1993	2,310	160	140	2,610	1,420	580	10	2,010	260
1994	2,220	155	235	2,610	1,175	310	115	1,600	650
1995	1,370	50	90	1,510	1,420	310	310	2,040	1,150
1996	1,360	90	80	1,530	1,000	230	360	1,590	480
1997	1,470	180	480	2,130	1,852	498	1,260	3,610	236
1998	1,625	857	599	3,081	1,462	1,108	1,100	3,670	240
1999	1,236	694	422	2,352	1,280	1,051	760	3,091	244
2000	1,262	809	437	2,508	1,145	1,456	846	3,447	324
2001	1,422	677	300	2,399	1,019	1,258	684	2,961	476

1.4 Factors Affecting Population Status

1.4.1 Harvest

1.4.1.1 Below Bonneville

Lower Columbia River sturgeon populations collapsed at the beginning of the 20th century due to excessive harvest exploitation and a lack of regulations protecting broodstock fish (sturgeon 6 feet and greater). The sturgeon population rebounded after a maximum size limit of 6 feet was implemented in 1950. The population was stable during the 1970s, when harvest of 3-6 foot fish averaged about 30,000. A significant increase in harvest in the 1980s reduced the 3-6 foot population and consequently reduced recruitment to the broodstock population for future years. Management response in the 1990s focused on increasing future recruitment to the broodstock population by maintaining harvest levels that would provide annual growth in the legal harvest- sized population. To reduce the number of years in which sturgeon are subject to harvest, Washington and Oregon lowered the minimum size limit for white sturgeon to 42 inches, and the maximum to 60 inches.

History has shown that harvest can risk the health of the Columbia River sturgeon population unless the harvest is managed to protect broodstock and to pass enough younger sturgeon through the fishery to provide replacement broodstock for the future. Recent management policies adopted by Washington and Oregon assure long-term sturgeon health, but depend on adequate monitoring of the population status and the fisheries.

Interest in lower Columbia sturgeon sport fishing has increased dramatically in the past 25 years, rising from 60,000 angler trips in 1975 to over 200,000 in 1997, 1998, and 2000 (see Figure 1-8). Lower Columbia commercial sturgeon harvest has been stable compared to sport harvest over the same time period, primarily due to reductions in seasons due to salmon declines in the 1970s and 1980s and sturgeon catch allocations in recent years. Interest in commercial sturgeon fishing is also increased due to reduced opportunity for salmon and a stable market for sturgeon landings compared to salmon landings.

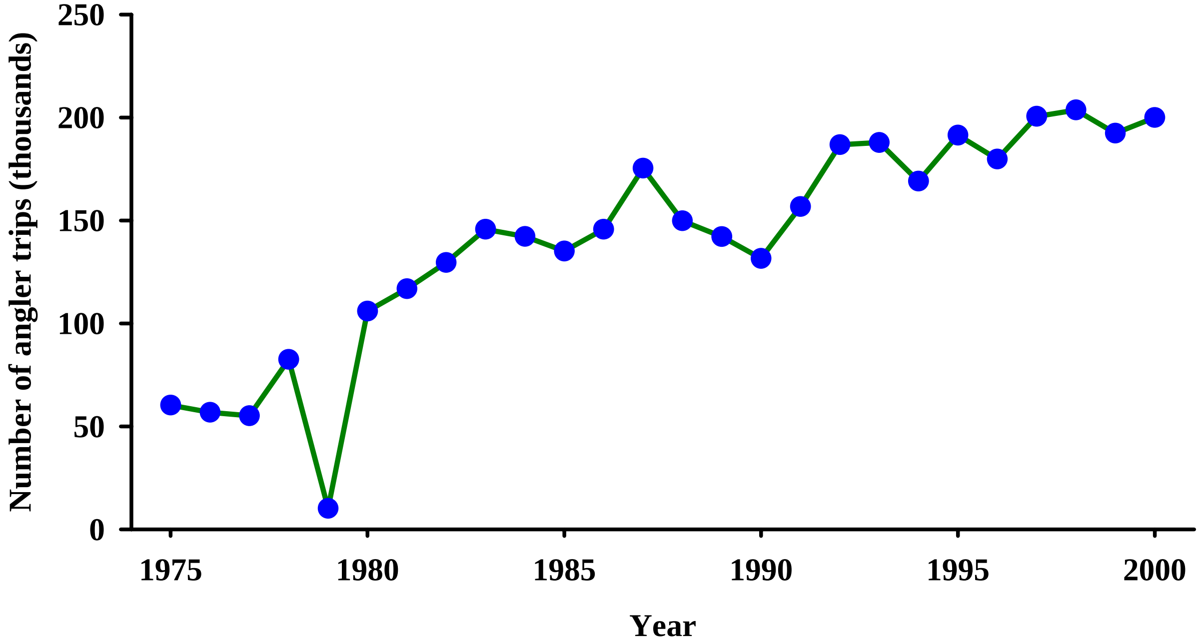

Figure 1-8. Sturgeon angler trips in Section 1-10 on the lower Columbia River, 1975–2000.

1.4.1.2 Above Bonneville

Intensive management of sturgeon fisheries upstream from Bonneville Dam is required by the reduced productivity of these impounded populations. These fisheries provide the only significant opportunity for Columbia River treaty Indian fishery access to sturgeon. Sport fisheries are managed for equal opportunity. Intensive management involves regular assessments of stock status, population modeling to identify sustainable fishing levels, fishery monitoring and in-season management to limit annual catches within prescribed guidelines, and an annual fishery management and allocation process. The objectives are similar to those in the lower river (i.e. ensure adequate broodstock escapement.

1.4.1.3 Ocean/Estuary

White sturgeon are not directly harvested in ocean fisheries but can be taken incidentally at very low numbers in trawl fisheries. Columbia white sturgeon are also harvested in Washington and Oregon Coastal estuaries. Oregon and Washington management agreements require that estuary fisheries do not increase from historical impacts to white sturgeon.

1.4.1.4 Supplementation

Unlike the proliferation of salmon supplementation hatcheries throughout the Columbia River Basin (NRC 1996; Lichatowich 1999; Brannon *et al.* 1999), white sturgeon supplementation hatcheries do not exist in the Columbia Basin. The slow and deliberate development of a few sturgeon conservation hatcheries responds to the competing forces of real and perceived hazards of salmon supplementation hatcheries and the need to demographically support remnant sturgeon populations in the Columbia and Fraser river basins experiencing ongoing recruitment failure. To date, the Kootenai Hatchery, which captively rears progeny of wild parents for release at ages between 3 days and 2 years for conservation purposes, constitutes the only program with a stocking history dating back to the early 1990s (Kincaid 1993; Duke *et al.* 1999; Ireland *et al.* 2002).

1.4.2 Water Development

Hydropower development at the Columbia River Basin scale has extensively fragmented numerous historically free-flowing river reaches (NRC 1996). Dams result in population fragmentation and reduce or eliminate migration and non-reproductive and genetically effective migration (gene flow) among historically connected areas.

Population fragmentation—Population fragmentation represents a critical threat to the status of many sturgeon taxa, including *A. transmontanus* (Jager *et al.* 2000, 2001). Fragmentation can reduce population size, and it has long been known that isolated populations lose neutral genetic variability at a rate directly proportional to how small they are (Wright 1931). Because fragmentation can reduce population or deme size, and small population size can negatively affect genetic variation, fragmentation can indirectly reduce genetic variability. An important additional effect of such river fragmentation on white sturgeon is the artificial creation of a series of relatively isolated sub-populations, with artificial population genetic characteristics (Jager 2000, 2001; Secor *et al.* 2002). Fragmentation by hydropower dams can also result in variable and reduced productivity in post-impoundment river reaches (Bevelhimer 2001).

Population and habitat fragmentation by dams is especially critical for a migratory species like *A. transmontanus* because it may also impose predominantly or exclusively downstream gene flow on a species that likely historically exhibited upstream and downstream migration and gene flow. Jager *et al.* (2001) simulated the effects of dam creation (fragmentation) on white sturgeon population viability and genetic diversity, using two different simulations of an individual-based genetic metapopulation model. The first simulation fragmented a 124-mile (200 km) river reach by sequentially building 1-20 dams. These dams resulted in the exponential decline in the likelihood of persistence, but failed to produce any extinction threshold indicative of any minimum river length required for theoretical population persistence. Adding more dams in this simulation consistently eroded genetic diversity within and among populations. The second simulation of Jager *et al.* (2001) evaluated the effects of various levels of upstream and downstream migration between river segments. Results of the second simulation supported the view that migration plays a major role in the viability of riverine fishes, such as white sturgeon, when their habitat is fragmented by dams. Likelihood of extinction was high for populations linked by high downstream and low upstream migrations, as is often the case in extremely fragmented systems.

1.4.3 Flow Alterations

Altered daily and seasonal river discharge and thermal regimes resulting from impoundment and dam operations also may alter migration, limit habitat availability, and affect timing, location and success of reproduction (Paragamian and Kruse 2001; Paragamian *et al.* 2001; Anders *et al.* 2002; Cooke *et al.* 2002; Jager *et al.* 2002; Secor *et al.* 2002). Parsley *et al.* (2001) simulated drawdown of a Columbia River reservoir and concluded that the quality and quantity of white sturgeon spawning habitat would increase as reservoir levels were lowered. However, these authors suggested this outcome was due to increased availability of suitable velocities for spawning (Parsley *et al.* 1993) despite a decrease in total area of the river (Parsley *et al.* 2001).

Important empirical correlations between water year; discharge characteristics during June, July and August; and recruitment measured during September in the lower Columbia River impoundments attest to the importance of flow alterations on white sturgeon recruitment (Counihan *et al.* in press). An understanding of a positive relationship between discharge (water years) and natural production of Columbia River white sturgeon has existed since the late 1980s (Beamesderfer and Nigro 1995). Furthermore, consistent annual recruitment in the lower Columbia River, in the Bonneville Dam tailrace, and downriver areas were associated with conditions representing good water years due to the artificial constriction of the Columbia River through Bonneville Dam; as such hydro development has artificially created what functionally amounts to white sturgeon spawning channels downstream from Bonneville Dam, resulting in reliable annual recruitment (L. Beckman USGS (retired), G. McCabe Jr. NMFS (retired), M. Parsley, USGS, Cook Washington. personal communication).

Flow alterations can also affect white sturgeon spawning and embryo hatching success, to the extent that flow they alter downstream thermographs. (See Sections 1.1.1 Spawn timing and conditions, and 1.1.2 Incubation for more detailed discussions of temperature effects on spawning and incubation success.)

1.4.4 In-Channel Habitat Conditions

Sturgeon are particularly abundant in deep-water habitats of the Columbia River subject to channel maintenance and dredging activities. Suction dredging in deep areas (66-85 ft [20-26 m]) in the lower Columbia River is known to seriously injure and kill juvenile white sturgeon (Buell 1992) but the magnitude of the population impact is unclear. Channel deepening also may affect sturgeon directly via entrainment or indirectly via habitat or food interactions, but the net effect is unclear and speculation continues.

1.4.5 Water Quality

Hypoxia (oxygen limitation) may have disproportionately negative effects on sturgeons relative to other fauna, due to their limited capacity to osmoregulate at low dissolved oxygen concentrations (Klyashtorin 1976; Secor and Gunderson 1998 as cited in Secor *et al.* 2002). Although hypoxic effects may be particularly important to sturgeons during their first year of life due to their increased sensitivity and reduced ability—especially incubating embryos—to escape anoxia environments (Secor and Niklitschek 2001).

1.4.5.1 Temperature

Refer to Section 1.1.2 Incubation for a review of the potential effects of temperature on white sturgeon spawning and incubation. Since system productivity and fish growth and metabolism are positively temperature-dependent, variation in temperature affects these parameters. However, such effects are expected to be more pronounced in the upper Columbia Basin areas due to the thermal tempering effects of the increasingly large water volumes in areas of the lower Basin.

Because sturgeons are poikilotherms (cold-blooded), the rates and timing of metabolic activities vary in latitudinal and upstream (elevational) perspectives. Natural spawning of white sturgeon in the lower Columbia River (downstream from Bonneville Dam) routinely began at 8°C, whereas spawning was routinely initiated in the three furthest downstream dam tailraces when water temperature reached at least 10°C (M. Parsley, USGS, Cook, Washington, personal communication). Spawning was also generally first observed downstream from Bonneville Dam, and at intervals close to 2 weeks subsequently at the upstream series of Columbia River dam tailraces (e.g. The Dalles, John Day, and McNary Dams). Natural spawning of white sturgeon was documented to occur as late as July or August in upstream reaches of the Columbia River in British Columbia (Colin Spence, BC Ministry of Land, Water and Air Protection, pres. Comm.). Kootenai River white sturgeon, found at elevations several thousand feet higher and farther north (~ 49th parallel) than lower Columbia fish, spawned in water as cold as 8°C, or slightly colder. Spawning also occurs earlier in the spring season in downriver areas compared to upriver areas of the Columbia River Basin.

Finally, thermal regimes also dictate the length of the growing season for sturgeons throughout the Columbia Basin. Length at age and condition factor measures generally increase in a downstream orientation throughout the Columbia River basin. These variables also parallel longitudinal clines in food production and availability, which are positively affected by thermal regimes (increasing temperatures).

1.4.5.2 Turbidity

Having evolved in reproductive association with the world’s large river systems, sturgeons successfully adapted and flourished in systems that seasonally became very turbid. Highly- developed sensory organs for chemical (olfactory) and mechanical reception, and the lack of well-developed vision in sturgeons (Moyle and Cech 1983; Long 1995) suggest that turbidity may not be an important limiting factor for sturgeon, since they have adapted in its presence. Furthermore, turbidity could serve a positive role in concealment of planktonic white sturgeon early life stages, thereby reducing the effectiveness of visual predation.

1.4.5.3 Dissolved Gas

Counihan *et al.* (1998) conducted laboratory experiments investigating the effects of dissolved gas supersaturation on white sturgeon yolk-sac larvae and found that signs of gas bubble trauma were evident in 1–2 day old fish after only 15 minutes of exposure at 118% supersaturation. Yolk-sac larvae exposed to total dissolved gas (TDG) levels of 118% experienced no mortality, though their behavior was significantly different from control groups. Because of the development of a bubble in the buccal cavity, these fish were unable to descend from the surface. Yolk-sac larvae exposed to TDG levels of 131% experienced 50% mortality after 13 d of exposure.

Hypoxia (oxygen limitation) may have disproportionately negative effects on sturgeons, relative to other fauna, due to their limited capacity to osmoregulate at low dissolved oxygen concentrations (Klyashtorin 1976; Secor and Gunderson 1998). Although hypoxic effects may be particularly important during the first year of life due to the increased sensitivity and reduced ability of sturgeons (especially incubating embryos) to escape anoxic environments (Secor and Niklitschek 2001), specific oxygen and gas exchange requirements for incubating white sturgeon embryos are currently unknown.

1.4.5.4 Chemicals

The following passage on water quality and pollutant sensitivity in white sturgeon yolk-sac larvae was taken directly from Parsley *et al.* (2002): White sturgeon yolk-sac larvae are sensitive to poor water quality and pollutants. Brannon *et al.* (1985b) reported that water quality parameters for chlorine and gas supersaturation might be more critical for white sturgeon than for salmonids. The anti-sap stain wood preservative Bardac 2280 (principal active ingredient 80% didecyldimethylammonium chloride, DDAC) a common wood preservative, has been found to be particularly toxic to white sturgeon yolk-sac larvae with a 24-hour 50% lethal concentration (LC50) value between 1 and 10 ppb. Spill at dams can cause supersaturation of atmospheric gases in waters during yolk-sac larval dispersal.

Kruse and Scarhechhia (2002) studied contaminant uptake and survival of white sturgeon embryos in the laboratory. Uptake of organochlorine pesticides, Aroclor 1200 series PCBs, and metals were assessed relative to embryo survival. Eight metal species and two organichlorine compounds (DDE and PCB Aloclor 1260) were detected in embryos, thus confirming that they were uptaken from the incubating environment. These authors concluded that copper and Aloclor 1260 in the rearing media had negative effects on survival of white sturgeon embryos.

No literature was available on direct roles of nutrients on white sturgeon. However, sturgeon in parts of the Columbia River basin characterized by increased nutrient availability and abundant food resources (e.g. lower Columbia River and estuary) typically exhibit increased growth rates, length, and condition factor values compared to fish in upstream and culturally denutrified reaches (e.g. Kootenai River system). Growth rates of juvenile white sturgeon in the farthest downstream impoundments appeared to surpass those of fish in the lower river until about age 7 or 8, when access to marine-derived nutrients and estuarine food resources appeared to provide higher growth rates than those in upstream impounded fish (M. Parsley, USGS, Cook, Washington, personal communication).

1.4.6 Species Interactions

1.4.6.1 Competition

Little information exists on interspecific competition with sturgeon. Sturgeon occupy a unique niche among Columbia River fishes. Intraspecific competition and density-dependent effects on population dynamics are likely to be much more important for white sturgeon than interspecific competition. The potential for intraspecific competition between juvenile and adult is limited by diet shifts as larger fish are able to capitalize on larger prey, particularly including adult lamprey, shad, and salmon.

1.4.6.2 Predation

In fresh water, predation appears to be an important issue only for early life stages of white sturgeon, before age 1. In the estuary and ocean, predation on juvenile and subadult sturgeon is also likely.

***Eggs*—**Demersal white sturgeon embryos are vulnerable to fish predation (Anders 1994, 1996; Miller and Beckman 1996; Parsley *et al.* 2002). During 1994 and 1995, 632 stomach content samples from predatory fishes collected from the Kootenai River (northern pikeminnow *Ptychocheilus oregonensis*, peamouth chub *Mylocheilus caurinus* and suckers *Catastomus spp.*) were analyzed (Anders 1996). As discussed above, of 428 naturally spawned white sturgeon eggs collected, 12.2% (52) were collected from 623 predatory fish stomach samples analyzed. Miller and Beckman (1996) reported the occurrence of one to 70 white sturgeon eggs in guts of four omnivorous fishes in the Columbia River. These authors noted that a single largescale sucker (*Catastomus macrocheilus*) consumed 70 white sturgeon eggs.

***Larvae/juveniles*—**Recent empirical research revealed species-specific predatory behavior by several Columbia River omnivorous fish species on YOY white sturgeon (Gadomski *et al.* 2000, 2001, 2002). Researchers reported that adult northern pikeminnow and channel catfish (16-24 in TL [400-600 mm]) ingested sturgeon juveniles up to about 5 in 5 in (120 mm). Similarly-sized adult walleye ingested almost no sturgeon juveniles. However, juvenile walleye (6-8 in [150-200 mm]) ate sturgeon larvae and juveniles up to about 1.6 in (40 mm). Prickly sculpins (4-8 in [100-200 mm]) ate sturgeon up to 2 in (50 mm). When rock substrate was available, fewer sturgeon larvae were ingested by sculpins. When equal numbers of alternate prey were available, sculpins presented with both sturgeon and goldfish ate more sturgeon. Pikeminnow with smaller sturgeon and coho salmon prey available consumed both about equally. When sturgeon and coho prey were both larger, more coho were ingested. (Gadomski *et al.* 2000, 2001, 2002). Thus, predation appears to be an important natural mortality factor, at least with white sturgeon age 0 and younger life stages. However, beyond age 0, body size and scute development appeared to function as successful anti-predatory mechanisms.

1.4.6.3 Ocean & Estuary Conditions

White sturgeon, like other Acipenserids, are able to move freely between freshwater, marine, and estuarine habitats without requiring developmental and age-specific physiological adjustment (e.g. smoltification). However, little is known about specific effects of ocean and estuary conditions on white sturgeon. Likewise, little is known about how variability in ocean conditions may affect white sturgeon. Seasonal and inter-annual variation in productivity and food availability resulting from dynamic ocean and estuary conditions likely affect white sturgeon diet and habitat use in these areas. However, no studies directly linking ocean and estuary conditions to measurable physiological performance or behavior of white sturgeon were found.

1.4.6.4 Food Source Abundance

Very little is known regarding the effects of food source abundance for white sturgeon in marine and estuarine environments, but, based on empirical growth studies of white sturgeon in the four Columbia River impoundments farthest downstream and in the lower Columbia River, annual juvenile growth rates in the impounded areas generally surpassed those in the lower Columbia River until approximately age 7 or 8. Following this age, mean annual growth rate in the lower Columbia River, possibly including the estuary, generally exceeded rates in the impoundments (M. Parsley, USGS, personal communication). This increase in relative growth rate for juvenile white sturgeon downstream from Bonneville Dam was thought to result from access to food items unavailable in the impoundments (e.g. eulachon) (DeVore et al. 1995; M. Parsley, J. Devore, personal communication).

1.5 References

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Volume III, Chapter 2
Green Sturgeon

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2.0 Green Sturgeon (*Acipenser medirostris*)

Green sturgeon is an anadromous species that spawn in several West Coast rivers but spend most of their life in near-shore marine and estuarine waters from Mexico to southeast Alaska (Houston 1988; Moyle *et al.* 1995). Significant spawning populations of green sturgeon have been identified in the Sacramento, Klamath, and Rogue Rivers. While green sturgeon do not spawn in the Columbia Basin, significant populations of subadults and adults are present in the estuary during summer and early fall. Green sturgeon are occasionally observed as far upriver as Bonneville Dam. Reasons for concentrations in the Columbia River are unclear because no spawning occurs in that system and all of the green sturgeon stomachs examined to date have been empty. These fish may be seeking warmer summer river waters in the northern part of their range.

NOAA Fisheries (formerly know as NMFS) completed a status review for green sturgeon in 2003 and determined that listing under the ESA was not warranted at this time.

2.1 Life History & Requirements

Adult green sturgeon typically migrate into fresh water beginning in late February (Moyle *et al.* 1995). Spawning occurs in deep turbulent river mainstems. Klamath and Rogue River populations appear to spawn within 100 miles of the ocean, while the Sacramento spawning run may travel over 200 miles. Spawning occurs from March–July, with peak activity from April–June (Moyle *et al.* 1995). Confirmed spawning populations in North America are in the Rogue, Klamath, and Sacramento Rivers (Moyle *et al.* 1995).

Specific spawning habitat preferences are unclear, but eggs likely are broadcast over large cobble where they settle into the cracks (Moyle *et al.* 1995). The adhesiveness of green sturgeon eggs is poor compared to white sturgeon (Van Eenennaam *et al.* 2001), which may be explained by the reduced thickness of the outer layer of the chorion of green sturgeon eggs (approximately half the thickness of that in white sturgeon; Deng *et al.* 2001). Optimum flow and temperature requirements for spawning and incubation are unclear, but spawning success in most sturgeons is related to these factors (Dettlaff *et al.* 1993). Temperatures above 68°F (20ºC) were lethal to embryos in laboratory experiments (*Cech et al.* 2000).

Green sturgeon larvae are distinguished from other sturgeon by the absence of a swim-up or post-hatching pelagic stage. They can be distinguished from white sturgeon by their size (longer and larger), light pigmentation, and size and shape of the yolk-sac (Deng *et al.* 2001). Larvae hatched in the laboratory are photonegative, exhibiting hiding behavior (Deng *et al.* 2001), and after the onset of exogenous feeding, green sturgeon larvae and juveniles appear to be nocturnal (Cech *et al.* 2000). This development pattern and behavior may be an adaptation suited for avoiding downstream displacement.

Juvenile green sturgeon grow rapidly reaching 2 feet within 2–3 years (Nakamoto and Kisanuki 1995). Juveniles appear to spend up from 1–4 years in fresh and estuarine waters and disperse into salt water at lengths of 1-2.5 feet. Green sturgeon are benthic feeders on invertebrates including shrimp and amphipods, small fish, and possibly mollusks (Houston 1988).

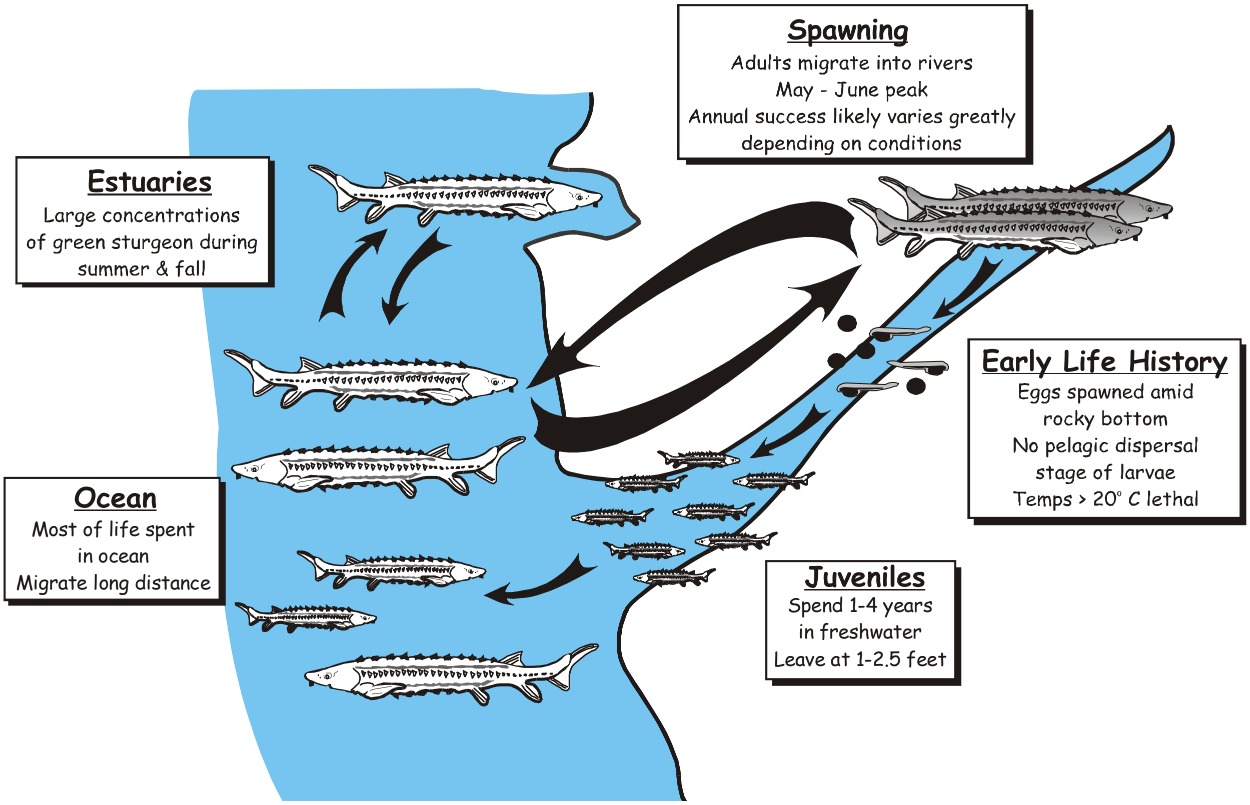

Figure 2-1. Conceptual model of green sturgeon life cycle and limiting factors

2.2 Population Identification

Based on a review of the limited genetic data available for green sturgeon, NMFS identified two distinct population segments. A northern DPS includes Klamath and Rogue spawning populations. A southern DPS includes the Sacramento spawning population. Fish using the Columbia River apparently include individuals from both population segments.

An Asian form of green sturgeon (*A. mikadoi*) also has been described and is known to spawn only in the Tumnin River (Artyukhin and Andronov 1990; Birstein et al 1997 as cited in Birstein and Bemis 1997).

2.3 Status & Abundance Trends

Available time series data were limited, but were not consistent with a modern decline in green sturgeon abundance. Time series data on green sturgeon abundance and size composition are limited to fishery landing statistics; these do not provide a consistent index of green sturgeon abundance. Columbia River harvest per unit effort and size composition data suggest an increasing rather than decreasing trend in green sturgeon abundance. Current data indicate that:

- green sturgeon still spawn in most systems where they were historically present,
- significant numbers of spawners are present in several systems, and
- geographic range of spawning green sturgeon is currently stable or increasing.

The wide distribution of green sturgeon, large numbers seasonally observed in some areas, and projections based on demographic rates suggest that total green sturgeon numbers are at least in the tens of thousands.

2.4 Limiting Factors

Spawning populations of green sturgeon are potentially affected by runoff and water management in natal rivers. Physical factors affecting green sturgeon habitat use in the lower Columbia River and estuary are unclear.

Like all sturgeon, green sturgeon populations are highly susceptible to fishing mortality. Significant catches or concentrations of green sturgeon have been reported in widely-scattered areas throughout the broad distribution of this species. Directed green sturgeon fisheries do not occur anywhere in their range, but small incidental harvest occurs in several areas. Annual harvest data are available from the Columbia River where salmon and white sturgeon fisheries intercept green sturgeon that gather in the estuary in large numbers during late summer and early fall. The majority of the green sturgeon harvest occurs in commercial gill net fisheries. Green sturgeon landings in the Columbia River declined steadily from a peak harvest of 6,400 in 1986, which occurred during expanded salmon fisheries for a very large fall chinook and coho return (Figure 2-2). Rather than an index of green sturgeon abundance, decreases are the direct result of increasing restrictions on white sturgeon and salmon fisheries since 1986.

Like the white sturgeon, green sturgeon probably recovered slowly following collapse of excessive 19th century fisheries. Gradual recovery is consistent with harvest patterns of green sturgeon in Columbia River fisheries: green sturgeon landings were identified beginning in 1938 and they show a generally increasing trend until the 1960s with variable harvest but no obvious trend after 1960. Recent fluctuations in green sturgeon landings likely result from a combination of annual variation in occurrence in the Columbia estuary and in salmon or white sturgeon fisheries. Where total harvest is expressed as catch per unit effort based on fish tickets logged by commercial fish buyers, catch rates have been variable and possibly increasing over the same period that total green sturgeon landings have decreased. Increases might have been even more pronounced but for the fact that the legal-size slot for retention was reduced from 48-72 inches to 48-66 inches TL in 1993. In addition, average size and frequency of large green sturgeon have been increasing.

No supplementation or conservation hatchery programs currently exist for green sturgeon throughout their range (Joel VanEnennaam, University of California, Davis, personal communication).

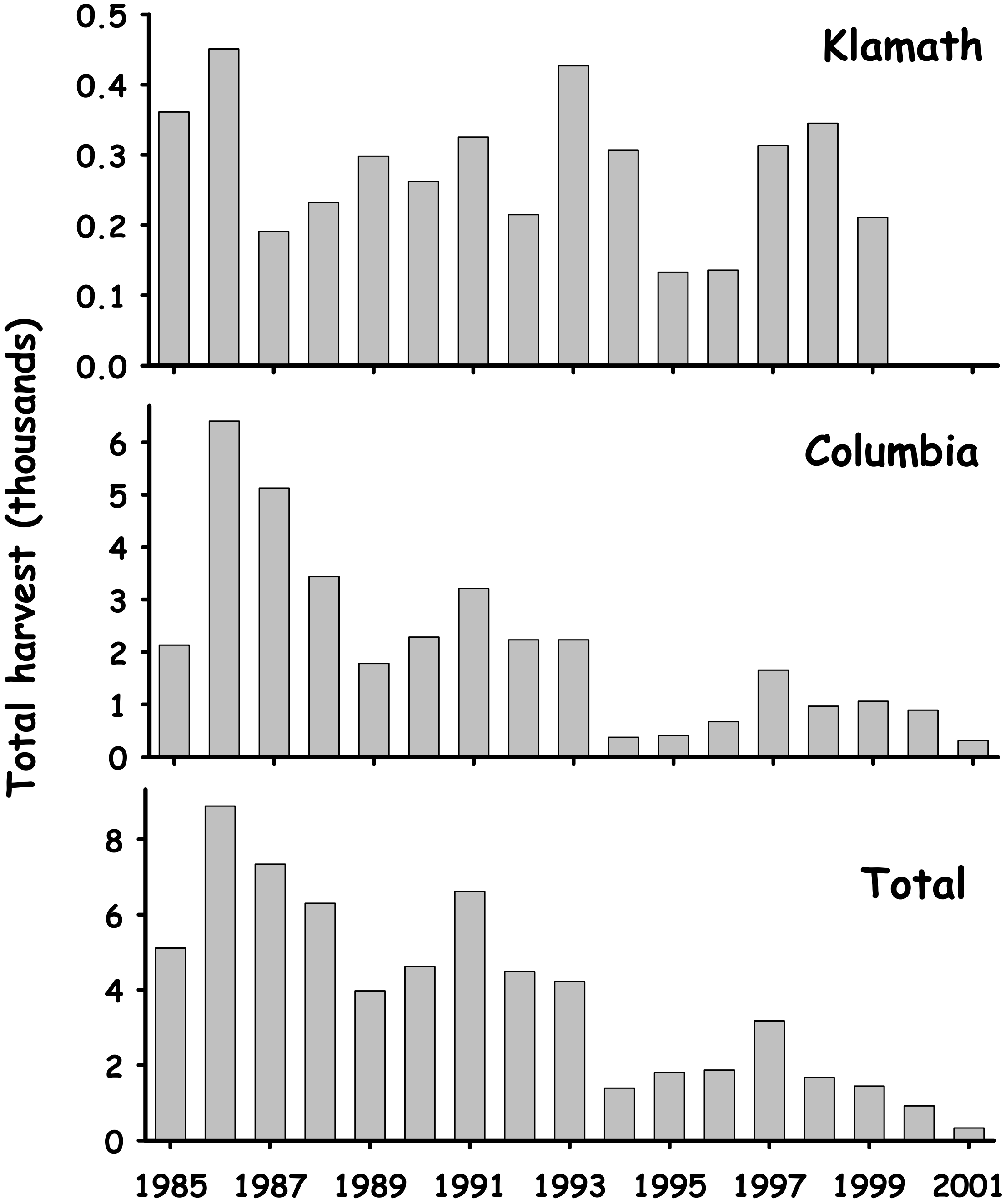

Figure 2-2. Recent annual harvest of green sturgeon as reported by Farr et al. (2002). Columbia includes Oregon and Washington sport and commercial fishery harvests. Klamath includes Yurok and Hoopa subsistence fishery harvests. Total also includes Oregon coastal sport, Washington coastal sport, commercial, Tribal, and Oregon and Washington trawl. The total does not include California sport harvest which is believed to be minor (Schaffter 2000).

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Pacific Lamprey

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3.0 Pacific Lamprey (*Lampetra tridentata*)

The anadromous and parasitic Pacific lamprey (*Lampetra tridentata*) is a native inhabitant of the Pacific Northwest. Pacific lampreys, largest of the native lampreys, are an important part of the Columbia basin in a cultural, utilitarian and ecological sense. Native Americans harvested Pacific lamprey at numerous natural barriers throughout the Columbia basin for subsistence, ceremonial and medicinal purposes.

When European settlers arrived they began harvesting Pacific lamprey in great numbers at select locations such as Willamette Falls, where 7 tons of lamprey were harvested in 1913 (Close et al. 1995). In the mid-1900s, lamprey became commercially important with harvest at Willamette Falls approaching 200 tons in 1946 (Ward 2001). Lampreys were used for vitamin oil, protein food for livestock, poultry, fish bait and fish meal (Mattson 1949). Recent year harvest at Willamette Falls is primarily for tribal treaty, collection for anticoagulants, subsistence use, bait for sturgeon fishing, and teaching specimens (Close et al. 1995).

Two other species of lamprey, the river lamprey (*L. ayresii*) and western brook lamprey (*L. richardsonii*) coexisted with Pacific lamprey in the Columbia River basin (Kan 1975). Western brook lampreys have been observed on Oregon and Washington stream (Jackson et al. 1997). However, distinction of ammocoetes of these three species is difficult. Little is know about the river lamprey.

Pacific lampreys are ecologically important to the Pacific Northwest. As an anadromous species that dies after spawning, lampreys provide an important vessel for carrying marine-derived nutrients to the freshwater environment. Numerous marine species use lamprey as an important food item. Near the mouth of Klamath River, it was estimated that 82% of sea lion feeding observations were of Pacific lamprey (Jameson and Kenyon 1977). In fresh water, at least 7 aquatic and five avian species prey on lamprey.

Understanding of Pacific lamprey population status in the lower Columbia is hindered by lack of data. Very little research has focused on Pacific lamprey distribution, abundance, productivity, migration survival, and habitat association. However, limited available data suggest that Pacific lamprey populations in the Columbia basin have been declining since the construction of the hydroelectric network of dams on the mainstem Columbia River. Adult lamprey counts at each of the mainstem dams are markedly lower than counts during the mid-1900s, and growing evidence indicates that Pacific lamprey have great difficulty surviving downstream passage at dams and migrating upstream past dams (Close 2000).

3.1 Distribution

The parasitic Pacific lamprey is found primarily along the Pacific coast and near coastal islands. Its range in North America extends from Unalaska Island (Aleutians) to Baja California. From the Aleutians, the range of Pacific lamprey extends to the eastern Asian coast and as far south as the Uuhutu River, Japan (Scott and Crossman 1973). Pacific lamprey are common within the Columbia basin, and extend inland to Chief Joseph and Hells Canyon dams (Close et al. 1995), and their historicaldistribution is considered to be similar to that of Pacific salmonids (Close et al. 1995). Although populations currently exist in the Snake and Clearwater River drainages, Clair (2003) reported that Pacific lamprey may be nearing extinction in the Snake River basin.

The Pacific lampreys are anadromous and thus utilize both freshwater and marine environments. In fresh water, lampreys use small to mid-sized streams for spawning. Juvenile lampreys use these environments for rearing for up to 7 years before migrating to the ocean. Little is known of distribution of Pacific lamprey once they reach the ocean, but it is believed that they move off-shore quickly, and have been caught in high seas sampling (Kostow 2002).

There are some accounts of landlocked Pacific lamprey. Hubbs and Miller (1948) describe populations of landlocked Pacific lamprey in Oregon and California. Other landlocked populations include Cultus Lake (Vladykov and Kott 1979), and in Cowichan Lake (Hart 1973). In landlocked populations, it is reported that Pacific lamprey mature to the parasitic life stage and prey upon resident fish populations (Zanadrea 1961; Scott and Crossman 1973; McPhail and Lindsey 1970; Hart 1973).

3.2 Life History Characteristics

3.2.1 Freshwater Existence

The timing of the return of anadromous Pacific lamprey from the ocean to spawning grounds seems to be prolonged, but is poorly understood. Beamish (1970) reported that Pacific lamprey return to fresh water from May to September. In the lower Columbia River, Kan (1975) reported that returning adults have been seen as early as February. At Willamette Falls, peak numbers passing the falls occurred in May and June (Kostow 2002). In contrast, Scott and Crossman (1973) suggest that spawning migration from the ocean begins in July to September. Adult lampreys enter Cedar Creek in the Lewis River basin in southwest Washington between June and November (Stone et al. 2001; 2002).

Also in Cedar Creek, researchers found that movement of adults past traps occurred in two pulses, one in the late-spring/early summer and another in late summer/early fall (Stone et al. 2002). It is unclear whether lampreys from these two movements left the ocean at similar times, or if a portion of the fish held lower in the basin prior to moving past the traps in the fall. Starke and Dalen (1995) analyzed data from the Columbia River and determined that there are two Pacific lamprey runs past Bonneville Dam, with one occurring in late May to early June, and another in late July to early August. Stone et al. (2002) found that this movement pattern may be related to flow, but Starke and Dalen (1995) suggested that these movements indicate the existence of multiple sub-runs of lamprey. Feeding ceases during upstream migrations (Scott and Crossman 1973). Lampreys move into their spawning tributaries where they over-winter prior to spawning in the spring (Beamish 1980; Kostow 2002; Stone et al. 2001, 2002). During this time, they are dormant and hide within the substrate (Scott and Crossman 1973).

The length of lamprey upon return to fresh water depends on the length of time spent in the marine environment. Beamish (1970) reported that returning adults commonly range from 5 to over 28 in (13-72 cm). These measurements were taken at spawning, and likely underestimate lengths upon leaving the ocean. In Cedar Creek in the Lewis basin, the mean length of adult lamprey was 21 in (53 cm) with a range of 19 to 24 in (48 to 61 cm) in 2001 (Stone et al. 2002). The previous year, lengths were similar with a mean of 22 in (55 cm) and a range of 18-24 in (46-61 cm) (Stone et al. 2001).

Homing abilities and mechanisms of Pacific lamprey are not well understood. Sea lamprey (*Petromyzon marinus*) do not home to natal streams as salmon do (Bergstedt and Seelye 1995), but home based on response to pheromones released by larval lamprey (Bjerselius et al. 2000). Specific references regarding homing in Pacific lamprey are conflicting. Hardisty and Potter (1973) found no conclusive evidence that Pacific lamprey home to natal streams. Beamish (1980) reported, based upon lengths of Pacific lamprey returning to streams, that at least some adults return to native streams. These latter studies only comment on homing to natal streams, and do not consider the potential for homing similar to that of sea lamprey.

Spawning of Pacific lampreys occurs in the spring when temperatures approach 43°F (8.5°C) (Pletcher 1963; Carl et al. 1977; Lee et al. 1981; Kostow 2002). In the Willamette and on the Oregon Coast, spawning takes place primarily between February and May (Kostow 2002). In Cedar Creek, spawning of lampreys in both 2000 and 2001 occurred primarily from April through late June/early July. Average daily temperatures during this period in 2001 ranged from 45 to 61°F (7 to 16°C) (Stone et al. 2001; Stone et al. 2002). In British Columbia populations spawning may begin in April, and extend until late July (Beamish 1980).

Pacific lampreys spawn in low gradient streams in sandy gravel at the head of riffles and in pool tailouts (Kan 1975; Carl et al. 1977; Scott and Crossman 1973; Stone et al. 2002). Some authors have noted Pacific lamprey spawning in lentic environments. In Babine Lake, British Columbia researchers noted the presence of 14 lamprey redds and spawning lamprey pairs in shoreline environments (Russell et al. 1987). Redds are constructed by either moving rocks with body motions, or by grasping individual gravels and cobbles with their grasping mouthparts (Scott and Crossman 1973; *US v Oregon* 1997). Redds are 16-24 in (40-60 cm) in diameter, and are located in water depths less than 3 ft (1m) (Pletcher 1963; Kan 1975; Russell et al. 1987). Pacific lampreys are highly fecund, with females possessing up to 34,000 eggs. Most evidence indicates that lamprey die within 2 weeks of spawning (Moffet and Smith 1950; Scott and Crossman 1973; Beamish 1980). However, Michael (1980) reported evidence of seaward migration of adult lamprey after spawning, and repeat spawning on the Olympic Peninsula. Similar observations have been made by ODFW personnel for Oregon waters (Kostow 2002).

Egg incubation is influenced by temperature, and may last 10 to 14 days (Kostow 2002). After hatching but prior to the stage at which they begin to metamorphose into adults Pacific lamprey are known as ammocoetes. After hatching, ammocoetes burrow into sand/silt substrate downstream of their nests (Scott and Crossman 1973). Ammocoetes will spend approximately 6 years rearing in fresh water, during which time they remain burrowed in fine substrates filter feeding on algae (Kostow 2002). Although relatively little is known about Pacific lamprey rearing habitat requirements, Claire (2003) reported that ammocoetes density was not significantly correlated with water depth and coarse substrates but did increase with fine substrates and canopy cover (shading) of the stream reaches studies in Idaho. Ammocoete density also decreased with increasing water column velocity in this study.

Prior to the adult life stage, Pacific lamprey are not parasitic and do not have parasitic mouthparts. Ammocoetes possess an oral hood and their eyes are undeveloped. At hatching, ammocoetes are minute and grow to about 0.4 in (1 cm) in their first year (Scott and Crossman 1973). Survival and growth at early life stages are related to temperature. In laboratory experiments, survival decreased at temperatures of 72°F (22°C) compared to 50°F (10°C), 57°F (14°C), and 64°F (18°C) (Meeuwig et al. 2002). Juvenile Pacific lamprey were found in Idaho at stream temperatures up to 79°F (26^{o}C), however, interstitial space and substrate temperatures were generally 2 degrees cooler (Claire 2003). As ammocoetes grow, they move gradually downstream (primarily at night; Claire, 2003) and continue to burrow and feed in fine substrates (Kostow 2002). Older ammocoetes generally occupy the lower portions of river basins and flood plains.

As metamorphosis approaches, ammocoetes are typically 4.8 to 12 in (12 to 30 cm) in length and are known as macrothalmia (the physiological equivalent of a salmon smolt). Pacific lamprey transform from ammocoetes to macrothalmia in July to October (van de Wetering 1998, Stone et al. 2001, Stone et al. 2002). At the macrothalmia stage, Pacific lampreys begin their migration to the sea and develop parasitic features characteristic of the adults. During this transformation, lampreys survive on lipid reserves and do not feed (Kostow 2002). Downstream migration of ammocoetes and macrothalmia in Idaho’s Clearwater River basin occurred primarily at night from mid-March to the end of May, with a limited number of out-migrant observed during September and October (Claire 2003). These results were similar to a study of the abundance and freshwater migrations of pacific lamprey in a tributary of the Fraser River in British Columbia (Beamish and Levings 1991).

There are some discrepancies in reports of downstream migration timing. In the lower Willamette River outmigration peaked in May, though no monitoring occurred in the winter (Kostow 2002). Studies in the Umatilla have shown that outmigration peaked in the winter and early spring (Kostow 2002). In Cedar Creek in the Lewis basin, peaks in macrothalmia movement occurred in February and June/July where macrothalmia movements were typically correlated to discharge (Stone et al. 2002). There was also a downstream movement of ammocoetes in February through July (Stone et al. 2002). Peak ammocoete movements in February were temporally correlated with high flows, but peak movements in late spring were not. Fish that moved during periods of increased river discharge were significantly shorter than those that did not (Stone et al. 2002). It is unclear if the difference in size between those whose movements were correlated to discharge were smaller because of discharge influences or smaller because their movement occurred earlier in the year. In landlocked populations, macrothalmia finish their metamorphosis into a parasitic adult and spend their adult life preying on resident fishes (Scott and Crossman 1973).

3.2.2 Marine Existence

Adult marine Pacific lampreys prey on fishes and marine mammals. Accounts of marine residence time differ with estimates ranging from 12 to 20 months (Scott and Crossman 1973) and others estimating up to 40 months (Kan 1975; Beamish 1980; Richards 1980; Lee et al. 1981). Feeding at sea is accomplished by attaching to fishes. The toothed tongue penetrates scales and skin and fluids are consumed. (An illustration of the mouth of the Pacific lamprey is provided below.) Feeding is supported by the production of an anticoagulant which prevents the host’s blood from coagulating (Scott and Crossman 1973).

Pacific lamprey prey include salmon, steelhead, rockfish, cod, sable fish, halibut, flounder, Pacific ocean perch, pollock, hake, and whales (Pike 1950; Pike 1951; Beamish et al. 1976; Beamish 1980). Predation rates by lamprey on specific stocks can be extremely high. Beamish et al. (1976) reported that 10% of 145 pollock from Dixon Entrance in British Columbia had lamprey wounds. Prey are weakened by lamprey predation, but many survive as evidenced by survivors with lamprey scars (Carl et al. 1977). Williams and Gilhousen (1968) examined lamprey scarring on salmon entering the Fraser River and found that 20% of pink salmon and 66% of sockeye salmon sampled had lamprey scars. They estimated that mortality due to lamprey wounds after fish moved onto the spawning grounds was 1.6-1.8%.

Figure 3-1. Mouth of parasitic Pacific lamprey.

Little is known about marine movement of Pacific lamprey. Some authors have suggested that lamprey move considerable distances, and that migration distance may be correlated to lamprey length (Pletcher 1963; Beamish 1980). Larkins (1964) recorded catches of Pacific lamprey during high seas sampling, suggesting that lamprey move considerable distances off-shore.

3.2.3 Population Demographics

There is evidence that suggests Pacific lamprey populations occur in clusters. Observations on Clear Creek in the Clackamas River basin and in Bear Creek in the Rogue River basin showed that lamprey tended to congregate in certain areas. In both Clear Creek and Bear Creek, lampreys were found in high concentrations, but when using the same sampling method the same year in adjacent tributaries relatively few lampreys were found (Kostow 2002). This spatial pattern of population distribution would increase the susceptibility of lamprey to significant losses under localized disturbances such as landslides, chemical spills, forest fires, and other natural and anthropogenic perturbations.

3.3 Status & Abundance Trends

3.3.1 Abundance

The status of lamprey populations in the lower Columbia basin is not well understood for several reasons: 1) Many observations of lampreys in fresh water are of juveniles, and it is difficult to distinguish juveniles of the various lamprey species; 2) Data on lampreys is typically only collected incidental to research on salmonids, and thus data gathered regarding lampreys is typically not analyzed or presented and 3) There are very few historical data sets for lampreys (Kostow 2002).

Counts at mainstem Columbia River dams suggests that Pacific lamprey runs in the Columbia basin have declined considerably. However, counts at dams are considered unreliable for estimating lamprey abundance. In the past, counts were only conducted during the day, and evidence suggests that primary lamprey movement is at night. Recent fish counts that employed lights at counting windows indicated that lamprey display avoidance of these lighted areas (Ocker 2001). Also, lampreys often struggle passing fish ladders and are often seen floating downstream past viewing windows. Though today these fish are considered in daily passage counts, this has not have always been the case, and some lamprey may have been double counted (Kostow 2002).

Despite these limitations, lamprey counts at dams are one of the few long term consistent data sets available. Records of lamprey passage at Bonneville Dam are available from 1938 to 1969. Monitoring was discontinued until it began again in 1997 and continues to date. Average annual lamprey passage between 1938 and 1969 was 109,000 with Bonneville Dam counts ranging from 26,000 to 380,000. Since counts were reinitiated in 1997, lamprey passage has averaged 39,000 and ranged from 19,000 to 100,000. (Figure 3-2 shows annual counts of Pacific lamprey passing Bonneville Dam.) These counts indicate that abundances of lamprey moving to the interior Columbia basin are reduced from the mid-1900s. However, in 2002, lamprey counts at Bonneville were much higher than the previous 5 years, and similar to the average for the period of 1938–69.

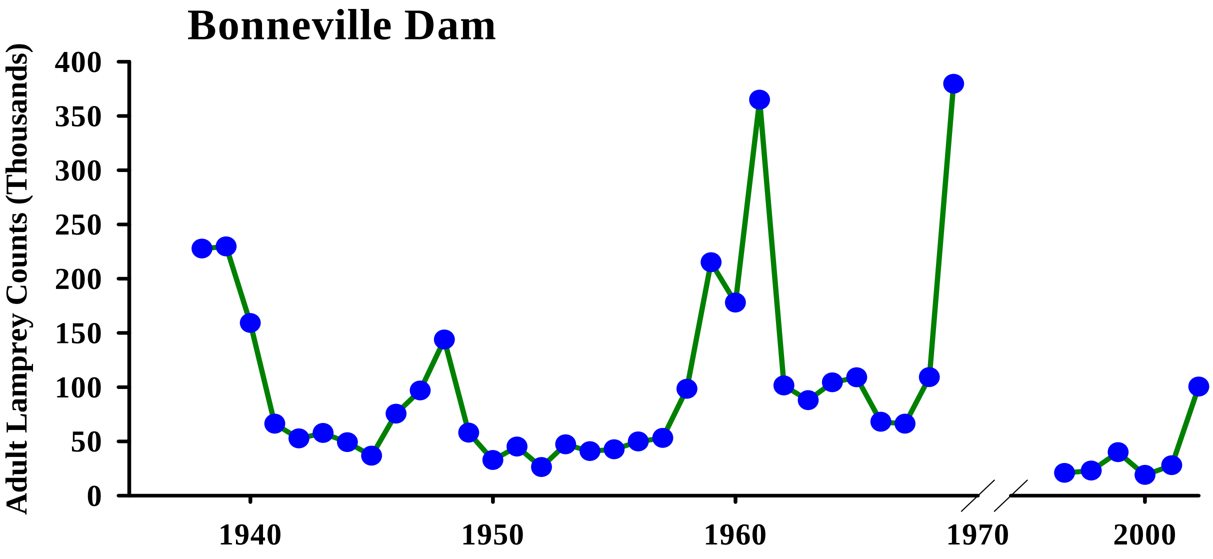

Figure 3-2. Annual counts of Pacific lamprey passing Bonneville Dam, 1938–69 and 1997–2002.

Counts at Bonneville reflect the variable nature of lamprey abundance. From 1967–69, counts expanded by 600% at Bonneville Dam, and wide fluctuations were common between 1938 and 1969. More recently, Bonneville passage increased from 19,000 in 2000 to just over 100,000 in 2002. These fluctuations suggest a cyclic survival pattern similar to salmonids, likely dependent on variable freshwater and marine environmental conditions.

Evidence from the lower Columbia River also suggests that lamprey abundances may be on the decline. Observations at Willamette Falls in the 1800s indicate that hundreds of thousands lamprey passed the falls annually (Kostow 2002). During the 1940s, nearly 400,000 pounds (18,140 kg) of lamprey were harvested in a single year. Recent year harvest opportunity at Willamette Falls has been reduced by time and area regulations compared to the 1940s and the fishing interest level has reduced. A significant reduction in recent year lamprey landings at Willamette Falls likely reflects recent year fishery restrictions, reduced fishing effort, and reduced lamprey abundance. (Figure 3-3 illustrates the Willamette Falls lamprey harvest.)

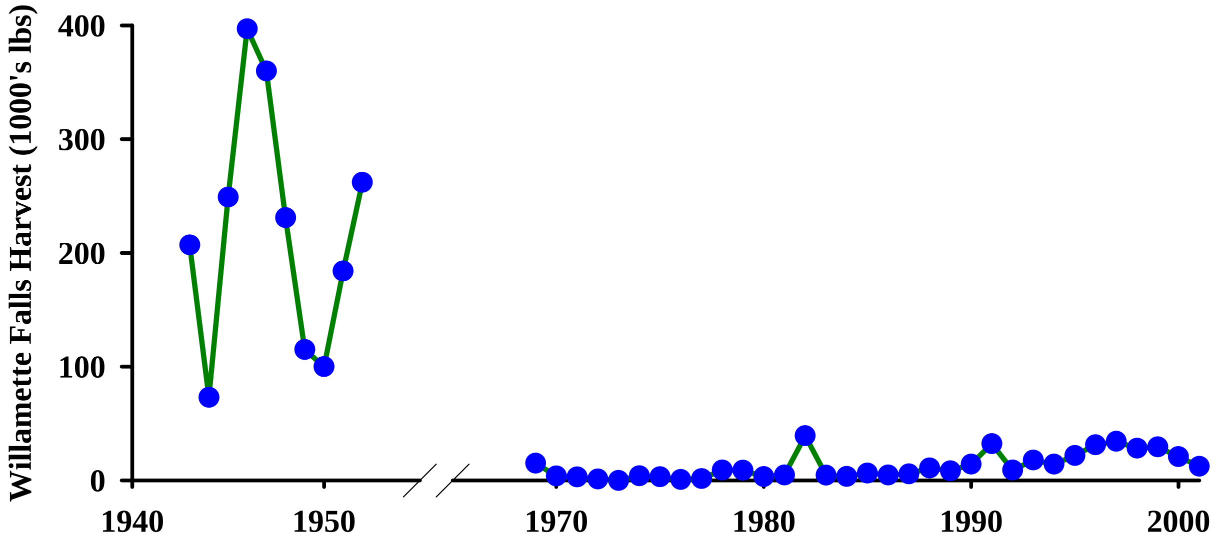

Figure 3-3. Harvest of adult Pacific lamprey at Willamette Falls, 1943–52 and 1969–2001. Data from Ward (2001).

Kostow (2002) compared Gnat Creek data from the 1950s and 1960s with recent data from Scappoose Creek and concluded a reduction in lamprey abundances. ODFW monitored an adult weir on Gnat Creek (tributary to the lower Columbia) in the 1950s and 1960s (Willis 1962). Each year, hundreds to thousands of adult lamprey were caught moving upstream and downstream past the weir along with thousands of juveniles. Willis (1962) noted additional adult lamprey moving upstream around the weir that were not being counted. Scappoose Creek is a slightly larger tributary to the lower Columbia, compared to Gnat Creek. Adult lampreys have been counted at a trap facility on Scappoose Creek since 1999, and fewer than 400 adults have passed during the 3 years of monitoring. This comparison suggests that abundance of adult lamprey in the lower Columbia may have declined since the 1960s.

Despite data indicating decreasing abundances, evidence suggests that Pacific lampreys are well distributed throughout the lower Columbia basin. In the last 2 years, WDFW personnel have observed adult lamprey in screw traps in Mill, Germany, and Abernathy Creeks in the lower Columbia River drainage (Patrick Hanratty, WDFW, personal communication). Populations successfully pass Willamette Falls, are still found in the Clackamas River, and

ODFW crews have encountered adult lamprey in various locations in the lower Columbia and Willamette River basins (Kostow 2002).

3.3.2 Productivity

Information regarding productivity of lower Columbia lamprey populations was unavailable.

3.4 Factors Affecting Population Status

3.4.1 Harvest

Historically, tribes in the Columbia basin harvested lamprey for food, ceremonial, medicinal, and trade purposes. Harvest occurred at natural barriers throughout the Columbia basin (US vs. Oregon 1997). With the addition of dams throughout the basin, and decreases in lamprey abundance, these conditions and opportunities are no longer present. Today, harvest occurs primarily at Willamette Falls and at Sherars Falls on the Deschutes River.

Since the early 1980s through 2000, harvest was allowed 7 days a week but restricted to June through August. The harvest period occurs at the tail end of the lamprey run , and its impact is inherently limited by the method and time of day of catch (*US v. Oregon* 1997). In 2001, in response to concern for the status of Willamette basin lamprey, the Oregon Fish and Wildlife Commission further reduced harvest by restricting the number of fishing days in the week and limited the season to 6 weeks (Ward 2001). Tribal subsistence opportunity also occurred within the same 6 week period. Landings of lamprey and the number of permits issued for harvest at Willamette Falls from 1980–2001 are displayed in Table 3-1.

Harvest of Pacific lamprey in the lower Columbia River basin is not regulated with the exception of Willamette Falls, and given its limited nature is not likely a major causal factor of decline for Pacific lamprey in the lower Columbia basin. At Willamette Falls, current regulations restrict harvest to certain portions of falls by hand or with hand operated tools, and only during the daytime (Kostow 2002), and in the late spring after the majority of the lamprey run has passed over the falls.

3.4.2 Supplementation

There are no Pacific lamprey supplementation programs in the lower Columbia basin.

Table 3-1. Lamprey harvest at Willamette Falls, 1980–2001 (Non-commercial landings are not known prior to 2001*) .

Year	Commercial Landings (lbs)	Value ($)	Number of permits		
			Commercial	Personal	Indian
1980	3,223	970	3	—	—
1981	4,666	2,008	2	—	—
1982	39,169	26,681	2	—	—
1983	4,482	582	4	80	70
1984	3,391	856	3	55	68
1985	6,381	2,233	3	43	54
1986	4,740	1,659	4	75	90
1987	5,633	1,100	10	28	40
1988	10,896	1,634	14	6	25
1989	8,366	2,950	7	9	27
1990	14,203	3,562	8	6	19
1991	32,221	9,017	13	13	11
1992	9,089	2,381	15	12	6
1993	17,858	4,028	12	26	9
1994	7,884[a]	2,819	4	20	21
	6,376[b]	1,734			
1995	14,097[a]	17,622	8	23	20
	7,800[b]	4,091			
1996	23,008[a]	23,008	4	13	11[c]
	8,256[b]	5,366			
1997	15,546[a]	15,870	3	22	23
	18,696[b]	13,823			
1998	14,580[a]	14,967	7	15	33
	13,638[b]	13,647			
1999	3,330[a]	3,516	7	22	7
	26,119[b]	27,598			
2000	4,314[a]	4,530	4	34	20
	16,624[b]	18,455			
2001	12,276[b]	14,608	8	16	12

* Table from Ward (2001). a = Indian landings and values, b = Non-Indian landings and values

3.4.3 *Water Development*

3.4.3.1 Dams

Despite evidence of sustained landlocked populations, the majority of evidence suggests that dams have significant deleterious effects on Pacific lamprey. The construction of a dam on the outlet of Elsie Lake, British Columbia caused the apparent extinction of Pacific lamprey above the dam. For 7 years after dam construction, occurrences of lamprey attacks on resident trout increased and then suddenly ceased. Since this is the estimated life span of ammocoetes, it was interpreted that the conversion from an anadromous life history to the resident form was unsuccessful (Beamish and Northcote 1989). Researchers have found that it is difficult to keep metamorphosed parasitic lamprey in fresh water (Potter 1970; Potter and Beamish 1977).

There is substantial evidence indicating that Columbia River dams have had a negative effect on Pacific lamprey populations. Hammond (1979) suggested that construction of hydroelectric dams in the Columbia system have caused a significant decrease in Pacific lamprey populations. Upstream passage efficiency of adult lamprey between 1997 and 2000 at Bonneville, The Dalles and John Day Dams have been estimated to be 38-47%, 50-82%, and 27-55% respectively (Moser et al. 2002). In the Clackamas River basin, data from several screw traps indicated that Pacific lamprey are not restricted to streams below North Fork Dam (Kostow 2002). Upstream passage efficiency is shown in Figure 3-4.

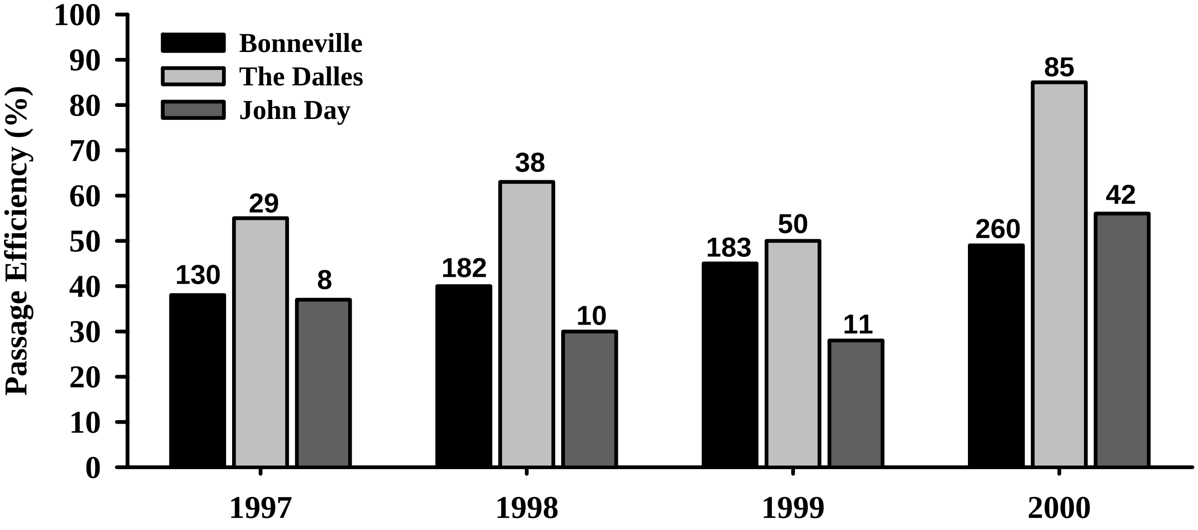

Figure 3-4. Overall passage efficiency for Bonneville, The Dalles, and John Day Dams in 1997–2000. Number of lampreys that approached the dam are shown above the bar. Graph adapted from Moser et al. 2002.

Pacific lamprey are weak swimmers, and passage at dams proves to be difficult. Areas of fish ladders that are most difficult include areas where gratings have to be crossed, where water velocity is high, and where there is lighting, as Pacific lamprey are nocturnally active and negatively phototaxic. Although Pacific lamprey were reported to have difficulty passing through hydropower facilities (Vella et al.1997), the preferred passage tactic of lamprey at hydropower dams is to grasp onto surfaces to rest, and then surge forward a short distance before resting again (Ocker et al. 2001). In areas where there is no surface to cling to (gratings), or where the water velocity is too high, passage may be impossible. Moser et al. (2002) concluded that the ability to find attachment sites is important to lamprey passage through areas of high velocities including fishways.

The lower Columbia basin may provide a critical refuge area for Pacific lamprey unable to access the interior Columbia basin due to poor passage efficiencies at mainstem hydropower projects (Kostow 2002).

Negative effects on downstream migrants have also been noted. Lamprey migrate downstream as macrothalmia and ammocoetes. They may pass under fish screens on dams (Long 1985) or may pass through bypass screens which have openings larger than the diameter of lamprey. Kostow (2002) suggested that the highest potential for mortality comes from the fish screens. Kostow (2002) states, "Anecdotal observations by biologists working on mainstem dams on the Columbia and Snake Rivers during the 1970s and 1980s indicated that juvenile lampreys impinged on the perforated plates that blocked various openings across the forebay faces of the dams and on the juvenile bypass screens in huge, but undocumented numbers." Such observations have been documented to some extent by Stark and Dalen (1995).

3.4.3.2 Other Migration Barriers

Other migration barriers may also contribute to the Pacific lamprey decline in the lower Columbia River basin. In Oregon's Alsea River basin, a presence/absence survey by van der Wetering showed that no Pacific lamprey were found above most road culverts (Kostow 2002). Other small barriers that may hinder lamprey populations include hatchery weirs, tide gates, or diversion structures (Kostow 2002).

3.4.4 Flow Alterations

As with other anadromous fish, lamprey undergo a metamorphosis as part of their migration to the sea. This metamorphosis takes place within a given physiological window. Dams in the Columbia River and its tributaries disrupt and slow river flow and alter thermal conditions. It has been observed that increased discharge initiates the downstream movement of macrothalmia (Beamish and Levings 1991; Stone et al. 2002). Lamprey are weak swimmers, and typically rely on flow to carry them towards the ocean. The reduction in velocity in reservoirs may delay migration for juvenile lamprey, and disrupt the synchrony of physiological development and downstream migration.

Dams and land use patterns have altered flow regimes to some degree in most Columbia basin tributaries. Kostow (2002) suggested that increased peak flows may mobilize cobble substrate disrupting Pacific lamprey nests. Increased flows may also flush fine sediments that have accumulated and provided habitat for ammocoetes.

3.4.5 In-Channel Habitat Conditions

3.4.5.1 Channel Maintenance & Dredging

Lamprey burrow in fine sediment river bottoms throughout their migration to the ocean (Kostow 2002). Dredging is common in the lower Willamette and Columbia Rivers, and may pose a threat to downstream migrating lamprey. Beamish and Youson (1987) discovered that they could find River lamprey by filtering through dredging spoils from the lower Fraser River. They estimated that only 3-26% of lamprey that passed through a dredge survived.

3.4.5.2 Floodplain Development

Stone et al. (2002) found that juvenile lamprey density was negatively associated with gradient and gravel substrate, and positively associated with percent fines in the substrate in Cedar Creek, Washington. This habitat preference suggests that juvenile lamprey likely favor lower basin, low gradient reaches for rearing. It is typically these portions of basins that experience the most development. With development may come flow alterations, water chemistry influences, channel morphology alterations, thermal alterations, and chemical pollution. With the lack of understanding of Pacific lamprey ecology, it is difficult to say how these alterations may influence lamprey populations. However, such changes are generally assumed to not benefit Pacific lamprey.

3.4.6 Water Quality

3.4.6.1 Temperature

Increased stream temperatures from land use practices and hydro modifications may also affect the survival of juvenile Pacific lamprey. Van de Wetering and Ewing (1999) found that lamprey mortality begins as temperatures reach 28°C, whereas Meeuwig et al. (2002) determined that survival begins to decrease as temperatures reach 22°C. Elevated temperatures may adversely affect lamprey survival through increased metabolic rates and decreased stream microbial activity (van de Wetering and Ewing 1999). However, increased temperatures increase growth rates of juvenile lamprey (Meeuwig et al. 2002), and at various times in the life cycle may benefit juvenile lamprey. In other fish species, increased size has been correlated to increased survival.

3.4.6.2 Dissolved Gas

Very little information is available regarding the effects of dissolved gases on Pacific lamprey. Stone et al. (2002) found that at the reach scale, lamprey presence was positively associated with dissolved oxygen in Cedar Creek, Washington.

3.4.6.3 Chemicals

From the late 1940s through the 1980s, the Oregon Fish Commission used rotenone in basins throughout the state to eliminate non-game species including Pacific lamprey (US vs. Oregon 1997). Kostow (2002) reported that lamprey in Oregon were easy to kill with rotenone. This practice no longer occurs today, but with up to 7 year-classes of Pacific lamprey present in fresh water at any one time, the intentional fish kills of the mid-1900s likely severely impacted Pacific lamprey populations.

Chemical spills in lower Fifteen Mile Creek and in the John Day River in Oregon killed thousands of lamprey, empirical confirmation of the susceptibility of lamprey to localized chemical accidents. Pacific lamprey juveniles likely use lower river basins including the Willamette and Columbia Rivers extensively. These habitats tend to accumulate toxins, and may potentially build up lethal concentrations in the substrate occupied by juvenile lamprey. The Environmental Protection Agency (EPA) recently detected high levels of PCBs in lamprey collected from the Columbia River (Kostow 2002).

3.4.7 Species Interactions

3.4.7.1 Competition

Little is known regarding competition between Pacific lamprey and other species. Scott and Crossman (1973) speculated that Pacific lamprey compete with other Pacific coast lamprey including River lamprey and Western Brook lamprey. However, Stone et al. (2002) showed that Pacific lamprey and Western Brook lamprey spawned in different habitats of Cedar Creek, Washington, with Pacific lamprey preferring larger substrate.

3.4.7.2 Predation

During feeding observations of Stellar sea lions near the mouth of the Klamath River, 82% were on Pacific lamprey. Two sea lions whose stomach contents were examined contained solely Pacific lamprey (Jameson and Kenyon 1977). Roffe and Mate (1984) revealed that the most abundant dietary component of sea lions and seals was Pacific lamprey. Other predators of lamprey have included sperm whale (Pike 1950), fur seals (Hubbs 1967), spiny dogfish and sablefish (Beamish 1980). Lampreys are a valuable source of nutrition for these predators. Caloric values for lamprey range from 5.92-6.34 kcal/g wet weight (Whyte et al. 1993) compared to 1.26-2.87 kcal/g for salmon (Stewart et al. 1983).

Adult Pacific lampreys have been targeted by avian and terrestrial predators as well (Roffe and Mate 1984). Beamish (1980) cites a report of a mink seen with a Pacific lamprey in its mouth.

The author of this report has witnessed the predation on juvenile lamprey by gulls at Willamette Falls. It is generally believed that predation on the juvenile lamprey life stages by salmonids is uncommon. Experiments have indicated that their low incidence in the diet of salmonids may result from secretions from the skin of the lamprey (Scott and Crossman 1973, Pletcher 1963). Juvenile rainbow trout prey on lamprey eggs and larvae. Non-salmonid freshwater piscivores include channel catfish, white sturgeon, northern pikeminnow, minnows, sculpin and logperch (Close et al. 1995). Close et al. (1995) suggested that the presence of abundant outmigrating juvenile lamprey may provide a buffer for predation by other species on juvenile salmon.

3.4.8 Ocean & Estuary Conditions

The abundance of Pacific lamprey may be positively correlated to their food base in the ocean. Recently, ocean conditions have improved as evidenced by increasing runs of salmonids in the Columbia basin as well as on the Oregon and Washington coasts. In addition, harvest of salmonids has been curtailed increasing the potential prey base in the ocean. In 2002, counts of lamprey were the highest seen in recent years at dams on the mainstem Columbia. Since lamprey feed on salmon, it would stand to reason that increases in adult lamprey populations would occur some years after increases in salmonid populations.

Increased ocean productivity and protection of marine mammals have resulted in increased populations of these species which prey on lamprey (Close et al. 1995; BioAnalysts 2000). Also, harvest on non-salmonid species in the ocean may be depleting a portion of the prey base of Pacific lamprey.

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Volume III, Chapter 4
Eulachon

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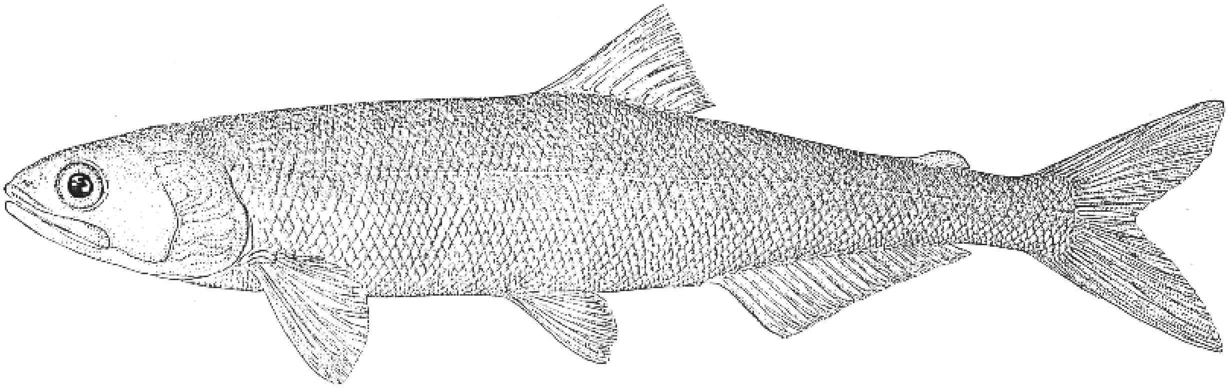


4.0 Eulachon (*Thaleichthys pacificus*)

Eulachon or smelt (*Thaleichthys pacificus*) are a small, anadromous forage fish inhabiting the Northeastern Pacific. Eulachon are a member of the family *Osmeridae*, which comprises several species including another anadromous smelt, the longfin smelt, *Spirinchus thaleichthys*. Eulachon differ from longfin smelt by having shorter pectoral fins (shorter than head length) and fewer gill rakers on the first arch (Wydoski and Whitney 1979). The name *Thaleichthys pacificus* is derived from the Greek words *thaleia* meaning rich, *ichthys* meaning fish, and *pacificus* meaning of the pacific (Hart 1973).

Eulachon fill a unique niche in the Northwest fishing community because of the timing of their runs and value as a food source. British Columbia tribal members even named the eulachon "salvation fish" because eulachon begin returning to rivers during bleak winter months providing sustenance until spring and summer. Eulachon were also called "candlefish" because of the high oil content of eulachon they could be dried and burned like candles (Scott and Crossman 1973).

Eulachon range geographically from Monterey Bay, California, to the Bering Sea and the Pribilof Islands. Eulachon spend most of their adult life in saltwater and little is known about this stage of life. The Columbia River is the site of the largest commercial eulachon fishery; other spawning locations are the Fraser and Nass Rivers in British Columbia. Approximately thirty rivers support eulachon runs in North America (Hay et al. 1997). Before the construction of Bonneville Dam in 1938, eulachon were reported as far upstream as Hood River, Oregon (Smith and Saalfeld 1955). Washington rivers outside of the Columbia River basin with known eulachon spawning runs include the Naselle, Nemah, Wynoochee, Bear, Quinault, Queets and Nooksack rivers (WDFW 2001).

In Washington and Oregon, eulachon support a commercial fishery in the lower Columbia River and tributary rivers: Grays, Cowlitz, Kalama, Lewis and Sandy. Commercial eulachon runs have been recorded in the Columbia River since 1894. Annual harvest in the Cowlitz River, a major tributary, has varied over time from no fish harvested to just over 3,000,000 pounds in 1976 (WDFW 2001).

Commercial production (harvest) of eulachon decreased considerably in the mid-1990s prompting Washington, Oregon, and British Columbia fishery managing agencies to reassess their eulachon management framework and increase research activity (WDFW2001). Commercial eulachon harvest in the main-stem Columbia River was recorded as only 235 pounds in 1994, the smallest harvest since 1935 (Figure 4-1) (WDFW 2001). The decline in eulachon abundance from British Columbia to California has generated more eulachon research than 100 years of commercial fishing (Moffitt et al.2002). Reasons for the decline in eulachon numbers remain unknown, but have been attributed to changes in climate, ocean productivity, and increased bycatch in shrimp trawl commercial fisheries. In 1999–2001, the commercial harvest of eulachon has increased. Eulachon are anadromous and like salmon are susceptible to similar impacts during their spawning cycle. Logging, land development, dredging, predation, water quality and fishing can affect eulachon production and survival rates (WDFW 2001).

Sport harvest of eulachon takes place in tributaries of the lower Columbia River. There is some sport fishing in the mainstem of the Columbia, but the majority of sport fishing takes place in the Cowlitz River. Effort and harvest data are not collected on sport harvest of eulachon. The amount of sport harvest is considered similar to that of the commercial tributary catch (WDFW 2001).

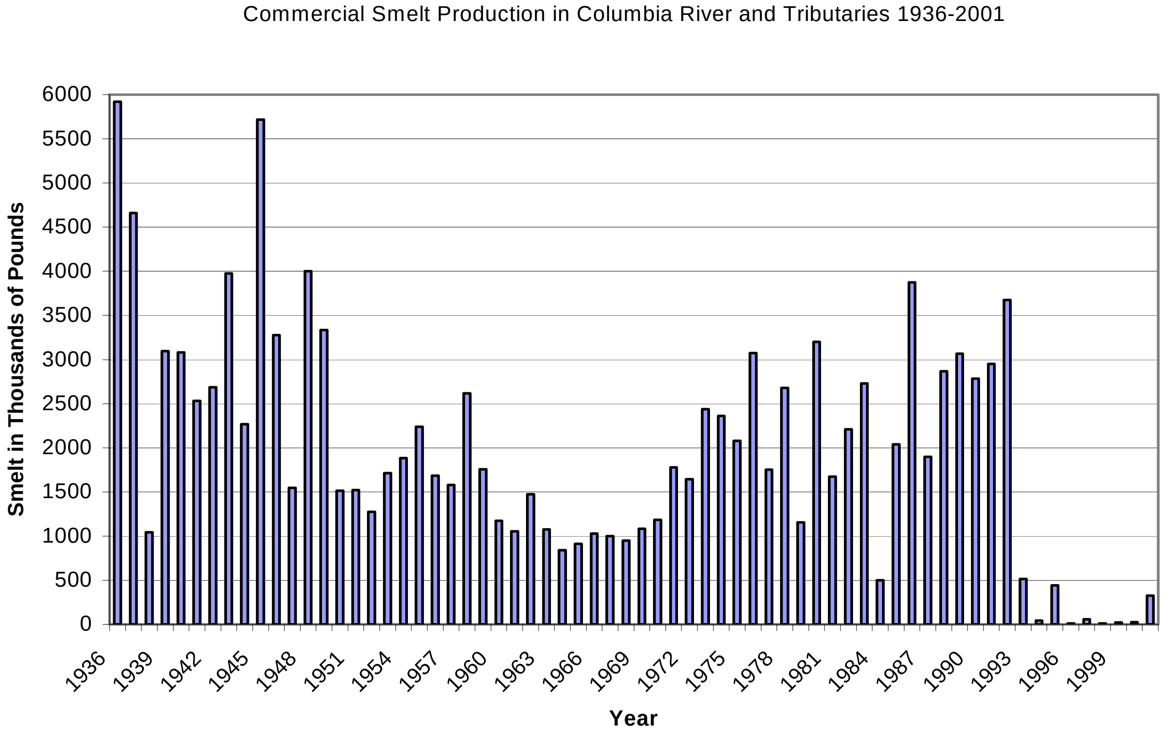
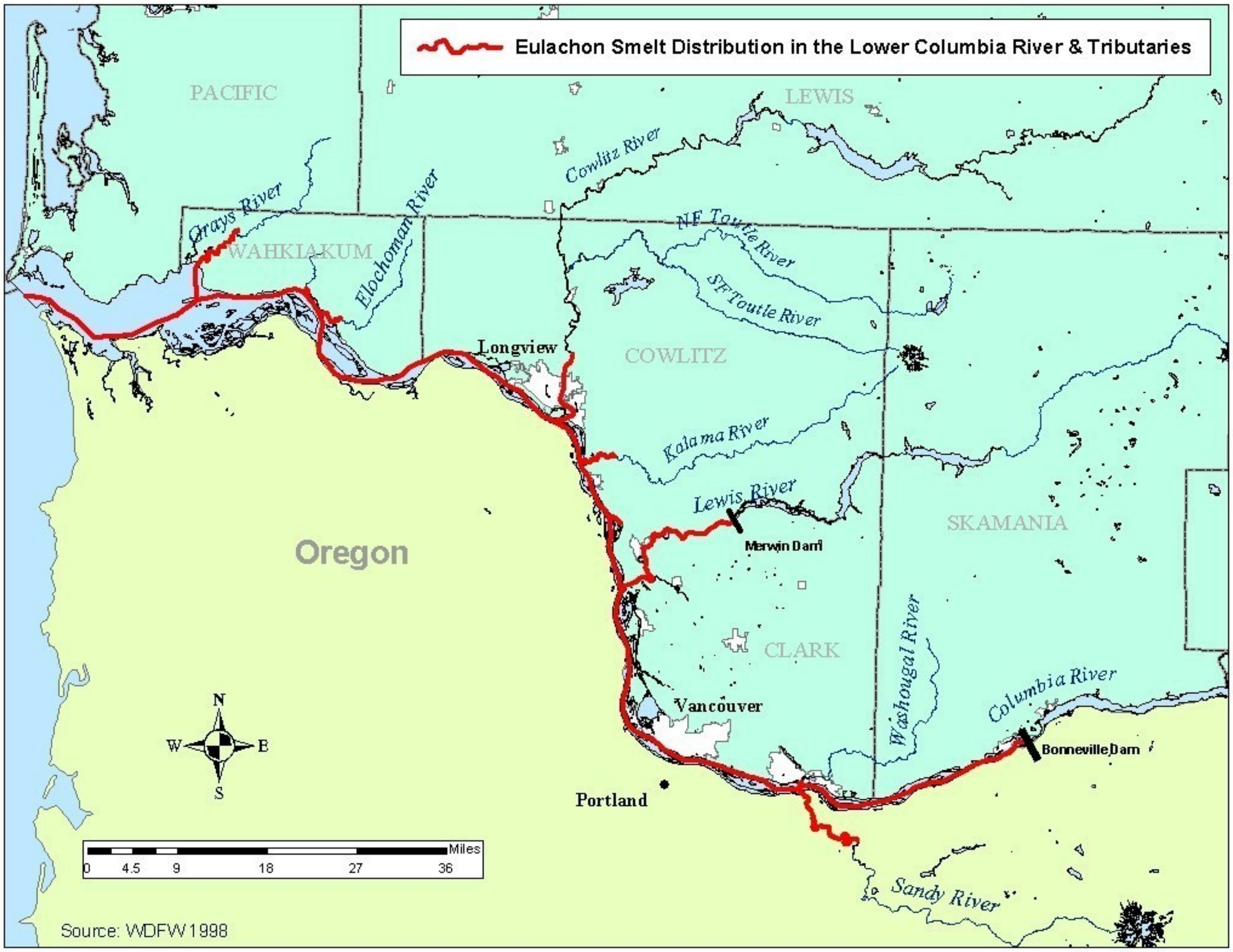


Figure 4-1. Eulachon commercial landings in the Columbia River and tributaries 1936–2001.



Source: WDFW 1998

Figure 4-2. Eulachon distribution in lower Columbia River and tributaries.

4.1 Life History & Requirements

Eulachon are anadromous. They spawn and their eggs hatch in fresh water and grow to maturity in the sea where, as juveniles and adults, they feed mainly on euphasids, a small shrimp-like crustacean sometimes called krill.

As the spawning season approaches, eulachon gather in large schools off the mouths of their spawning streams and rivers. Males usually outnumber the females during the spawning migration. Researchers in a study of the Copper River Delta, Alaska, eulachon population found that between 1998–2002 males composed a mean of 68% of samples (Moffitt et al. 2002).

Eggs are broadcast over sandy gravel bottoms where they attach to sand particles. Newly hatched young are carried to the sea with the current. After three to four years at sea, they return as adults to spawn. After spawning, the majority of eulachon die.

Table 4-1. Annual timetable for eulachon presence in Columbia River and tributaries during individual life stages.

	Eulachon Presence in Columbia River System											
Month	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**
Adult	X	X	X	X	X	X						X
Egg	X	X	X	X	X	X						
Larvae	X	X	X	X	X	X	X					

4.1.1 Spawning Conditions

Eulachon typically enter the Columbia River system from December to May with peak entry and spawning during February and March (WDFW 2001). Eulachon spawn in the main tributaries of the Columbia River and in the mainstem of the Columbia River. Water temperature plays an important role in upstream migration for spawning eulachon. Past studies have shown that the optimum water temperature for upstream migration is 40F (Smith and Saalfeld 1955). The colder the water, the longer the delay for spawning runs. Spawning eulachon enter Northern British Columbia Rivers in March through April or May (Garrison and Miller 1982).

4.1.2 Incubation

Eulachon spawn primarily at night. Eggs are shed, fertilized and abandoned. The average egg size is 1 mm. Each female deposits approximately 17,000 to 60,000 eggs, depending on size of female. (Morrow 1980) Fertilized eggs have an outer membrane, which separates from an inner cover and remains attached at a small area forming a short stalk or peduncle. This peduncle is adhesive and attaches to particles of coarse sand or other river substrate like pea-sized gravel or sticks (Smith and Saalfeld 1955). Eulachon eggs have been observed in water from 8 to 20 feet in depth. Water temperature influences the length of time to hatching. In temperatures of 6.5-9.0°C, eggs will hatch in about 22 days. At colder temperatures of 4.4-7.2°C, as found in the Cowlitz River, eulachon eggs will hatch in 30 to 40 days (Garrison and Miller 1982).

4.1.3 Larvae & Juveniles

Newly hatched larvae are transparent and 4-7 mm in length. They have poor swimming ability and migrate downstream at the mercy of river currents. Eulachon fry have been recorded to within 20 miles seaward of the Columbia River mouth. The result of several plankton hauls conducted in 1946 showed no fry had developed beyond yolk-sac stage; therefore, it is probable no feeding occurs in fresh water during outbound migration (Smith andSaalfeld 1955). After the yolksac is depleted eulachon will feed on pelagic plankton. Stomach samples of juvenile eulachon contained euphausiids (Barraclough 1964).

4.1.4 Adult

Eulachon spend the majority of life in salt water and little is known about this saltwater phase. Typically, eulachon return to spawn at 3-5 years of age. Recent studies indicate that age proportion of spawning eulachon can vary. In the Copper River Delta, Alaska, spawning eulachon ages varied as follows; in 1998, 89.4% were age 5; 9.5% age 4 and 0.3% age 3. In year 2000, 3.3% were age 5; 48.2% year 4; 48% age 3. In 2001, 1.3% were age 5; 42% age 4; 55% age 3 years. In 2002, 1.2% were age 5; 96.1% age 4; 1.4% age 3. Data for 1999 was not included due to insufficient sampling trials (Moffitt et al. 2002).

Otoliths have been collected from 1987–2000 runs in the Lower Columbia River and its tributaries. Only those otoliths from 1992–1998 have been examined and preliminary readings indicate that Columbia River eulachon returns (1992–1998) comprised 26-66% age 3 fish, 28-49% age 4 fish and age 5 fish made up 5-25% of the run (WDFW2001) (Table 4-2).

Table 4-2. Age composition of Columbia River eulachon, 1992-98 (WDFW 2001)

	Age Composition			Average Length (mm) by Age		
Year	**3**	**4**	**5**	**3**	**4**	**5**
1992	26%	49%	25%	169.4	189.3	190.8
1993	39%	39%	22%	164.4	159.4	149
1994	66%	28%	6%	178.7	177.4	164.8
1995	41%	46%	13%	171.3	181	197.5
1996	56%	39%	5%	168.5	179.4	170.2
1997	60%	33%	7%	165.4	170.5	162.8
1998	56%	37%	7%	173.5	181.5	175.9
Average	*49%*	*39%*	*12%*	*170.2*	*176.9*	*173.0*

Eulachon feed on plankton in salt water, but stop feeding when returning to fresh water. The homing instinct of eulachon (returning to birth streams) has not been established. Eulachon larvae are flushed out to sea shortly after hatching leaving little time for imprinting (Moffitt et al.2002). Returns to tributary rivers can vary from year to year, with some rivers having no return run of eulachon. Most, but not all, eulachon die after spawning. A few live spent adult eulachon have been observed downstream of spawning locations (Garrison and Miller1982). Whether this indicates long-term survival following spawning is unknown.

Adult eulachon are 15 to 20 cm long with a maximum recorded length of 30 cm. They are a brown to dark bluish color on the back, fading to silvery white on the belly. Males are slightly longer and heavier then females (Morrow 1980). The sex ratio of spawning adults is an average of 4.5 males to 1 female in the Columbia River and tributaries supporting eulachon. The male to female ratio has been recorded as high as 10.5 males to 1 female in the Cowlitz River (Smith and Saalfeld 1955).

Eulachon rear in near-shore marine areas from shallow to moderate depths. At sea juvenile eulachon may grow from 23 mm to lengths of 46-51 mm within eight months (Barraclough 1964). Eulachon will move into deeper water, up to depths of 625 m, as they grow (Allen and Smith 1988). Eulachon are an important link in the food chain between zooplankton and larger organisms. Small salmon, lingcod and other fish feed on small larvae near river mouths. As eulachon mature they are eaten by many predators including; halibut, cod, dogfish, sharks, seals, sea lions, porpoise, finback whales, killer whales, gulls, ducks and other sea birds (Garrison and Miller1982). Their value to the marine system is due to their high energy fat content and large biomass. Eulachon are high in oil (total lipid), ranging from 16.8% to 21.4% (Payne et al. 1999).

4.1.5 Movements in Fresh Water

Movements in freshwater are restricted to anadromous spawning cycles. Adult eulachon enter freshwater to spawn from December to March and the young migrate downstream shortly after hatching.

A study of larval distribution in the lower Columbia River found that eulachon larval density was greater in the lower portion of the water column (Howell et al. 2002). Larval densities in bottom samples were greater then densities in mid-water and surface samples. Mechanisms controlling eulachon larval distribution are not clearly understood. Plankton net sampling was concentrated in the Columbia River downstream of the Cowlitz River with one sampling location above the confluence of the Cowlitz and Columbia Rivers. Larval density varied throughout the season, but larval numbers peaked between April 2 and April 18. (Howell et al. 2002). The purpose of the study was to evaluate potential effects of proposed channel-deepening operations in the Columbia River. (More on this study will be covered under 4.4 Factors Affecting Population Status)

An associated study by the same group (Romano et al.2002) found that in 2001 eulachon spawning habitat within the Columbia River was larger then previously assumed by earlier studies (Smith and Saalfeld 1955). Eulachon larvae were found between Price Island and the mouth of the Kalama River (Howell et al. 2002). Previous larval distribution studies did not find any eulachon larval above the Cowlitz River (Smith and Saalfeld 1955).

During the 2001 study, adult eulachon migrated upstream to the Bonneville Dam and entered all major lower Columbia River tributaries (Grays, Elochoman, Kalama, Lewis and Sandy). Observations by researchers conclude that the strength of a eulachon spawning run varies throughout the course of a single season (Howell 2002).

4.1.6 Ocean Migration

Information about eulachon ocean migration is minimal. The current data about eulachon marine habits comes from information gathered from the Canadian shrimp trawl fisheries. Eulachon are commonly caught as bycatch in marine shrimp trawl fisheries. Harvest and mortality rates due to handling are unknown. In the British Columbia shrimp trawl fisheries, the eulachon bycatch has been estimated as high as 27% of the biomass caught (Hay et al.1999).

In a study measuring the Southern British Columbia offshore biomass, there was a strong positive correlation ($r^2=0.34, P<0.01$) between the offshore biomass and Columbia River eulachon catches. This study surmises that it seems probable most eulachon captured offshore of Vancouver Island spawn in the Columbia River (Hay et al.1999).

4.2 Population Identification & Distribution

Although eulachon are found throughout the northeast Pacific Ocean, genetic relationships among populations are unclear. Eulachon predecessors are believed to have survived through historical glacial changes in North America. Over the past two million years (Pleistocene Epoch) most of North America was covered with ice that advanced and retreated over the land through cycles of cooling and warming (McPhail and Lindsey 1970).

Freshwater fish were strongly affected by advancing glaciers. Species either became extinct or moved into ice-free glacial refugia. Anadromous fish, like eulachon, which require fresh water to spawn, would also have used the ice-free refuge areas as habitat in the cool climate (McPhail and Lindsey1970). The most recent cold period in North America was the Wisconsinian glacial period. Evidence from mtDNA suggests that populations of eulachon are derived from a single Wisconsinian glacial refuge, during the Pleistocene Epoch. While many private mtDNA haplophytes were found, over 97% of the total variation was found within populations. Mclean et al’s analysis of eulachon mtDNA suggests that there is little genetic difference among eulachon from distinct freshwater locations and that eulachon might be considered as one meta-population.

Eulachon is the only member of the genus *Thaleichthys*. There are no other species or subspecies having a different life history.

4.3 Status & Abundance Trends

Eulachon are listed as a state candidate species on WDFW's SOC list. According to WDFW Policy M-6001, a species will be considered for designation as a state candidate if sufficient evidence suggests that its status may meet the listing criteria defined for state endangered, threatened, or sensitive.

In 1999, the NMFS received a petition to list the Columbia River populations of eulachon as an endangered or threatened species and to designate critical habitat under the ESA. The NMFS determined the petition did not present enough substantial evidence to warrant the listing of eulachon. (Fed Reg 64(226)).

4.3.1 Abundance

No quantitative stock assessment of eulachon is conducted. The best available long-term data on Columbia River eulachon returns are historical commercial landings in the Columbia River and its tributaries. Unfortunately, commercial landings are a poor index of eulachon run size because the economic market can dictate the harvest amount. Commercial fishing may cease or slow down once the market has been saturated and prices of eulachon decreased (WDFW 2001).

In 1994, WDFW initiated eulachon larval sampling in the Cowlitz River and other lower Columbia River tributaries. The long-term objective is to develop a relational index of eulachon production in the lower Columbia basin that can be used to assess annual variation in spawning and recruitment. Larval sampling conducted from 1994 through the present (2003) still needs further evaluation through a broad range of run sizes before being used as an assessment tool (WDFW 2001).

4.3.2 Productivity

Currently, there is no accurate measurement of eulachon productivity. Researchers believe eulachon abundance is influenced by ocean productivity within the first year at sea. Developing reliable eulachon forecasting techniques may included examining ocean productivity indices such as Southern Oscillation Index, sea surface temperature profiles and Oyster Condition Index. Another useful relationship to investigate for evidence of eulachon productivity is the survival of other anadromous species with a 3-year spawning cycle (WDFW 2001).

4.3.3 Supplementation

There are no supplemental hatchery programs for eulachon in the Columbia River or its tributaries. Experimental artificial propagation of eulachon has been conducted to observe the influences and water temperature and substrate on eggs and larval development (Howell et al 2002; Smith and Saalfeld 1955).

4.3.4 Harvest

The harvest of eulachon in the Columbia River mainstem is regulated by Washington and Oregon within the guidelines of the Columbia River Compact. The states must mutually approve the fishing regulations for eulachon. Sport and commercial fishing in the Columbia River tributaries is regulated by the individual states.

4.3.5 Commercial Fishery

Washington commercial anglers are required to have a Columbia River smelt license to fish commercially for eulachon (RCW 77.65.200 (1)(g)). Oregon does not require a separate smelt license, but anglers must have a commercial fishing license and commercial fishing boat license (WDFW 2001).

Columbia River-caught eulachon are sold for bait in the sport sturgeon fisheries and also as a fresh food fish. Eulachon fishing in the Columbia River drops off after the fish have entered the Cowlitz River and other tributaries. Typical commercial fishing gear used are the 2-inch bobber gill nets and, not as commonly, diver gill nets and otter trawl. Trawl vessels greater than 32 feet are prohibited upstream from Tongue Point. Commercial eulachon fishing is limited to dip nets in the tributaries (WDFW 2001).

4.3.6 Sport Fishery

The majority of sport fishing for eulachon takes place in the tributaries using dip net gear, although the mainstem is also open for sport fishing. In general, both states manage the tributary fisheries consistent with the mainstem fisheries (WDFW 2001). Most sport harvest is in the Cowlitz River. Neither Washington nor Oregon requires an angling license for eulachon. There are restrictions on gear. In Washington the size of the dip net bag frame must not exceed 36 in (WAC 220-16-028). Oregon has no bag frame restriction. Washington's 2003 sport regulations allowed a maximum daily bag limit of 20 pounds from February 12–March 31, up from the standard 10-pound bag limit early in the year. Oregon's eulachon sport fishing regulations for 2003 allowed a 25-pound limit per day (WDFW 2001).

4.3.7 Tribal Fishery

Native Americans have fished for eulachon in the Columbia River tributaries for centuries. At present, members of the Yakama Nation fish for eulachon for subsistence purposes in the lower Cowlitz River using dip net gear. The annual catch of eulachon by Yakama tribal members is minimal (WDFW 2001).

4.4 Factors Affecting Population Status

4.4.1 Water Development

Hydropower development on the Columbia River has decreased the available spawning habitat for eulachon. Prior to the completion of Bonneville Dam, eulachon were reported as far upstream as Hood River, Oregon (Smith and Saalfeld 1955). Similar developments on tributary rivers, like the Cowlitz, also may have decreased spawning habitat.

4.4.2 In-Channel Habitat Conditions

Eulachon freshwater spawning habitat can be affected by in-channel conditions. Eulachon are broadcast spawners with highly adhesive eggs that attach to coarse sandy substrates. Dredging has the potential to impact adult and juvenile eulachon (Larson and Moehl 1990). In a 2001 study, researchers found that the sand wave movements in near-shore areas of dredging operations in the lower Columbia River made the substrate too unstable for the incubation of eulachon eggs. Recommendations presented suggested that channel-deepening operations be scheduled to avoid eulachon spawning areas during peak spawning times (Romano et al. 2002). The same recommendations have been echoed in the Washington and Oregon Eulachon Management Plan concerning dredging activities in tributaries to the Columbia River. The plan also recommended further investigation into the theory that the development by the Corps of the Sediment Retention Structure on the Toutle River may have caused poor eulachon returns in the Cowlitz River during 1994–99 (WDFW 2001).

4.4.3 Water Quality

Information concerning the effects of contaminants on eulachon remains minimal. Eulachon have been shown to be sensitive to pollutants in fresh water (Smith and Saalfeld 1955). There also is some evidence that, because of their high lipid content, eulachon may be susceptible to accumulating contaminants during spawning runs. But, this last idea has been refuted based on the fact that eulachon do not spend much time in fresh water either as adults or juveniles (WDFW 2001).

4.4.4 Species Interactions

Predators and scavengers accompany large runs of eulachon as they enter the rivers to spawn. The sight of many birds fishing for eulachon is not uncommon on tributary rivers. Avian predators include bald eagles, mergansers, cormorants and eight species of gulls (WDFW 2001). Eulachon have a high energy density from being extremely high in lipids. Eulachon oil is also comparatively high in vitamin A and iron. The effort exerted by predators to capture eulachon is relatively low because eulachon are weak swimmers and concentrate in low-velocity waters. This combination of attributes makes eulachon an energy efficient meal for predators and scavengers (Marston et al. 2002).

The California sea lion, Northern sea lion, Harbor seal, Harbor porpoise, and Dall’s porpoise also prey on Lower Columbia River eulachon. In a study of gastrointestinal contents of stranded marine mammals in the Columbia River estuary, eulachon made up 43.8% (by occurrence) in California sea lions and 40% (by occurrence) in the stomach contents of Harbor seals. By comparison, Pacific herring (*Clupea pallasi*) composed 12.5% and 14% stomach contents by occurrence, respectively (Jeffries 1984).

A white sturgeon (*Acipenser transmontanus*) feeding study in the lower Columbia River resulted in the following information regarding predation on eulachon eggs. Two size classes of white sturgeon from two separate river locations were examined for the feeding study. Size class I measured 5.6-13.7 in (144-350 mm) FL (fork length) and size class II measured 13.7-28.5 in (351-724 mm) FL. Stomach content samples were taken from sturgeon at RM 95 (RKm 153) (Woodland) and RM 131 (RKm 211) (Skamania). Summary numbers represent percentage of total Index of Relative Importance (%IRI) during May–June 1988.

Table 4-3. Percent of total index of relative importance for eulachon eggs observed in white sturgeon stomach samples (McCabe et al. 1993).

Location	Size I (144-350mm)	Size II (351-724mm)
Woodland	2%	12%
Skamania	25%	51%

4.4.5 Ocean & Estuary Condition

Juvenile and adult eulachon spend most of their life at sea; it is unknown for what portion of their lives or how long juveniles stay in estuaries before migrating further seaward. Information about the impacts ocean and estuary conditions have on eulachon remains limited. Larval sampling has mainly taken place in fresh water. It is believed that young larvae in estuaries and near-shore ocean areas are sensitive to marine pollution and runoff from agriculture and urbanization. If conditions in a river are not right, eulachon may choose a different stream for spawning or return another year (PSMFC 1996).

4.5 Inventory & Assessment of Existing Management Plans

4.5.1 Washington & Oregon

Washington and Oregon jointly regulate commercial and sport eulachon fishing in the mainstem of the Columbia River. Recreational and commercial fishing in the Columbia River tributaries are managed by the individual states. To meet management needs, each state regulates the tributary fisheries consistent with the mainstem fisheries (WDFW 2001).

The Joint State Eulachon Management Plan, developed in 2001, is intended to guide eulachon fishing regulations in the Columbia River basin. The function of the management plan is to provide for three levels of fishing based on:

- parental run strength as indexed by commercial and sport fisheries data,
- juvenile production as indexed by larval sampling data, and
- estimates of ocean productivity as indexed by environmental measures and the abundance of other fish species.

These fishing regulations would be in effect through the January to March timeline. Any adopted fishing level may be modified in-season based on data collected from sport or commercial fisheries (WDFW 2001).

Level one fisheries are the most conservative level with a presumed harvest rate of 10% or less based on indications of a poor run or uncertainty of run strength. Under the level one fisheries commercial and sport fishing would be limited to one 12–24 hour fishing period per week for the Columbia and Cowlitz Rivers. The purpose of the level one fisheries is to develop a fishery database and collect information of the variability of eulachon runs while minimizing the risk of overexploiting the run.

Level two fisheries are recommended when fishery data indicates a promising abundance in spawning return, yet it is unknown whether the run is moderate or strong. Commercial and sport fishing would be open two to three days per week in the Columbia and Cowlitz Rivers. Depending on the level of abundance in these rivers, consideration would be given to opening commercial and sport fishing in other tributaries of the Lower Columbia River. Fishery monitoring data would be used to decide if an increase to level three or a decrease to level one is warranted (WDFW 2001).

Level three fisheries are recommended when there are very positive indicators of strong abundance and productivity and a very low risk of overexploitation. Commercial fishing would be open up to four days a week in the Columbia River and all tributaries. Sport fishing would be open in all tributaries four to seven days a week. Daily bag limits could be increased from 10 pounds per person to 15 to 25 pounds per person under the level three fisheries plan (WDFW 2001). In the current fishing year (2003) eulachon fishing in the Columbia River and tributaries was set at the level three fisheries. The daily limit was increased to 20 pounds per person and fishing was open 7 days a week through March 31, 2003 (WDFW News Release).

The conservation policy guidelines for the Washington and Oregon Eulachon Management Plan incorporates the following; use of a precautionary approach to resource management, maintain healthy populations of eulachon while assuring the integrity of the ecosystem and habitat, and to consider best scientific information while striving to improve the information base of eulachon (WDFW 2001).

4.5.2 Yakama Nation

Management plans for a tribal subsistence fishery for eulachon on the Cowlitz River and other Washington tributaries is being coordinated annually between members of the Yakama Nation and WDFW. The annual plan specifies that smelt taken by Yakama Nation members will be used for subsistence purposes only and may not be sold commercially. The parties will consult to determine the appropriate levels of subsistence fishing based on abundance and conservation needs (WDFW 2001).

4.5.3 British Columbia Eulachon Fishery

The Department of Fisheries and Oceans of Canada (DFO) has the Pacific Region Eulachon Integrated Fisheries Management Plan (IFMP) as a tool for regulating eulachon fisheries in British Columbia. Information to be considered in the management of Columbia River eulachon is that the IFMP includes an annual offshore index of eulachon biomass on the West Coast of Vancouver Island. These juvenile eulachon are believed to be both Fraser and Columbia River stock (DFO 2002).

4.6 Inventory & Assessment of Existing Restoration & Conservation Plans

The Washington and Oregon Eulachon Management Plan makes these recommendations for further conservation and research into Columbia River eulachon runs.

- Initiation of an observer program to determine eulachon bycatch in Washington and Oregon marine trawl shrimp fisheries.
- Consistent implementation of fishery monitoring and larval sampling activities to assess eulachon abundance and productivity.
- Evaluate and utilize abundance forecasting techniques. Abundance of other anadromous species and forage fish may provide insight into eulachon abundance.
- Define and characterize the critical habitats of eulachon, specifically understanding spawning areas and physical factors that affect freshwater survival.
- Establish one or more spawning sanctuaries on the Cowlitz River, which is the largest spawning tributary in the Lower Columbia River.

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Volume III, Chapter 5
Northern Pikeminnow

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5.0 Northern Pikeminnow *(Ptychocheilus oregonensis)*

In Washington, the northern pikeminnow *(Ptychocheilus oregonensis)* is found in the Columbia River system and coastal and Puget Sound drainages (Wydoski and Whitney 1979). The species fares well in stream, river, and lake-like habitats, and has flourished in the mainstem Columbia River and its many tributary systems following development and varying land uses (Parker et al. 1995; NRC 1996).

Intensive predation by northern pikeminnow on juvenile Pacific salmon *Oncorhynchus* spp. has been well-documented throughout the lower Columbia River basin (Rieman et al. 1991; Vigg et al. 1991; Ward et al. 1995; Ward et al. 2002), where extensive hydropower development has greatly increased the vulnerability of migrating juvenile salmonids to predation (Raymond 1979; Rieman et al. 1991). Concern about this predation led to the development of a large-scale management program for northern pikeminnow (Beamesderfer et al. 1996; Friesen and Ward 1999; Ward et al. 2002).

5.1 Distribution

The northern pikeminnow is a cyprinind native to the Pacific slope of western North America from Oregon north to the Nass River in British Columbia (Wydoski and Whitney 1979, Simpson and Wallace 1982). A map is provided in Figure 5-1.

Northern pikeminnow have successfully evolved in a range of dynamic lentic and lotic ecosystems and successfully adapted to their varied habitat conditions. Their plasticity allowed them to flourish despite construction and operation of the Columbia Basin hydropower system (NRC 1996). Beamesderfer (1992) attributed the widespread distribution and resiliency of northern pikeminnow to their relatively broad spawning and rearing habitat requirements. Furthermore, the wide range of prey types available in the lower Columbia River (Poe et al. 1991) appears suitable to support a trophic generalist. Parker et al. (1995) reported considerable variation in life history parameters of northern pikeminnow in the mainstem Columbia and Snake Rivers, further supporting the species' adaptability.

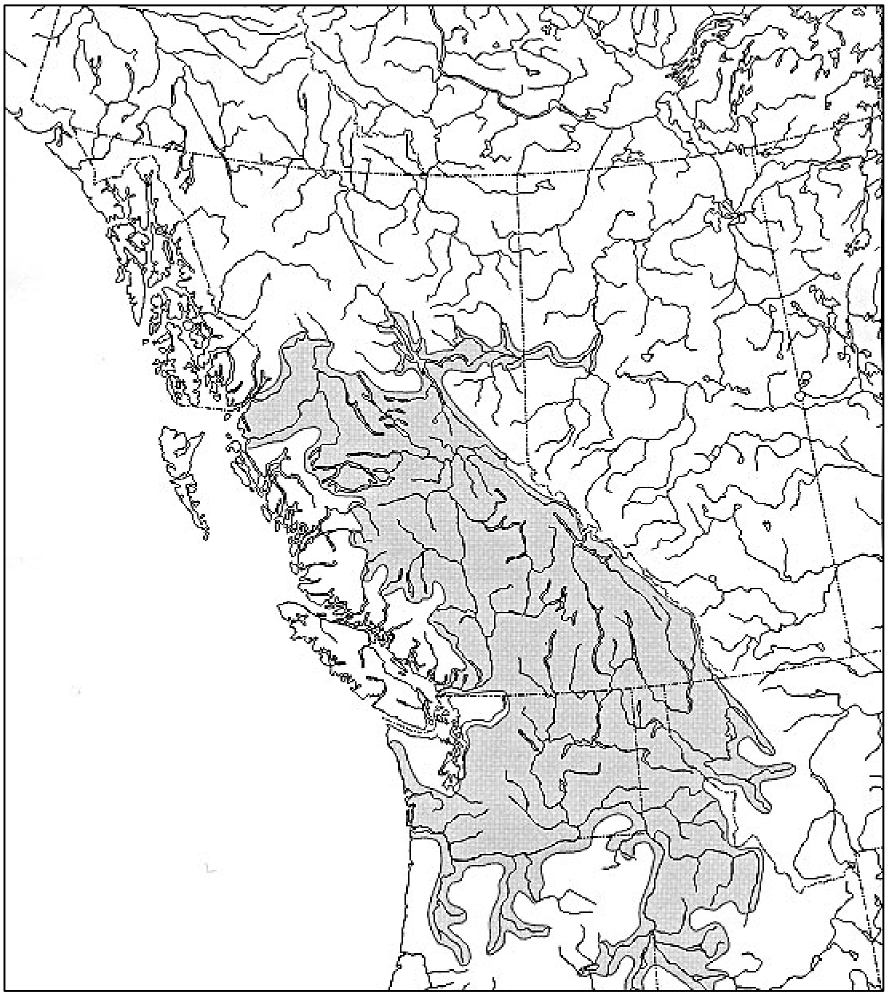

Figure 5-1. Geographical range of northern pikeminnow.

Overall objectives of the program were: 1) determining the significance of predation in Columbia River reservoirs by indexing predator abundance and comparing the index with consumption (of what) indices, 2) implementing a predator control plan, beginning with a test fishery in John Day Reservoir in 1990, and 3) evaluating the predator control program.

5.2 Life History Characteristics

5.2.1 Size & Mortality

Northern pikeminnow are large, long-lived, slow-growing predaceous minnows *(Cyprinidae)* whose unexploited populations are typically dominated by large, older individuals. In the Columbia River, maximum fork length, weight, and age are approximately 23½ in (600 mm), 5½ pounds (2.5 kg), and 16 years; annual mortality rates were reported to range from 12-31% (Rieman and Beamesderfer 1990; Parker et al. 1995). However, the maximum age of 16 years may be an underestimate based on possible underaging (Dave Ward, ODFW, personal communication). Individuals 15 in (380 mm) in length and greater constituted 12-59% of the population with FL > 9 3/4 in (250 mm) (Parker et al. 1995). Ward et al. (1995) reported that differences in life history trait expressions of northern pikeminnow among reservoirs and between free-flowing areas and impounded reaches of the Columbia and Snake Rivers underscore their ability to adapt.

Sexual maturity occurs at sizes of 8-14 in (200-350 mm) and corresponding ages of 3–8 years, with males typically reaching initial maturity before females (Beamesderfer 1992; Parker et al. 1995). Spawning generally occurs during June and July in large aggregations that broadcast eggs over clean rocky substrate in slow-moving water at a range of depths in rivers, lake tributaries, lake stream outlets, and shallow and deep littoral areas (Beamesderfer 1992). Wydoski and Whitney (1979) reported spawning over gravel areas in stream and gravel beach areas in the lake. Parker et al. (1995) reported that individual fecundity averaged about 25,000 eggs/female, whereas Wydoski and Whitney (1979) published a fecundity range of 6,700 to 83,000 eggs per female. They also reported that eggs hatch in 7 days at 65°F water, and that the young become free swimming within 14 days.

The diet of northern pikeminnow varies with their size (Ricker 1941; Falter 1969; Olney 1975; Buchanan et al. 1981). In the Columbia River, invertebrates dominate the diets of northern pikeminnow that are smaller than 11.8 in (300 mm) FL, with fishes and crayfish increasing in importance as fish size increases (Thompson 1959; Kirn et al. 1986; Poe et al. 1991, 1994). Salmonids, sculpins (*Cottus* spp.), trout perch *(Percopsis transmontana)*, and suckers *(Catostomous* spp.*)* are common prey items of northern pikeminnow (Poe et al. 1991). Salmonids are generally an important diet item only for large, old northern pikeminnow (Vigg et al. 1991), and the consumption rate of juvenile salmonids increases exponentially as the size of the northern pikeminnow increases (Beamesderfer et al. 1996). Consumption rates of juvenile salmonids by northern pikeminnow correlate positively with how abundant salmonids are; in other words, the more salmonids there are, the more the northern pikeminnow eat until the pikeminnow reaches satiation (Thompson 1959; Buchanan et al. 1981; Poe et al. 1991; Vigg et al. 1991; Tabor et al. 1993; Henchman 1986; Vigg 1988; Petersen and DeAngelis 1992).

Newly-emerged larval northern pikeminnow in the Columbia River drift downriver in the nighttime hours of July. The highest overall densities of drifting northern pikeminnow larvae were collected below Bonneville Dam. In reservoirs, the highest densities of drifting larvae occurred in tailrace areas. The period of larval drift was brief, with larvae recruiting to shallow sand or fine sediment shoreline areas to rear. Age-0 northern pikeminnow rearing in littoral habitats of the upper John Day Reservoir had significantly greater growth and lower mortality in June–September 1994, a year with low flows, abundant instream vegetation, and high near-shore water temperatures.

5.2.2 Population Dynamics & Demographic Risk

Population is affected by competition from other species as well as from other members of their own species. While northern pikeminnow represent the only native piscivorous salmonid predators in Columbia River reservoirs, numerous non-native predatory fish species have been introduced into the Columbia Basin (e.g. walleye, smallmouth bass *Micropterus dolomeui,* and channel catfish *Ictalurus punctatus*).

Beamesderfer et al. (1996) found a negative correlation between concurrent year classes of walleye and northern pikeminnow, and suggested that walleye might influence (reduce) northern pikeminnow numbers by predation. Furthermore, interactions and population dynamics among native and non-native fish species and subsequent ecological responses are difficult to predict accurately (Beamesderfer et al. 1996). Potentially inextricable changes in species abundance due to the program to remove northern pikeminnow may further confound such investigations.

5.3 Status & Abundance Trends

5.3.1 Abundance

Northern pikeminnow abundance in the Columbia River downstream from its confluence with the Snake River is highest in the approximately 186 miles (300 km) from the estuary to the Dalles Dam (2,580-3,020 fish/km), and decreases significantly in the 100 miles (161 km) from the Dalles Dam to McNary Reservoir (550-690 fish/km; abundance is shown in Figure 5-2.

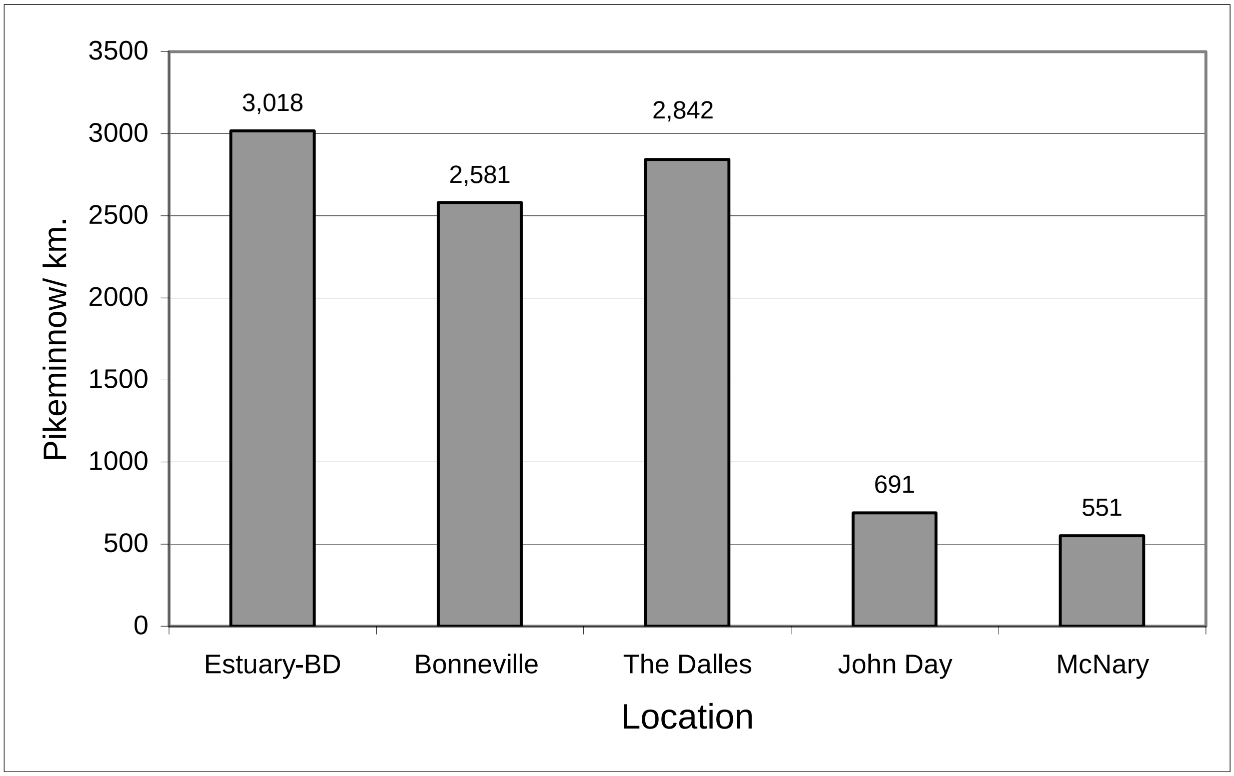

Figure 5-2. Estimated abundance of northern pikeminnow in the Columbia River downstream from its confluence with the Snake River (data from Beamesderfer et al. 1996).

However, a longitudinal trend in abundance was not noted in the four lower Snake River reservoirs. In the lower Snake River, northern pikeminnow were most abundant in Little Goose and Lower Monumental reservoirs (1,065 and 1,000 fish/km respectively) and least abundant in Ice Harbor Reservoir (255 fish/km), as shown in Table 5-1. The longitudinal northern pikeminnow abundance trend may be supported by similar trends of increasing food availability and habitat suitability in the same downstream orientation.

Table 5-1. Projected abundance of northern pikeminnow based on 1983–86 mark-recapture estimates in John Day Reservoir (Beamesderfer and Rieman 1991).

Location	Distance (km)	Fish km	Northern pikeminnow abundance (000s)
Estuary to Bonneville	224	3,018	676
Bonneville Reservoir	74	2,581	191
The Dalles Reservoir	38	2,842	108
John Day Reservoir	123	691	85
McNary Reservoir	98	551	54
Ice Harbor Reservoir	51	255	13
Lower Monumental Reservoir	46	1,065	49
Little Goose Reservoir	60	1,000	60
Lower Granite Reservoir	85	341	29
Total	*NA*	*NA*	*1,265*

Figure 5-3. Estimated abundance of northern pikeminnow upstream from the confluence of the Columbia and Snake Rivers (data from Beamesderfer et al. 1996).

5.3.2 Productivity

Northern pikeminnow populations in the Columbia River basin do not appear to be facing any demographic risks. On the contrary, natural production appears to be strong, as reflected by the continuing rigorous prosecution of the program for their removal. The program (Ward et al. 2002) represents the most robust management activity affecting the northern pikeminnow population. Although millions of pikeminnow have been removed from the Columbia and Snake River populations, the need for removal and control continues, indicating their high productivity in these areas. The management fisheries harvested 201,164 northern pikeminnow 200 mm FL in 2002 (Takata and Friesen 2003). Beamesderfer (1992) attributed the widespread distribution, resiliency, and productivity of northern pikeminnow to their relatively broad requirements for spawning and rearing habitat.

As large fish have been removed, the size structure of northern pikeminnow populations has decreased, and no compensation in reproduction or growth has been observed (Knutsen and Ward 1999; Zimmerman et al. 2000). Similarly, no trends of increased predation, reproduction, or growth of walleye or smallmouth bass have been observed (Ward and Zimmerman 1999; Zimmerman 1999; Friesen and Ward 2000). ODFW expects to continue to collect information on population dynamics of northern pikeminnow, walleye, and smallmouth bass along with predation indexing.

5.3.3 Harvest

Since 1990, when a focused pikeminnow control and management program was implemented, over 1.7 million northern pikeminnow have been removed from the lower Columbia and Snake Rivers, with annual exploitation since 1991 averaging over 12% of fish >250 mm FL (Table 5-2). Evaluating the program involves monitoring how many and what proportion of northern pikeminnow are harvested annually for each fishery, and how their removal affects the rate at which they take other fish. The program:

- compares predation indices before and after sustained implementation of the program,
- describes the response of northern pikeminnow to sustained removals, and
- describes the response of other predators (walleye and smallmouth bass) to sustained removals of northern pikeminnow.

Table 5-2. Catch and exploitation rate in the Northern Pikeminnow Management Program, 1990–2001. Includes only fish >250 mm FL (minimum size changed to approximately 200 mm FL in 2000).

Year	Sport Reward		Dam Angling		Site Specific		Other	
1990	4,681	(—)	11,005	(—)	—	(—)	1,648	(—)
1991	153,508	(8.5%)	39,196	(2.2%)	—	(—)	7,366	(—)
1992	186,095	(9.3%)	27,442	(2.7%)	—	(—)	8,766	(—)
1993	104,536	(6.8%)	17,105	(1.3%)	—	(—)	3,460	(—)
1994	129,384	(10.9%)	15,938	(1.1%)	9,018	(1.2%)	—	(—)
1995	199,788	(13.4%)	5,397	(0.3%)	9,484	(1.9%)	—	(—)
1996	157,230	(12.1%)	5,381	(0.3%)	6,167	(0.5%)	—	(—)
1997	119,047	(8.8%)	3,517	(0.1%)	2,806	(0.5%)	—	(—)
1998	108,372	(11.1%)	3,175	(0.1%)	3,035	(0.3%)	—	(—)
1999	114,687	(12.5%)	3,559	(0.0%)	1,604	(0.1%)	—	(—)
2000[a]	121,519	(11.9%)	423	(0.0%)	554	(0.0%)	—	(—)
2001[a]	153,577	(16.0%)	2,751	(0.0%)	518	(0.0%)	—	(—)
2002	200,533				712			

[a] Although minimum size in the sport-reward fishery was decreased to approximately 200 mm FL (9 in total length) in 2000, for comparison purposes, totals for 2000 and 2001 reflect catch of fish >250 mm only. Catch of fish 200-250 mm FL totaled 67,945 in 2000 (6.6% exploitation), and 87,317 (10.6%) in 2001.

As shown in Table 5-2, annual exploitation rates since 1991 average over 12% of northern pikeminnow >250 mm FL, and the minimum goal of 10% exploitation has been met or exceeded in 9 of 11 years. All fisheries target large, piscivorous northern pikeminnow, with mean fork lengths of just under 10 in (346 mm) in the sport-reward fishery, 15 ¾ in (401 mm) in the dam-angling fishery, and 16 in (409 mm) in the site-specific fishery (Friesen and Ward 1999).

Recommendations from a review and audit of the Northern Pikeminnow Management Program (NPMP) (Hankin and Richards 2000) included decreasing the minimum size of northern pikeminnow eligible for reward from 11 inches to 9 inches total length (similar to a reduction from 250 mm to 200 mm FL), and this change was made in 2000. Exploitation rate of fish 9-11 inches total length was 6.6% in 2000 and 10.6% in 2001, resulting in overall exploitation estimates of 10.9% and 15.5%.

5.4 Factors Affecting Population Status

5.4.1 Northern Pikeminnow Management Program History

Intensive predation by northern pikeminnow on juvenile Pacific salmon *Oncorhynchus* spp. has been well-documented throughout the lower Columbia River basin (Rieman et al. 1991; Vigg et al. 1991; Ward et al. 1995; Ward et al. 2002) (Figure 5-4 andFigure 5-5).

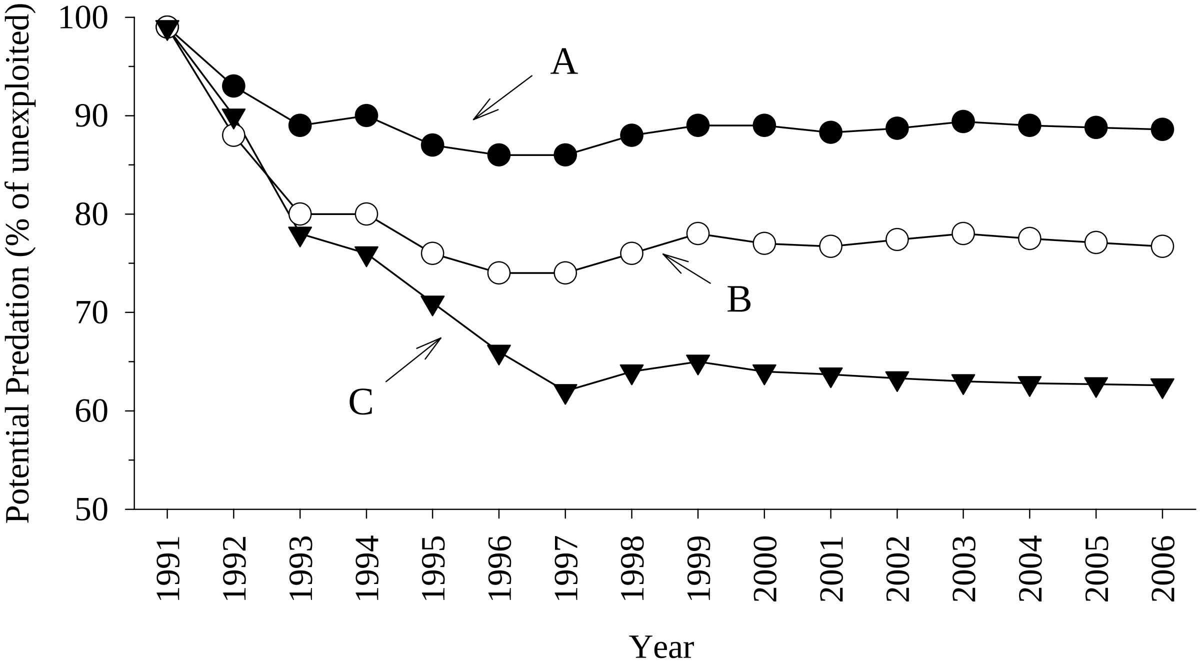

Figure 5-4. Maximum (A), median (B), and minimum (C) estimates of potential predation on juvenile salmonids by northern pikeminnow relative to predation prior to implementation of the Northern Pikeminnow Management Program. Trends after 2002 indicate predicted predation in future years if exploitation is maintained at mean 1996–2002 levels (from Takata and Friesen 2003).

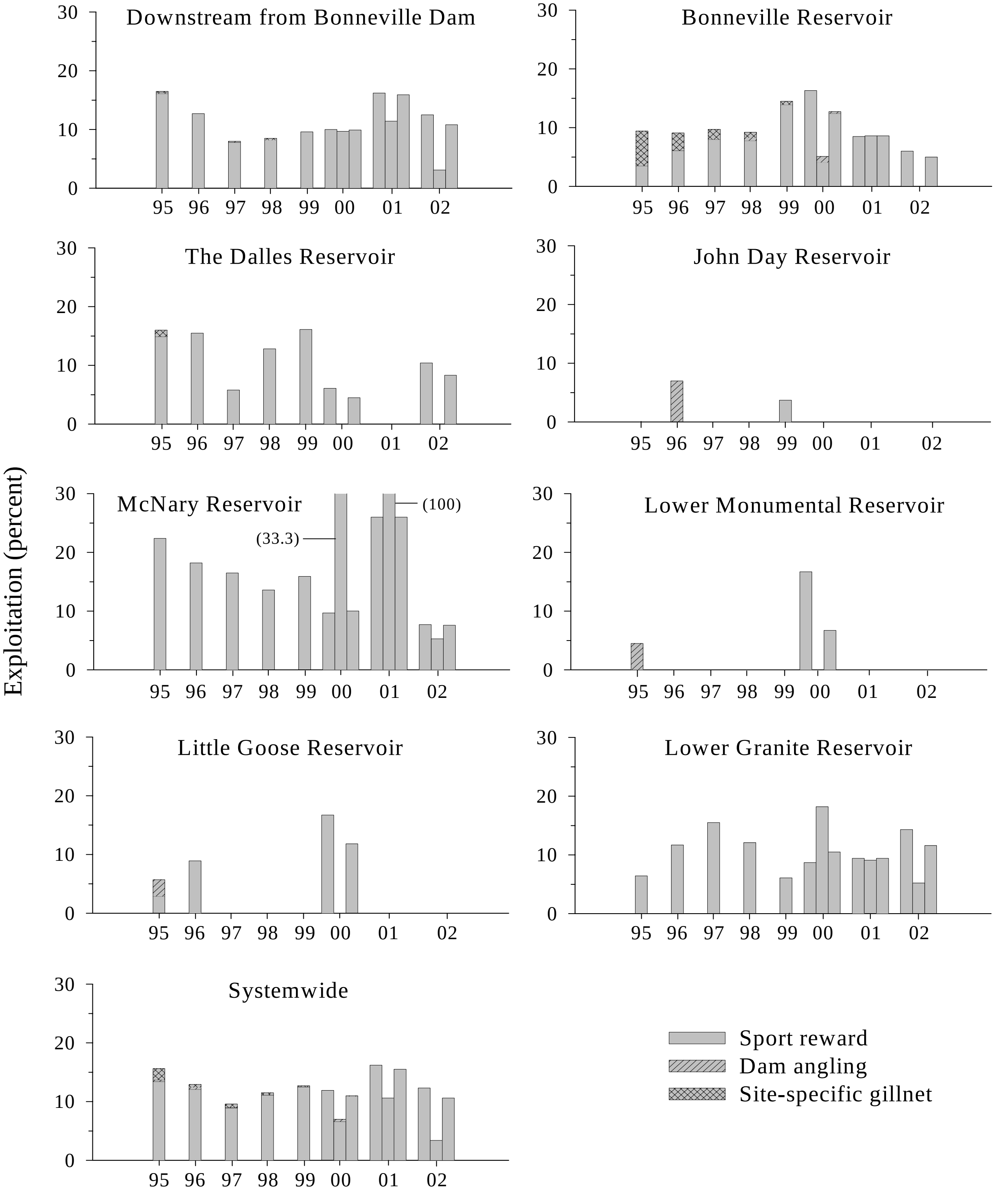

Figure 5-5. Exploitation of northern pikeminnow 250 mm FL by reservoir/area and fishery, 1995–2002. For 2000–2002, vertical bars, from left to right, show exploitation for northern pikeminnow 250 mm FL, 200-249 mm FL, and 200 mm FL. Exploitation rates were not corrected for tag loss in 2000–02 (from Takata and Friesen 2003).

The NPMP was begun in 1990 based on findings from earlier work in John Day Reservoir (Poe et al. 1991; Vigg et al. 1991; Beamesderfer and Rieman 1991; Rieman et al. 1991) and on the potential for successful predation management (Rieman and Beamesderfer 1990). The project's overall objectives were to:

- determine the significance of predation in Columbia River reservoirs through indexing predator abundance and integration with consumption indices.
- implement a predator control fishery development plan, beginning with a test fishery in John Day Reservoir in 1990.
- initiate an evaluation of the Predator Control Program.

Developing northern pikeminnow fisheries required adaptive management. Harvesting techniques were tested from 1990–93 and the most successful were continued. Because they were not able to harvest significant numbers of northern pikeminnow, lure trolling, purse seining, electrofishing, trap-netting, and tribal and commercial long-line fisheries were discontinued.

Adaptive management also has been required for the continued success of the fisheries. The sport-reward fishery was relatively unsuccessful in 1990 until the reward was raised from $1 to $3 per northern pikeminnow. From 1991–94, anglers were paid $3 for each northern pikeminnow at least 11 inches total length (similar to 250 mm FL). Tagged fish were worth $50.

In 1995, the reward changed from $3 to a tiered reward system based on total number of fish caught and this resulted in increased participation. The reward paid to successful anglers was $4 for the first 100 fish caught in the season, $5 for each fish from 101-400, and $6 per fish when catch exceeded 400. Tagged fish were still worth $50. In mid-2001, rewards were temporarily increased to $5, $6, and $8, with tagged fish increasing in value to $1,000. This was an attempt to increase exploitation as one means of partially offsetting poor migration conditions for juvenile salmonids. In 2002, the tiered reward system returned to $4, $5, and $6, with tagged fish being worth $100. The number and locations of registration stations have also changed over the years, depending on trends in effort and catch. Current locations of the stations maximize the efficiency of the fishery.

The dam-angling and site-specific fisheries have used adaptive management to maximize catches while decreasing costs. Dam-angling is concentrated in the dams' tailraces where catch per effort is highest. The site-specific fishery is also concentrated in areas where catch per effort is highest. Lessons learned through the NPMP potentially could be used for understanding and limiting predation mortality caused by other species of predators.

ODFW, the National Biological Service (NBS), and WDFW conducted initial predation indexing from 1990–93. Indexing was conducted in lower Columbia River reservoirs (1990) and downstream from Bonneville Dam (1992). Indexing was conducted before significant removals of northern pikeminnow in each area (Parker et al. 1995).

Test fisheries for northern pikeminnow initiated in John Day Reservoir in 1990 included a public sport-reward fishery, a tribal long-line fishery, and an agency-operated dam-angling fishery (John Day and McNary Dams). The dam-angling fishery also was conducted at Bonneville, The Dalles, and Ice Harbor Dams. The success of the sport-reward fishery led to implementation of the fishery throughout the lower Columbia and Snake Rivers in 1991. Dam-angling was also successful in 1990, leading to its implementation at the four lower Columbia River dams in 1991.

The long-line fishery was expanded to include Bonneville and The Dalles reservoirs in 1991. The long-line fishery was discontinued after 1991 due to lack of participation.

Other technologies for removal of northern pikeminnow were tested from 1990–93, including lure trolling, purse seining, electrofishing, trap-netting, and commercial long-lining, but none proved effective. In 1994, a site-specific gill net fishery to remove northern pikeminnow near hatchery release points and tributary mouths was implemented and considered successful. Implementation of new test fisheries was discontinued after 1994, leaving sport-reward, dam-angling, and site-specific fisheries as the methods of northern pikeminnow removal.

Since 1990, over 1.7 million northern pikeminnow have been removed from the lower Columbia and Snake Rivers with annual exploitation since 1991 averaging over 12% of fish >250 mm FL (Table 5-2). Evaluation of the program consists of monitoring the exploitation rate and size of northern pikeminnow harvested annually for each fishery, and monitoring the effects of observed exploitation rates on predation. Monitoring the effects of exploitation includes the elements described above in Section 5.3.3 Harvest and in Table 5-2.

Predation by northern pikeminnow was indexed throughout the lower Columbia and Snake Rivers each year from 1990–96 and in 1999. Indices of predation were consistently lower from 1994–96 than from 1990–93 (Zimmerman and Ward 1999). Whether piscivory by surviving northern pikeminnow has changed since implementation of the program has not been fully resolved (Zimmerman 1999; Petersen 2001).

Predation by resident fishes is known to be a substantial cause of juvenile salmonid mortality, especially in dam tailraces and at outfall locations. Funded by the ACOE, predation studies are being conducted in some areas near dams. For example, conditions in The Dalles Dam tailrace are unique compared to other projects on the Columbia or Snake Rivers. This dam has a complex basin with a series of downriver islands where predators reside. Studies have been conducted to examine the behavior of predators and estimate the relative densities of northern pikeminnow and smallmouth bass in The Dalles Dam tailrace, and to apply habitat models for these predators (Martinelli and Shively 1997; Petersen et al. 2001).

Recent studies show a relatively high number of smallmouth bass compared to northern pikeminnow in The Dalles Dam tailrace (Petersen et al. 2001). Habitat models developed for northern pikeminnow use water velocity, depth, distance to shore, and bottom substrate type as independent variables. Fitted equations were used in GIS to predict the relative quality of northern pikeminnow habitat throughout The Dalles Dam tailrace for three flow conditions (Petersen et al. 2001). Future work will attempt to improve the northern pikeminnow models by testing some assumptions and adding new data from radio-tagged predators. Habitat models also will be developed for juvenile salmonids and smallmouth bass in The Dalles Dam tailrace using recent, or planned, telemetry studies. These models will be linked to computational fluid dynamics (CFD) models of the tailrace, providing a flexible tool for management decisions. Future studies may also include work in the John Day Dam tailrace to examine predator and prey behavior in response to dam operation and to evaluate the juvenile salmonid bypass.

5.4.2 NPMP Review

In September 1999, the NWPPC[1] recommended that future funding of the NPMP depend on an independent review of the program. Completed in April 2000 (Hankin and Richards 2000), the review reported on the justification for the program and its biological performance, examined the program's cost-effectiveness, and outlined principal findings and recommendations concerning biological and economic issues. Although the review made recommendations for a program that could achieve objectives at a reduced long-term cost, reviewers found that studies suggest that the impact of northern pikeminnow predation is much likely greater than what it may have been prior to construction of dams. (The report found the review task was greatly simplified by the number of papers published in fisheries journals.)

The report also included several recommendations to increase the efficiency and reliability of program evaluation. These recommendations have been implemented and include:

- adopting a scale-age validation study.
- minor changes to the predation model.
- better estimate of the Force of Natural Mortality.
- statistical consultation to review methods of estimating exploitation rates, natural mortality, and northern pikeminnow abundance.
- reduction of the WDFW staff by one permanent full-time position and reducing one permanent full-time position to a 9-month career seasonal position.

A decrease in the minimum size of fish eligible for rewards also was recommended; this was implemented in 2000 as described above.

Economic recommendations included decreasing the costs associated with dam-angling and site-specific fisheries, and reducing the number of agencies involved in program oversight. These recommendations were implemented in 2000. Dam-angling and site-specific costs continue to decrease. The number of agencies involved in the program has decreased because:

- program oversight formerly shared by the PSMFC and the Columbia Basin Fish and Wildlife Authority (CBFWA) is now conducted solely by PSMFC.
- coordination of dam-angling and site-specific fisheries by Columbia River Inter-Tribal Fish Commission (CRITFC) has been eliminated.
- reduction in scope of dam-angling and site-specific fisheries has eliminated all tribes other than the Yakama Indian Nation.

Additional recommendations included conducting further study of the tiered reward system, and exploring the possibilities of increasing rewards by decreasing promotion costs.

5.4.3 Harvest

Under continued implementation of the large-scale predator removal program for northern pikeminnow, harvest is likely the primary determinant affecting northern pikeminnow status in the Columbia Basin, certainly in terms of management activities. Since 1990, a controlled harvest program has been in place to maintain a desired exploitation rate of northern pikeminnow as a means to increase survival of juvenile outmigrating salmonids in the Columbia River basin (Ward et al. 2002). Since 1990, over 1.7 million pikeminnow have been removed

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[1] Now the Northwest Power Planning and Conservation Council

from the lower Columbia and Snake rivers, with annual exploitation since 1991 averaging over 12% of fish >250 mm fork length (Table 5-2). Thus, harvest likely plays the biggest role in affecting future northern pikeminnow population status in the Columbia River Basin.

5.4.4 Recruitment

As discussed above, Beamesderfer et al. (1996) found a negative correlation between concurrent year classes of walleye and northern pikeminnow, and suggested that walleye might influence (reduce) northern pikeminnow numbers by predation. Walleyes are a predator of salmonids, but to a much lesser extent than northern pikeminnow (Rieman et al. 1988). Thus, management favoring walleye might provide a net benefit in salmon survival. Although physical variables are known to influence walleye year-class strength (Busch et al. 1975; Koonce et al. 1977; Serns 1982), similar relations have not been demonstrated for Columbia River walleye stocks (Connolly and Rieman 1988). Species interaction potentially affecting recruitment could subsequently affect northern pikeminnow. However, the effect of recruitment limitation as a function of species interaction is assumed to be minimal compared to affects of harvest under the northern pikeminnow removal program (Ward et al. 2002).

5.4.5 Species Interactions

5.4.5.1 Predation

Perhaps because northern pikeminnow are a predator on outmigrating juvenile salmonids, and because since 1990, they have been the target of a large-scale predator removal program (Beamesderfer et al. 1996; Ward et al. 2002), relatively little attention and research have been focused on predation *on* northern pikeminnow. However, due to their robust demographic trends and relatively high and stable abundance, predation on northern pikeminnow does not appear to limit their production. WDFW instituted a predator management (tiger musky introduction) program in the Cowlitz (Mayfield Lake) and Lewis (Merwin Lake) river systems to reduce the abundance of northern pikeminnows (Jack Tipping, WDFW).

5.4.5.2 Competition

A similar argument can be made for northern pikeminnow regarding the potential negative effects of competition within and between species. If these competitive mechanisms reduce or limit natural production, their effects appear to be masked by productivity and reproductive potential that require ongoing pikeminnow removal to suppress the effects of their predation on outmigrating juvenile salmonids. In the Columbia River reservoirs, northern pikeminnow represent the only native piscivorous salmonid predators. However, numerous nonnative predatory fish species have been introduced into the Columbia Basin (e.g. walleye, smallmouth bass *Micropterus dolomeui,* and channel catfish *Ictalurus punctatus*), and interactions and dynamics among native and non-native fish species, and the subsequent ecological responses are difficult to accurately predict (Beamesderfer et al. 1996). Ward et al. (1995) reported that direct and indirect competitive interactions may affect northern pikeminnow habitat use, prey availability, or juvenile survival, which may in turn contribute to differences in growth, mortality, or recruitment of northern pikeminnow among areas and reservoirs studied.

Furthermore, Ward and Zimmerman (1999) described the response of smallmouth bass density, year-class strength, consumption of juvenile salmonids, mortality, relative weight, and growth to sustained removals of northern pikeminnow in the lower Columbia and Snake Rivers. However, resulting density, consumption, mortality, and growth rate estimates were similar to those determined before northern pikeminnow removals.

5.4.6 Water Development

5.4.6.1 Dams

Northern pikeminnow have successfully evolved in a range of dynamic lentic and lotic ecosystems, and have successfully adapted to varied habitat conditions within those systems, Thus, the species potential for adaptation has allowed northern pikeminnows to flourish despite construction and operation of the Columbia Basin hydropower system. Dams, and specifically the orientation and creation of protected habitat and increased prey availability (e.g. the juvenile fish bypass outlet of Bonneville Dam) create favorable areas for high densities of northern pikeminnow and salmon smolts that make the predation of northern pikeminnow more efficient, thus perhaps *enhancing,* rather than limiting, pikeminnow population growth and productivity (B. Muir, NOAA Fisheries, personal communication). In the lower Columbia River Basin, reservoirs impounded by dams in the Cowlitz and Lewis river systems (Mayfield Lake and Merwin Lake respectively) likely contributed to increased abundance and production of pikeminnow (J. Tipping reports, WDFW mid-90s).

5.4.6.2 Flow Alterations

Interannual variation in water-years, and flow alterations within and among years, may have relevant effects on northern pikeminnow population status, year-class production, and effectiveness of pikeminnow predation on juvenile salmonids in the Columbia River basin. Mesa and Olson (1993) determined prolonged swimming performance of two size-classes of northern pikeminnow, and reported that water velocities from 3.28 to 4.27 ft/sec (100 to 130 cm/sec) may exclude or reduce predation by northern pikeminnow around juvenile bypass outfalls at Columbia River dams, at least during certain times of the year. Furthermore, these authors recommend that construction or modification of juvenile bypass facilities place the outfall in an area of high water velocity and distant from eddies, submerged cover, and littoral areas for the same reason.

5.4.7 Water Quality

5.4.7.1 Temperature

Northern pikeminnow tolerate a wide range of temperatures.

5.4.7.2 Turbidity

Increased turbidity was demonstrated to reduce efficiency of visual predation of YOY white sturgeon by northern pikeminnow in controlled predation studies (Gadomski et al. 2000, 2001, 2002).

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Volume III, Chapter 6
American Shad

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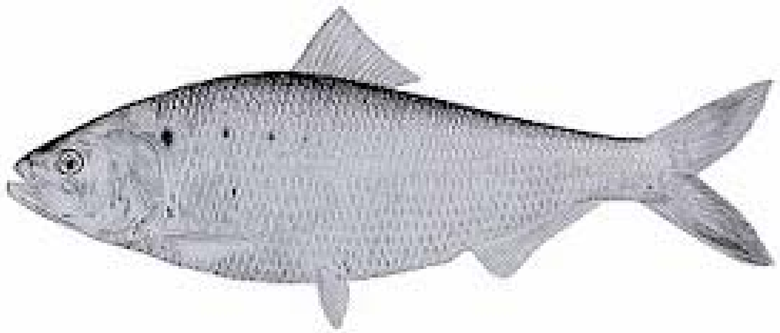


6.0 American Shad *(Alosa sapidissima)*

6.1 Introduction

American shad (*Alosa sapidissima*) is a member of the herring family Clupeidae and has a herring-like body, large and deep, with a row of dark spots on the side decreasing in size towards the posterior (Hart 1973). American shad can grow up to 30 inches long and weigh 12 pounds. In the Columbia River, American shad average 3-5 pounds (Wydoski 1979). Shad are native to the Atlantic coast and were introduced to the Pacific coast in the early 1870s. [Records document commercial landings in the Columbia River in 1885.] Shad have become abundant and well established in the Columbia River and tributaries, including the Snake River and the Willamette Rivers (WDFW and ODFW 2002). Shad are now found from southern California to Kodiak Island, Alaska (Hart 1973).

Extensive biological and ecological information exists about American shad in its native Atlantic coast habitat. Similar information is lacking about American shad in the Columbia River system. Shad migrate upstream with adult salmon through fishways in the Columbia River dam system. Competition between salmonids and American shad for passage through fishways has resulted in some dams being modified to allow for better shad access to upstream habitat (Wydoski 1979). American shad have migrated past Bonneville, The Dalles, John Day, McNary, and Priest Rapids Dams and as far as the Lower Granite Dam on the Snake River (WDFW and ODFW 2002).

On the Atlantic coast, American shad are a popular commercial and sport fish. In the Columbia River, American shad are under utilized in the commercial and sport fisheries. Between 1990–2000, approximately 4% of the annual American shad run was harvested (WDFW and ODFW 2002), rising to approximately 6% in 2001 and 7% in 2002 (WDFW and ODFW 2003).

6.2 Life History & Requirements

American shad are anadromous with the ability to return to spawn for several years. They live most of their life in marine waters where little is known of their life history on the Pacific coast. Returning American shad begin entering the Columbia River in May. The timing of spawning runs is temperature-dependent. The peak movement of shad through the Bonneville Dam occurs during temperature ranging from 16.5-19° C (Leggett 1973). In the Columbia River system, spawning occurs in June–August (Wydoski 1979).

American shad spawn in groups of one female and one to several males. Eggs are semi-buoyant and develop quickly, hatching within 6–10 days. Young fish remain in the river until late fall when they out-migrate to marine water. American shad are plankton feeders in fresh water and, as adults, will filter food (like mysid shrimp) through gill rakers. American shad spend 3–4 years at sea before becoming sexually mature (Morrow 1980).

6.2.1 Spawning Conditions

American shad may spawn immediately on entering fresh water or may migrate upstream several miles to spawn. Water temperature influences the timing of spawning runs. The farther North in latitude, the later in the year shad will spawn. In the Columbia River American shad spawn in June–August when water temperatures are 15.5-18.3° C (Wydoski 1979) and where water velocity is less than 0.7 m/s (Ross et al. 1993).

Spawning usually occurs in the evening over fine gravel in shallow water. One female may emit eggs over a period of several days before all eggs have been dispersed (Olney et al. 2001). The number of eggs depends on size of female. Estimates of fecundity range from 116,000-616,000 per female (Morrow 1980). Fecundity of Hudson River American shad—an original source of Pacific coast shad—range from 116,000-468,000 (MacKenzie et al. 1985).

6.2.2 Incubation

Fertile American shad eggs are about 3.5 mm in diameter, pale pink to amber in color, semi-buoyant and non-adhesive. The eggs are shed over sandy pebbly substrate where they gradually drift downstream while developing to maturity. Eggs can be found at any depth during spawning season, but are most numerous near the bottom (Moyle 2002). Eggs hatch in 6–8 days at temperatures ranging from 14-17° C. Colder water increases the length of time to hatching (Morrow 1980).

6.2.3 Larvae & Juveniles

Newly-hatched shad are 9-10 mm in length. The yolk sac is absorbed in 4–5 days. Within 10–12 days after hatching, shad larvae will begin feeding primarily on copepods and chironomid larvae and in 3–4 weeks will reach approximately 2 cm long, with fully-developed fins. Juvenile shad stay in fresh water for several weeks before moving seaward in late fall. American shad grow to a size of 3.7 to 11.2 cm before leaving fresh water (Morrow 1980).

A study of habitat use by premigratory American shad in the Delaware River suggests that juvenile shad are habitat generalists and use submerged aquatic vegetation (Ross et al. 1997). A study of American shad in the Columbia River showed that shad larvae were found in the main channel, shorelines, sloughs, and backwater habitats (Petersen et al. In press). American shad larvae were found to be more abundant in the heavily vegetated backwaters of the lower Columbia River and are denser at night (Gadomski and Barfoot 1998).

6.2.4 Adult

American shad in the Columbia River average 1.5-2.3 kg and 43-55.8 cm long (Wydoski 1979). Shad can live to be 11 years old. Females mature between 4–6 years, and grow slightly larger than males. Male shad mature between 3–5 years (Morrow 1980, Wydoski 1979). Shad are plankton feeders, filtering copepods and mysids through their gill rakers. Most shad have a diel vertical migration as they follow their principal food source (MacKenzie et al. 1985).

6.2.5 *Movements in Fresh Water*

American shad flourished on the West Coast and in the Columbia River after their introduction in the late 1800s. The reasons for shad abundance vary, but it is strongly believed that the freshwater habitat created by dam reservoirs is ideal for spawning and rearing (Petersen et al. In press, Monk et al. 1989). The Dalles Dam was built in 1956, the adult shad count at Bonneville Dam (downstream) increased from a 22–year average of 15,475 fish (1938–59) to 329,850 fish in the period 1960–64 (Wydoski 1979) (Figure 6-1).

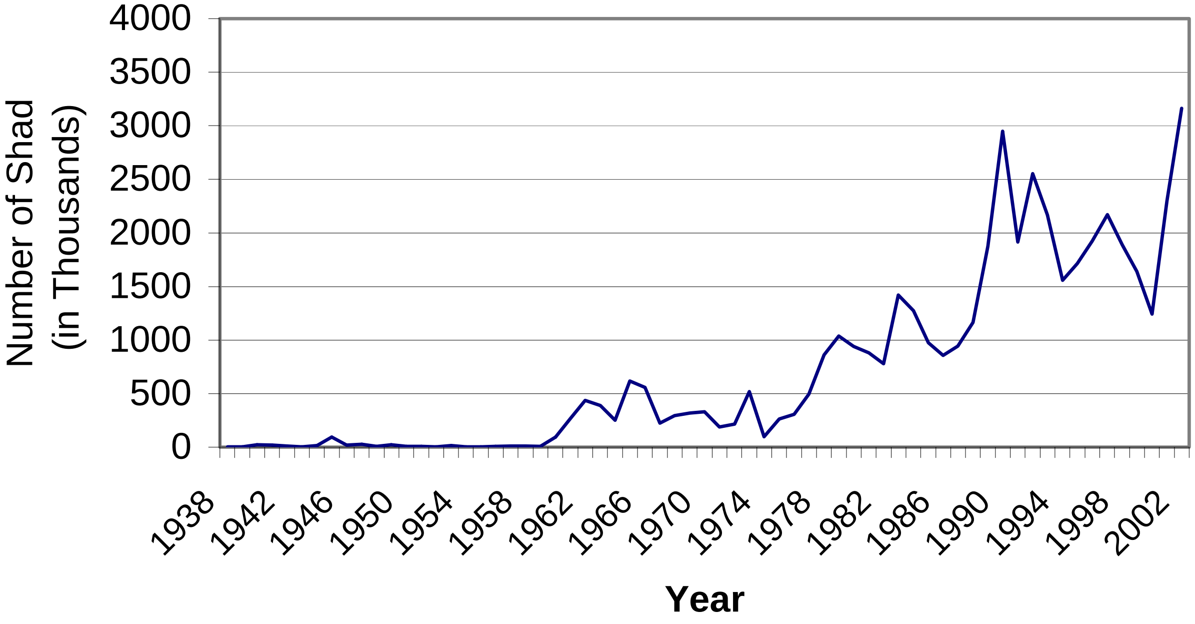


Figure 6-1. Minimum numbers (in thousands) of American shad passing Bonneville Dam, 1938–2002 (WDFW and ODFW 2002)

American shad move upstream during the same spawning run time as spring chinook, and shad have migrated upstream as far as the Priest Rapids Dam, past Ice Harbor Dam into the Snake River. The sheer abundance of shad returning upstream causes crowding at fish ladders designed for the passage of Pacific salmon. The annual peak of upstream shad migration occurs during May–July. In June 2003, approximately 4,025,000 adult shad migrated through the Bonneville Dam fishway. During the same month, 73,600 chinook and 26,400 sockeye passed through Bonneville Dam fishway (DART). The American shad run size is probably higher than the dam counts because many adults spawn below Bonneville Dam (Petersen et al. In press).

Fish ladders at some dams have improved the passage of American shad. When the John Day Dam became operational in 1968, American shad were either reluctant or unable to pass through the submerged orifices in the dam's two fish ladders. This resulted in large numbers of shad dying and blocking the fish barrier screens (Monk et al. 1989). Modifications in 1970 to the fish ladders reduced the flow velocity and created surface passage weirs. Similar modifications were made to Bonneville Dam in 1974. There were no significant differences in the upstream migration of salmonids after the changes were made to the fishways (Monk et al. 1989).(Figure 6-2).

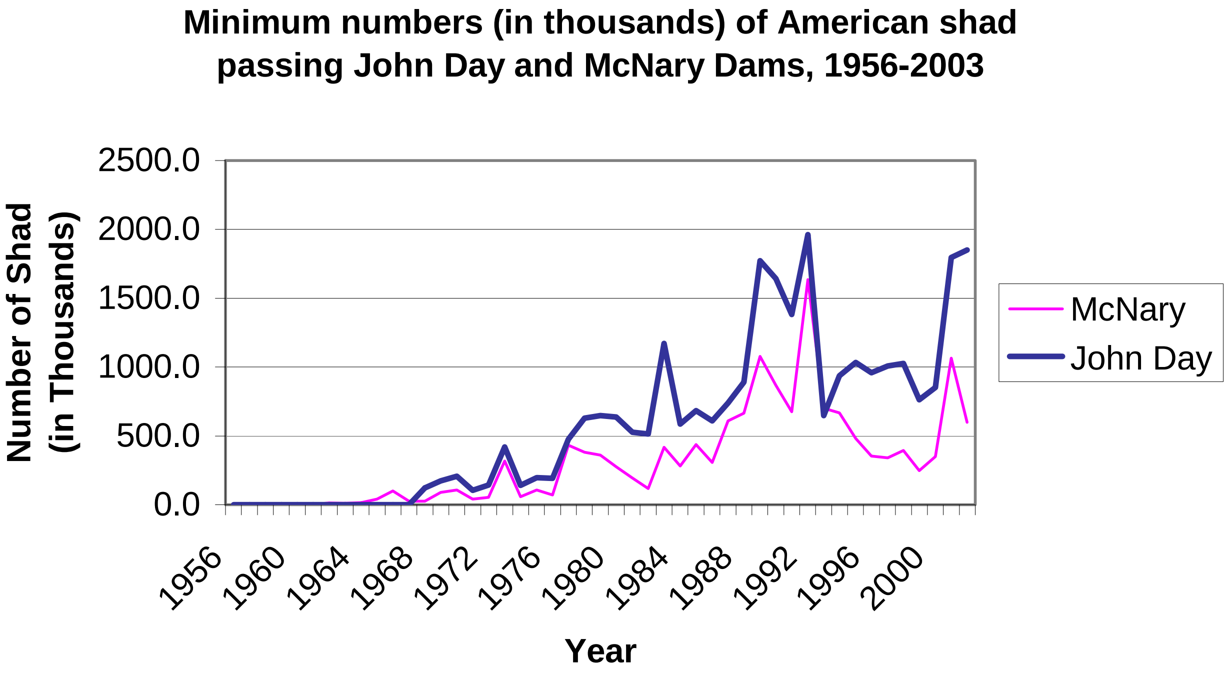


Figure 6-2. Minimum numbers (in thousands) of American shad passing through John Day and McNary Dams, 1956–2003 (DART 2003).

6.2.6 Ocean Migration

Once American shad have spawned, they begin feeding again on their return to salt water (Morrow 1980). Little is known of the ocean stage of shad life on the Pacific coast. Tagging studies on the Atlantic coast have shown American shad migrated 2,400 miles annually from the St. John's River in Florida north to Nova Scotia (Leggett 1973). It is believed that Columbia River shad follow similar migration patterns: a northward migration in summer and a southward migration in winter (Leggett and Whitney 1972). Following their introduction into the Sacramento River in 1871, shad appeared at Vancouver Island, British Columbia in 1876 and by 1904, had migrated as far as Cook Inlet, Alaska (Morrow 1980).

6.3 Population Identification & Distribution

The American shad introduced into the Sacramento River came from a hatchery on New York's Hudson River. In 1885, shad were introduced into the Columbia River from the Susquehanna River in Pennsylvania (WDFW and ODFW 2002). However, shad had been observed in the Columbia River earlier. It is believed they were transported northward by the Davidson Current that flows from San Francisco to Vancouver Island (Ebbesmeyer and Hinrichsen 1997). Shad have become very abundant in the Columbia River system, with as many as 4 million shad estimated in 1990 (WDFW and ODFW 2002).

6.3.1 Life History Differences

Only one species of shad (*Alosa sapidissima*) is found in the Columbia River and there are no other species or subspecies with different life histories on the Pacific coast.

6.3.2 Genetic Differences

Little is known about American shad genetic variations in the Columbia River. In a study of American shad developmental physiology, differences between stocks from the Columbia

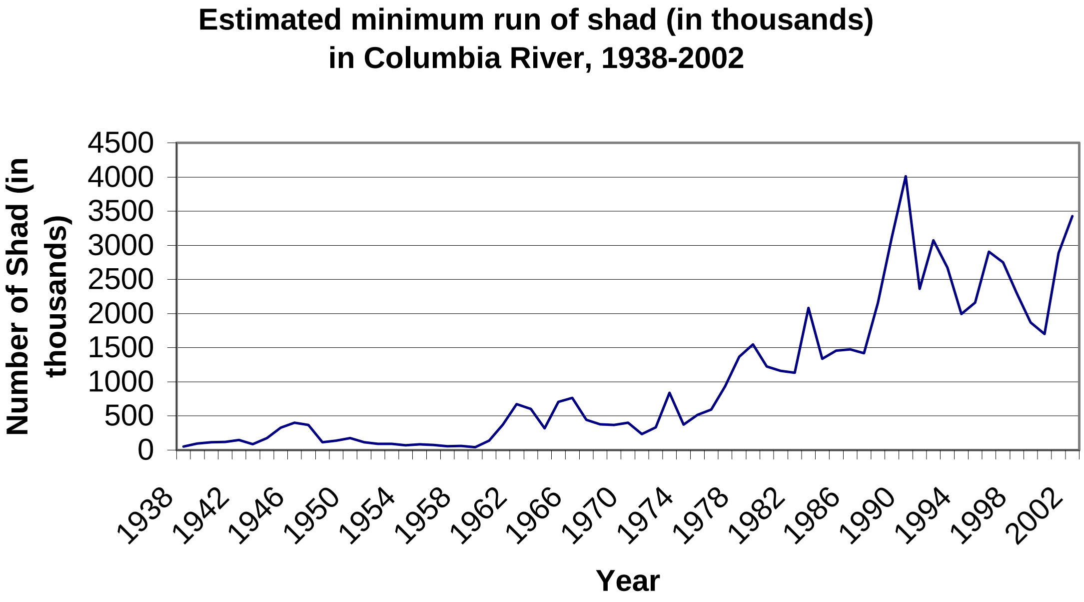
River (Pacific coast) and the Delaware River (Atlantic coast) were observed as part of an effort to reestablish American shad runs in the Susquehanna River (Rotifers et al. 1992). Biologists collected American shad eggs from the Columbia River and Delaware River to use in production at the Van Dyke Fish Hatchery in Thompsontown, Pennsylvania. Columbia River American shad grew significantly faster, attained greater final weight, and were more tolerant of changes in temperature and salinity than the Delaware River shad. Electrophoresis analysis of the Columbia River and Delaware River stocks revealed allelic differences at one locus (creatine kinase). The researchers suggested that the genetic variation might be due to natural selection in the Columbia River populations after introduction to the Pacific coast. The researchers also suggested that fish managers further investigate genetic variations of American shad stocks before using out-of-basin stocking for restoration projects (Rottiers et al. 1992).

6.4 Status & Abundance Trends

6.4.1 Abundance & Productivity

American shad are well established in the Columbia River and its tributaries, including the Snake River. In 2002, 81,373 shad passed the Ice Harbor Dam on the Snake River and 4775 shad passed Priest Rapids Dam. Since 1977, the estimated minimum run sizes of shad in the Columbia River have been over 1 million fish. In 1990, the estimated minimum run size was a record high of 4 million fish (WDFW and ODFW 2002) (

Figure 6-3).



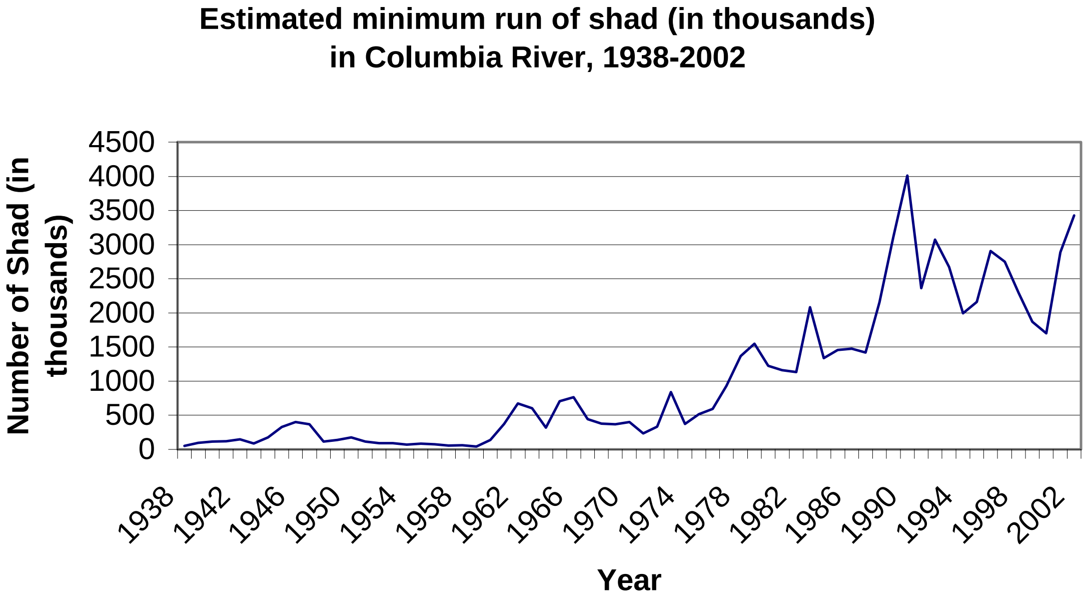


Figure 6-3. Estimated minimum run of American shad in Columbia River, 1938–2002.

6.4.2 Supplementation

Numerous hatcheries for American shad exist on the Atlantic coast. No supplementation hatcheries or projects for American shad exist in the Columbia River.

6.4.3 Harvest

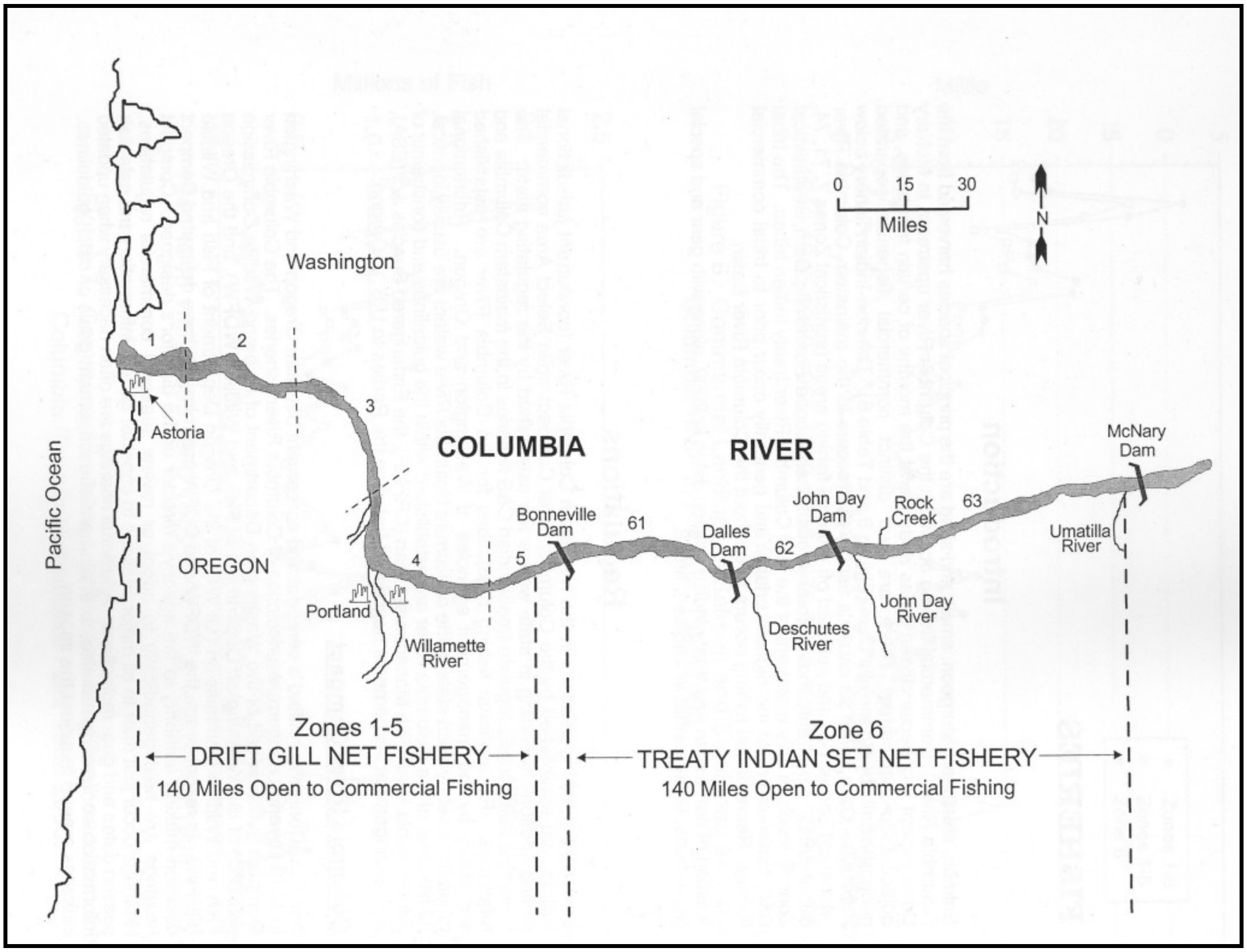


Figure 6-4. Commercial fishing zones on Columbia River below McNary Dam (WDFW and ODFW, 2002).

Commercial harvest of American shad in the Columbia River mainstem is jointly regulated by Washington and Oregon within the guidelines of the Columbia River Compact. Non-treaty commercial fishing takes place downstream from Bonneville Dam, while Treaty fishing takes place from upstream of Bonneville Dam to McNary Dam (Figure 6-4). American shad are taken by gill nets in the commercial fishery. A typical season runs May–June. The commercial shad fishery has been limited because the run coincides with spawning runs of spring chinook, summer chinook, sockeye, and summer steelhead. Since 1996, shad fishing has been restricted to a daily period from 3–10 pm and is restricted to shorter, shallower nets to limit the capture of salmonids (WDFW and ODFW 2002). Figure 6-5 shows the overlap of shad and salmonid species passing through Bonneville Dam during May–August (WDFW and ODFW 2003).

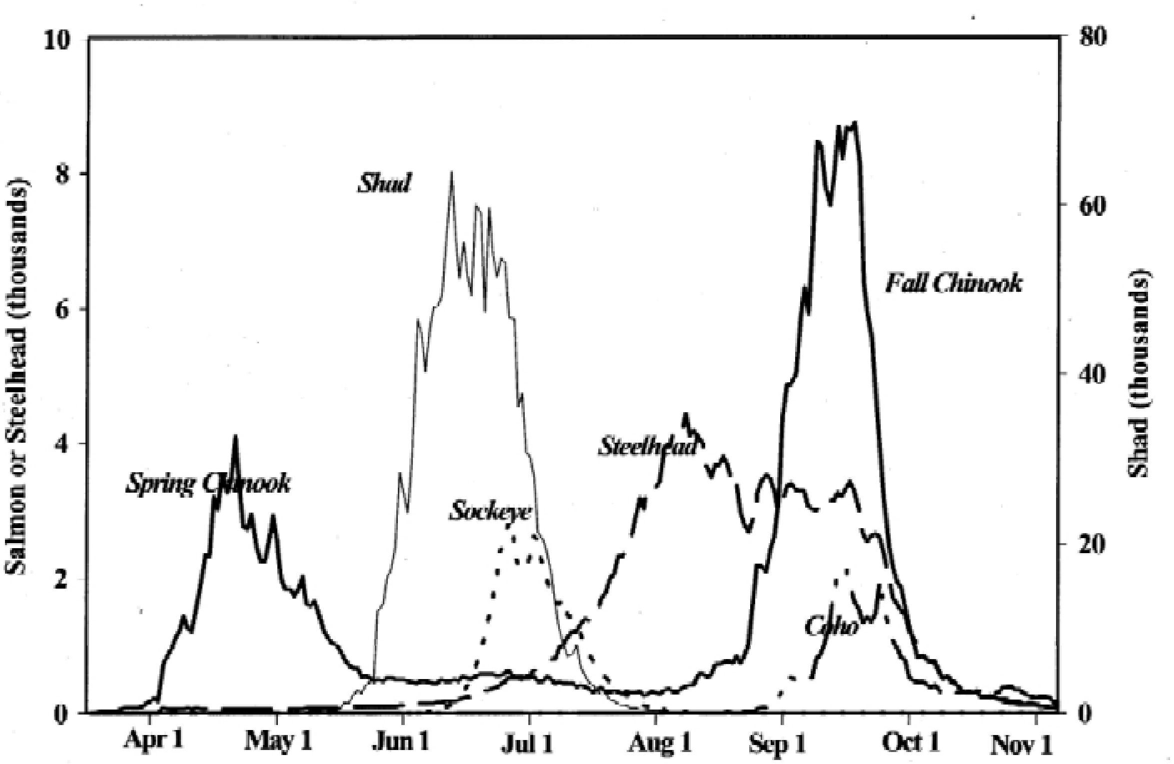


Figure 6-5. Average daily counts of salmon, steelhead, and shad at Bonneville Dam, 1986–2001 (WDFW and ODFW, 2003).

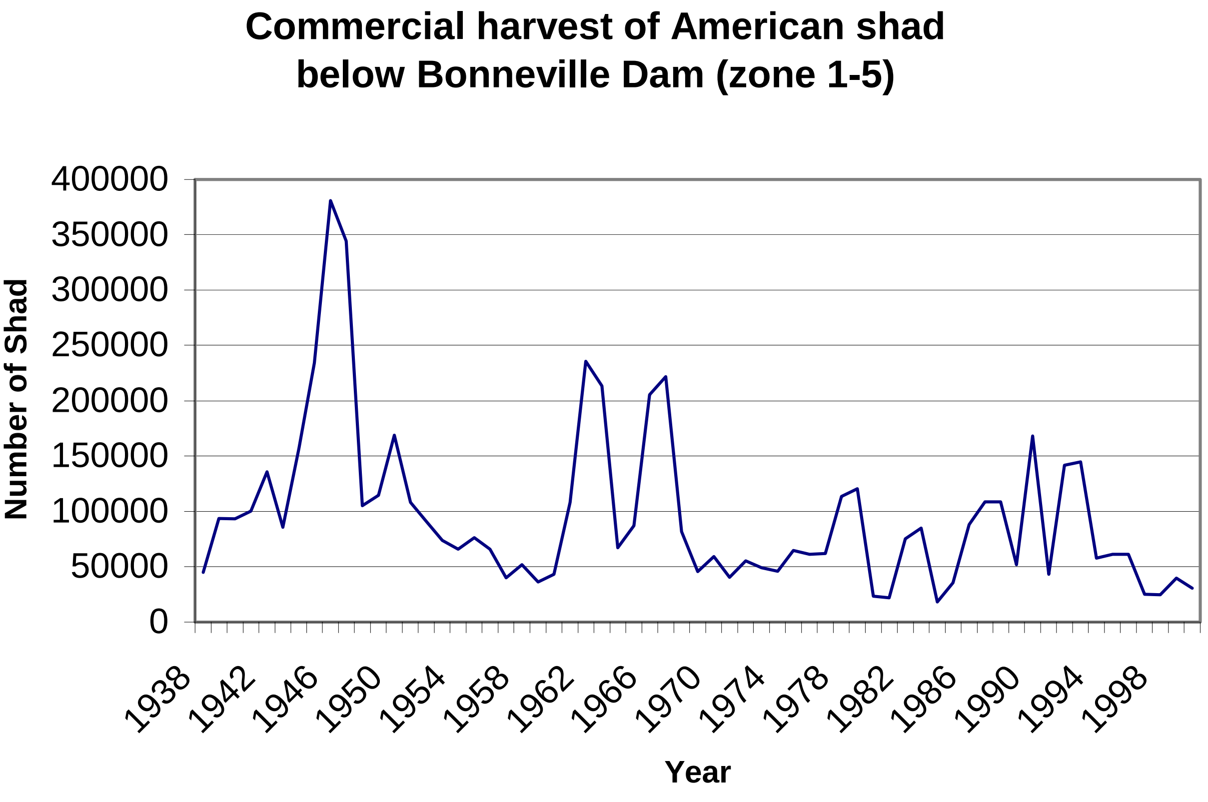


Figure 6-6. Commercial harvest of American shad, 1938–2000 (WDFW and ODFW, 2002).

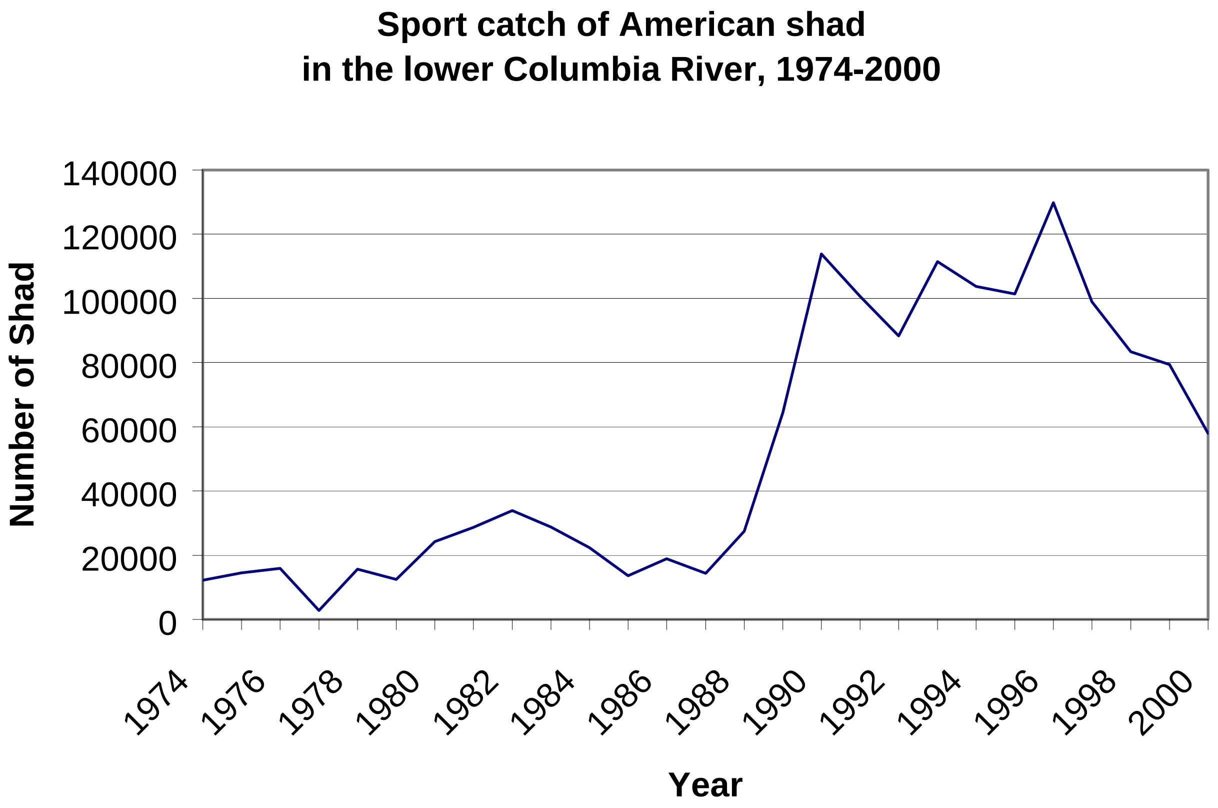


Figure 6-7. Sport catch of American shad, 1974–2000 (WDFW and ODFW, 2002).

The combined commercial and sport non-treaty harvest for 2001 and 2002 were 163,400 and 211,200 shad, respectively (Figure 6-6 and Figure 6-7).

6.5 Factors Affecting Population Status

6.5.1 Water Development

Hydropower development on the Columbia River increased habitat for American shad by creating reservoirs and backwater habitats. Improved access to upstream spawning areas on the Columbia River has positively affected American shad populations. Before the John Day Dam was modified in 1972, an average of 18% of the shad run successfully passed upstream. After modification, shad passage increased to an average of 73% of estimated run (Monk et al. 1989). Estimated run sizes of shad in the Columbia River have been over 1 million fish since 1977 (WDFW and ODFW 2002).

6.5.2 In-Channel Habitat Conditions

It is unknown how American shad may be affected by in-channel conditions. Shad eggs are semi-buoyant and drift downstream (Moyle 2002) or may settle within the river substrate (Morrow 1980). Shad larvae will migrate downstream in late fall.

6.5.3 Species Interactions

Due to the abundance of American shad in the Columbia River, system studies have been launched to investigate species interactions between shad, salmonids, and other fish species such as northern pikeminnows, smallmouth bass, and walleye (Petersen et al. In press). A pattern is slowly emerging that may show the existence of American shad is changing trophic relationships with the Columbia River. One study found that in the lower estuary (up to RKm 62) shad diet overlapped with subyearling salmonid diets, which may indicate competition for food. Juvenile shad and salmonids also utilize similar heavily-vegetated backwater habitats (McCabe et al. 1983). Another study examined the abundance of shad as prey on the faster growth rates of northern pikeminnows, which in turn are significant predators of juvenile salmonids (Petersen et al. In press).

Commercial harvest of American shad has been restricted because the spawning run coincides with depressed runs of summer and spring chinook, sockeye, and summer steelhead (WDFW and ODFW 2002).

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Walleye

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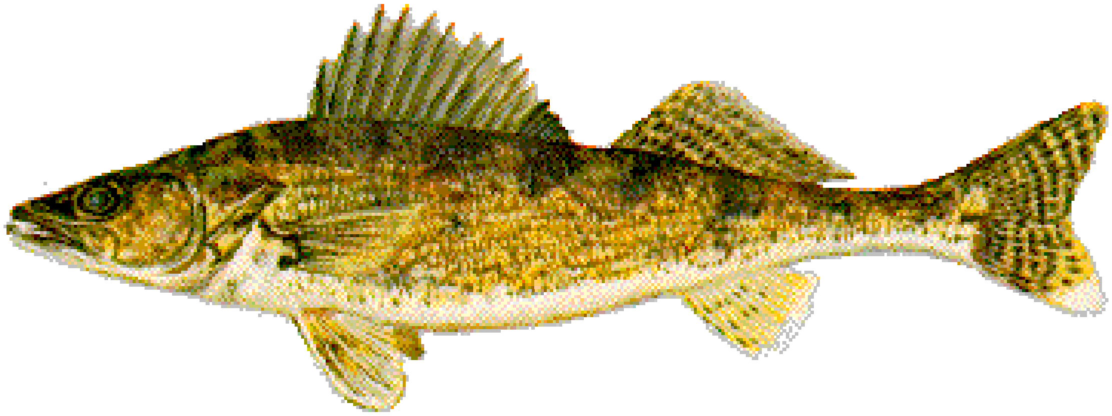

7.0 Walleye *(Stizostedium vitreum)*

7.1 Introduction

The 71 families of fishes in the order Perciformes can be found throughout North American and Europe (Scott and Crossman 1998). The family Percidae (the perches) is made up of two subfamilies, nine genera, and 121 species. The Percids can be distinguished by two well-separated dorsal fins. In North America, Percids are found in warm temperate to cold subarctic lakes and streams (Scott and Crossman 1998). Although the family is distributed circumpolarly, most species are confined to North America (Scott and Crossman 1998).

7.2 Distribution

Walleye *(Stizostedium vitreum)* [1] are native to the Great Lakes and the upper Mississippi River basin. They are found only in fresh water, as illustrated by the map in Figure 7-1 (Scott and Crossman 1998).Walleye also have been introduced along the East Coast and to most states west of their natural range (Scott and Crossman 1998).

Over the past 40+ years, the walleye (*Stizostedium vitreum*) has become one of Washington's most popular and valued game fish species. It is still unclear when the walleye were first introduced into Washington. The first theory has USFWS releasing walleye fry from Lake Oneida (New York) into Lake Roosevelt (Williams and Brown

Figure 7-1. Original distribution of walleye in North America[1].

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[1] The walleye illustration is by Virgil Beck, courtesy of the Wisconsin Department of Natural Resources (DNR).

1983). The second theory has unknown sources planting walleye in the 1930s into Devil's Lake; when Devil's Lake was inundated by the Columbia basin irrigation project, these walleye found their way into Banks Lake and the Columbia River (Beamesderfer and Nigro 1989). Although the origin of the first walleye introductions into Washington is uncertain, since 1960, walleye have become widely dispersed throughout the Columbia River basin, including all of the major reservoirs of the Columbia basin irrigation project (Figure 7-2).

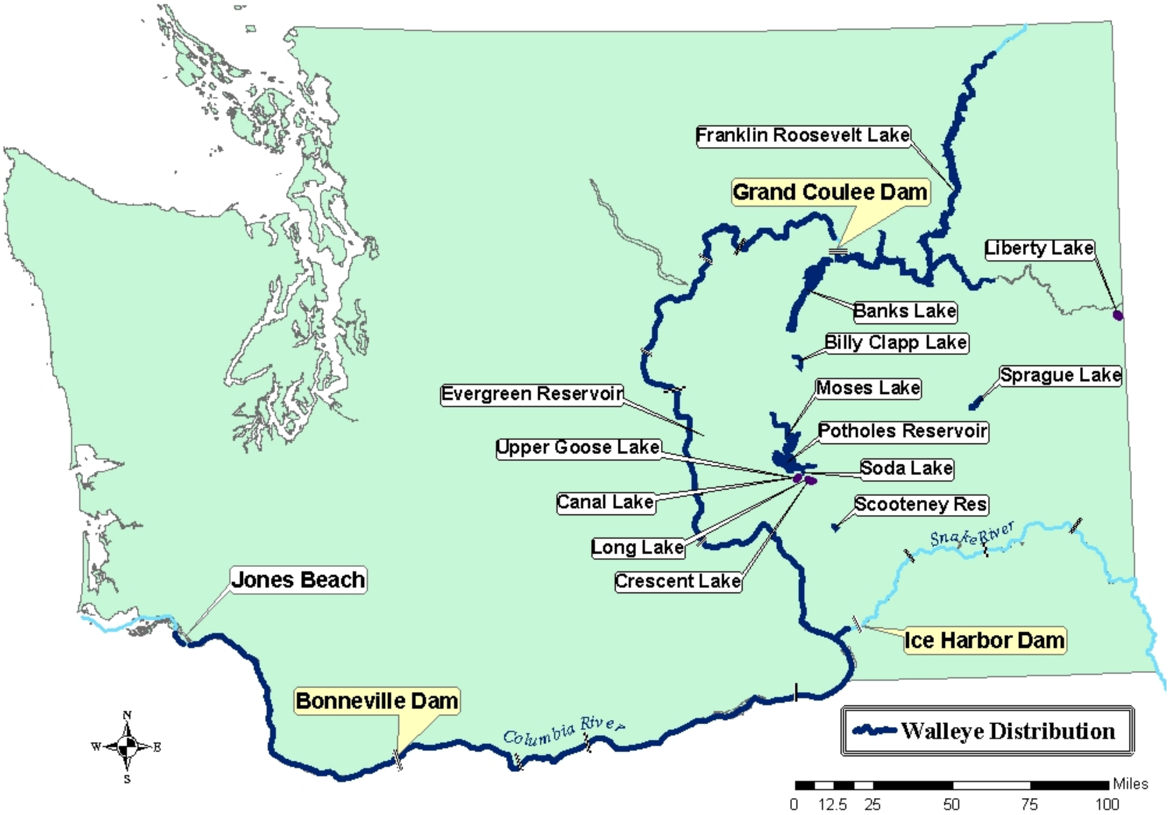

Figure 7-2. Walleye distribution in Washington

Until the early 1980s, walleye management efforts focused on documenting the distribution of expanding populations, regulating harvest, and monitoring the catch from major fisheries. Before 1974, Washington had no regulations for legal catch or size limits for walleye. In 1974, the daily catch limit was set at 15 fish with no more than 5 over 20 inches (51 cm) long (Tinus and Beamesderfer 1994) (see Table 7-1 outlining sport fishing regulations). The fishery focused on Lake Roosevelt and, to a lesser extent, on other Columbia River reservoirs above Rocky Reach Dam, Banks Lake, and Potholes Reservoir. In the early 1980s, the relatively new walleye fisheries of the lower Columbia River reservoirs began to attract national attention and experience a rapid increase in angling pressure and harvest.

From 1973–82, the average size of walleye caught in Lake Roosevelt—by far Washington's most productive and popular walleye fishery—declined from 18.5 to 13.5 inches (47 to 34 cm) (Nigro et al. 1983). At the same time, walleye fisheries in Columbia River reservoirs immediately downstream from Lake Roosevelt experienced a similar decline in average fish size and catch rate (Williams and Brown 1983). The decline of these established walleye fisheries and the desire to protect newly-emerging walleye fisheries from overharvest prompted WDFW to reevaluate walleye management and harvest regulations during the early 1980s.

As a result, in 1986 the walleye catch limit for the lower Columbia River was reduced to 5 fish per day with an 18-inch (46 cm) minimum size. In 1990, modified regulations for the lower Columbia kept the same minimum size, but only one fish could exceed 24 inches. Growing concern over increased predation by walleye on young salmonids migrating through and rearing in the lower Columbia River led to the modification of sport fishing regulations for 2000 to allow increased harvest of smaller walleye.

The presence or absence of suitable early rearing habitat plays a major role in the ability of Washington's walleye populations to sustain levels adequate to support expanding recreational demand. The most important components of good early rearing habitat include a relatively stable water level and temperature, and the presence of nutrient-rich nursery areas adjacent to spawning areas where newly hatched walleye fry can find plankton and develop swimming proficiency.

While lack of early rearing habitat appears to be the major factor limiting walleye production in the Columbia River, other habitat conditions are important as well. Some include availability and access to spawning habitat; suitable water temperature for growth and development; and an adequate food supply. In some cases, it may be possible to enhance walleye populations in Washington by implementing habitat improvement measures such as stabilizing water levels, providing more off-channel rearing habitat, and improving forage conditions.

Table 7-1. History of WDFW sport fishing regulations for the Lower Columbia River *

	Daily Bag Limit	Size Minimum	Size Maximum
1970–73	none	none	none
1974–85	15	none	no more than 5 >20 in
1986–89	5	18 in	none
1990–99	5	18 in	not more than 1 >24 in
2000–present	10	none	no more than 5 >18 in no more than 1 >24 in

* Information on state regulations is from Tinus and Beamesderfer (1994), and WDFW regulations, 1994–2002.

7.3 Life History & Requirements

7.3.1 Spawn Timing & Conditions

Walleye normally spawn from late March through early May, depending primarily on water temperatures. The preferred spawning temperature range is 4.4-10°C. The males arrive at the spawning grounds before the females and tend to stay a little later (Scott and Crossman 1998; Wydoski and Whitney 1979). Spawning generally occurs in water less than 15 feet deep over a variety of substrates such as flooded vegetation, coarse gravel, and boulders. Although walleye do not have a restricted home range, they tend to spawn in the same location each year (Wydoski and Whitney 1979). Walleye have been known to spawn along shoreline areas of lakes and reservoirs, but most often prefer moderately-flowing streams (Becker 1983).

7.3.2 Incubation

Egg development varies with water temperature (Wydoski and Whitney 1979). Depending on the water temperature, eggs can hatch after 7 (>12.8°C) to 26 (4.4°C) days. Above Bonneville Dam, walleye spawning areas tend to be on the windward side of the impoundment where wave action helps keep the water free of silt, which can suffocate eggs. For the same reason, walleye tend to spawn in areas of moderate current below Bonneville Dam (Steve

Jackson, WDFW, personal communication). If there is too much wind or current, the eggs can be washed ashore (Rook 1999) or preyed upon by various species cohabitating the area, although this is not thought to be significant (Becker 1983; Steve Jackson, WDFW, personal communication).

7.3.3 Larvae & Juveniles

The yolk-sac of walleye fry is relatively small and is usually fully absorbed within 2 to 3 days (Becker 1983). For that reason, the survival of walleye fry depends largely on their first 3 to 5 days of life (Becker 1983). Newly-hatched fry do not develop paired fins for several weeks after hatching, restricting their mobility to vertical swimming movements utilizing the whip-like action of their tails. Because of their limited mobility, early rearing habitat must be located close to spawning areas. Walleye fry start out utilizing zooplankton and progress rapidly to larger forms of invertebrates and small fish within the first few months of life. From that point on, their diet is composed almost exclusively of fish (Becker 1983). The dietary transition from invertebrates to fish coincides with a change from a surface to a bottom habitat (Scott and Crossman 1998).

It is believed that this period in life history of walleye most limits their reproductive success in the Columbia River. Lower Columbia River reservoirs typically are shallower, warmer, and more productive than those of the mid-Columbia. However, even with these apparent advantages, reproductive success in the lower Columbia River is highly variable, most likely because of the effects of high flows and extreme fluctuations in water level and temperature during and after spawning (Rieman and Beamesderfer 1988). These conditions coincide with spring run-off and are at times aggravated by the operation of mainstem dams for hydropower production and/or smolt passage (Beamesderfer and Nigro 1989). Although the fry are subject to predation by other species of fish, the flushing of prey items out of the rearing area due to flow and water level changes is thought to affect fry more significantly (Steve Jackson, WDFW, personal communication).

7.3.4 Adult

Walleye have been found to live longer than 15 years. The oldest reported walleye taken in Washington waters was taken from Banks Lake, and was estimated to be 19 years old (Lucinda Morrow, scientific technician, WDFW, April 3, 2003 personal communication).

Growth rates for walleye in Washington generally exceed those reported for walleye in its native range (Becker 1983). On the average, Washington walleye attain a length of 5-7 inches (13 cm) at age 1, 10-14 inches (25-36 cm) at age 2, 15-18 inches (38-46 cm) at age 3, 16-20 inches (41-51 cm) by age 4, 17-22 inches (43-56 cm) at age 5, 19-25 inches (48-63 cm) at age 6, and 20-26 inches (51-66 cm) at age 7 (Fletcher 1992; Williams and Brown 1983, Nigro et al. 1983, Connolly and Rieman 1988). As expected, the fastest growth occurs in the lower Columbia River and in some of the warmer, more productive habitats of the Columbia Basin irrigation project, while the slowest growth rates occur in colder, more densely-populated waters like Lake Roosevelt (Nigro et al. 1983; Williams and Brown 1983; Connolly and Rieman 1988).

Adult walleye prefer to inhabit areas where the water temperature is around 77°F (25°C), but can be found in water temperatures as low as 32°F (0°C) and as high as 90° F (32.2°C) (Wydoski and Whitney 1979). In the lower Columbia River reservoirs, walleye are most abundant in tailraces, somewhat less abundant in mid-reservoir, and least abundant in forebays (Zimmerman and Parker 1995). Downstream from Bonneville Dam, walleye can be found as low as RKm 137, but they are most numerous from RKm 178 to 234 (Zimmerman and Parker 1995).

Walleye can tolerate a variety of environmental conditions, but prefer shallow, turbid areas (Scott and Crossman 1998). Because walleye have a special layer of the eye (*tapetum lucidum*, see Ali and Anctil 1968 cited in Scott and Crossman 1998) that is sensitive to bright daylight (Scott and Crossman 1998), in habitats with very clear water or during periods of the year where there is intense daylight, walleye most often feed at dawn, dusk, and night. In addition to daily movements in response to light intensity, walleye also move annually for spring spawning and daily and seasonally according to water temperature and prey availability (Scott and Crossman 1998). In open water, walleye travel in loose aggregations and schooling is common when feeding and spawning (Becker 1983).

In Washington, first spawning occurs at ages 2 or 3 for males and ages 3 or 4 for females (Williams and Brown 1983), and appears to be mainly size- rather than age-dependent. Female walleye will deposit between 25,000-40,000 eggs per pound of body weight (Becker 1983).

Spawning occurs at night and usually involves a group of one female and up to two males, or two females and up to six males (Scott and Crossman 1998). Walleye are broadcast spawners and exhibit no parental care. Some form of courtship behavior takes place before spawning (Scott and Crossman 1998); the following description of courtship and spawning behavior is taken from Ellis and Giles (1965).

> *Overt courtship began by either males or females approaching another of either sex from behind or laterally and pushing sideways against it or drifting back and circling around pushing the approached fish backwards. The first dorsal fin was alternately erected and flattened during these approaches. The approached fish would either hold position or withdraw. Approaches and contact of this sort appeared to be the preliminary essentials of courtship and were promiscuous, i.e., there was no continued relationship between any particular pair of fish. Activity increased in frequency and intensity and individuals began to make preliminary darts forward and upward. Finally one or more females and one or more males came closely together and the compact group rushed upward. At the surface the group swam vigorously around the compound until the moment of orgasm when swimming stopped and the females frequently turned or were pushed violently onto their sides. This sideways movement by the females was taken as an indicator of spawning even when no eggs or milt were seen. On one occasion during orgasm a male was clearly seen to have the first dorsal fin fully erected.*

When spawning is ready to take place, the group heads to shallow water (Scott and Crossman 1998). Most females release the majority of their eggs in one night, while males can spawn over a longer period (Ellis and Giles 1965). The egg diameter is 0.05-0.08 in (1.5-2.0 mm) and they have an adhesive surface (Scott and Crossman 1998). After release, the eggs attach to one another and to adjacent vegetation or streambed material. After an hour or two, they water-harden, lose their adhesive properties, and settle onto weedmats or drop into crevices in the substrate (Scott and Crossman 1998) for protection from predators.

Adult walleye are predominantly piscivorous, but are opportunistic feeders and will consume crustaceans and insects if the opportunity is presented (Gray et al. 1984; Zimmerman 1999). Suckers, minnows, sculpins, and salmonids are the walleye's most important prey items (Gray et al. 1984; Zimmerman 1999) (Table 7-2), but they also will become cannibalistic if prey is scarce (Scott and Crossman 1998).

Table 7-2. Prey items of walleye from lower Columbia River.

Scientific name	Common name
Family/*Genus species*	
Catostomidea/*Catostomus sp.*	suckers
Cottitdae/*Cottus sp.*	sculpins
Cyprinidae	
Acrocheilus alutaceas	chisel mouth
Mylocheilus caurinus	peamouth
Ptychocheilus oregonensis	northern pikeminnow
Richardsonius balteatus	redside shiner
Percopsidae/*Percopsis transmontana*	sand roller
Salmonidae/*Oncorhynchus sp.*	salmon

7.3.5 Movements

Above Bonneville Dam, walleye move up the reservoir during the spring (March and April) and as summer progresses, move back down the reservoir (Beamesderfer and Nigro 1989). The upriver movement may be a spawning migration (Colby et al. 1979). Walleye also will move into an area below an impassable dam to spawn (Scott and Crossman 1998).

Individual walleye can be highly mobile (Beamesderfer 1989). In 1984–86, mark and recapture studies were conducted in the John Day Reservoir from March to September (Nigro et al. 1985a; Nigro et al. 1985b; Beamesderfer et al. 1987). The range of movement for individual walleye during the entire season was 3 to 70 miles (5 to 113 km), with average daily movement of 0.2 to 1.9 miles (0.4 to 3 km). Beamesderfer and Nigro (1989) stated that 68% of the walleye were recaptured 0.3 miles (0.5 km) from the point of release, and 20% were recaptured at least 3.7 miles (6 km) away.

7.4 Factors Affecting Population Status

7.4.1 Harvest

The reported commercial harvest of walleye in tribal net fisheries between 1993–2002 ranged from 662 to 3,667 lbs. (300 to 1663 kg) per year with a mean of 2,118 lbs. (961 kg) (Table 7-3). Because walleye in the lower Columbia River exhibit highly variable reproductive success (Rieman and Beamesderfer 1988), population size is relatively low (Beamesderfer and Rieman 1988). Additionally, net fisheries are selective for large walleye (Hallock and Fletcher 1991). Tribal harvest therefore remains an important consideration in the management of lower Columbia River walleye populations. However, the overall impact of these commercial fisheries on the lower Columbia River walleye populations remains unknown.

Sport fishing for walleyes has occurred in the lower Columbia River since the early 1980s (Tinus and Beamesderfer 1994). Estimates of harvest and effort for the sport fishery were calculated for the years 1982–93 for Bonneville Pool and below Bonneville Dam from angler surveys (Tinus and Beamesderfer 1994). Unfortunately, surveys were not conducted every year or in every month, nor did they necessarily represent all areas of the impoundment (Tinus and Beamesderfer 1994). However, it is apparent that harvest rates are low because for the years 1982–93, the average minimum harvest of walleye was 423 fish per year (Tinus and Beamesderfer 1994). From 1991–2002, creel survey data for walleye was collected during sturgeon creel surveys at Bonneville Pool and below Bonneville Dam (Dennis Gilliland, WDFW, personal communication; Eric Winthrop, WDFW, personal communication). The data collected at Bonneville Pool (1993–2001) suggests that the low exploitation rate continues (Table 7-4). Although 44% of the total catch was harvested, the catch per unit effort (CPUE) was less than one fish per angler trip (0.77).

The walleye tournaments are catch-and-release fishing with some low-level mortalities, but these tournaments have a negligible effect on the walleye population. CPUE is low (average=0.05 from 1999–2001) and the percent of the fish caught that are released alive is quite high (average=96.8% from 1999–2001).

7.4.2 Supplementation

One hatchery in Washington (Ringold Hatchery) has the facilities for rearing walleye. The walleye population in the lower Columbia River is healthy and there are issues with walleye interaction with salmonids. Therefore, there are no plans for supplementation of walleye in the lower Columbia River.

Table 7-3. Commercial harvest of walleye from the Zone 6 fishery, 1993–2002*.

Year	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.	Total
1993	240	132				298	1,464	659	117			2,910
1994	105	18					105	230	204			662
1995	854	1,858					398	557				3,667
1996	84	410						1243				1,737
1997	783	964					182	553				2,482
1998	618	443					38	211				1,310
1999	238	1,193						119	17			1,567
2000	1,252	1,723		360			64	196				3,595
2001	334	838	251	190	56	67	35	108		19		1,898
2002	296	670	215	59		13	27	80				1,360
Total	*4,804*	*8,249*	*466*	*609*	*56*	*378*	*2,313*	*3,956*	*338*	*19*		*21,188*

* The Zone 6 Fishery is the Columbia River between Bonneville Dam and McNary Dam.

7.4.3 Water Development

7.4.3.1 Dams

Hydropower development affected the walleye population in the Columbia River basin positively. The numbers of walleye in the free-flowing portion of the lower Columbia River are lower than those in the impoundment areas. By creating pools and reducing water flow, the dams have actually created habitat that is more suitable for walleye.

7.4.3.2 Flow Alterations

Flow alterations in the lower Columbia can limit walleye production (Beamesderfer and Nigro 1989; Connolly and Rieman 1988; Corbett and Powles 1986; and Mion et al. 1998). Decreased water flows can decrease habitat suitable for both spawning and rearing, and can strand eggs after spawning. High flows can wash eggs ashore or downstream, can also flush out zooplankton as food for larval walleye, and can displace larval walleye from nearshore and backwater rearing areas. Larval walleye also can suffer in times of high discharge because an increase in suspended sediments and turbulence can damage the fish.

Table 7-4. Walleye harvest, catch, and effort estimates from creel surveys at Bonneville Pool (1993–2001) and below Bonneville Dam (1991–2002)

	Bonneville Reservoir			Below Bonneville Dam		
Year	**Harvest**	**Catch**	**Trips**	**Harvest**	**Catch**	**Trips**
1991						
1992						
1993	82	180	1,009			
1994	206	1,190	797			
1995	852	1,297	1,231			
1996	288	406	653			
1997	60	75	248			
1998	219	415	597			
1999	183	244	702			
2000	127	238	575			
2001	39	676	341			
2002				46	63	
Total	*2,056*	*4,721*	*6,153*			

7.4.4 In-Channel Habitat Conditions

7.4.4.1 Channel Maintenance & Dredging

Below Bonneville Dam, walleye have been found downstream to the limit of the saltwater intrusion zone—normally about 31 miles (50 km) upstream from the mouth of the Columbia River. Under low flows, this area can be as far upstream as 20 miles (33 km) (Jimmy Watts, ODFW, personal communication). It is unclear what the impacts of dredging will have on walleye.

7.4.4.2 Water Quality

Low oxygen levels can have a deleterious affect on walleye and on walleye embryo development (Niemuth et al. 1959; Priegel 1970).

7.4.4.3 Temperature

Walleye can tolerate a wide temperature range (32-90°F [0-32.2°C]), though they prefer the warmest water (77°F [25°C]) (Wydoski and Whitney 1979). Lower water temperatures can inhibit egg and larval development. Higher temperatures will lead to increased metabolism and increased predation.

7.4.4.4 Turbidity

Turbidity probably would benefit walleye; although they prefer clear water (Ali and Anctil 1968), they seem to reach their greatest abundance in large, shallow, turbid lakes (Scott and Crossman 1998). Since walleye have very sensitive eyes, turbidity would reduce the amount of sunlight passing through the water, enabling the walleye to inhabit shallower areas of the lower Columbia where prey items are more likely to occur, and allowing feeding throughout the day instead of only at twilight or during the night (Scott and Crossman 1998).

7.4.4.5 Dissolved Gas

Dissolved gas supersaturation can be detrimental to walleye because the increased gases can create gas bubbles under the skin, fin rays, and gills (Becker 1983). The capillaries within the gills can then become obstructed and blood prevented from flowing through (Becker 1983).

The result would be mortality caused by respiratory failure (Becker 1983). Walleye inhabiting the tailraces below McNary and Bonneville Dams can be subject to an increase in dissolved gases during spillover events.

7.4.4.6 Chemicals

Mercury occurs naturally in aquatic ecosystems and methylated mercury (methylmercury) is highly bioavailable for aquatic organisms. Methylmercury is accumulated quickly, but slowly depurated, which allows it to be biomagnified in higher trophic levels (Beckvar et al. 1996). Fish-eating predators tend to have the highest levels of methylmercury (Beckvar 1996). Methylmercury can affect reproduction, growth, behavior, and development in walleye.

7.4.5 Species Interactions

7.4.5.1 Competition

Little information exists on competition between walleye and other species in the lower Columbia River. However, Scott and Crossman (1998) mention that yellow perch and smallmouth bass (*Micropterus dolomieui*) compete with walleye for food.

7.4.5.2 Predation

Becker (1983) stated that there is little evidence of significant predation on walleye eggs by other species of fish, although it does occur (Colby et al. 1979; Corbet and Powles 1986). If carp are spawning where walleye eggs have been deposited, they can disturb the area (Becker 1983) by dislodging eggs that resettle on the silty bottom where they can die from lack of oxygen. Walleye fry are preyed upon by other fishes and larger invertebrates in the same habitat. They also can be cannibalized by larger walleye (Scott and Crossman 1998). Without many enemies, adult walleye are one of the top predators in their habitat. Predation most likely would occur from fish-eating birds and mammals (Scott and Crossman 1998).

7.5 Status & Abundance Trends

7.5.1 Abundance

Zimmerman and Parker (1995) captured walleye from RKm 137 and above. In July 1982, NMFS field personnel using a beach seine caught a walleye at Jones Beach (RKm 75) (Dawley et al. 1985). Walleye abundance for Bonneville Pool and below Bonneville Dam has not been estimated, and Zimmerman and Parker (1995) were unable to calculate density indices for walleye. However, extrapolations from research conducted on the John Day Pool give insight to the abundance of walleye in the Bonneville Pool (Steve Jackson, WDFW, personal communication). Therefore, walleye abundance in the Bonneville Pool is probably similar to that of the John Day Pool, estimated during 1983–86 at 15,000 fish (Tinus and Beamesderfer 1994).

7.5.2 Productivity

The lower Columbia River walleye population is self-sustaining (Tinus and Beamesderfer 1994) and the carrying capacity of the lower Columbia River walleye habitat is unknown. The condition of lower Columbia River walleye was evaluated by calculating relative weights (Tinus and Beamesderfer 1994) and the mean relative weight was 99%. An analysis of 113 walleye populations in 27 states and Canadian provinces (Murphy et al. 1990) revealed that 1/20 of these populations had a mean relative weight greater than 99%. Successful recruitment coincides with years of lower than average flows, while poor recruitment coincides with years of higher than average flows (Connolly and Rieman 1988).

7.5.3 Supplementation

There are no supplementation programs or efforts in the lower Columbia River.

7.5.4 Harvest

The current sport fishery harvest regulations for walleye in the lower Columbia River (Bonneville and below Bonneville Dam) is a 10 fish limit with no more than 5 fish over 18 inches and no more than one fish over 24 inches. Exploitation rates for the walleye sport fishery are low (Tinus and Beamesderfer 1994). The mean harvest per unit effort (fish per hour) for walleye below Bonneville Dam (from the dam to 35 miles downstream) from 1982 to 1993 was 0.322 and for Bonneville Pool, 0.085 (Tinus and Beamesderfer 1994). Creel survey data collected by WDFW from Bonneville Pool from 1993–2001 also suggests the low exploitation rate is continuing.

Since walleye have become established in Washington, fishing tournaments have become popular. The first recorded walleye tournament was held in 1994 and the first walleye tournament held on the lower Columbia River was in 1999 below Bonneville Dam (Divens 2001).

As walleye populations expanded into the lower Columbia River reservoirs where treaty tribes traditionally operated net fisheries for salmon and steelhead, walleye were caught and sold by tribal fishermen (Fletcher 1987). The harvest and sale of walleye taken in tribal fisheries first became a concern for both state and tribal fish managers in the mid-1980s. The issue was addressed in 1988 as part of the ongoing negotiations under US v Oregon. At that time, the court approved a settlement among Oregon, Washington, and Columbia River treaty tribes known as the Columbia River Salmon Management Plan. As part of this agreement, the right of treaty tribes to sell walleye caught incidental to legally-authorized fisheries for salmon and steelhead was affirmed (Fletcher 1987).

7.6 Inventory & Assessment of Existing Management Plans

Draft Warmwater Fish Management Plan (currently for WDFW internal discussion only).

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Volume III, Chapter 8
Smallmouth Bass

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8.0 Smallmouth Bass *(Micropterus dolomeiui)*

8.1 Introduction

Smallmouth bass *(Micropterus dolomeiui)*[1] belong to the order Perciformes and family Centrarchidae (the sunfishes). Perciformes can be found throughout North America and Europe (Scott and Crossman 1998). More than 70 families of fishes comprise this order. The family Centrarchidae contains 10 genera and 30 species and they are normally grouped as sunfishes, crappies, and basses (Scott and Crossman 1998). The dorsal fin is made up a spiny-rayed portion and a soft-rayed portion—distinctive to centrarchids (Scott and Crossman 1998). Unlike percids, the two parts of the dorsal fin are closely connected. Some Centrarchids are the most colored and attractive North American warmwater fishes (Scott and Crossman 1998). Centrarchids were originally restricted to North America but have been introduced elsewhere. They inhabit slow-moving streams and the shallow areas of warm, rocky, and vegetated lakes (Scott and Crossman 1998).

The original North American distribution of smallmouth bass extended from Minnesota and southern Quebec, south to the Tennessee River drainage, and as far west as eastern Oklahoma (Becker 1983). Because of their popularity with anglers, smallmouth bass have been extensively transplanted throughout the continental United States, and are now found in virtually every corner of the mainland US, with the single exception of the extreme southeast (Becker 1983) (Figure 8-1).

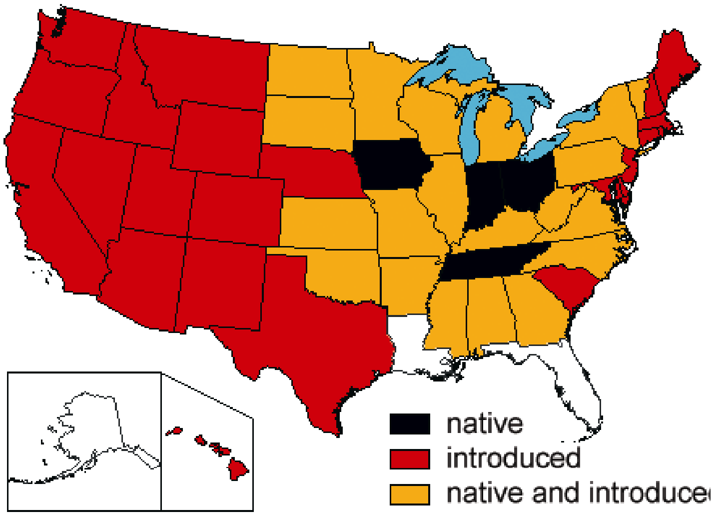

Figure 8-1. Range of smallmouth bass in the US (map courtesy of USGS)

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[1] The illustration of the smallmouth bass at the beginning of this chapter is by Virgil Beck, courtesy of the Wisconsin Department of Natural Resources.

The first documented introduction of smallmouth bass in Washington occurred in 1924, when a shipment of smallmouth arrived from the "east" and was released by a private individual into a small lake on Blakely Island in the San Juans (Lampman 1946). Other early plants were made into Lake Washington in 1925 and into the Yakima River in 1926 (Lampman 1946). Since then, smallmouth have become widely distributed across Washington, and significant populations are now found in a fairly large number of Washington streams and lakes, including the Columbia, Snake, Yakima, Okanogan, and Grande Ronde Rivers, and Lake Washington, Lake Sammamish, Lake Whatcom, Lake Stevens, Lake Osoyoos, Moses Lake, Potholes Reservoir, and several other smaller lakes on either side of the Cascade Mountains (Wydoski and Whitney 1979) (Figure 8-2). WDFW has expanded the range of smallmouth bass in Washington through a program of selective transplantation aimed at increasing fishing opportunity and success rates for this highly-prized sport fish. In recent years, smallmouth bass have been successfully transplanted into Banks Lake (1981), Mayfield Lake (1982), Lake Whatcom (1983), Palmer Lake (1983), and Lake Goodwin (1984) (Fletcher 1986).

It is important to have properly managed smallmouth bass populations in Washington to satisfy a growing public demand for recreational fishing opportunities and harvest. There is concern that smallmouth bass may negatively impact other native species, specifically salmonids. Smallmouth bass and salmonids have overlapping habitats. However, in 1985 WDFW completed an exhaustive evaluation of the interaction between smallmouth bass and native salmonid populations in the Northwest. Fletcher (1991) found that there was no clear evidence of reduced salmonid survival as a result of smallmouth bass interaction. However, smallmouth bass may negatively impact other native fish species.

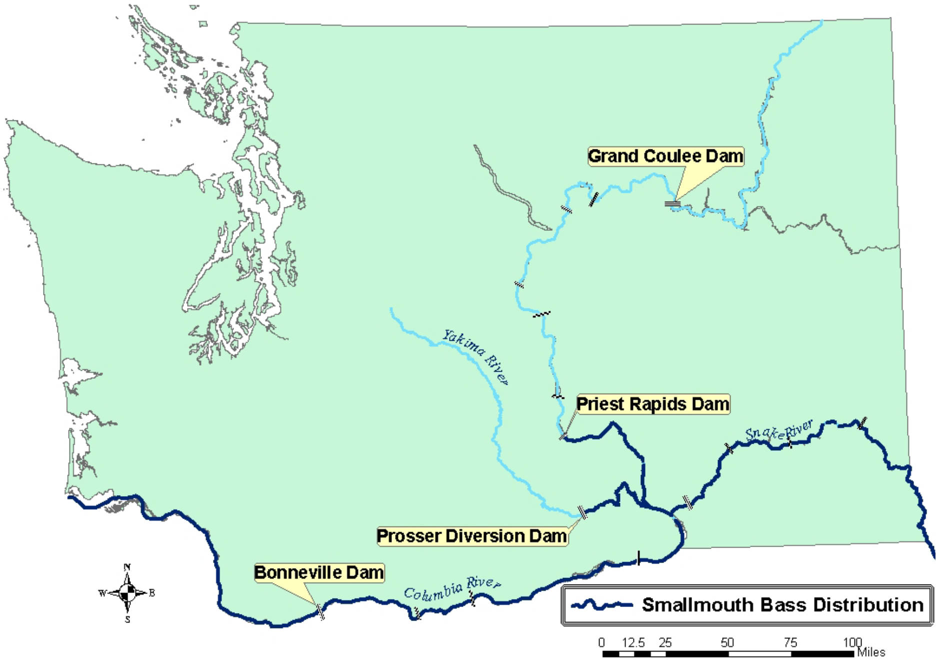

Figure 8-2. Distribution of smallmouth bass in the Columbia and Snake Rivers.

Before the 1930s, managing smallmouth bass and other non-indigenous warmwater game fish species primarily aimed at increasing their availability to Washington anglers. During this period, little was known of the biology, life history, or habitat requirements of this recently-introduced species, or how it was adapting to Washington's cooler water ecosystems.

Before 1935, bass fishing regulations were established by individual county governments and varied widely (Zook 1993), ranging from no catch or size limit to surprisingly conservative regulations. These included spring spawning closures and somewhat restrictive catch and size limits. Statewide fishing regulations were not established for bass and other warmwater game fish species until 1935, a year after the Department of Game was established (Zook 1993).

From 1935-50, the catch and possession limits for bass in Washington were 20 fish, not to exceed 10 pounds plus one bass.

The subsequent history of bass fishing regulations is summarized in Table 8-1. Current statewide harvest regulations for smallmouth bass went into effect in April 2001. They include a daily catch limit of five bass, only bass less than 12 inches (30 cm) or greater than 17 inches (43 cm), with no more than one bass over 17 inches (43 cm). The possession limit allows for two daily catch limits to be retained. There is no minimum size limit for bass.

A small, but growing body of lakes and rivers in the state are managed with special harvest regulations for bass. These regulations often include a slot size limit (bass between either 12-15 inches [30-38 cm] or 12-17 inches [30-43 cm] being protected from harvest), catch and release stipulations, or restrictive size and/or daily limits. The objective of protected length or slot limit regulation is to increase the number of larger bass in the population, for either recreational or predation reasons (Gablehouse 1986).

Sport fishing regulations covering the lower Columbia River have essentially followed the statewide regulations (Table 8-1). Whereas the statewide regulations changed in 2002, the lower Columbia regulations have remained virtually unchanged since mid-April 1992. Only the possession limits have changed.

8.2 Life History Requirements

8.2.1 Spawn Timing & Conditions

Smallmouth spawn along shoreline areas of large rivers and lakes, preferring gravel and rubble, but also utilizing sand and large rock as spawning substrate when necessary (Becker 1983). Spawning activity is normally initiated when the water temperature reaches the 60-65°F (15.6-18.3°C) range. However, smallmouth bass spawning activity has been reported at water temperatures as low as 53°F (11.7°C) (Becker 1983). Most smallmouth bass spawning occurs between mid-May and late June in Washington (Fletcher 1982). Spawning is usually located in 2-5 feet (0.6-1.5m) of water and adjacent to a log, boulder, or other submerged cover.

8.2.2 Incubation

The number of smallmouth bass eggs per nest ranges from 2,000 to 10,000. The eggs are a light amber to pale yellow in color, demersal, adhesive, and 1.2-2.5 mm in diameter (Scott and Crossman 1998). It normally takes somewhere between 9.5 days at 55°F (12.8°C) to 2.25 days at 75°F (23.9°C) for eggs to hatch. Flooding, which results in a rapid drop in water temperature and/or excessive siltation, and excessive lowering of the water level during spawning are the two most common habitat-related reasons for reproductive failure (Becker 1983).

Table 8-1. History of bass sport fishing regulations for Washington and the lower Columbia River.

Year	Statewide Daily Bag Limit	Size		Lower Columbia River
		Minimum	Maximum	
1935	20 fish not to exceed 10 lbs. plus one bass	none	none	same as statewide regs.
1951	15 fish not to exceed 10 lbs. plus one bass	none	none	same as statewide regs.
1956	none	none	none	same as statewide regs.
1963	10 fish not to exceed 20 lbs. plus one bass	none	none	same as statewide regs.
1976	10	none	no more than 3 > 17"	same as statewide regs.
1980	10 fish possession limit	none	no more than 3 > 17"	same as statewide regs.
1992	5	none	no more than 3 > 15"	same as statewide regs.
2001	5	none	no more than 3 > 15"	same as statewide regs.
2002	5	only fish <12" and >17" with no more than one bass >17"		no more than 3 > 15"

8.2.3 Larvae & Juveniles

Newly-hatched smallmouth bass fry may remain in a tightly grouped ball in the nest for up to 15 days after hatching, after which they leave the nest and begin to disperse. Young smallmouth bass begin their carnivorous existence very early in life, feeding on midge larvae, Daphnia, and other small crustaceans even before their yolk-sac is completely absorbed. The yolk-sac is absorbed in approximately 6-15 days (State of Iowa DNR, 2001). By the time they reach 3 inches in length, juvenile smallmouth bass are actively feeding on crawfish, other bass fry, and almost any other suitably-sized life form that swims or floats (Becker 1983).

8.2.4 Adult

Smallmouth bass prefer medium to large rivers and large, clearwater lakes (Becker 1983). They are most often associated with gravel or rocky substrate, but thriving populations of smallmouth bass are found in a number of Washington waters with little or no rocky habitat, most notably Lake Sammamish (Wydoski and Whitney 1979). In these cases, rooted aquatic vegetation and other forms of natural or artificial cover play an especially important role in providing the required spawning, rearing, and feeding cover (Becker 1983). Rock outcroppings, boulders, logs, aquatic vegetation, and in some cases, artificial structures such as bridge pilings and boat docks protect juveniles from predation and concentrate forage for feeding adults. Smallmouth bass prefer water temperatures in the 70-80°F (21.1-26.7°C) degree range (Wydoski and Whitney 1979). During the long hot days of summer, smallmouth bass will seek deeper, cooler water (Scott and Crossman 1998). In the lower Columbia River above Bonneville Dam, smallmouth bass can be similarly distributed throughout the reservoir (Zimmerman and Parker 1995). Below the dam, smallmouth bass are fewer in number and are basically found down river as far as RKm 71 (Zimmerman and Parker 1959). Smallmouth bass have a home range and do not travel long distances (Wydoski and Whitney 1979; Scott and Crossman 1998; Becker 1983).

The diet of adult smallmouth bass consists primarily of fish, crawfish, and aquatic insects. Of the fish species eaten, various cyprinids, perch, and sunfish are the most common fish species consumed by adult smallmouth bass (Becker 1983). In the lower Columbia River, smallmouth bass primarily preyed upon sculpins, cyprinids, suckers, and sand rollers (Zimmerman 1999). Crayfish were the most important non-fish food item consumed by smallmouth bass (Ward and Zimmerman 1999). Zimmerman (1999) also found that, during the spring and summer, smallmouth bass consumed more fish in the lower Columbia River below Bonneville Dam than above. Smallmouth bass also prey upon juvenile salmonids (Gray et al. 1984; Beamesderfer and Rieman 1988; Connolly and Rieman 1988; Rieman and Beamesderfer

1988; Rieman et. al. 1991; Ward and Zimmerman 1999; Zimmerman 1999). Salmonid consumption by smallmouth bass was somewhat similar below and above Bonneville Dam (Zimmerman 1999).

The growth rate of Washington smallmouth bass is, on average, below that reported for most areas within the native range for the species (Fletcher et al. 1993; Becker 1983). However, growth rates in Washington exhibit a high degree of variability, and in some areas of the state—including portions of the Snake and Columbia Rivers—growth exceeds the reported averages for those areas. On average, smallmouth bass in Washington attain a length of 2-4 inches (5-10cm)at age 1, 3-7.5 inches (8-19 cm) by age 2, 4.5-11 inches (11-28 cm) by age 3, 6-13.5 inches (15-34 cm) at age 4, 10-15 25-38 cm) inches by age 5, 12-15 inches (30-38 cm) at age 6, 13-17 inches (33-43 cm) at age 7, and 14-18.5 inches (35-46 cm) by age 8. Smallmouth bass have attained a reported maximum age of 13 years and weight of 8 pounds, 12 ounces in Washington (Fletcher 1982).

Smallmouth bass normally attain sexual maturity at a length of 10-12 inches, between three and five years of age for most Washington waters (Fletcher 1982). Spawning activity is normally initiated when the water temperature reaches the 60-65°F (15.6-18.3°C) range. However, smallmouth bass spawning activity has been reported at water temperatures as low as 53°F (11.7°C) (Becker 1983). Most smallmouth bass spawning occurs between mid-May and late June in Washington (1982).

Smallmouth spawn along shoreline areas of large rivers and lakes, preferring gravel and rubble, but also utilizing sand and large rock as spawning substrate when necessary (Becker 1983). Spawning usually takes place in 6-10 days (Scott and Crossman 1998). The male selects and constructs the nest, usually located in 2-5 feet (0.6-1.5m)of water adjacent to a log, boulder, or other submerged cover. The nest is approximately 2-6 feet (0.6-2m) in diameter. Scott and Crossman (1998) report that some males will return to the same nest in subsequent years and that over 85% return to within 150 yards of where they nested the previous year. A lot of nest building takes place in the early morning. After it has been built, the male awaits a female (Becker 1983). Female smallmouth may spawn in more than one nest and with several different males. Ordinarily, male smallmouth spawn with only one female at a single nest site (Becker 1983). After spawning—a process lasting from 15 minutes to over 2 hours—the female leaves the nest while the male remains to care for the eggs. The male smallmouth usually guards the nest for 2-8 days after hatching, but there are times when the male will continue to guard the brood for up to 30 days after hatching (Becker 1983). Predation by various species of cyprinids, catfish, and sunfish (same and other species) is common and in some situations results in the total loss of eggs or fry (Becker 1983).

8.2.5 Movements

Smallmouth bass are essentially non-migratory and the adults rarely school (Becker 1983). The majority of studies have revealed that smallmouth limit their movements to 0.83-8.33 Km (Scott and Crossman 1998). Mark and recapture studies done on smallmouth bass in John Day Reservoir (Nigro et al 1984; Nigro et al. 1985) also found that smallmouth bass exhibited limited movement. Smallmouth are most active at dawn and dusk (Becker 1983; Todd and Rabeni 1989). Their movements respond to spawning, wanting to remain in their preferred temperature range, prey availability, and cover (Horning II and Pearson 1973; Becker 1983; Todd and Rabeni 1989; Scott and Crossman 1998). They will seek deeper depths during the day to avoid bright light and find a more tolerable water temperature. During the winter, smallmouth will seek deeper depths and become semidormant (Becker 1983; Scott and Crossman 1998).

8.3 Status & Abundance Trends

8.3.1 Abundance

Zimmerman and Parker (1995) found similar densities of smallmouth bass in the forebay, mid-reservoir, and tailraces in lower Columbia River impoundments. Densities of smallmouth bass were lower in the free-flowing portion of the Columbia River and they also were found as far as RKm 71. In recent creel surveys conducted by ODFW, anglers reported catching smallmouth on the downstream side of Puget Island (Jimmy Watts, ODFW, personal communication), just downstream of the location reported by Zimmerman and Parker (1995). Because of saltwater intrusion, smallmouth bass are probably not found much farther downstream than Puget Island (Jimmy Watts, ODFW, personal communication).[2]

8.3.2 Productivity

The carrying capacity of the lower Columbia River smallmouth bass habitat is unknown. Parker and Zimmerman (1995) reported that the factors affecting smallmouth bass proliferation in the Columbia and Snake Rivers had not been studied. Since smallmouth bass are not stocked in the lower Columbia River and they continue to be caught and harvested recreationally, there must be some natural reproduction occurring.

8.3.3 Supplementation

There are no supplementation programs or efforts in the lower Columbia River.

8.3.4 Harvest

There is no commercial harvesting of smallmouth bass. Current sport fishery harvest regulations for smallmouth bass in the lower Columbia River (Bonneville Pool and below Bonneville Dam) include a limit of five fish with no more than three fish over 15 inches in length. Exploitation rates for the smallmouth bass fishery are very low. Data from WDFW's Volunteer Angler Diary Program revealed that from 1993-2002, three smallmouth bass fishing trips were taken in the lower Columbia River; a combined total of three fish (data was only available for fish ≥10") were caught in a combined effort of 14 hours for a catch per unit effort (CPUE) of 0.21. The state average for CPUE for smallmouth bass for the 11-year period from 1990-2001 was 0.59.

Competitive fishing is not unique to the bass fishery. Fishing contests are very visible and important to the sport of bass fishing in North America. On the national level, competition is almost synonymous with the sport for many anglers. However, in Washington, the level of competitive bass fishing by comparison is relatively subdued. Most bass clubs sponsoring tournaments in Washington believe fishing contests are for enjoyment and to promote the sport and conservation ethic.

The results of bass fishing contests held in Washington since 1978 show a low of 38 contests in 1983 and a high of 178 in 2002. As the number of bass fishing contests began to increase, fish managers and some anglers began to be concerned about their potential impacts. In 1984, WDW undertook a study and concluded that bass fishing contests—at the current or projected level of future activity—did not have a significant impact on Washington's bass resources (Fletcher 1986).

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[2] As far upstream as RKm 33 daily and, under low flows, as far upstream as RKm 50.

In 1987, WDW changed fishing contest rules and began to require permits for all clubs and/or sponsors. Most contests target *bass* rather than smallmouth bass. Since 1987, 41 fishing contests have been held in the lower Columbia River—all but one below Bonneville Dam—and 31 reported catching smallmouth bass (Table 8-2). The yearly number of contests ranged from 0-9. Although these tournaments are catch and release fishing, some mortality occurs (99.8% of the fish released alive). Compared to other fishing contents around the state, the CPUE for the lower Columbia River is very low. For 1990-2001, the CPUE for the lower Columbia River was 0.10, while for the rest of the state it was 0.59.

8.4 Factors Affecting Population Status

8.4.1 Harvest

8.4.1.1 Tournament Fishing

Since 1987, 41 bass fishing contests (registered with WDFW) have been held in the lower Columbia River (Table 8-2). Smallmouth bass were caught in 34. Almost all were released alive. The CPUE ranged from 0.01 to 0.24 fish per hour. The catch rate does appear to be lower in the lower Columbia River than in other parts of the Columbia River basin, and is fairly comparable to the average catch rate for the entire state. For example, in 2001, the CPUE for the lower Columbia River was 0.17 and the statewide average was 0.20, while the average for five Columbia and Snake River impoundments was 0.29.

Table 8-2. Bass fishing contests held in the lower Columbia River, 1987-2001.

Year	Contests	Smallmouth Bass Caught	Fish Caught	Pounds	% Released	CPUE
1987	0	0				
1990	5	2	20	1.5	100	0.05
1991	2	1	1	2.5	100	0.01
1992	4	2	42	7.8	100	0.04
1993	2	1	24	36.4	100	0.24
1994	2	2	3	1.9	100	0.01
1995	1	1	2	4.6	100	0.01
1996	3	3	37	65.8	100	0.12
1997*	3	3	188	353.2	100	0.15
1998	2	2	15	9.0	100	0.07
1999	1	1	1	1.0	100	0.01
2000	9	9	307	343.3	98	0.09
2001	7	7	190	381.4	100	0.17
Total or Average	41	34	830	1208.4	99.8	0.08

* one tournament was held in Bonneville Pool

8.4.1.2 Sport Fishing

From 1991 to 2002, creel survey data for smallmouth bass was collected during sturgeon creel surveys at Bonneville Pool and below Bonneville Dam (Dennis Gilliland, WDFW, personal communication; Eric Winthrop, WDFW, personal communication). The data collected at Bonneville Pool (1993-2001) suggests that the low exploitation rate continues (Table 8-3). A total of 31,981 smallmouth bass were caught in the sport fishery from 10,237 trips during 1993-2001 in Bonneville Pool by Washington anglers. Smallmouth were not caught in three years (1997, 1999, and 2000). The reported harvest was 6,410 fish (range 5-28%, average = 20%) and the average CPUE was 3.12 fish per trip (range 0-4.63).

Table 8-3. Smallmouth bass harvest, catch, and effort estimates from creel surveys conducted at Bonneville Pool (1993–2001) and below Bonneville Dam (1991–2002).

	Bonneville Pool			Below Bonneville Dam		
Year	Harvest	Catch	Trips	Harvest	Catch	Trips
1991				*	*	*
1992				*	*	*
1993	1,764	6,365	1,993	*	*	*
1994	1,739	8,543	2,782	*	*	*
1995	1,227	10,784	2,330	*	*	*
1996	767	2,808	1,384	*	*	*
1997	0	0	102	*	*	*
1998	9	198	237	*	*	*
1999	0	0	118	*	*	*
2000	0	0	15	*	*	*
2001	904	3,283	1,276	*	*	*
2002				295	1,686	
Total	6,410	31,981	10,237	295	1,686	

* Data was not available for inclusion in this report

8.4.2 Supplementation

There is one hatchery in Washington State (Meseberg Hatchery) with the facilities for rearing smallmouth bass. The smallmouth bass population in the lower Columbia River is self-sustaining and there are issues with the interaction of bass with salmonids. Therefore, there are no plans for supplementation of smallmouth bass in the lower Columbia River.

8.4.3 Water Development

Dam construction has inadvertently created habitat suitable for smallmouth bass. The once free-flowing Columbia is now a series of large slow-flowing pools. Preferred habitat structures became submerged and water temperatures have risen toward those favored by bass.

8.4.4 Flow Alterations

Two life stages of smallmouth bass are most vulnerable to current flows (Simonson and Swenson 1990). Of the two, the fry stage is probably the more sensitive to increases in velocity. While fry are developing in the nest, high velocities can displace them and result in catastrophic mortalities. When the young have dispersed from the nest and become independent, their first year survival also can be greatly affected. Shuter et al. (1980) found that growth during the first summer was important for overwintering survival. High stream flows can lower foraging activity because the smallmouth bass can be forced to occupy areas that are subpar for feeding and growth. Because they may have to expend more energy for positioning, their fitness could be suboptimum for overwinter survival and increased mortality could result. Increased flows also could lead to an increase in turbidity. Rankin (1986) observed that when the current velocity increased, smallmouth bass decreased their foraging behavior and just moved through the habitat instead.

8.4.5 In-Channel Habitat Conditions

As mentioned above, smallmouth bass prefer inhabiting clear lakes and streams; turbidity would affect their presence or absence in a given body of water. Although dredging probably will have a negative affect on smallmouth bass, its impact on smallmouth bass population(s) is uncertain.

8.4.6 Water Quality

Smallmouth bass prefer inhabiting waters with pH values greater than 6.3 (Johnson et al. 1977). In waters where the pH value falls between 5.5 and 6.0, reproduction is limited or altogether absent (Kane and Rabeni 1987). Waters with pH levels less than 5.5 can impact smallmouth bass densities or prevent their colonization (Kane and Rabeni 1987). YOY smallmouth bass are less pH tolerant and prone to slower growth rates and overwintering mortalities in acidified environments (McCormick and Leino 1999).

8.4.6.1 Temperature

Long term (> 7 days) exposure of smallmouth bass juveniles to temperatures below 5°C impairs survival (Horning II and Pearson 1973), and mortality occurs at temperatures above 36°C (Mundahl 1990). During the summer, juvenile smallmouth bass prefer to inhabit areas that are slightly warmer than the preference of adult bass (Barans and Tubbs 1973; Horning II and Pearson 1973; Coutant 1977). Water temperature fluctuations can impact the growth and distribution of smallmouth bass (Hubert and Lackey 1980; Wrenn 1980; Serns 1982; Shuter et al. 1985; King et al. 1999; Zweifel et al. 1999; Stefan et al. 2001). Patton and Hubert (1996) found growth was impaired at 20°C and Oliver et al (1979) found that at 10°C, the fish became inactive and stopped feeding. Sowa and Rabeni (1995) observed that the density of lotic fish greater than 100 mm (TL) decreased as maximum summer temperatures increased from 22-34°C. In some circumstances, smallmouth bass will remain in an area with a water temperature above their comfort range if food and cover are available (Bevelhimer and Adams 1991).

8.4.6.2 Turbidity

Smallmouth bass generally prefer clear water with Secchi depths greater than 3 meters (Johnson et al. 1977). Turbidity will affect their presence or absence in or within a given body of water (Bayley and Peterson 2001). Smallmouth bass prefer water with nephelometric turbidity unit (NTU) readings of 1.6 or less. Smallmouth bass are somewhat intolerant of waters having NTU readings between 1.6 and 4.0 (Whittier and Hughes 1998). High levels of turbidity can reduce the visual search range of smallmouth bass (Paragamian and Wiley 1987), which would decrease foraging efficiency and result in poor growth.

8.4.6.3 Dissolved Gas

At the northern end of its range, smallmouth bass inhabit lakes with dissolved oxygen (DO) levels of 2.9 to 6.5 mg/L (Johnson et al. 1977). The early life stages of smallmouth bass are most susceptible to low DO levels. The survival of swim-up fry may decrease by 20% if DO levels are around 4 mg/L and the water temperature is 20°C. If the DO levels were to decrease further to 2.5 mg/L, almost all the swim-up fry would die within one week of exposure. The hatching of smallmouth bass would be averted when DO levels fall below 1.5 mg/L (Siefert et al. 1974).

8.4.6.4 Chemicals

Acidification of water causes aluminum to be released from the sediments (Horne and Goldman 1994), and this can result in deformities, reduced activity, and abnormal swimming behavior, which in turn would increase the susceptibility of smallmouth bass to natural mortality (Kane and Rabeni 1987).

There is also the possibility of bioaccumulation of methylmercury. Since smallmouth bass are piscivorous, there is the possibility of bioacummulating methylmercury from predation on fish. Please refer to section .1.1.1.4.6.4 in the walleye species overview chapter.

8.4.7 Species Interactions

8.4.7.1 Competition

The smallmouth bass has the opportunity to interact intra- and interspecifically at various levels of its life history for food and spawning sites (Vander Zanden et al. 1997; Scott and Crossman 1998). Competing for food with other fishes is not a serious limiting factor (Scott and Crossman 1998). Juvenile smallmouth bass may compete with other non-game fishes or even themselves for food and shelter (Schlosser 1987; Easton and Orth 1992).

8.4.7.2 Predation

The fry and juvenile stages of smallmouth bass are susceptible to predation by older smallmouth bass, other species of fish (i.e., northern pikeminnow Zimmerman [1999]), crayfish, frogs, and birds (Beck 1983).

8.5 Inventory & Assessment of Existing Management Plans

Draft Warmwater Fish Management Plan (currently for WDFW internal discussion only).

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Volume III, Chapter 9
Channel Catfish

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9.0 Channel Catfish *(Ictalurus punctatus)*

9.1 Introduction

Channel catfish *(Ictalurus punctatus)*[1] belong to the order Siluriformes and family Ictaluridae (bullhead catfishes). Ictalurids are representative of exclusively soft rayed families of North American origin (Becker 1983). In the United State and Canada, there are six genera and 39 species of bullheads (Robins et al. 1991). A spinous ray in the dorsal and pectoral fins (Becker 1983) eight barbels, which look like whiskers and are sensitive to touch and chemical stimuli (Becker 1983). In Washington, the channel catfish is the only ictalurid that has a deeply forked tail (Wydoski and Whitney 1979).

Channel catfish are restricted to fresh waters and to some, brackish waters (Scott and Crossman 1998). The original North American distribution of channel catfish appeared to be from the St. Lawrence River and its tributaries in Quebec, south along the western Appalachian Mountains to southern Georgia and Florida (Lake Okeechobee), west through the Gulf states to eastern Texas and northern Mexico, and northwest throughout eastern New Mexico to Montana (Missouri River drainage), east to the Red River system in Manitoba, southwestern Ontario, southern Minnesota, Wisconsin, and Michigan through Ontario and Quebec at the level of Lake Nipissing (Wydoski and Whitney 1979; Scott and Crossman 1998) (Figure 9-1). Channel catfish have been widely introduced outside this native range and can be found almost everywhere in the United States (Scott and Crossman 1998).

Channel catfish reportedly were first introduced into Washington in 1892 at Clear Lake (Lampman 1946). Skagit County stocked into an unnamed privately owned farm pond near Vancouver and into Deer Lake in Spokane County (Lampman 1946). Additional releases of channel catfish were made in various lakes and streams across the state in the ensuing years, as all forms of catfish (predominately bullheads) became abundant and popular in the region with sport and commercial anglers alike (Lampman 1946).

While bullhead catfish thrived in many of Washington’s lakes and streams following these early introductions, channel catfish abundance and distribution was limited by their very specific spawning habitat requirements and the region’s generally low water temperatures. Today, naturally reproducing populations of channel catfish are found only in the Snake, Columbia, and lower reaches of the Yakima (Prosser Dam to the mouth), Tucannon, and Walla Walla Rivers (Doug Fletcher, WDFW, personal communication; Paul Hoffarth, WDFW, personal communication) (Figure 9-2).

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[1] The channel catfish illustration at beginning of this chapter is courtesy of the Michigan Department of Natural Resources, Fisheries Division.

Figure 9-1. North American distribution of channel catfish (map courtesy of Worldwaters.com, Bend, Oregon).

In the last decade, WDW has introduced channel catfish into a number of Washington lakes, attempting to increase predation on overabundant forage fish populations and/or add diversity to mixed-species fisheries. Plans to expand this program are currently under consideration. Of the 30 lakes planted in Washington since 1982, Fazon Lake, Sprague Lake, and McCabe Pond (Kittitas County), Swofford Pond (Lewis County), Gissberg Ponds (Snohomish County), Harts Lake (Pierce County), and Kress Lake (Cowlitz County) have produced the most significant fisheries (Steve Jackson, WDFW, personal communication).

Prior to the 1930s, the management of channel catfish and other non-indigenous warmwater game fish species in Washington was primarily directed at increasing their geographic distribution, availability to anglers, and, in some cases, in regulating harvest. During this period, little was known of the biology, life history, or habitat requirements of these species in Washington's coldwater streams and lakes.

9.2 Life History & Requirements

9.2.1 Spawn Timing & Conditions

Channel catfish spawning is initiated when water temperatures reach 75°F, with 80°F the optimal spawning temperature (Becker 1983).

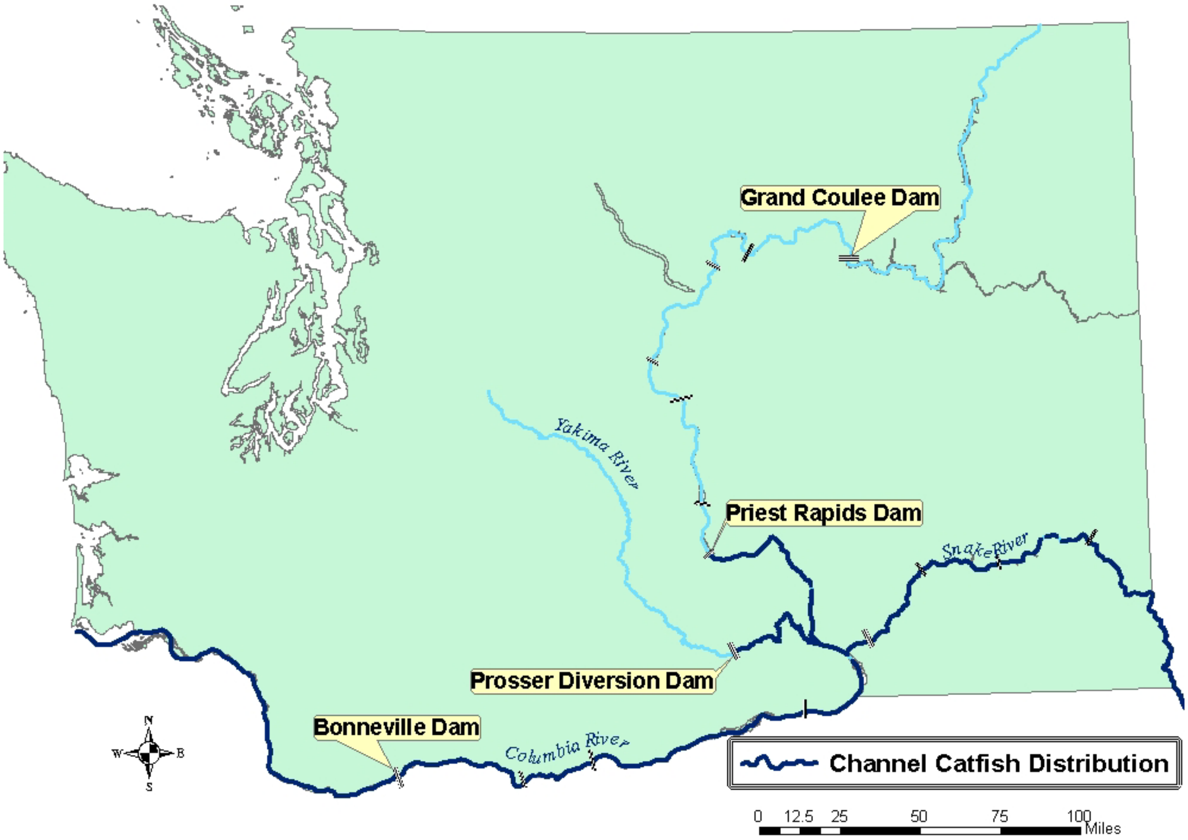

Figure 9-2. Distribution of channel catfish in the Columbia River basin. Distribution below Bonneville Dam is questionable.

Channel catfish are true cavity spawners. A dark cavity or crevice in the substrate, hollowed log, undercut bank, overhanging rock ledge, or similar habitat is required for successful reproduction. The clearer the water, the more formidable the nesting cavity must be to provide the apparent security needed to initiate spawning activity (Becker 1983). In very turbid water, nest may be made directly on the bottom in the mud (Becker 1983). A current or rocky substrate is not necessary for spawning (Becker 1983).

9.2.2 Incubation

Channel catfish eggs are demersal and adhesive, sticking to one another a the bottom of the nest. The eggs average 3.5-4.0 mm in diameter before they are laid (Scott and Crossman 1998). At 26.7°C, incubation takes 7-8 days. During incubation, the male will guard the nest and also during this time, will fan the eggs and loosen up the egg mass (Wydoski and Whitney 1979; Becker 1983) is to provide aeration and water circulation.

9.2.3 Larvae & Juveniles

Upon hatching, channel catfish fry remain in a tightly grouped mass at the bottom of the nest. The newly hatched fry have a large yolk-sac and depending on the water temperature, it can take up to 10 days to be fully absorbed, which after, therefore make trips to surface and begin to feed (Scott and Crossman 1998; Wellborn 1988). They rely on the protective cover of the nesting cavity through the early stages of development. The male continues to guard the nest during this time.

Prior to dispersal, the fry may travel in schools anywhere from several days to or for weeks (Mansueti and Hardy 1967).

After dispersal, juvenile catfish begin feeding in quiet, shallow areas, usually over sand bars, around drift piles, or among rocks, before progressively seeking out moderately flowing rocky riffle areas or more turbulent areas near sand bars. They also rely on rock and woody debris for protective cover. Some juveniles seek shelter during the winter under boulders in swiftly flowing water (Becker 1983). Young channel catfish feed primarily on aquatic insects or on bottom-dwelling arthropods. When channel catfish reach a length of 4-5 inches, they become more omnivorous, feeding on a variety of insects, small fish, and plant material (Becker 1983). They rarely feed during the winter (Becker 1983).

9.2.4 Adult

Channel catfish are found in a variety of habitats including clear fast-moving streams, and moderately eutrophic lakes and reservoirs, and are commonly found in large rivers below dams (Scott and Crossman 1998). Adults are usually found associated with boulders or submerged logs or in deep pools during daylight hours, and feeding in the shallows at night (Becker 1983). Protective cover is an especially critical habitat requirement for channel catfish.

Adult channel catfish have been known to forage on an incredible variety of food organisms including: frogs; crawfish; clams; snails; worms; pond weeds; seeds; snakes; and birds; in addition to the more traditional forage of fish, insects, and algae (Becker 1983). They normally detect their food by touch and smell (Becker 1983). Feeding activity is generally greatest at night and channel catfish feed only at water temperatures above 15.6°C. Channel catfish rarely feed in the winter and there is some evidence that the adults do not feed during the breeding season (Becker 1983). Members of the cyprinid (minnow) family are the most common fish species consumed by channel catfish, although they are known to prey on perch, sunfish, and even on salmonids (Becker 1983; Poe et al. 1988).

The age and size at which channel catfish reach sexual maturity vary greatly. No age and growth data is available for channel catfish in Washington, although 7-year old catfish as large as 28 pounds have been taken from ponds stocked by the Department of Wildlife (Doug Fletcher, WDFW, personal communication). The average calculated length at annulus formation (spines) for channel catfish from the Mississippi River was reported to be 3 inches at age 1, 6.5 inches at age 2, 9.5 inches by age 3, 12 inches at age 4, 14.5 inches by age 5, 17 inches by age 6, 19.5 inches at age 7, 21.5 inches by age 8, and 24.5 inches at age 9 (Becker 1983). Few channel catfish live more than 8 years, although large specimens up to fifteen years or older are occasionally taken. The maximum size recorded for channel catfish is 58 pounds, from the Santee Cooper Reservoir in South Carolina. The Washington record sport-caught channel catfish weighed in at 32.5 pounds and was caught from a borrow pit adjacent to the Naches River near Yakima (Doug Fletcher, WDFW, personal communication).

Depending on the habitat, channel catfish may or may not migrate to moving water or rivers to spawn (Scott and Crossman 1998). The male channel catfish selects and prepares the nest by cleaning away debris with vigorous finning action. The male may also secrete mucous to make a smooth surface within the nest (Wydoski and Whitney 1979; Scott and Crossman 1998). The female takes no part in preparing the nest, nor care for the young (Becker 1983). The male waits at the nest for a ripe female to spawn with (Davis 1959). Becker (1983) reports that the largest or oldest fish spawn first, and the smallest fish are the last to spawn. A female channel catfish lays approximately 4,000 eggs per pound of body weight (Becker 1983). Spawning can last up to six hours (Clemens and Sneed 1957). Upon completion of spawning, the male eventually drives the female away (Clemens and Sneed 1957).

9.2.5 Movements

Becker (1983) reports that there are downstream and upstream migrations by channel catfish, although data on channel catfish movements in the lower Columbia River is lacking.

9.3 Status & Abundance

9.3.1 Abundance

Although channel catfish have inhabited Washington waters for more than a century, their abundance and distribution remain very limited. Early transplantation efforts succeeded in establishing self-sustaining populations in only a few areas, and recent attempts to expand their distribution and increase their abundance have only slightly increased their range or numbers (Doug Fletcher, WDFW, personal communication). This species does, however, have the potential to provide additional fishing opportunity in Washington through the use of artificial production.

The low abundance and limited distribution of channel catfish in Washington is primarily attributable to relatively low water temperature and lack of suitable spawning habitat. There are only a few waters in Washington that have both suitable habitat and sustained water temperatures of 23.9°-26.7°C or higher.

Zimmerman and Parker (1995) did not capture any fish below Bonneville Dam, but there have been reports of fish being present in the Multnomah Channel and as far downstream as Puget Island (Jimmy Watts, ODFW, personal communication). Zimmerman and Parker (1995) however, did capture channel catfish above the dam in the (Bonneville Pool). The number of fish captured was very low, and most were found in the tailrace below The Dalles Dam.

9.3.2 Productivity

Because Washington waters do not provide the minimal spawning habitat requirements for successful channel catfish reproduction, population levels are generally low and statewide distribution is limited. Washington’s streams are typically steep, cold, fast moving, and generally devoid of good cavity nesting habitat.

Channel catfish are present but are not stocked in the lower Columbia River, so their presence indicates that some natural reproduction is occurring.

9.3.3 Supplementation

There are no supplementation programs or efforts in the lower Columbia River.

9.3.4 Harvest

There is no commercial harvesting of channel catfish.

Prior to 1983 there were no regulations governing the statewide catch, possession, minimum size limit, or maximum size limit for channel catfish in Washington. In 1983 a four-fish limit was established for Fazon Lake (Whatcom County). In 1988 the daily limit for Fazon Lake was decreased to two fish. A five-fish daily limit was established at Sprague Lake (Adams and Lincoln Counties) in 1987 and was in place until 2000, when it came under statewide rules. In 1998, limits were established statewide (for lakes, ponds and reservoirs), a five-fish daily limit with a minimum size of 12 inches and no more than one fish greater than 24 inches. In 2002, the statewide regulations were modified and the size limits were dropped.

The most productive channel catfish fisheries in Washington are found in areas where natural reproduction occurs, namely the lower Yakima Rivers and throughout the Washington portion of the Snake and mid (downstream of the Tri-Cities area) Columbia Rivers.

9.4 Factors Affecting Population Status

9.4.1 Harvest

There is limited fishing for channel catfish in the lower Columbia River and it is probably restricted to the upper Bonneville Pool. Data from the Volunteer Angler Diary Program has not shown any anglers targeting channel catfish, nor catching any in the lower Columbia River. There is no creel survey information that is specific to channel catfish above Bonneville Dam (Dennis Gilliland, WDFW, personal communication. However, from 1991 to 2002 creel survey data for channel catfish was collected during sturgeon creel surveys conducted below Bonneville Dam (Eric Winthrop, WDFW, personal communication).

Table 9-1. Channel catfish harvest, catch, and effort estimates from below Bonneville Dam creel surveys (1991-2002)

Year	Harvest	Catch	No. Trips
1993	*	*	*
1994	*	*	*
1995	*	*	*
1996	*	*	*
1997	*	*	*
1998	*	*	*
1999	*	*	*
2000	*	*	*
2001	*	*	*
2002	27	210	*
Total	27	210	*

* Data not available for inclusion in this report

9.4.2 Supplementation

Until now, channel catfish needed for stocking purposes have been imported from other states. In the future, it may become cost-effective to establish an in-state source of fingerling channel catfish for stocking projects, depending on the outcome of ongoing research efforts. There is no supplementation of channel catfish in the lower Columbia River, nor is there any plan to do so in the future.

9.4.3 Water Development

9.4.3.1 Dams

Dams may provide increased suitable spawning habitat as well as more favorable water temperatures. Bonneville Pool has a greatly reduced water velocity compared to the lower Columbia River unimpeded section. This may allow the water to warm up to temperatures more favorable for channel catfish. There may be some habitat available to channel catfish that would otherwise be unavailable if it were free flowing.

9.4.3.2 Flow Alterations

Channel catfish can be found in a variety of habitats (Becker 1983). They occur in rivers, lakes, and reservoirs. Channel catfish can be found inhabiting clear, rocky, swift-flowing streams, to slow-flowing, silty streams. They are often found in the tailraces of power dams. Currently, there is insufficient research on channel catfish in the lower Columbia River to determine how they would be affected by flow alterations.

9.4.3.3 Channel Maintenance & Dredging

A study by Zimmerman and Parker (1995) did not find any catfish below Bonneville Dam. If channel catfish are actually present, it is unclear what the impacts of dredging would have on channel catfish.

9.4.4 Water Quality

Low oxygen levels can have a harmful affect on the early life stages of channel catfish (Becker 1983). If oxygen levels are too low, it can affect embryonic and larval development or even be lethal. The lethal or deleterious affects of low oxygen levels are dependent upon temperature (Becker 1983). Adult catfish are more tolerant of waters with below optimal oxygen levels (Becker 1983, Smith 1985.

9.4.4.1 Temperature

In most cases, Washington waters do not provide acceptable temperatures for successful reproduction. Lower temperatures would also result in decreased metabolism and decreased prey consumption. Conversely, warmer temperatures would increase metabolism and result in increased prey consumption. An increase in metabolism could result depending on the time of year, in an increase in consumption of juvenile salmonids (Poe et al. 1988; Vigg et al. 1988).

9.4.4.2 Turbidity

Although catfish prefer clean water, in some instances, turbidity could be beneficial to channel catfish. As previously mentioned, catfish prefer to spawn in cavities, crevices, or some area where there is some protective structure.

9.4.4.3 Chemicals

Since channel catfish are omnivorous, there is the possibility of bioacummulating methylmercury from predation on fish. Mercury occurs naturally in aquatic ecosystems and methylated mercury (methylmercury) is highly bioavailable for aquatic organisms. Methylmercury is accumulated quickly, but slowly depurated, which allows it to be biomagnified in higher trophic levels (Beckvar et al. 1996). Fish-eating predators tend to have the highest levels of methylmercury (Beckvar 1996). Methylmercury can affect reproduction, growth, behavior, and development in walleye. Borton et al. (1996) studied the effect of high-substitution bleached kraft mill effluent (HS BKME) on three species of fish, one of which was channel catfish. In the experiment, channel catfish were exposed to 0, 4, and 8% HS BKME by volume in experimental streams for 263. The channel catfish were not adversely affected by the effluent and their productivity increased.

9.4.5 Species Interactions

9.4.5.1 Competition

The channel catfish probably competes for food with other bottom dwelling feeding animals and with some predatory species of fish in its habitat (Scott and Crossman 1998).

9.4.5.2 Predation

Adult channel catfish do not have many predation issues as a result of their long spines (Becker 1983). However, young channel catfish are vulnerable to predation by insects, other fish, and fish-eating birds. There may be some cannibalism on young catfish by adults 12 inches or greater in length (Bailey and Harrison 1948).

9.5 Inventory & Assessment of Existing Management Plans

A draft Warmwater Fish Management Plan is currently for WDFW internal discussion only.

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Western Pond Turtle

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10.0 Western Pond Turtle *(Clemmys marmorata)*

10.1 Introduction

The western pond turtle *(Clemmys marmorata)* is listed by Washington as an endangered species. It was petitioned in 1992 for federal listing, but in 1993 the USFWS found that listing was not warranted.

This highly aquatic turtle occurs in streams, ponds, lakes, and permanent and ephemeral wetlands. Although pond turtles spend much of their lives in water, they require terrestrial habitats for nesting. They also often overwinter on land, disperse via overland routes, and may spend part of the warmest months in aestivation on land. Pond turtles are generally wary, but they may be seen basking on emergent or floating vegetation, logs, rocks, and occasionally mud or sand banks. Nesting occurs from May to mid-July in soils with scant vegetative cover. They usually nest within 325 ft (100 m) of water, but occasionally up to 1,300 ft (400 m) away. Western pond turtles are long-lived, with some reaching an estimated maximum life span of 50 to 70 years, and require more than 10 years to attain sexual maturity.

The historic range of the western pond turtle extended from the Puget Sound lowlands in Washington south to Baja California. Western pond turtles were essentially extirpated in the Puget lowlands by the 1980s. Their present range in Washington is thought to be composed of two small populations in Skamania and Klickitat Counties, and a small pond complex in Pierce County where they were recently reintroduced from captive bred stock. The total number of western pond turtles in known Washington populations is estimated at only 250-350 individuals. Additional turtles may still occur in wetlands that have not been surveyed in western Washington and the Columbia Gorge.

The western pond turtle is declining throughout most of its range and is highly vulnerable to extirpation in Washington. They are still abundant in northern California and southern Oregon wherever there are relatively few people. The species requires a continued recovery program to ensure its survival in the state until sources of excessive mortality can be reduced or eliminated.

The initial cause of the decline in western pond turtle numbers in Washington may have been commercial exploitation for food. Western pond turtle populations cannot be sustained under exploitation, due to their low rate of recruitment and lower densities at the northern portion of the range. Pond turtles never recovered from this decline, in part, due to concurrent or subsequent alteration and loss of habitat. Wetlands were filled for residential and industrial development, particularly in the Puget Sound region. Dam construction and water diversion projects reduced available habitat and isolated populations. Introduced predators such as bullfrogs and warm-water fish probably took a toll on hatchlings and young turtles. Human disturbance may have kept females from crossing over land to lay eggs, or may have reduced the amount of time spent basking, which in turn, may be important for egg maturation. Loss of lakeside emergent wetland vegetation to grazing and trampling may have made habitat less suitable for hatchlings and juveniles. Successional changes through fire suppression on native grasslands may have resulted in excessive shade on nesting grounds.

10.2 Taxonomy

The western pond turtle has been known variously as the Pacific pond turtle, western mud turtle, Pacific mud turtle, Pacific terrapin, and Pacific freshwater turtle. It is a member of the order Testudines and the family Emydidae.

The type specimens of the western pond turtle were collected during the U.S. Exploring Expedition in 1841 in the vicinity of Puget Sound, and were described by Baird and Girard (1852) as *Emysmarmorat*a. The first use of the combination *Clemmys marmorata* was by Strauch (1862). Based upon examination of 158 specimens from throughout the range of the species (Washington and Nevada excluded), Seeliger (1945) divided the species into two subspecies: the northwestern pond turtle *(C. m. marmorat*a) and the southwestern pond turtle *(C. m. pallid*a). The northwestern subspecies is found from the Sacramento Valley, California northward to Puget Sound. The southwestern subspecies is found from the vicinity of Monterey, California southward to Baja California Norte. The area of the San Joaquin Valley, California is considered a zone of intergradation.

In 1992, the Washington Department of Wildlife supported a study of genetic variation within western pond turtle populations using DNA fingerprinting. Gray (1995) found that turtles in the Columbia Gorge region of Washington and Oregon had very high genetic similarity within sites and significant genetic divergence between sites. She concluded this was an indication of lack of dispersal and gene flow between sites. Her results indicated a significant genetic difference between northern populations in Washington and Oregon, and southern California populations. She found no genetic subdivision between turtles from the Puget Sound region and the Willamette Valley, and a small genetic subdivision between Puget Sound/Willamette Valley turtles, and Columbia Gorge turtles. She also stated that the level of genetic variation within the Puget Sound region may have been overestimated due to small sample sizes.

Janzen *et al.* (1997) used a mitochondrial DNA technique to evaluate the molecular phylogeography of the western pond turtle. They found low levels of genetic differences among populations of northern pond turtles. They conducted a more detailed analysis of turtles in Oregon, and found that there were small genotypic differences within Oregon populations of turtles. Of particular note, turtles in the Willamette Valley were slightly different from turtles in the Columbia Gorge in Oregon.

Considering the work of both Gray (1995) and Janzen *et al.* (1997), there is an indication that the Willamette Valley turtles are more similar to Puget Sound turtles than Columbia Gorge turtles. In addition, these genetic studies support morphological differences suggested by Holland (1992) between Columbia Gorge and Puget Sound turtles.

10.3 Description

The western pond turtle is a medium-sized turtle, dark brown or olive above without dark reticulations or streaking and a yellowish plastron (underside), sometimes with dark blotches in the centers of the scutes (Storm and Leonard 1995). Maximum size varies geographically, with the largest animals (210 mm or 8.2 in) occurring in the northern part of the range. Turtles become sexually mature at a carapace (dorsal or top shell) length of about 120 mm (Nussbaum *et al*. 1983). Large animals may exceed 1 kg (2.2 lb) in mass. In a series of 45 adults from Klickitat County the mean weight of males was 554 g (1.2 lb) and the mean weight of non-gravid females was 504 g (1.1 lb) (D. Holland, unpubl. data). Non-gravid females of a given carapace length are usually significantly heavier than males (Holland 1985a). Hatchlings are 1.0-1.22 in (25-31 mm) in length and weigh from 0.11-0.25 oz (3-7 g) (D. Holland and F. Slavens, unpubl. data).

Color varies geographically and with age. In general, animals in the northern part of the range are darker in overall coloration. The ground color of the carapace (dorsal or top shell) is generally dark brown or black, but may be reddish in a small percentage of females. In some extremely old males the melanin in the carapace becomes mottled in appearance. The carapace may be unmarked, or may possess a series of fine black radii or lines extending outward from the growth center of each shield. These lines may be darker than the ground color of the carapace and often surround small yellow-gold flecks. The plastron is generally cream to yellow in color, with varying degrees of black or brown mottling (Storm and Leonard 1995).

Head and neck coloration varies sexually and geographically and changes during the life cycle. Small animals and females typically have dark flecks or rosette-like markings (often referred to as a "paisley print") on the head, sides of the neck, and throat (Storm and Leonard 1995). Females tend to retain these markings throughout life, whereas males usually become progressively darker on the head and sides of the neck, while the throat becomes white or cream-yellow. Hatchlings are generally dark brown-olive, with prominent mottling on the head and neck.

Western pond turtles are sexually dimorphic. In general, the female has a smaller head, less heavily-angled snout, relatively higher and rounder carapace, and a thinner tail. Males have a slightly concave plastron. In northern populations, males reach a larger maximum size than females (Holland and Bury 1998).

10.4 Distribution

10.4.1 North America

The western pond turtle historically ranged from the vicinity of Puget Sound in Washington south to the Sierra San Pedro Martirs in Baja California Norte (Figure 10-1). Most populations occurred west of the Sierra/Cascade crest.

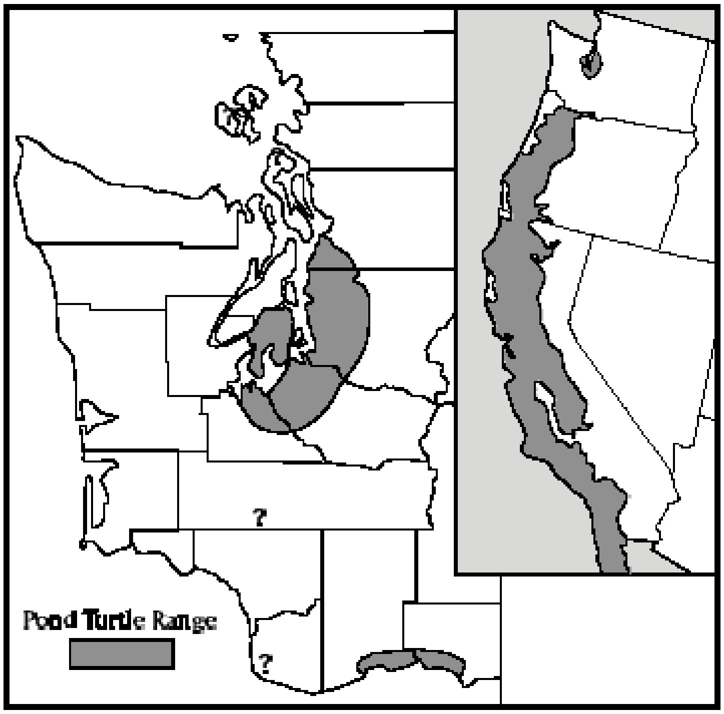

Figure 10-1. Approximate historic range of the western pond turtle in Washington (circa 1850), and in North America (inset). ?= records that may have resulted from human transport.

10.4.2 Washington

10.4.2.1 Past

Fossils assigned to this species are known from Pleistocene strata in the vicinity of White Bluffs in south-central Washington (Brattstrom and Sturn 1959 in Gustafson 1978), indicating that western pond turtles once had a wider range than present. Additional populations along the Columbia River may have been lost in recent decades because dam construction may have eliminated suitable habitats within the last 70 years.

Areas surrounding Puget Sound probably were incapable of supporting western pond turtles prior to approximately 10-11,000 years ago due to extensive glaciation. D. Holland (pers. comm.) suspects turtles in this area arrived about 9-10,000 years ago through the Puget Trough from the Willamette drainage, and suggests a pyroclastic event from Mount Rainier about 4,700 years ago isolated the Puget Sound population. Reconstructing the range of this species is difficult because turtles are often transported by people and they may be found in areas where native populations have never existed. Western pond turtle populations have been documented from the south Puget Sound lowlands (Suckley and Cooper 1860) and the Columbia River Gorge (Slater 1962). Reliable records of western pond turtles suggest a broad distribution in the western half of Washington State, but most records pertain to single turtles, either collected or observed, and some unknown subset of these records probably reflect human translocations.

There are 19 western pond turtle specimens from Washington in museum collections (**Error! Reference source not found.**). Sixteen are from the Puget Sound area (Figure 10-2) (representing eight distinct locations), two are from the vicinity of Lyle, Klickitat County, and one is from San Juan Island, San Juan County. The turtles collected in San Juan and Clark Counties may have been moved by people, because they were quite distant from any known populations. A. Flynt obtained the first record of the species from the Columbia River Gorge when he found the turtles near Lyle (Slater 1962).

Table 10-1. Western pond turtle specimens collected in Washington. Information assembled from Milner (1986), Holland (1991b), and WDFW data.

Locality	County	Date	Collector	Comments	Specimen[a]
Steilacoom, Puget Sound		1841	US Exploring	Holotype	USNM 007700
Puget Sound		1841	Expedition	Cotype	USNM 008800
Puget Sound		1841	"	Cotype	USNM 00759400
Puget Sound		1841	"	Cotype	USNM 00759500
Puget Sound		1841	"	Cotype	USNM 00759600
Puget Sound		1841	"	Cotype	USNM 00131830
Fort Steilacoom		1853-1860	J. G. Cooper	Gravid female	MCZ 42200
Lk Washington, Tacoma	Pierce	Apr 1891			ANSP 3986
Talbot Marsh, McChord AFB	Pierce	23 May 1937	J. R. Slater	Imm. female	PSM 3020
Talbot Marsh, McChord AFB	Pierce	18 April 1939	H. Myhrman		PSM 3621
Meridian Lake	King	20 Jun 1948	W. Hagerman		PSM 4992
Long Lake	Thurston	10 May 1950	S. M.		PSM 6300
Sportsman's Lake	Pierce	12 Oct 1951	H. Myhrman		PSM 4971
Meydenbauer Bay, Lk Wash	King	9 Aug 1952	M. Johnson		PSM 8189
Bay Lake	Pierce	23 Mar 1956	Anon.		CRCM 57-244
San Juan Island	San Juan	26 Aug 1960	J. Berger	Transported?	UI 48370
W of Lyle	Klickitat	7 June 1960	A. Flynt		AMNH 84331
W of Lyle	Klickitat	7 June 1960	A. Flynt		PSM 8233
Lake Washington	King	1963	Anon.		UWBM 20332
Salmon Crk, Vancouver Lk	Clark	1963	E. Nelson	Found dead	CCC
Salmon Crk N of Kid Valley	Lewis	10 June 1993	B. Bicknell	Photo voucher	UWBM

[a] Holdings as follows: American Museum of Natural History, New York (AMNH); Academy of Natural Sciences, Philadelphia (ANSP); Clark Community College (no museum), Vancouver, Washington (CCC); Charles R. Conner Museum, Washington State University, Pullman (CRCM); Museum of Comparative Zoology, Harvard University, Cambridge, Massachusetts (MCZ); James R. Slater Museum of Natural History, University of Puget Sound, Tacoma, Washington (PSM); University of Illinois, Urbana-Champaign (UI); National Museum of Natural History, Smithsonian Institution, Washington, D.C. (USNM); Thomas Burke Memorial Washington State Museum, University of Washington, Seattle (UWBM).

There are recent reports of pond turtles from south Puget Sound (Table 10-2), as well as recent reports of two animals from the vicinity of Vancouver Lake, Clark County (**Error! Reference source not found.**). A 1993 photograph of an animal was taken in Lewis County (**Error! Reference source not found.**). D. Blackburn (D. Holland, pers. comm.) was shown the shell of an animal found dead in 1989 in this area and one specimen exists in the collection of Clark Community College. The Lewis County and San Juan Island records came from areas where no populations have ever been found. Another record consists of a plastron fragment found during archaeological excavation of 500 year old shell middens at Cornet Bay on the north end of Whidbey Island (Weasma 1991). This location is 50 miles north of the northernmost historical records, and may have been transported and eaten at the site. A conservative interpretation of historic distribution relies on accounts that give observations of turtles in significant numbers and locations with multiple records. However, with little information available, an accurate depiction of the species' distribution is not possible.

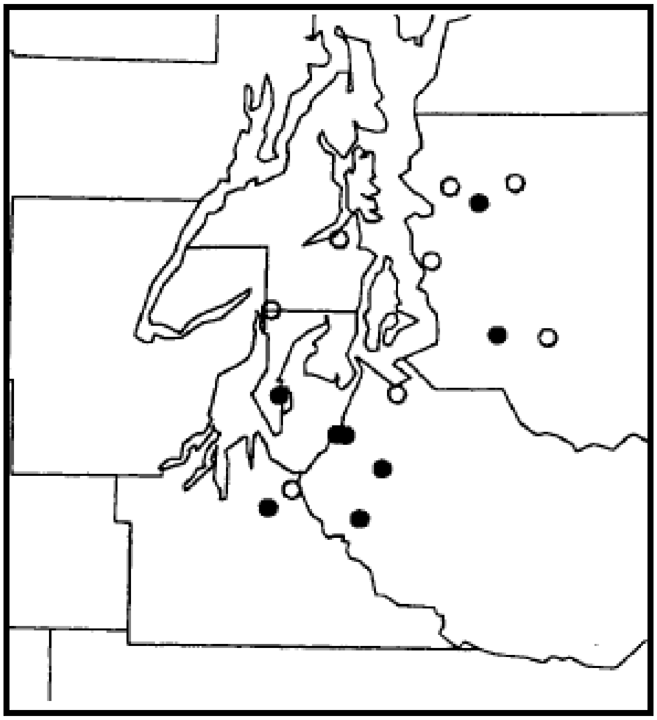

Figure 10-2. Museum records (solid dots) and recent sightings (open dots) of western pond turtles in the Puget Sound region.

Table 10-2. Western pond turtle sight records in the Puget Sound lowlands, Washington, 1980–1997.

Locality	County	Date	Observer	Comments
Kitsap Lk roadside	Kitsap	1983/84	L. Bomstead	Taken to pond at Bomstead home.
Crossing hwy 901[a]	King	30 Oct 1987	J. Pryal	Released in Lk Sammamish
Tacoma City waterway[b]	Pierce	15 July 1987	J. Slipp	Adult male
Lake Garret, Brien	King	July 1988	M. Jolivet	Old bullet hole in carapace, died
Port Orchard[b]	Kitsap	May 1991	G. McIntyre	Died 30 June, 1991
McAllister Crk, under I-5[a]	Thurston	May 1991	R. Van Deman	Released at Nisqually NWR
Wapato Creek, Fife[b]	Pierce	May 1992	S. Siebers	Adult female
Kent Kangley Rd[b]	King	7 July 1992	M. Flatt	Adult male, estimated 5 years
Under Hwy 520, Lk Wash	King	21 June 1992	L. Shaftel	Adult female, 7 inches long
Koeneman Lk (Fern Lk)	Kitsap	1 June 1992	D. Payne	Adult male

[a] Photos examined by WDFW biologist

[b] Captive Breeding Program

10.5 Life History & Habitat Requirements

10.5.1 Behavioral Characteristics

10.5.1.1 General

Western pond turtles spend a considerable amount of time engaged in thermo-regulatory behavior. When out of water, turtles seek warmth from the sun in an activity known as emergent basking. Emergent basking has been noted in all months of the year in some areas, but generally increases in frequency through the spring to a peak in early to mid-June. Emergent basking declines in summer until September, when another peak is observed. Turtles also seek thermally-suitable micro-habitats in the water to engage in an activity known as aquatic basking (Holland

1985a). In these situations turtles are typically found concealed in or under masses of floating vegetation or algae, or in shallow water relatively close to shore. This behavior varies by site and season. In general, aquatic basking peaks in early to mid-July in most areas and declines by early September.

Western pond turtles often engage in aggressive interactions while emergent basking (Bury and Wolfheim 1973). A ritualized intention to bite called the “open-mouth gesture” is the most common aggressive behavior. Aggressive behavior seems to maintain spacing on basking sites and may be used to settle disputes over preferred sites (Bury and Wolfheim 1973).

Western pond turtles are wary, with a well-developed sense of sight and a moderate sense of hearing (Holland 1985a). The initiation of escape behavior varies with the individual and circumstances, but often occurs when a perceived threat is 325 ft (100 m) distant or more. If turtles are surprised in shallow water with no nearby refuge they may remain motionless. Turtles surprised while engaged in aquatic basking simply withdraw their heads and limbs and remain motionless. Pond turtles can be habituated to human presence through repeated interaction (R.B. Bury, pers. comm.).

In the spring, early summer, and autumn most turtle activity is diurnal. Nocturnal activity primarily occurs in the summer (Holland and Bury 1998). During the summer the species may be most active in early morning and evening, and inactive during the heat of the day. Western pond turtle activity may persist throughout the year in some parts of their range.

10.5.1.2 Overwintering

“Overwintering” refers to periods of reduced or no activity during the winter and may include periods of a hibernation-like state of reduced physiological activity. Western pond turtles overwinter from mid-October or November to March or April. Pond turtles may overwinter on land as much as 1,600 ft 500 m) from the nearest watercourse, and they sometimes change sites during the season (Holland 1994, Slavens 1992a). During a study in California, 10 of 12 pond turtles overwintered at upland sites (Reese and Welsh 1997). Preliminary observations from turtles in a pond environment suggest that juveniles overwinter in the water (Slavens 1995). Turtles that overwinter under water may change sites and may form aggregations. Holland (1994) observed pond turtles in winter swimming under ice, and recorded an aggregation of 43 turtles in a one square meter area in the Willamette Valley of Oregon. Aggregations in shallows under ice were also described at Old Fort Lake in Pierce County in the mid 1800s by an employee of the Hudson’s Bay Company (see Strahle 1994). Stream-dwelling pond turtles may be more likely to overwinter on land than pond-dwelling turtles (R.B. Bury, pers. comm.)

10.5.1.3 Aestivation

Aestivation is an inactive state that turtles may enter in the hottest weeks of the year or to avoid short-term drought conditions or drying of a water body. During a telemetry study in Washington, pond turtles moved onto land and burrowed under logs or leaves and remained inactive for days or months (Slavens 1995). One female went onto land 5 times between 9 August and 1 October and returned to water after 2-9 days each time. Aestivation merged with hibernation for one female who was in the same upland location from 11 Aug - 2 Feb, and was next found in the water on 29 March (Slavens 1995).

10.5.1.4 Foraging

Western pond turtles locate food by sight or by smell, and appear to spend considerable amounts of time foraging. Under normal conditions feeding behavior is solitary. However, large numbers of individuals may gather at a vertebrate carcass and aggressive interactions are common under these conditions (D. Holland, unpubl. data). Western pond turtles are apparently incapable of swallowing in air, so food must be swallowed in the water. Animals normally forage along the bottom of water bodies, searching carefully in submerged leaf litter and other detritus. They may also forage on items on the surface or feed in the water column under special circumstances (Holland 1985b). Nocturnal foraging has been observed during the summer months in central California (Holland 1985a).

10.5.1.5 Diet

Western pond turtles are dietary generalists. They prey heavily on aquatic invertebrates, such as the larvae of beetles, stoneflies, caddisflies, dragonflies and other insects (Bury 1986, Holland 1994). Bury (1986) notes that pond turtles in a stream environment in northern California may occasionally take small fish and frogs. Holland (1985a) found two vertebrate prey items in over 500 stomach flushings of animals from the central coast of California, but it is thought that these were scavenged because turtles frequently feed on carrion. Scavenging has been noted on the carcasses of various mammals, birds, reptiles, amphibians and bony fishes. Where bullfrogs (*Rana catesbeiana*) occur with western pond turtles, there is no evidence that turtles feed on either larval or post-metamorphic bullfrogs, although they may feed on their carcasses (D. Holland, pers. comm.). Unpalatable elements in the skin of bullfrogs may deter predation by pond turtles.

Use of plants appears to be limited except in the case of post-partum females, who may ingest large quantities of cattail (*Typha spp*.) or bulrush (*Scirpus spp*.) roots at certain seasons (Holland 1985a). Water lily pods and alder (*Alnus spp*.) catkins are also eaten (Holland and Bury 1998). In certain circumstances, turtles may eat large quantities of filamentous green algae (Holland 1991b), which they ingest while trying to eat live prey (R. B. Bury, pers. comm.).

10.5.1.6 Home Range

Western pond turtles in a stream environment in northern California had average home ranges of about 2.47 ac (1 ha) for adult males, 0.62 ac (0.25 ha) for adult females, and 1 ac (0.4 ha) for juveniles (Bury 1979, Holland and Bury 1998). Considerable overlap in home ranges of individuals of both sexes occurred in this area. Preliminary information from the Columbia Gorge indicates turtles may have larger home ranges in Washington (Slavens, pers. comm.).

10.5.1.7 Movements

Most western pond turtles are somewhat sedentary, although they are capable of moving significant distances and occasionally travel several hundred meters in just a few days (Bury 1979). Daily movements in a California stream averaged 150 m/day for males, and 21 m/day for gravid females (Holland 1994, summarized in Holland and Bury 1998). In an Oregon lake, daily movements averaged 640 ft/day (194.5 m/day) for males, 606 ft/day (185.5 m) for gravid females, and 616 ft/day (188.7 m) for non-gravid females. Some turtles have moved over 3.1 mi (5 km) within a stream (Holland 1994).

Most movements on land are associated with nesting, overwintering or aestivation, although other types of movements also occur. Gravid females typically make multiple trips onto land (Reese and Welsh 1997, K. Slavens, pers. comm.). Reese and Welsh (1997) reported travel to overwintering sites as far as 0.3 mi (500 m) from a California river, and speculated that overwintering away from the river may have been an adaptation to avoid winter flooding. Slavens (1995) reported movements between wetlands in Washington: a male turtle that was captured and then released in a different wetland moved 0.5 mi (800 m) back to the original site. Male turtles have been encountered moving overland in spring, possibly searching for females. K. Slavens (pers. comm.) reports capturing males several times in different ponds during April of the same year. A juvenile turtle moved 656 ft (200 m) between ponds either overland or through a stream, and another juvenile was observed moving overland between ponds. Some turtles move between ponds on an annual basis, moving to larger ponds as water levels recede (Slavens 1995).

Females may move considerable distances from the water to nest. In Washington distances of up to 614 ft (187 m) are known (Holland 1991a), but distances of 65-325 ft (20-100 m) are more typical (Slavens, unpubl. data). Distances as little as 10 ft (3 m) and as great as 1,300 ft (400 m) away from, and 300 ft (92 m) above the watercourse have been recorded (Storer 1930, Holland 1994).

10.5.1.8 Interspecific Relationships

The western pond turtle occurs sympatrically with the western painted turtle (*Chrysemys picta belli*) in northern Oregon and at one of the two Columbia River Gorge localities in Washington (Nordby 1992; D. Holland, pers. comm.). The two species are frequently observed basking together, and they may utilize the same prey base. In areas where thermally desirable emergent basking sites are limited, competition for these sites may occur when population densities are high (Bury and Wolfheim 1973, D. Holland, unpubl. data). Competition for available prey may occur between western pond turtles and introduced fish species (Holland and Bury 1998). Western pond turtles may also interact with introduced turtles (D. Holland, pers. obs.) and other animal species. Aggressive interactions with two-striped garter snakes (*Thamnophis hammondi*) and several species of birds have been noted in California (Holland 1985a). Western pond turtles are preyed on by a variety of species.

10.5.2 Habitat Requirements

10.5.2.1 Aquatic

The western pond turtle is associated with a variety of aquatic habitats, both permanent and intermittent. They are found from sea level to approximately 4,500 ft (1,375 m), but all records for Washington are below 985 ft (300 m) in elevation. The name western "pond" turtle is something of a misnomer, as ponds are relatively scarce throughout most of the range of this species, and the turtles are more often associated with rivers and streams. They are usually rare or absent in reservoirs, impoundments, canals, or other bodies of water heavily altered by humans. However, in Washington and many areas of Oregon the species is found in ponds and small lakes.

Historically, western pond turtles occurred in large numbers in the warm, shallow lakes and sloughs on the floor of the San Joaquin and Sacramento valleys of California (Holland 1991b). This species reaches its highest densities in the few remaining areas that approximate these habitat conditions.

Western pond turtles inhabit some of the larger rivers within their range (e.g., the Sacramento, Klamath, and Willamette), but are usually restricted to areas near the banks or in adjacent backwater habitats where the current is relatively slow and abundant emergent basking sites and refugia exist. They may be found in slower moving streams where emergent basking sites are available, but generally avoid heavily shaded areas. In some areas of California, intermittent streams hold sizeable populations. Turtles are also known to use ephemeral pools. They tolerate brackish water, and along the California coast they often coexist with brackish-water fish species such as sculpins (*Leptocottus armatus and Cottus sp*.) (Holland 1991b).

Habitats used by western pond turtles may have a variety of substrates including solid rock, boulders, cobbles, gravel, sand, mud, decaying vegetation, and combinations of these. In many areas turtles are found in rocky streams with little or no emergent vegetation. In other areas they occur in slow-moving streams or backwaters with abundant emergent vegetation such as cattails or bulrush (*Scirpus spp*.) (Holland 1991c). In certain coastal streams of California they occur in areas with no emergent vegetation but abundant submerged vegetation, most typically ditch grass (*Ruppia maritima*). In the northern parts of the range, pond lilies (*Nuphar spp*.) or arrow weed (*Sagittaria spp*.) are often the dominant aquatic macrophytes. In disturbed habitats large mats of filamentous algae may be the only aquatic vegetation present. Dense growths of woody vegetation along the edges of a watercourse may shade potential emergent basking sites, and make habitats unsuitable for pond turtles.

Turtles have been observed to be active in water temperatures as low as 37°F (1-2°C) and as high as 100°F (38°C) (Holland 1991c, Holland 1994). In general, turtles avoid prolonged exposure to water above 95°F (35°C). Visibility through water in areas inhabited by turtles may range from less than 6 in (15 cm) to more than 33 ft (10 m).

10.5.2.2 Basking Sites

Western pond turtles spend a considerable amount of time basking, and they are more abundant in habitats that have basking sites (Holland and Bury 1998). Turtles may use a variety of sites for emergent basking, such as rocks, sand, mud, downed logs, submerged branches of near-shore vegetation, and emergent or submerged aquatic vegetation. Turtles are also known to bask on planks, barrels, abandoned autos, the carcasses of large mammals, and other items.

10.5.2.3 Refugia

Western pond turtles are associated with areas that contain underwater refugia, and are rarely found more than a few meters from a refuge of some sort (Holland and Bury 1998). These refugia may consist of rocks of various sizes, submerged logs or branches, submerged vegetation, or holes or undercut areas along the bank. When escaping, turtles swim rapidly toward the bottom of the water body and hide in or under nearby refugia. In some cases animals attempt to burrow into the substrate. Turtles also occasionally hide in thick vegetation or holes at the edge of the watercourse.

10.5.2.4 Uplands

Western pond turtles use upland areas for dispersal, to nest, to overwinter, and to aestivate. Overland movements may occur during spring and fall migrations to and from upland overwintering sites, or may be in response to drying of the water body, or for other reasons not presently understood (Holland 1991b). Males may make overland movements in search of females (K. Slavens, pers. comm.). In a California study, Reese and Welsh (1997) reported use of terrestrial habitats by male turtles in 10 months of the year, and by females in all months. Many turtles overwinter on land at sites up to 0.3 mi (500 m) from the water. Overwintering sites tend to have a deep layer of duff or leaf litter under trees or shrubs, and some turtles return to the same site each year (Holland 1994, Holland and Bury 1998, K. Slavens, pers. comm.). Reese and Welsh (1997) reported that 10 turtles overwintered at upland sites a mean distance of 666 ft (203 m) from the water. Turtles burrowed into deep leaf or needle litter at sites beyond the riparian zone in woodlands with 15-90% canopy cover. Most of the overwintering sites were on relatively cool north or east facing slopes.

In Klickitat County, Washington, 10 of 15 overwintering sites were on slopes of 5-15°. Nine of these had an east or west aspect, and one had a north aspect (K. Slavens, pers. comm.). Of the five remaining sites; one was a west slope of 25°, and four were on south, east, or north slopes of 40-45°. One site was only 3 ft (1 m) from the high water mark in March. All the sites had 80-90% shrub and tree canopy coverage. Virtually all overwintering sites were beneath or near Oregon white oak (Quercus garryana). Two turtles were dug in under logs, and the remaining 13 were under small shrubs (K. Slavens, pers. comm.).

Reese and Welsh (1997) reported that gravid females in their study were highly terrestrial, though the presence of researchers may have affected turtle activities. They noted that during the nesting season, the air temperature was consistently higher than the water temperature. They speculated that female turtles may use uplands prior to oviposition for its thermal advantage. Most nest sites discovered have been in dry, well-drained soils with significant clay/silt content and low slope (< 15°) (Holland 1994, Reese and Welsh 1997).

In Washington, pond turtles use wetlands that have open uplands, such as oak-pine savanna, prairie, or pastures. Human-caused fires may have been beneficial to turtles historically by maintaining open areas for nesting. Suppression has resulted in an increase in the distribution and cover of coniferous trees such as Douglas fir (Crawford and Hall 1997). A reduction in fires since white settlement has dramatically altered native grassland habitat. In the south Puget Sound region, less than 10 percent of historical grassland habitat remains (Crawford and Hall 1997). Successional changes in grassland and oak woodland habitat may have played a major role in the decline of western pond turtles.

Turtles usually nest in open areas with good sun exposure that are dominated by grasses and herbaceous vegetation, with few shrubs or trees close by. Exposure varies, but typically is south or southwest (Holland 1991b). The distance from water for 275 nests in California averaged 149 ft (45.6 m) (range 5-1,326 ft (1.5-402 m) (Holland and Bury 1998). In Washington, nest sites have all been less than 614 ft (187 m) from the water (Holland 1991a). Some female turtles seem to exhibit nest site fidelity (Holland and Bury 1998). The degree of nest site fidelity exhibited in an area may be related to the relative abundance of nesting habitat (K. Slavens, pers. comm.).

10.6 Population Dynamics

10.6.1 Reproduction

Courtship and mating behavior have been observed from February to November (Holland 1988, Holland and Bury 1998). Age and size at development of secondary sexual characteristics varies geographically (D. Holland, unpubl. data), but these are generally evident in both sexes by the time an animal reaches 110 mm (4.3 in) carapace length. The time required for males to achieve sexual maturity is not known, but is thought to be at least 10-12 years in Washington. In a sample of 10 gravid females from the Klickitat County population, the smallest animal was 5.63 in (143 mm) carapace length, and approximately 14-17 years of age (Holland 1991c).

However, females as small as 4.3 in (111 mm) with an approximate age of 6-7 years have been observed carrying eggs in southern California (Holland 1994).

When preparing to lay eggs, females typically leave the water in late afternoon or early evening and travel distances generally up to 325 ft (100 m) to nest (Slavens 1995). Females moisten the soil around the nest by urinating prior to digging the nest chamber. Excavating the flask-shaped nest may require several hours and the female commonly remains on or near the nest site overnight.

In Washington’s Columbia Gorge populations, most females that were monitored in successive years nested each year (K. Slavens, pers. comm.). Holland and Bury (1998) report that in northern areas, most females only deposit eggs in alternate years. In central and southern California females produce eggs every year and two clutches in some years (Holland and Bury 1998). Double-clutching by wild females has been observed in Washington during 1996, 1997, and 1998 (K. Slavens, unpubl. data). In Washington, clutches have been laid between May 31 and July 9 (n=41) with a peak in mid-June (Figure 10-3). Clutch size ranges from 2-13 eggs and is positively correlated with body size. Mean clutch size for 36 wild nests from Washington was 6.64 (SD ±1.57, range 2-10) (F. & K. Slavens, WDFW, unpubl. data).

Eggs average 1.34 in (34 mm) in length, 0.83 in (21 mm) in diameter, and 0.28-0.35 oz (8-10 g) in weight (Holland 1994). Hatching rate of fertile eggs in the Pacific Northwest seems to be dependent on the weather during the incubation period. Unusually cold wet weather can cause total nesting failure (Slavens 1995). In 10 nests in Washington, incubation time varied from 95 to 127 days (Holland 1991a, Slavens 1995). Incubation time in captivity is 73-132 days (Lardie 1975, Feldman 1982). Hatchlings from Washington average 0.18-0.25 oz (5-7 g) in weight and 1.1-1.2 in (27-31 mm) in carapace length (F. Slavens, unpubl. data). In southern California, some hatchlings leave the nest in early fall. Field observations to date indicate that in the northern parts of the species’ range, hatchlings overwinter in the nest (Holland 1994, Reese and Welsh 1997, F. & K. Slavens, pers. comm.).

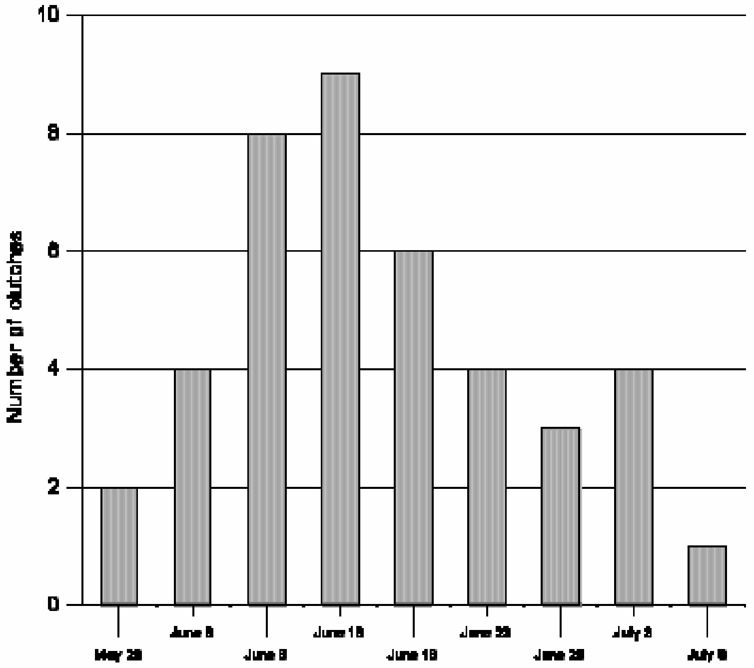

Figure 10-3. Timing of western pond turtle oviposition in five day intervals in the Columbia River gorge, 1990–97 (K. Slavens unpubl. data).

10.6.2 Mortality

Preliminary analyses by D. Holland (unpubl. data) suggest mortality is high in the younger (less than 6-8 years) age classes. Under undisturbed conditions possibly only 10-15% of the animals that hatch in a given year survive until the end of the first year. Survival from the first to second and second to third year is similarly low, but increases slightly by the fourth and fifth years (Holland unpubl. data). R. S. Bury (pers. comm.) reports only slight mortality after year 3 for pond turtles in a northern California study. Survivorship apparently continues to increase until the attainment of sexual maturity. Exact rates of turnover in the adult portion of the population are not known, but probably average 3-5% per year in undisturbed populations (D. Holland, unpubl. data). The average life expectancy of adults is not known. The maximum age achieved by animals in the wild is at least 42 years (D. Holland and R. B. Bury, unpubl. data). The estimated maximum life-span based upon an extrapolation from known adult growth rates is 50-70 years (D. Holland, unpubl. data), which approximates that of the ecologically similar Blanding's turtle (*Emydoidea blandingi*) in Michigan (Congdon *et al.* 1993).

10.6.2.1 Predation

Bullfrogs prey on juvenile western pond turtles (Moyle 1973). Bullfrogs are native to the eastern United States, but have become abundant and widely distributed in the west since their introduction to Idaho in the 1890s, and to Oregon in the 1920s (Lampman 1946). They currently are found throughout the range of the western pond turtle (Bury and Whelan 1985). Bullfrogs may be an important predator on hatchlings because both frequent shallow water habitat. Holland (1991b) has observed a reduction in the abundance of juvenile western pond turtles in areas with bullfrogs (Figure 10-4). Predation by bullfrogs and other predators may be responsible for the lack of juveniles in many pond turtle populations. Largemouth bass (*Micropterus salmoides*), another widely introduced species, also preys on juvenile pond turtles (Holland 1991b). However, observations by Holland (1991b) indicate that the impact of bass may not be as important as that of bullfrogs, perhaps because bass do not frequent the shallows as much as bullfrogs.

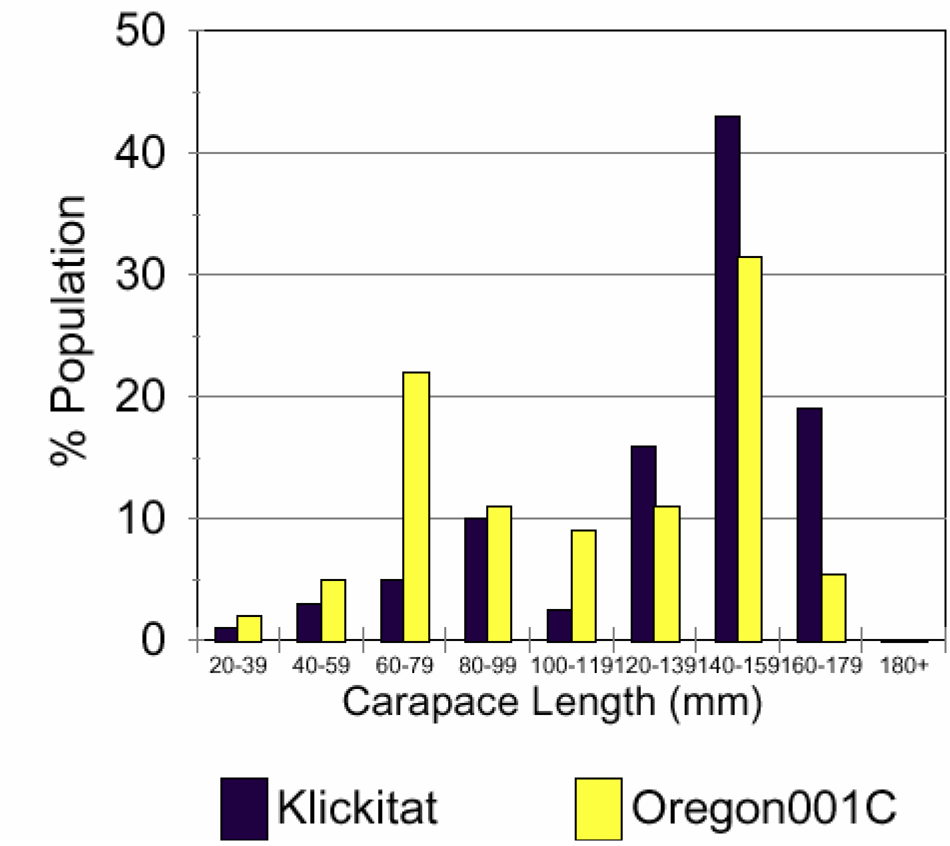

Figure 10-4. Size class composition of western pond turtles where bullfrogs are present (Klickitat County, n=84, 1987-90) and absent (Oregon site 001C, n=54, 1991) (from Holland 1991b:42).

Black bears (*Ursus americanus*) and coyotes (*Canis latrans*) completely eliminated a southern California pond turtle population when drying of a stream forced overland movement by the turtles (S. Sweet, pers. comm. to D. Holland). Raccoons (*Procyon lotor*) preyed on two adults from the Klickitat County population in 1991-92. In 1992, 97 of 106 western pond turtle nests monitored in Oregon were depredated, probably by raccoons or skunks (Holland 1993). River otters (*Lutra canadensis*) are known to prey on western pond turtles (Manning 1990). Holland and Bury (1998) reported 10 pond turtle carcasses and over 20 live turtles with missing limbs along a 3 km stretch of stream and attributed this to river otters. Holland (1994) lists five additional known predators: bald eagle (*Haliaeetus leucocephalus*), osprey (*Pandion haliaetus*), gray fox (*Urocyon cinereoargenteus*), mink (*Mustela vison*), and dog.

Suspected predators include bobcat (*Lynx rufus*), great blue heron (*Ardea herodias*), lack-crowned night-heron (*Nycticorax nycticorax*), golden eagle (*Aquila chrysaetos*), red-shouldered hawk (*Buteo lineatus*), giant garter snake (*Thamnophis gigas*), two-striped garter snake, California red-legged frog (*Rana aurora draytonii*), rainbow trout (*Oncorhynchus mykiss*) and channel catfish (*Ictalurus punctatus*) (Holland 1994). Predation by humans may take the form of wanton shooting, capture by hook and line fishing or entanglement in nets, collection for the pet trade (Bury 1982; D. Holland, pers. obs.) or collection for food (M. P. Hayes and S. Sweet, pers. comm. to D. Holland).

10.6.2.2 Drought

A prolonged drought in California (1985-1990) apparently resulted in declines of up to 85 percent in some pond turtle populations and the outright elimination of others (D. Holland, unpubl. data). Drought may function as a direct mortality factor by eliminating the habitat or prey base required by turtles for survival. Without adequate body fat reserves normally produced by late-season feeding, turtles may be unable to survive the stress of overwintering.

10.6.2.3 Disease and parasites

A syndrome similar to upper respiratory disease caused a decline in the Klickitat County population in 1990. The causal agent is not known with certainty, but may have been a virus or mycoplasma, transmitted from one or more introduced turtles. Western pond turtles essentially have evolved in isolation from most other turtle species for most of their history. Non-native species may introduce pathogenic agents to which western pond turtles have never been exposed, or have inherent resistance to. If this is the case, the introduction of non-native species, particularly from unhealthy captive situations, (e.g., red-eared slider) may have catastrophic consequences for western pond turtle populations. A herpes-like virus has been reported to kill captive western pond turtles in California (Frye et al. 1977).

The effect of parasites on western pond turtle populations is unknown. The only known ecto-parasites are leeches (Placobodella sp.) found on specimens from northern California and central Oregon (Holland 1991b). Endo-parasites include nematodes (Bury 1986) and lungworms (Holland 1991b). Ingles (1930) reported a trematode (Telorchis sp.) from western pond turtles. Thatcher (1954), reported 7 species of helminthes from western pond turtles.

10.7 Population Density

Western pond turtles may reach the highest densities of any emydid turtle and may have been the dominant element of the vertebrate biomass of some aquatic communities on the west coast (Holland 1991b). They historically occurred in large numbers in warm shallow lakes and sloughs such as in the San Joaquin and Sacramento valleys of California. Pond turtle densities range widely, from a low of five turtles per acre (2/ha), to, at one site, an estimated 3,700 turtles /acre (1,500/ha) (Holland 1991b). Pond turtle densities in a few sites have been calculated at over 1,000 turtles/ha (405 turtles/ac) of water surface (Holland 1991b), but typically are found at much lower densities.

10.8 Population Status

The western pond turtle is declining in numbers throughout its range, particularly in Washington, northern Oregon, southern California, and Baja California. It is now common only in a fraction of its original range (Holland and Bury 1998).

10.8.1 Past

The western pond turtle was once abundant in California, Oregon, and locally in Washington. They were commercially marketed for food with annual sales in San Francisco in the 1890s averaging $18,000 (Smith 1895). Exploitation and habitat destruction dramatically reduced pond turtle populations.

No historic data are available on the size or dynamics of populations in the Columbia River Gorge or Puget Sound regions. Cooper (1860) noted that turtles were "common in freshwater ponds and rivers west of the Cascades," a point questioned by Storer (1937), who stated that if this was the case, "specimens should be forthcoming." Three possibilities may explain the scarcity of specimens: 1) the wary nature of the species precluded extensive collection; 2) the species was uncommon or present in low numbers due to various limiting factors at the terminus of its range; and/or 3) a dramatic reduction in the size of population(s) in this area occurred prior to the initiation of extensive collecting efforts that began with the activities of J. Slater in the 1930s.

10.8.1.1 Puget Sound

Western pond turtles were probably locally common to abundant in the south Puget Sound area. Edward Huggins (Strahle 1994) gave an account of securing dozens of turtles that could be seen “congregated in large numbers” under the ice at Old Fort Lake, near Dupont, Pierce County in the 1860s. Indians recall accounts of gathering turtle eggs at Nisqually Lake where turtles were abundant, and the Nisqually name for the lake translates to “place where the turtles came from.” The Puget Sound population was apparently large enough to support commercial collecting activities for the restaurant trade during the late 1800s (M. Jennings, pers. comm. to D. Holland). Due to the low rate of recruitment, pond turtle populations cannot sustain the increased mortality to adults from exploitation. Holland (1991c) examined historic localities in the Puget Sound area and concluded that commercial collection could easily have eliminated or severely reduced populations in certain habitats. Habitat alteration probably also played a significant role in localized declines. By the 1930s populations were probably severely reduced from levels present 50-100 years earlier (Owen 1940).

Pond turtles appear to have continued to decline throughout this century. In King County, turtles were seen during the 1950s at each end of Sammamish Slough, and one was collected at Lake Meridian (Milner 1986). Meydenbauer Bay near Bellevue supported turtles in the 1960s. In Seattle, individuals were reported near Northgate, in West Seattle, and at Haller Lake during the 1960s and 1970s. In Pierce County, western pond turtles were found on and around Fort Lewis and McChord Air Force Base. Spanaway and Halvorson marshes, and Muck, Sequalitchew, American, Lewis, Spanaway, Bay, and Chambers lakes all had resident turtles (Milner 1986). The south Tacoma swamp and Talbot Marsh on McChord Air Force Base both supported turtles. Another turtle site, the headwaters of Murray Creek, was inhabited in the 1940s but was altered and channelized later. A specimen was collected in 1951 at “Sportsman’s Lake”, which Milner (1986) identified as Shaver Lake, but this location is not certain. Turtles observed at several other Pierce County sites were not identified to species (Milner 1986). In Thurston County, a pond turtle was collected at Long Lake, eggs were collected from a site north of Olympia, and adults were observed in Lacey during the 1940s and at Patterson (also called Pattison) Lake in 1969. In Kitsap County one was found near Kitsap Lake in the early 1980s (Milner 1986).

Western pond turtles had effectively been extirpated in the region by the 1980s. Surveys of 56 wetlands by Milner (1986) failed to find any western pond turtle populations in the Puget Sound area. An old male with a bullet hole in its carapace was found at Lake Garret in Burien in 1988, but died in captivity. Another adult pond turtle was found crossing a road under I-5 near McAllister Creek in May 1991. This turtle was released on Nisqually National Wildlife Refuge. This individual, which was identified by a chipped shell, was recaptured in April 1992 (Vicencio and Van Deman 1992). Extensive surveys and incidental sightings in 1991-92 revealed a few isolated individual turtles in King, Kitsap, and Thurston Counties; no turtles were detected in Mason County (Table 10-3) (Nordby 1992).

Table 10-3. Results of 1992 western pond turtle surveys in Washington (compiled from Nordby 1992).

County	Sites surveyed	Sites with pond turtles	Adult pond turtles	Juvenile pond turtles	Painted turtles	Red-eared Sliders
Skamania	39	10	23	2	288	—
Klickitat	6	2	33	8	4	—
Clark	15	0	—	—	150	—
King	14	1	1	—	5	5
Pierce	7	0	—	—	—	7
Kitsap	5	1	1	—	3	—
Thurston	1	1	1	—	—	—
Mason	1	0	—	—	—	—
Total	*88*	*15*	*59*	*10*	*450*	*12*

Forrester and Storre (1992) reported three possible western pond turtle sightings at Sequalitchew Lake on Fort Lewis, Pierce County in 1991, but follow-up surveys found no pond turtles (R.B. Bury, pers. comm.). Subsequent surveys of 24 wetlands on Fort Lewis involved a total of 303 site visits and 258 trap nights, but no additional western pond turtle sightings were recorded (Stringer 1992, Bury 1993). An animal seen on several occasions near Stan Sayres hydroplane pits on Lake Washington was reported to be a western pond turtle (P.D. Boersma and S. Andelman, pers. comm.); follow-up trapping at this location in 1995 produced many introduced turtles, but no western pond turtles. In recent years, WDFW staff obtained five western pond turtles: near Tacoma (1987), Port Orchard (1991), Fife (1992), Ravensdale (1992), and Kid Valley (1993). These individual turtles have to be considered of unknown origin, because they may have been transported to the area by people from Oregon or California. Genetic comparisons with museum specimens may determine their origins. No functioning populations were found anywhere in the Puget Sound region and it was concluded that the species was effectively extirpated from the area.

10.8.1.2 Columbia Gorge

The Klickitat population was estimated to total about 108 turtles in 1986 (Zimmerman 1986). At the beginning of 1990, the Klickitat County population was estimated to number between 60-80 animals (Holland 1991a). Subsequent data indicate the 1990 population was over 96 turtles. Measurements of carapace lengths indicated the population was moderately adult-biased, with about 78% of the animals over 4.5 in (120 mm) (Holland 1991a), compared to 55-70% under normal circumstances (Holland and Bury 1998). This indicated that recruitment may be low and the population may be in decline (Holland 1991a). Head-starting of juvenile turtles was initiated in an effort to augment juvenile survival.

In early to mid-1990, the Klickitat County population was decimated by an unknown pathogen, and at least 36 animals died. To curb the spread of the disease and to treat sick animals, 32 adult and subadult animals were removed from the ponds and treated at the Woodland Park (WPZ) and Pt. Defiance zoos. Based on spring 1991 surveys, at least 45 turtles survived the 1990 disease outbreak at the Klickitat County lake/pond complex (Slavens 1992a). Fourteen of the turtles that were treated at WPZ survived. These, along with 17 head-started juveniles, were returned to the wild in July and August 1991. After the release of an additional 9 head-starts in 1992 and 4 known mortalities (2 killed by a raccoon, 1 by an auto, 1 unknown; Slavens 1992b) the Klickitat population was estimated at 70 in 1992. In 1994, 52 turtles were captured in the pond complex and 12 were observed at the lake. This mark and re-capture effort resulted in a 1994 estimate of 117 turtles in the Klickitat population (Slavens, pers. comm., in Scott 1995a).

The Skamania County population was surveyed repeatedly between 1990 and 1994 (Scott 1995b). During 1992 surveys, 26 turtles were detected at 12 sites, and during 1994, 39 turtles were found at over 14 different sites. The 1994 estimate for Skamania and Klickitat Counties combined was 156 turtles (39 in Skamania County, 117 in Klickitat County).

10.8.2 Present

Only about 250-350 western pond turtles are known to remain in the wild in Washington (Table 10-4). Most of these are in the Columbia Gorge, and approximately half are young turtles from the head-starting and captive rearing program. Of 21 head-started turtles released at a Puget Sound reintroduction site in Lakewood in 1996-1997, at least 14 turtles remained alive in spring 1998. An additional 5 were released in summer 1998. Two old males of unknown geographic origin were released to wetlands at Northwest Trek in 1996. A few additional scattered old adults may remain in the Puget Sound area, but no reproductive populations have been found.

Table 10-4. Numbers of western pond turtles at three locations in Washington, 1998

Location	# turtles present, 1996	Releases, 1997–98 (head starts and captive bred)	Total
Columbia River Gorge			
Klickitat	89[a]	87	176
Skamania	49[b]	60	109
Puget Sound lowlands			
Lakewood	15[c]	11	<26
Total	*153+*	*158*	*311±*[d]

[a] 1996 estimate using the Jolly-Seber mark-recapture method (Bender, unpubl. data).

[b] High count from 1984 - 1994 basking surveys (Scott 1995)

[c] Headstarts

[d] Plus unknown number of unmarked turtles, minus at least 3 known mortalities.

10.9 Habitat Status

A number of factors have contributed to the decline in habitat for western pond turtles. Wetland draining, filling, and development eliminated much habitat during the past century. Milner (1986) reported that several historic western pond turtle sites were altered or the shoreline was developed in the past 50 years. Many have been dredged, channelized, filled, or drained. According to conservative estimates, 33-50% of wetlands present during pre-settlement times were lost in Washington (Canning and Stevens 1990). Wetland losses in urbanized areas around Puget Sound were 90-98%. Historical analysis indicates an 82% loss of wetlands for Pierce County, and 70% for the Lake Washington area (Boule *et al.* 1993). Diversion of water for irrigation and other purposes has also eliminated or altered turtle habitat. The construction of dams and creation of reservoirs has been detrimental to western pond turtles by altering water flow in drainages, inundating habitat behind dams and reservoirs, and creating habitat suitable for the spread of non-native species (bullfrogs, warmwater fishes) that are harmful to western pond turtles. Additionally, dams and their associated reservoirs may have fragmented populations by creating barriers to dispersal (Holland 1991b). On the Trinity River, California, a dam increased sedimentation, decreased water temperatures, increased canopy cover, and increased water velocities (Reese and Welsh 1998a). These factors may negatively impact juvenile turtles (Reese and Welsh 1998b). Grazing or trampling of emergent vegetation may have modified aquatic and riparian habitats to the extent that they became less suitable for hatchlings and juveniles (Holland 1991b).

The harmful effects of habitat alterations were not limited to watercourses, because western pond turtles nest and overwinter in the uplands. Urban, residential, and agricultural development of upland habitats within 325 ft (100 m) of waterbodies effectively eliminated historical nesting areas. Pond turtles need open sunny locations for basking and nesting. Fire suppression may have reduced the amount of habitat available by allowing the invasion of Douglas fir into Puget prairies and oak-pine woodlands and increasing the shading of the forest floor.

The area occupied by known populations of western pond turtles in Washington totals slightly over 1.5 mi (2.4 km). One population is restricted to a complex of ponds in Skamania County and the other occurs in a lake and pond complex in Klickitat County. Most of the Klickitat County habitat has been purchased by WDFW. The Skamania County habitat is in private and U.S. Forest Service ownership and most of it is within the Columbia River Gorge National Scenic Area. The two populations are separated by a road distance of about 17 mi (27 km).

10.9.1 Klickitat County

The Klickitat County lake site can be characterized as moderately disturbed. The lake was slightly modified within the last 20-30 years to increase its size and water storage capacity. The area surrounding the lake was historically grazed by livestock, which has been limited in recent years. The area immediately surrounding the lake shows signs of prolonged human use in the form of a small abandoned pump-house, vehicle track-ways, and footpaths. A road located about 16 ft (5 m) above and 35-50 ft (10-15 m) east of the eastern shore of the lake allows a direct view of the lake and potential access. Traffic on this road was observed to average 2-3 vehicles per hour during late May to early June 1990, and shooting in the general vicinity of turtles was observed at least once in this period. Similar activities were noted previously by the former landowner. To a limited extent, turtles appear to be somewhat acclimated to the presence of traffic on this road and the effects of disturbance of this type are unknown.

The lake has contained brown bullheads (*Ictalurus nebulosus*) since at least the 1940s and was stocked with largemouth bass and bluegill (*Lepomis macrochirus*) by a local sportsman in the late 1970s or early 1980s (D. Anderson, pers. comm.). The presence of fish encouraged recreational use of the lake by local fishers, which was permitted by the landowner until summer 1990. Recreational fishing may have had a significant effect on the turtle population through incidental capture and interruption of normal basking activities.

Large numbers of bullfrogs also occur at the lake and may be responsible for the apparent lack of recruitment in this population. Despite control efforts, frogs and tadpoles remain in the lake. Emergent vegetation in the lake is limited. There are relatively few areas where water depth and clarity allow for growth of emergents. Emergent basking sites for turtles in the form of logs or fallen trees were very limited at this location before artificial rafts were installed in 1991-92.

Immediately south of the lake site are five ponds that hold the majority of the turtles in this population. The ponds, like the lake, are located in a mixed oak/pine/grassland habitat, with Oregon white oak and ponderosa pine (*Pinus ponderosa*) dominant. Bullfrogs exist in all ponds on this site and brown bullheads are present in the two largest ponds described below. While the area around the lake lacks surface rock formations, numerous basalt outcrops and rock piles are present throughout the pond site. Of the five ponds that hold turtles on this site, one is artificial (permanent) and four are natural (two are permanent ponds formed by natural basaltic sinkholes and two are ephemeral). Studies of the turtles in this population have revealed only one instance of movement between the pond complex and the lake. A turtle that had been translocated from one of the ponds to the lake moved 800 m back to the pond complex. However, it is probable that turtles historically moved freely within this system. The lake and pond complex are considered an ecological unit.

The artificial pond was created by excavation in the mid-1970s. It is relatively shallow, with a depth in most areas of about 3-5 ft (1-1.4 m), and currently has a thick growth of native and exotic water lilies. There is a small patch of emergent vegetation in the form of cattails on the north shore and it is surrounded by a fringe of willows (*Salix sp.*) and oaks. The area immediately around the pond is lightly disturbed.

The four natural ponds are located south of the artificial pond. Three of them occur along the base of a small basalt bench. This area was historically used for seasonal cattle grazing which was discontinued in the early 1980s. The two westernmost ponds are permanent but the smaller eastern pond is ephemeral, and frequently dries up by July. All of these ponds are used by turtles. The eastern pond is less than 6.5 ft (2 m) deep, covers about 500-600 ft (150-180 m) and has abundant emergent vegetation in the form of rushes and sedges, as well as seasonal growths of aquatic angiosperms (*Ranunculus sp*.). There is a horse trail about 66 ft (20 m) south of the south shore of the pond. This area can be categorized as lightly disturbed.

The middle and western ponds are considerably larger, at 0.6-0.8 ac (0.25-0.32 ha) and 1 ac (0.4 ha), respectively, and deeper, at about 8.2 ft (2.5 m). They support abundant growths of pond lilies and arrow weed. Small patches of cattails exist on the south and west shore of the western pond. Emergent basking sites in the form of downed logs are present in both ponds, but are more abundant in the western pond. The area around both ponds can be characterized as lightly disturbed to undisturbed.

On the basalt bench above these three ponds, and about 325 ft (100 m) north-northwest of the western pond, is an ephemeral pond that is seasonally utilized by turtles. It is less than 3 ft (1 m) deep, covers more than 0.37 ac (0.15 ha), and has abundant emergent vegetation. This pond usually dries up by July.

10.9.2 Skamania County

The Skamania County population exists in an extensive lake, pond and wetland complex within a forest and pastureland environment. The area can be characterized as lightly to moderately disturbed. Nearly all of the wetlands contain bullfrogs. Some of these ponds were colonized by a native water fern (*Azolla sp.*) in 1997-98 that rapidly formed a thick mat over the ponds' surfaces. The affect that this habitat change will have on the turtles is unknown.

Although turtles have been seen in a number of places in this complex, there are four primary ponds occupied by western pond turtles. At least two of the ponds are artificial. These are relatively small, between 0.5-0.75 ac (0.2-0.3 ha), with mud substrates, abundant submerged vegetation, limited emergent vegetation and relatively few emergent basking sites.

Approximately 0.25 mi (0.4 km) west are two additional ponds. One covers less than 1,650 ft (500 m), is at least 13 ft (4 m) deep, and apparently holds water year-round. The area can be characterized as moderately disturbed. Approximately 500 ft (150 m) north of this pond is another pond of about (0.6-0.75 ac (2.5-3 ha) in area that holds both painted turtles and pond turtles. The origin of this pond is uncertain, as it may represent a natural pond that has been enlarged by diking. The grassland around the pond was moderately grazed until recently and there is little or no emergent vegetation on the periphery. Emergent basking sites in the form of downed logs are abundant.

10.9.3 Puget Sound

Many of the wetlands at Fort Lewis have a history of human alterations such as drainage, farming, and re-flooding. Stringer (1992) found that many wetlands have few or no natural emergent basking sites and most marshes have banks overgrown with reed canary-grass (*Phalaris arundinacea*) and cattails. Most wetlands also have populations of bullfrogs and/or introduced warmwater fish. Recreational use may be a limiting factor at some lakes. American and Sequalitchew lakes are heavily used for recreation and are surrounded by residential areas, though Sequalitchew has coves that are relatively isolated from disturbance (Forrester and Storre 1992). In spite of these factors, Bury (1993) believed several waterways on the military reservation had fair to excellent habitat conditions for western pond turtles and the reasons for their rarity or absence are unknown.

A small complex of three man-made ponds near Lakewood, Pierce County, was selected for the first re-introduction in the Puget Sound area. It is located in a 12+ ac (5 ha) fenced compound owned by WDFW. The ponds are fed by a small perennial creek. The ponds are small (less than 1 ha total area) and the reintroduction project is considered a pilot for future projects.

10.10 Conservation Status

10.10.1 Legal Status

In Washington, the western pond turtle was listed as a sensitive species by the Department of Wildlife in 1981. This status was changed to threatened in 1983. The pond turtle was classified under WAC 232-12-014 as an endangered species in November 1993. Unless allowed by special permit, western pond turtles may not be collected, harassed, possessed (live or dead), or sold.

The Columbia Gorge National Scenic Area Management Plan has placed a number of identified pond turtle habitats in categories that will protect them from development and alteration. Wetland protection regulations, such as Section 404 of the federal Clean Water Act that regulates the discharge of fill, also applies to wetland habitat of pond turtles.

In 1992 a petition to list the western pond turtle under the federal Endangered Species Act was denied by USFWS because although the turtle has declined and is affected by human activity, it still occurs in 90% of its historic range and is not in danger of extinction or likely to become so in the foreseeable future (USFWS 1993a). Though the western pond turtle is not protected under the federal ESA, it is a species of special concern for the Pacific Ecosystem office of the USFWS.

The western pond turtle is listed as sensitive in Regions 5 (California) and 6 (Washington and Oregon) by the USFS. The western pond turtle is considered a critical species by the Oregon Department of Fish and Wildlife (their designation with most concern for a species), and a species of special concern by the California Department of Fish and Game. In these states and Nevada, western pond turtles may not be taken without a scientific collecting permit.

10.11 Factors Affecting Continued Existence

10.11.1 Natural Factors

The western pond turtle has a long life span, requires 10 or more years to reach reproductive age, and has a low rate of recruitment. The vagaries of Pacific Northwest weather probably result in high variation in hatching success. The combination of these factors makes this species especially sensitive to any increase in chronic sources of mortality or other factors that affect reproduction and recruitment. Even relatively minor reductions in recruitment can affect the long term viability of a population, but due to the long life span of this species, changes of this nature may not be immediately evident. Turtles may persist in an area for extended periods even after the population is no longer successfully reproducing.

10.11.2 Habitat Loss and Degradation

In the Puget Sound region, the western pond turtle was reduced to near extirpation by historical habitat loss and exploitation (Holland 1991c). Though wetlands are now generally protected by regulation, there are few wetlands with suitable surrounding upland habitat for pond turtles left in Puget Sound. Human population increases and concomitant development will continue to alter or eliminate habitat for nesting, increase the rate of predation on nesting females, nests, or hatchlings, and/or expose hatchlings to hazardous post-hatching conditions. Though depredated nests have not (either with or without predator exclosures) been found in the Columbia Gorge study areas, predation on nests of other turtle species is higher near ecological edges (Temple 1987), such as those created by human activities.

Alteration of aquatic habitats, by water diversion projects or similar situations, may impose considerable hazard and hardship on moving turtles and result in higher than normal levels of mortality. Overland movements by western pond turtles increase their vulnerability to predators and other mortality sources. Vehicular traffic on roads that traverse western pond turtle habitat may be an important mortality factor.

10.11.3 Interspecific Relationships

Introduced species have changed the ecological environment in the region for pond turtles. As significant predators on hatchling and small juvenile western pond turtles, non-native species such as bullfrogs and warm water fish seem to reduce survivorship and alter recruitment patterns. Raccoons are major predators on turtles and turtle eggs (Christiansen and Gallaway 1984), and may be abundant in suburban areas due to the absence of larger native predators and the availability of refuse, pet food, and other man-associated food sources.

The introduction of opossums from the southeastern United States added another potential predator of turtle nests and hatchlings. Opossums are known to eat hatchling painted turtles and snapping turtles (Hamilton 1958; cited in Gardner 1982). Opossums seem to be particularly suited to the mix of urban/suburban/rural habitat that now exists in the Puget Sound area; they are now very abundant and may pose a serious problem for recovery efforts. Opossums are not presently a problem for the Columbia Gorge populations.

Sunfish compete for invertebrate prey. Carp muddy previously clear waters (Lampman 1946). This can influence the densities of zooplankton that can be important in the diet of hatchlings and young turtles (see Holland 1985b). Carp alter aquatic habitat when feeding on submerged and emergent vegetation. Introduced turtles, such as sliders, snapping turtles (*Chelydra serpentina*), and painted turtles (in western Washington) may compete with pond turtles and expose them to diseases for which pond turtles have no resistance. The potential for disease is greatly increased when sick pet turtles are released. In California, Oregon, and Nevada, a total of 17 species of exotic aquatic or semi-aquatic turtles have been found in pond turtle habitats (Holland and Bury 1998). Cattle trample and eat aquatic emergent vegetation that serves as habitat for hatchlings and they may crush nests. Dogs occasionally mutilate turtles (D. Holland, pers. obs.).

10.11.4 Disturbance

The western pond turtle appears to be relatively sensitive to disturbance. Disturbance may affect the frequency and duration of basking or foraging behavior, which may be particularly important for gravid females. Interruption of basking may lead to a delay in the maturation and deposition of eggs, leading to a decrease in hatching success or overwinter survival (Holland 1991c). Boat traffic and fishing may influence western pond turtle behavior or cause direct mortality.

10.11.5 Chemicals and Contaminants

The effect of biocontaminants on western pond turtles is largely unstudied. Bury (1972b) reported on the effects of a diesel spill on a California stream fauna. One western pond turtle was among the nearly 4,500 vertebrates killed and 30 pond turtles captured over 1 month after the spill had swollen necks and eyes, and sloughed off pieces of epidermis on their appendages. The 1993 Yonella Creek diesel spill in Oregon had negative effects on invertebrate food, habitat and health of western pond turtles. All 30 turtles recovered after the Yonella Creek diesel spill exhibited debilitating conditions that appeared to be the result of exposure to diesel fuel (USFWS 1993b). Given the long lifespan of turtles and their position as a tertiary consumer in the food chain, they may act as bio-accumulators of certain contaminants such as PCBs and heavy metals, a situation known to occur in other turtle species (e.g., common snapping turtle [Helwig and Hora 1983]).

Rotenone, a biodegradable substance extracted from a tropical plant, is commonly used in fishery management to eradicate fish species. Rotenone has been documented to kill amphibian adults and tadpoles, as well as turtles (Fontenot *et al*. 1994, McCoid and Bettoli 1996).

10.12 Inventory & Assessment of Existing Management Plans

10.12.1 Existing Management Strategies for the Bonneville Pool

- Western Pond Turtle Recovery Plan

The recovery plan identifies WDFW recovery goals for three populations of western pond turtle in the Bonneville Pool. Each of the three populations must reach at least 200 animals and meet conservation targets for age structure, reproduction, and habitat security. Currently, 80% of the recovery funding is provided by BPA. Uncertainty of future funding will be the limiting factor for achieving goals of the project. The WDFW is currently in its third contract year for this project. The minimum time anticipated for western pond turtle recovery is approximately 15 years.

10.12.2 Existing Management Strategies for the Lower Columbia River

- Provide for a fourth population of western pond turtles in the Columbia River Gorge below Bonneville Dam. A fourth population is needed to delist western pond turtle in Washington.
- Resurvey suitable western pond turtle habitat below the Bonneville Pool.

- Evaluate habitat for western pond turtle reintroduction project. Current reintroduction possibilities include the Steigerwald National Wildlife Refuge.
- Determine potential for existing population of western pond turtle in Vancouver lowlands (WDFW ownership).
- Determine feasibility of reintroduction of population in Vancouver lowlands.

10.13 Inventory & Assessment of Existing Restoration & Conservation Plans

10.13.1 Restoration & Conservation Projects: Columbia River Gorge Above Bonneville Dam

- Currently in the third year of BPA-funded western pond turtle recovery project in the Columbia River Gorge. This funding is part of Columbia River mainstem subbasin plan and is critical to maintaining the headstart program for augmentation of current populations in the Columbia River Gorge and the expansion of populations into suitable habitat. Key components are directed at implementing the WDFW recovery plan for the western pond turtle. Major focus is augmentation of existing western pond turtle populations in Skamania and Klickitat Counties. Reintroducing the third population at Pierce NWR to achieve conservation goals for downlisting western pond turtle in Washington. Included in the BPA project is funding for habitat improvement including improvement of nesting and basking habitat. In addition, WDFW and the USFS are involved with habitat acquisition projects in Klickitat and Skamania Counties. WDFW owns and manages approximately 250 acres of western pond turtle habitat in Klickitat County. The USFS has recently purchased approximately 100 acres of western pond turtle habitat in Skamania County.

10.13.2 Restoration & Conservation Projects: Below Bonneville Dam

- In the past five years, no work has been conducted on western pond turtles below Bonneville Dam.
- In 1992, WDFW conducted a comprehensive survey of wetland habitat below Bonneville Dam to the mouth of the Lewis River. No western pond turtles were found.

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Volume III, Chapter 11
Dusky Canada Goose

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11.0 Dusky Canada Goose *(Branta canadensis occidentalis, Baird)*

11.1 Introduction

The dusky Canada goose (*Branta canadensis occidentalis*) is a distinctive race of medium size (about 6 lbs. [3 kg]) and dark brown plumage that nests on Alaska's Copper River Delta, migrates through southeastern coastal Alaska and coastal British Columbia, and winters primarily in southwestern Washington and western Oregon. The population has been intensively managed since the 1950s with habitat preservation in the form of federal refuge creation and harvest regulations that reduced the harvest of dusky geese. Despite this, the size of the population has fluctuated considerably over the past three decades, with an overall decline since the late 1970s. Events on both the breeding and wintering grounds have affected the status of duskys. In 1964, an earthquake caused their nesting grounds to be uplifted an average of 7-ft (2 m), initiating decades of ongoing dramatically accelerated plant and animal succession and physiographic change. This succession has resulted in significant habitat changes, with associated effects on activities and populations of goose predators, and subsequent detrimental effects on productivity and numbers of geese.

Beginning in the early 1970s and increasing to the present, tens of thousands of Canada geese of several other races began wintering sympatrically with duskys. Only 25,000 Canada geese, the vast majority duskys, wintered in the Willamette Valley in 1973. Over 250,000 Canada geese, less than 10% duskys, winter there today. Harvest management that focuses on subspecies other than duskys became more complex and challenging in the face of this massive build-up of geese, particularly given the duskys' declining productivity and relatively high vulnerability to hunting. Habitat losses are now a critical threat to the long-term viability of the population. Crop depredations by geese that encourage farmers to plant alternative crops and increasing urbanization threaten goose habitat in their wintering areas. State agencies that manage harvest and habitat are under increasing pressure to reduce overall goose numbers while stabilizing or increasing dusky numbers. The dusky Canada goose is highly prized by consumptive as well as non-consumptive users, and revenue is generated by hunting and watchable wildlife activities that contribute to local economies.

11.2 Life History & Habitat Requirements

11.2.1 Life History

11.2.1.1 Diet

Food habits during nesting, brood rearing, and molt are poorly known. Geese were observed feeding on tidal mudflats in early spring and during brood rearing and molt (pers. obs.). They use spring melt pools in horsetail and sedge stands during the early spring melt, and they have been observed feeding on early sedge and horsetail shoots, and unfurled leaves of prostrate willow late in the melt (pers. obs.).

Duskys consumed at least 26 species of 13 families of plants on the eastern Copper River Delta during September and October (Hawkings 1982). Leaves were the most important component by volume overall, but seeds and roots increased in importance as fall advanced.

Although there have been no studies on diet of duskys during winter, it is of common understanding that they rely largely upon agricultural crops (e.g., Clark and Jarvis 1978, Pacific Flyway Council 1998). Federal and state refuge management directed towards providing food crops are likely a good reflection of the favored winter foods of duskys. Crops most commonly provided on federal and state lands, in descending order of acreage, include pasture grasses, moist soil (Fredrickson and Taylor 1982) vegetation, annual ryegrass, perennial ryegrass, fescue, Sudan grass/millet, clover and corn (Pacific Flyway Council 1998).

11.2.1.2 Reproduction

Arrival

The first dusky Canada geese arriving in spring on the Copper River Delta, Alaska are frequently observed in March (Isleib pers. comm.; Campbell and Rothe 1985; Crouse *et al.* 1996). Major influxes occur some time later, apparently depending upon weather conditions and the state of spring snowmelt. Records of major arrival dates range from 9 April (Campbell and Rothe 1986, Crouse *et al.* 1996) to 18-25 April (Bromley and Jarvis 1993).

Nest Initiation

Peak nest initiation (laying) dates have varied annually from 29 April - 5 May 1992 (Campbell 1992a) to early June, 1972 (Timm 1972, and Timm and Havens 1973). On the east Copper River Delta, initiation dates in 1978 and 1979 ranged from 6-31 May, with a peak from 6-11 May (Mickelson *et al.* 1980). Re-nesting occurs, especially in years of high early nest loss, causing the period of nest initiation to be much longer than for first nests, and in some years resulting in a bimodal distribution of initiation dates (e.g. Campbell *et al.* 1987, Grand *et al.* 1998). Duration of the initiation period was 30 and 39 days in 1974 and 1975 respectively (Bromley 1976) and averaged 38 days, with a maximum of 50 days from 1993-1995 (Crouse *et al.* 1996).

Incubation

Trainer (1959) determined a mean incubation period, i.e. "the number of days between the laying of the last egg and the hatching of the last egg," of 30.2 days (n=36, range=28-33 days). However, Bromley (1976), defining incubation as the period from the day after the last egg was laid to hatch of the first egg, found a mean of 27.4 days (n=21, range=25-31 days). Possibly the chilling effect of tidal flooding (Trainer 1959, Hansen 1961) slowed embryo development (e.g., Drent 1973, Bromley 1984) and led to a longer incubation period.

Constancy of incubation for successful dusky females was 89.5%, with twice the amount of recess time during the last third compared to the first two thirds of incubation (Bromley 1984). This pattern reflected changes in body weight, where steep declines were observed for the first two thirds of incubation, and no change thereafter, indicating a switch in emphasis from dependence on endogenous reserves to a much greater reliance on environmental food resources during the last phase of incubation. Constancy of incubation and recess frequency steadily declined during the first 13 days of incubation for unsuccessful nesting geese, while no change was detected during this period for successful nesters (Bromley 1984).

Hatch

Olson (1953,1954b) estimated peak hatch dates of 20-25 June 1953 and 22-27 June 1954 based on brood ages. Other peak hatch dates were about 1 July 1971, and 12-18 June, 1974 and 17-23 June 1975 (Bromley 1976). Crouse *et al.* (1996), for the years 1993-1995, noted earliest hatch dates of 4 June in 1994 and 1995, and 13 June 1993, with median hatch dates of 27 June 1993 and 17 June 1995.

Eggs

Eggs of dusky Canada geese on the Copper River Delta average about 2 in (56 mm) wide by 3 in (82 mm) long with a fresh egg mass of about 0.3 lbs (140-144 g) (Bromley and Jarvis 1993, Crouse *et al.* 1996). They are laid at the rate of one per day (Bromley 1976).

Mean annual clutch size has ranged from a low of 3.6 in 1971 to a high of 5.8 in 1965, with individual clutches of 2-8 eggs. Typically, low mean clutch size is observed in phenologically late springs (e.g. 1964, 1971, 1972) and high mean clutch size occurs in early springs (e.g., 1959, 1979).

Fledge

By 5 August 1953, about on- third to one-half of young observed could fly, whereas one quarter of the young were capable of flight on 5 August 1954 (Olson 1954b). An estimated one-half to three-quarters could fly by 12 August that year (Olson 1954a). Trainer (1959) estimated that 5% of goslings were still flightless on 19 August 1959, and Bromley (1976) noted a few flightless young as late as the second week of September.

11.2.1.3 Nesting

Nest Densities

Shepherd *et al.* (1967) established 15 random 24,000 yd^2 (2 ha) plots, in which they located 13 nests in 1966 (0.4/ha). Seven of these 15 plots hosted 27 nests (1.9/ha) in 1970 (McKnight 1971) and 20 nests (1.4/ha) in 1972 (Timm and Havens 1973). In an area overlapping part of Trainer’s study area, Bromley (1976) found a density of 0.8 nest/ha in 1975. Bromley (1976) in 20 random plots of 2 ha found a mean density of 0.6 nests/ha in 1974, and in larger non-random plots spanning low, medium and high density strata found densities of 0.3 to 1.3 nests/ha in 1975. Studies in the late 1960s through the 1970s reflected continually increasing densities of nests on the Copper River Delta.

Crouse *et al.* (1996), based on random plots within a 82 miles^2 (212 km^2) extensive study area, found average nest densities of 22.0 +4.3/km^2 (0.2/ha) from 1993 through 1995. In 1998, Youkey *et al.* (1998), in a repeat of this study, found 17.2 nests/km (0.17 nests/ha - uncorrected for detection rates and late-initiated nests) during the early search (i.e. probable re-nests found later not included). Youkey also examined nest detection rates by next-day repeat searches of plots with independent field personnel, and derived a detection rate of 83.2%+0.4% (95% CI) for nests on plots. Thus, a corrected mean density would be about 0.21 nests/ha. In an area overlapping with Trainer (1959) and Bromley (1976), Grand and Anthony (1998) found 412 nests in 13.5 km^2 (0.31/ha) within the high-density stratum in 1997. Based on these studies, nest densities apparently peaked in 1978, and declined thereafter to levels similar to those measured during the 1950s.

Renesting

Investigators have long suspected that re-nesting occurred, based on different ages of broods and observation of young broods late in the season (Olson 1953, 1954a), and based on patterns of nest initiation dates (Bromley 1976, Campbell *et al.* 1987, 1988; Campbell and Rothe 1989, Grand *et al*. 1998). Evidence became stronger with records of several second nests in the same bowls as those of earlier nests that had been destroyed (Campbell and Rothe 1989), and finally with the collection of 5 females in 1997 from relatively late-initiated nests, for which examination of ovarian condition provided certainty that at least 4 of the 5 were re- nesting (Grand and Anthony 1997). Grand *et al.* (1998) have attempted to model the extent of renesting based on nesting data during 1997 and 1998.

Nest Success

Although Mayfield-type nest success (Mayfield 1975) was not calculated until 1997 (Grand *et al*. 1998), at least until recent times even destroyed nests were easily detected, and were included in the apparent nest success estimates to alleviate the inherent bias in apparent nest success estimates. Clearly, nest success was much higher during the 1950s through the early 1970s than it has been since, and thus an overall declining trend is reflected. Note that in several years, nest success estimates are minimums, because late-initiated nests that typically are more successful than early-initiated nests were not followed to completion. Nevertheless, because nest success has been so low, net productivity of adults has declined over time.

The high degree of re-nesting in the 1980s and 1990s (Campbell *et al.* 1987, 1988; Campbell and Rothe 1989; Crouse 1995; Grand and Anthony 1997, Grand *et al.* 1998) may have mitigated, to some degree, the lowered success of individual females, particularly because nest success was higher in late-initiated nests compared to earlier ones in some years. Bromley (1976) and Bromley *et al.* (1995) found that most nest depredation on Canada geese occurs during the early stages of nesting. Although relatively low in absolute numbers, re-nests and late-initiated nests have higher success than do early nests.

11.2.1.4 Migration

Dusky Canada geese migrate along the Pacific coast of Alaska, British Columbia, and Washington. Based on collar observations, the islands in eastern Prince William Sound, Prince of Wales Island in southeast Alaska, and the Queen Charlotte Islands of British Columbia are important fall staging areas; although little is known about the habitats that dusky geese use in these areas or the length of time they use these areas in the fall or spring.

Fall

Relatively little is known about the distribution of duskys during migration. Hansen (1962) suggested that they migrate offshore, seldom stopping during the fall migration to wintering areas. Some areas used by geese during fall migration as determined from band returns include the southwest coast of Prince of Wales Island, Alaska; Graham Island, British Columbia: the northern tip of Vancouver Island, British Columbia; the west central coast of Vancouver

Island; and the southern interior of British Columbia (Hansen 1960). It is also clear that duskys use the Queen Charlotte Islands, British Columbia. Subsequent to their departure from Vancouver Island, or for those birds passing that point, the geese fly to Gray's Harbor and Willapa Bay, Washington (Chapman *et al.* 1969). Dawson (1909) reported that *occidentalis* migrate through, but were not a common winter resident in Puget Sound, Washington. From the southwest Washington coast, the majority of the population moves up the Columbia River to the mouth of the Willamette River where most turn south until settling in the central Willamette Valley, Oregon (Chapman *et al*. 1969).

Spring

Information is even more scant for the distribution of duskys during spring migration. Numbers briefly build on Sauvie Island, Oregon in the Lower Columbia River valley as the migration begins, and a subsequent surge in numbers is observed at Willapa Bay, Washington. A brief increase in numbers is again noted at the Queen Charlotte Islands, British Columbia. Large flocks have been observed in the Yakutat Bay, southeastern Alaska in spring (Gabrielson and Lincoln 1959, Petersen *et al.* 1981. Mickelson *et al.* (1980) and Hawkins (1982) found that most duskys migrated through the Copper River Delta without stopping during spring. Crouse (1994) surveyed the Copper and Bering river deltas weekly from 1 April through 1 May. Few geese were observed on 1 April, and numbers peaked on 23 April. Important use areas included Okalee Spit on 1 and 8 April, and coastal areas between the Edward and Bering rivers during the 16 April surveys. While geese were present on the Copper River Delta during surveys on 8 and 16 April, a large movement onto the area was noticed on the 23 April survey (Crouse 1994).

11.2.1.5 Mortality

Hansen (1962) was the first to estimate mortality rates of duskys. Using the composite dynamic method, he estimated the average annual mortality rate was 28.9% for adults, and 56.9% for juveniles for the period 1952-1960. Similarly, Chapman *et al*. (1969) analyzed data for 1952-1965, yielding average annual mortality rate estimates of 34.6%, 37.7% and 57.4% for adults, yearlings and juveniles, respectively. In recognition that the composite dynamic method can yield underestimates of survival rates (Seber 1972), Sheaffer (199321) re-analyzed and estimated survival rates for several periods during 1953 through 1990 for normal, leg-banded-only duskys, determining average annual adult survival rates of 65.8-77.2% and immature rates of 30.7-42.5%. Both Chapman *et al*. (1969) and Sheaffer (1993) also noted that adult survival rates increased over time in tandem with incremental restrictions in harvest regulations. Sheaffer (1993) also examined survival estimates based on resightings of marked birds for the period 1983-1990, and calculated mean annual adult survival rates of 78.8%. They were not significantly different from the rates estimated on the basis of band recoveries for the same period.

Following population modeling to simulate what was known of the dusky population from the early 1950s through 1989, Chapman *et al*. (1969) and Sheaffer (1993) concluded that large harvests were supportable by the population during the 1960s because the average recruitment rate was high. Subsequently, as recruitment rates declined during the 1980s, the population could not continue to increase despite modest increases in adult survival rates (Sheaffer 1993). Ultimately, very low rates of recruitment accompanied by moderate adult survival rates resulted in population decline. That is, as recruitment rates declined, population size became relatively more sensitive to small changes in adult survival. As severe restrictions were placed on hunting during the mid- to late-l980s, adult survival increased, leading to an apparent end to the decline.

Campbell and Griese (1987) estimated that over 85% of duskys were of breeding age, with 70% in the category of prime breeders aged 6-14 years. Longevity was subsequently noted to be at least 19 years of age, with evidence of breeding at 17 years of age (Campbell 1991b). Sheaffer (1993) concluded that the chances for the population to increase were favorable if recruitment and survival rates remained at or above levels typical of the late 1980s and early 1990s, noting also that the population would have a higher chance of increasing with greater variation in recruitment rates.

11.2.2 Habitat Requirements

11.2.2.1 Breeding Habitat

Almost the entire population of the dusky Canada goose nests on the Copper River Delta in south-central Alaska. Dramatically accelerated successional changes have occurred as a result of the uplift by the earthquake. By 1975, 23% of nests were in low shrub habitat (Eromley 1976), and an average of 46% were in low and tall shrub habitat by 1986 (Campbell 1990). Geese did not stop using sedge meadow and grass-forb nesting habitat even with the prolific colonization of those habitats by low and even tall shrubs.

During the late 1980s, beavers colonized the nesting area in great abundance, causing much of the area to be flooded (Campbell *et al*. 1988, Campbell and Rothe 1989, Campbell 1992b). Although this caused some nests to be flooded, an apparent decrease in large mammalian predators also may have resulted (Campbell *et al*. 1988, Campbell 1992). In recent years, no nests have been lost to flooding (Crouse *et al*. 1995).

Both Crouse *et al*. (1996) and Bromley (1976) concluded that geese preferred low shrub cover at nest sites: Eromley (1976) suggested that the geese may select for good visibility from nests and ease of escape if attacked. Campbell (1990) and Crouse *et al*. (1996) found that for 1982-86 and 1993-95, nests in all community types were equally susceptible to depredation.

11.2.2.2 Migration Habitat

Little is known of migration habitat for duskys. Hansen (1962) suggested that geese migrate offshore, stopping occasionally at the few accessible places en route. During early September to mid-October staging on the eastern Copper River Delta, geese use first salt marsh habitat, then freshwater meadow habitat (Hawkings 1982, Crouse 1994). Riverine habitats were more important than estuarine habitats. Bromley and Jarvis (1993) concluded that about half the energy costs of spring migration were derived from lipid reserves, with the remainder met through acquisition of food enroute.

11.2.2.3 Wintering Habitat

The lower Columbia River and the Willamette Valley provide ideal habitat for wintering Canada geese (Chapman *et al.* 1969, Cornely *et al*. 1985). Common agricultural practices, including dairy farming and production of ryegrass seed (Kimerling and Jackson 1985), yield high quality forage with high protein content (Riewe and Mondart 1985, and reviewed in Jarvis and Bromley 1998). Land acquisition for William L. Finley National Wildlife Refuge, Baskett Slough National Wildlife Refuge, and Ankeny National Wildlife Refuge were established in the 1960s and Ridgefield National Wildlife Refuge on the lower Columbia River was acquired in 1965 (Pacific Flyway Council 1998). Their mandate is to provide wintering habitat for dusky Canada geese. The Shillapoo and Vancouver Lake State Wildlife Areas in southwest Washington provide additional habitat. WDFW has expanded these areas through acquisitions.

Duskys tend to concentrate in and near federal and state wildlife refuges in the Willamette Valley, Oregon and the lower Columbia River in Washington and Oregon (Simpson and Jarvis 1979, Havel and Jarvis 1988, Lowe 1987 and Lowe pers. comm.). The largest proportion of wintering dusky Canada goose flocks consistently occur on and in association with William L. Finley National Wildlife Refuge, the southern-most of the valley refuges (Simpson and Jarvis 1979). Although there are seasonal differences related to the temporal proximity of spring and fall migration, the proportions of duskys are consecutively smaller at Sauvie Island, Ridgefield National Wildlife Refuge, and Willapa Bay National Wildlife Refuge (Havel and Jarvis 1988, Atkinson 1992).

Crop depredation has become a serious matter in the recent past. Landowners in Oregon and Washington have become unwilling to tolerate thousands of geese and the damage they cause to crops. A group of landowners, agency personnel, and others formed the Canada Goose Agricultural Depredation Working Group and developed a management plan to deal with increasing goose numbers and impacts on habitats (Pacific Flyway Council 1998). The plan outlines strategies to reduce Canada goose numbers, protect the dusky subspecies, improve goose habitat on public lands, outline critical habitats for acquisition, and quantify the dollar value of the crop losses among others. The future of goose wintering habitat and adequate dietary intake for geese is critical to future populations.

11.3 Population & Distribution

11.3.1 Historic

11.3.1.1 Population

The dusky population has been estimated annually in Washington and Oregon since 1953 (Hansen 1962), and has fluctuated from lows during the mid-1950s to highs during the late 1970s, and back to lows during the 1990s.

Kebbe (1958) reported winter inventories of duskys in western Oregon for 1947-58 (1962). Based on calculations of numbers of geese produced and harvested from studies by Olson (1953, 1954) and Trainer (1959) and compared with wintering ground studies (Kebbe in Hansen 1962), Hansen (1962, 1968) concluded that the best estimates of dusky numbers were the mid-winter aerial survey counts conducted in Oregon. However, he noted that counts did not include Washington and British Columbia, so he recommended adding 2,000 birds to annual estimates to account for this gap. Thus, total population estimates from 1953-60 ranged from 7,080 to 16,450 birds.

11.3.1.2 Distribution

Breeding Range

Early investigators considered all large dark Canada geese breeding along the Pacific coast, i.e. from Prince William Sound south to the Queen Charlotte Islands, British Columbia, to be occidentalis. Gabrielson and Lincoln (1959) considered the breeding range of duskys to include the region of Prince William Sound and Cook Inlet and inland through the Copper River drainage. They also noted that numerous flocks occurred in Yakutat Bay in spring.

Based on extensive aerial survey experience, Hansen (1962:303) delineated the breeding range of duskys, which extended “along the coast from the vicinity of the Bering Glacier on the southeast to Cook Inlet on the west, a distance of about 275 miles” (440 km). He noted that only small numbers occurred in Prince William Sound and the lower Susitna River, Cook Inlet, and even fewer near the confluence of the Bremner River with the Copper River. Hansen clearly defined two zones essentially devoid of breeding Canada geese because of habitat differences that helped to isolate and define dusky from Vancouver range and from Taverner's (or lessers) range.

Wintering Range

Baird *et al.* (1884) reported that *occidentalis* occurred south to California. Dawson (1909) noted that *occidentalis* migrated through the Washington coast area but was not a common resident in Puget Sound. Brooks (1917, 1923) reported large dark geese in interior British Columbia and on the coast. Jewett (1953) reported observations and harvest of duskys throughout the Willamette Valley and Sauvie Island from 1931-52. In *The Birds of Washington State*, Jewett *et al.* (1953) reported observations restricted to the coastal fringe, but concluded that the race was probably much more common than present information indicated. Finally, Hansen (1962) compiled an extensive set of information based on 1,129 recoveries of 3,943 duskys banded on the Copper River Delta, and on 164 recoveries of 3,593 Vancouvers banded in the vicinity of Glacier Bay, to demonstrate 1) that to a great extent, their ranges were discrete, and 2) that duskys wintered primarily in the Willamette Valley, Oregon.

11.3.2 Current

11.3.2.1 Population

As of January 2003, the total population size of dusky Canada geese is an estimated 16,724 (Table 11-1). The population estimate was obtained by multiplying the estimated number of neck-banded dusky geese by the ratio of unmarked to marked geese. The indirect estimate for 2002-03 of 16,724 dusky Canada geese was similar to the estimates from the previous two years (17,191 and 17,346, respectively) (Table 11-1). Duskys continue to maintain the increase in numbers since the previous low estimate in 1998-99. However, the winter indirect estimate has not followed increases exhibited by spring/summer surveys over the past few years (Drut and Trost 2003).

Table 11-1. Estimated total population size of dusky Canada goose, 1986- 2003 (Drut and Trost 2003).

Estimated number of neck-banded dusky geese (SE)	# unmarked # marked (SE)	Estimated total population of dusky geese (SE)
778.76 (50.78)	21.47 (1.93)	16,724 (1,856)

Year	Winter[a] Indirect Estimate (SE)	Breeding bird index	Spring/summer[b] Total goose index	%Young
1986-87	—	2,418	4,946	10.7
1987-88	—	2,121	4,528	9.8
1988-89	—	2,182	4,194	22.8
1989-90	12,438 (997)[c]	2,208	5,896	8.6
1990-91	19,768 (2,001)[c]	2,259	4,591	23.5
1991-92	17,996 (1,580)[c]	1,367	2,985	21.5
1992-93	—	2,250	5,637	21.3
1993-94	—	2,015	5,618	5.9
1994-95	7,948 (2,292)[d]	2,092	5,129	7.0
1995-96	18,175 (5,880)	1,668	3,199	3.9
1996-97	11,198 (1,711)	1,520	2,919	21.6
1997-98	21,280 (3,642)	1,759	3,587	10.8
1998-99	13,447 (1,679)	1,830	4,519	11.9
1999-2000	15,459 (2,459)	1,569	2,809	14.7
2000-01	17,346 (2,719)	1,276	2,343	22.7
2001-02	17,191 (2,820)	1,451	2,754	25.4
2002-03	16,724 (1,856)	1,599	3,444	30.5

[a] Survey conducted winter of 2nd year in series (i.e., 1987 for 1986/1987); [b] Survey conducted spring/summer of 1st year in series (i.e., 1986 for 1986/1987); [c] From Scheaffer 1993; [d] Survey conducted mid-March 1995; may have resulted in low estimate because of onset of migration. All other surveys conducted mid-winter.

11.3.2.2 Distribution

Breeding Distribution

The primary nesting range of *occidentalis* remains the Copper River Delta (Figure 11-1).

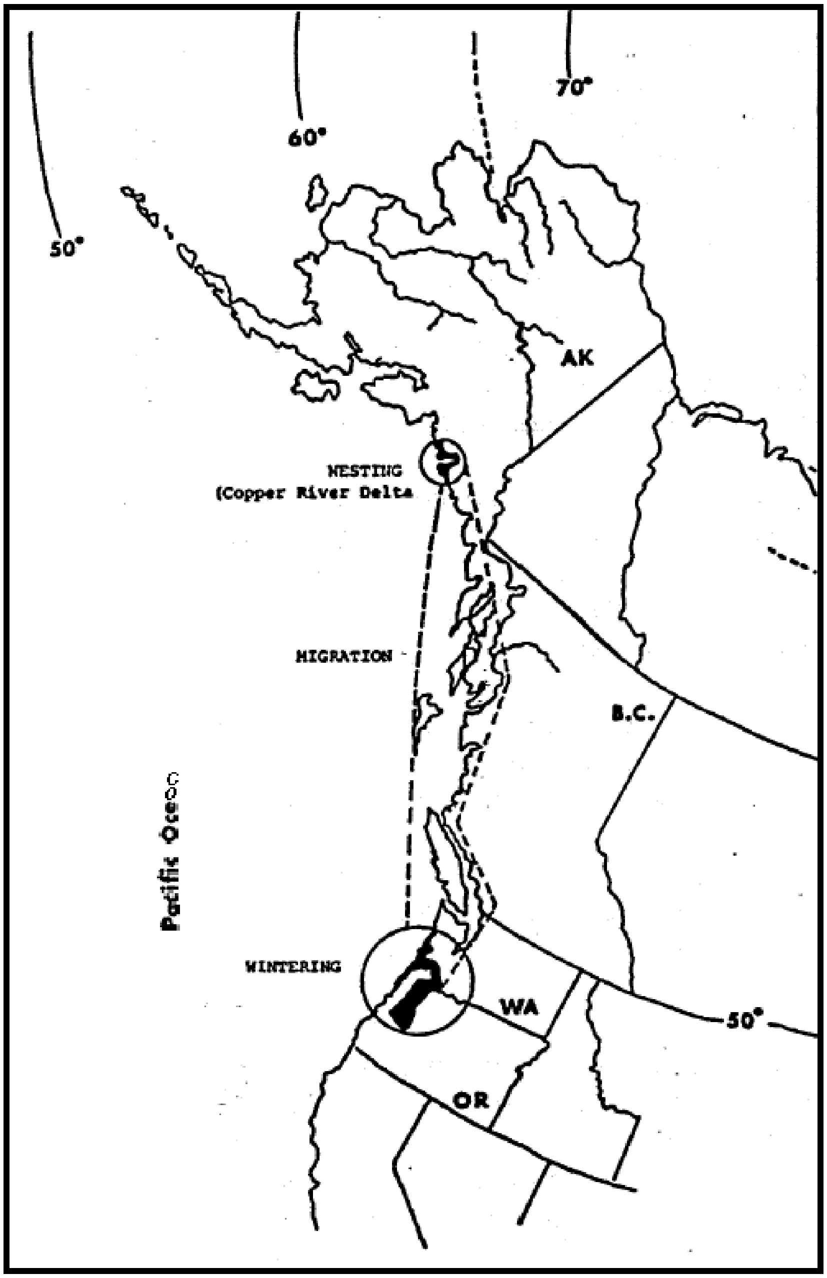

Figure 11-1. Primary range of dusky Canada geese (Pacific Flyway Council 1997)

Transplant to Willapa Bay, Washington

In accord with a previous USFWS policy to establish breeding Canada goose flocks, 41 dusky goslings were transplanted from the Copper River Delta to the Willapa National Wildlife Refuge in July 1958; 38 survived to 1961 when successful nesting began. This flock was free-flying by 1967 and was 407 birds in 1977 (Welch 1978). Refuge personnel now distinguish between migrant duskys and resident duskys and dusky hybrids (Atkinson 1987). About 120-175 resident birds are still associated with the refuge.

Wintering Range

In recent years, duskys have used Willapa Bay and Willapa National Wildlife Refuge, the lower Columbia River floodplain, the Woodland Bottoms, Vancouver Lowlands, and Ridgefield

National Wildlife Refuge in Washington (Figure 11-2). A high proportion of the population resides in the Willamette Valley and along the Columbia River from Portland to Astoria.

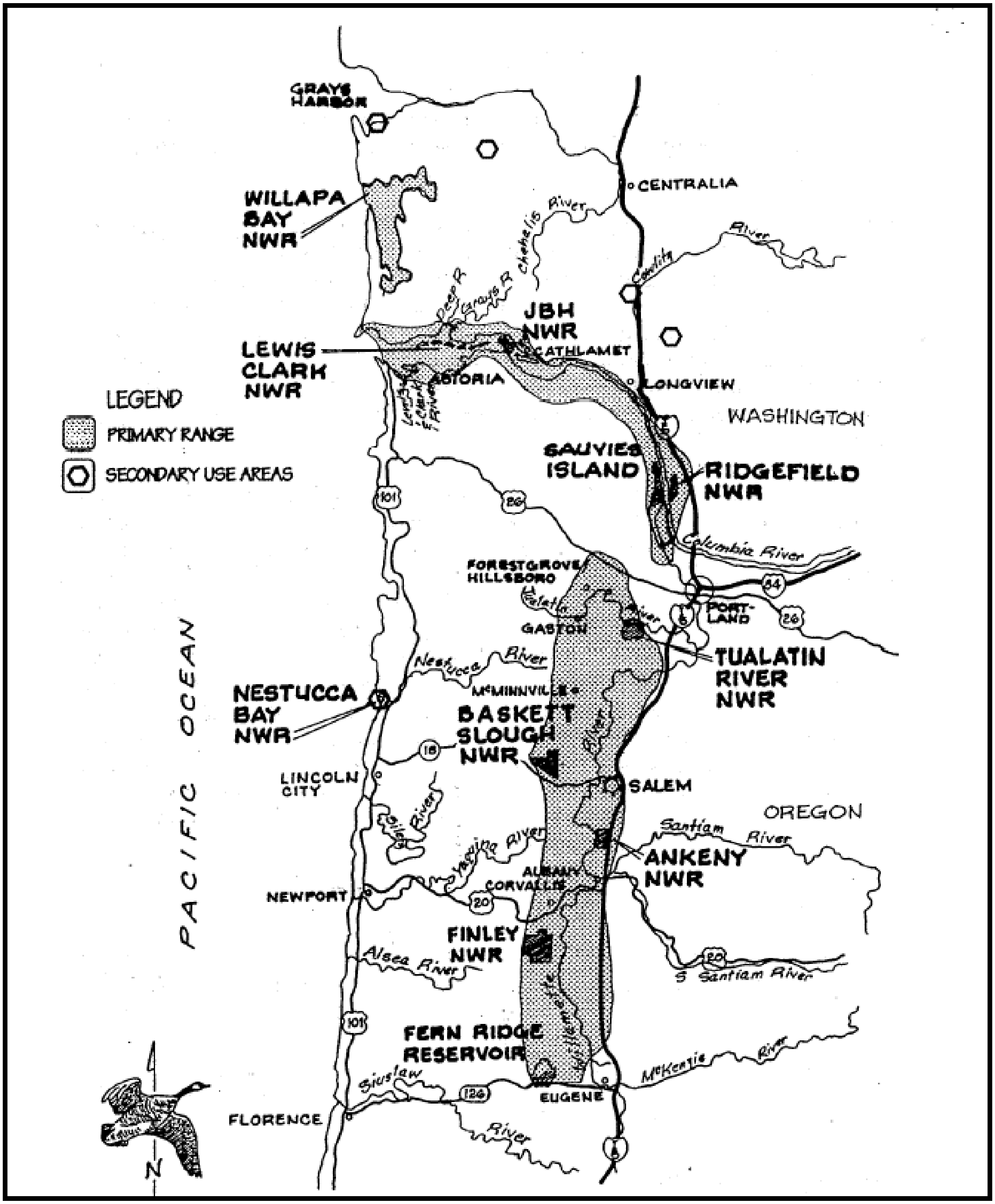

Figure 11-2. Winter distribution of dusky Canada geese (PFC 1997).

Cornely *et al*. (1998) identified 11 Washington and Oregon core areas used by wintering duskys in the mid-1980s.

11.4 Status & Abundance Trends

11.4.1 Status

The dusky Canada goose is classed as a migratory bird by federal regulation and a game bird by Washington rule. The Pacific Flyway and Washington Fish and Wildlife Commission regulate harvest.

11.4.2 Trends

Mid-winter indices from 1947 to the present indicate an increasing population to the mid-1970s (Figure 11-3). Jarvis and Cornely (1988), based on 3-year moving averages of mid-winter counts, concluded that the dusky population declined 8.3% per year from 1975-84, with most decline occurring since 1979. This decline continued through 1990, with widely fluctuating population estimates since that time (Pacific Flyway Council 1997, Drut *et al*. 1998).

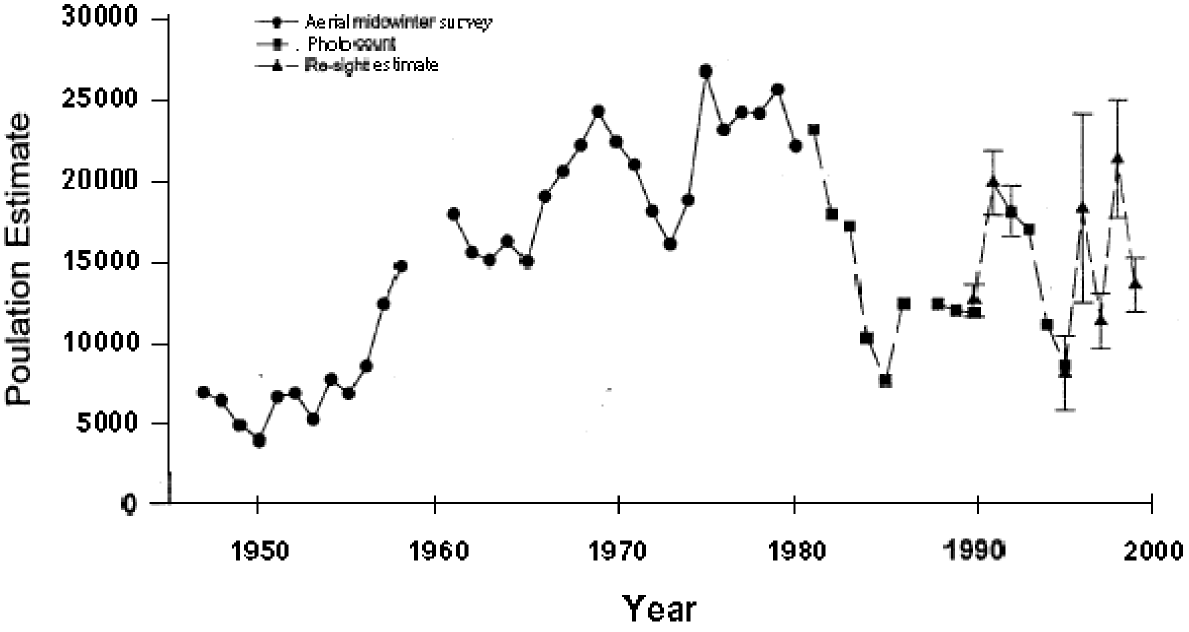

Figure 11-3. Winter population estimates of dusky Canada geese in Oregon and Washington, 1947-1999 (Bromley and Rothe 1999).

Similar to mid-winter indices, breeding ground surveys (Figure 11-4) from 1978–90 reflected a 50% linear decline over the 12-year period (Conant and Dau 1990), with a further decline in 1991 (Butler 1991). In an independent analysis considering both breeding population estimates and annual estimates of young produced, Stehn (1992) confirmed an average annual rate of population decrease from 1978-91 of 7-8% per year. However, the population rebounded in 1992 to a level similar to 1984 (Butler 1992, Conant and Dau 1990). In recent years, lower indices during 1995 and 1996 were offset by modestly higher indices during 1997 and 1998 (Eldridge *et al.* 1998). Assuming the early breeding ground surveys were equivalent to the current operational survey, a significant population decline occurred from 1978-85; however, the population has fluctuated around the level measured in the mid-1980s since then, without evidence of a further net decline.

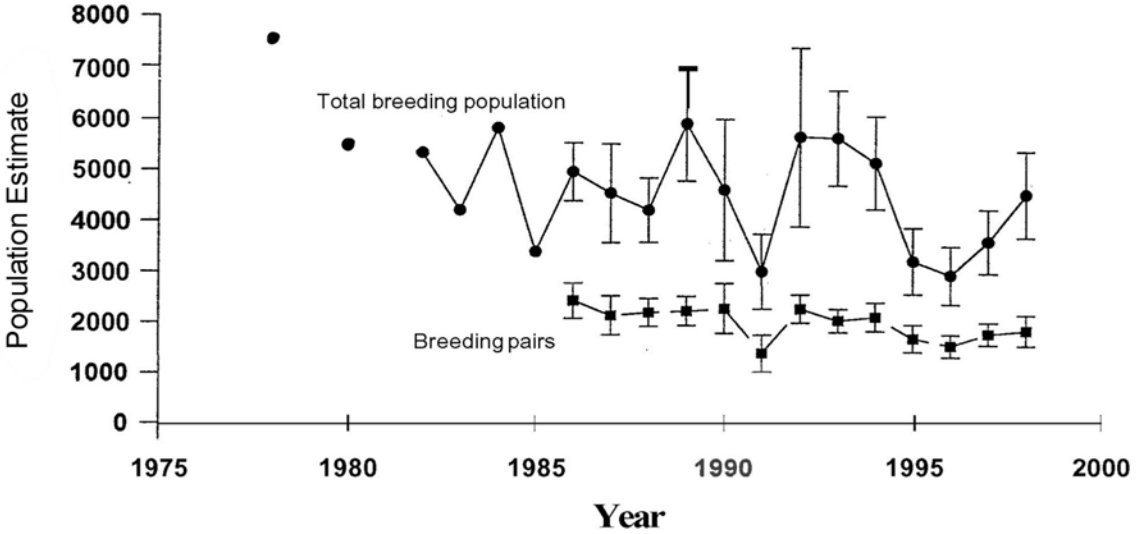

Figure 11-4. Breeding population estimates of dusky Canada geese on the Copper River Delta, Alaska, 1974-98 (Bromley and Rothe 1999).

11.4.3 Productivity

Productivity of duskys on the Copper River Delta has changed over the nearly three decades of study. During 1979, productivity dropped to a low level, and though moderate levels of production have occasionally occurred, years with peak production documented during the 1970s have not recurred. Over the long term, assuming no change in estimates caused by variation in methodology, production declined from the 1970s to the 1980s, with little or no change during the 1990s. Sheaffer (1993) assumed 80% of 2-year olds nested with nest success of 30% compared to 40% for 2- to 3-year old geese. These levels of nest success have only been realized in two of the past ten years.

11.4.4 Environmental Conditions

11.4.4.1 Habitat Distribution

Almost the entire population of dusky Canada geese nests on the Copper River Delta in south central Alaska. This has always been the case.

Dusky geese have always shown a high affinity for the Lower Columbia and Willamette Valley for wintering, finding winter forage in agricultural fields and wetland marshes. In recent years, duskys have consistently used Willapa Bay and Willapa National Wildlife Refuge, the lower Columbia River floodplain, the Woodland Bottoms, Vancouver Lowlands, and Ridgefield National Wildlife Refuge in Washington (Figure 11-2). A high proportion of the winter habitat is still found in the Willamette Valley of Oregon and along the Columbia River from Portland to Astoria. Other areas include Chehalis Bay, south Gray’s Harbor, Deep River, Wallace, and (to a lesser extent) Silver Lake and La Center, and possibly Port Susan Bay (Krueger 1995).

11.4.4.2 Habitat Status

Breeding

Breeding habitats historically were a mixture of USFS and Alaskan Native Corporations. Today, breeding habitat for the dusky Canada goose is nearly all owned by the USFS and managed as wildlife management area. Commercial uses are limited by permit and must be compatible with the primary focus of maintaining and enhancing wildlife habitat.

Wintering

Wintering habitats historically were a mixture of private ownerships in commercial operations to provide crop land and recreational uses. The lands were bought and sold with little regard for wildlife use. Today, wintering habitat is a mixture of public and private ownerships. Federal refuges and state wildlife areas protect many areas of high intensity dusky use. Private landowners control substantial portions of goose habitat in the Woodland and Vancouver areas. These private properties are not secure habitat and may be used for crop or industrial development that would be detrimental to the dusky and other goose subspecies.

11.5 Factors Affecting Population Status

11.5.1 Low Productivity on Breeding Grounds

The primary limiting factor for dusky Canada geese is low breeding ground recruitment. The "consensus opinion ... is that population growth is limited by poor recruitment directly related to the 1964 earthquake that significantly altered habitats on the primary breeding grounds." (Pacific Flyway Council. 1997. p. 23). "Continuing poor production [at Copper River Delta] has not only resulted in a population decline but has also lead to an unfavorable age structure...": (60% of population is older than 7 years old). At Copper River Delta "reduced capacity for goose production" has been due to high nest predation, which was less than 6% in 1959, but in the 1990's greater than 60%. Unfortunately, predator management options on the breeding grounds are limited, not cost-effective, or impractical (ibid. pp. 14-16).

The existence of 2 other breeding sites should be noted: 1) Middleton Island, which supports an increasing number of dusky Canada Geese (approximately 2000 currently) with high productivity; and 2) Prince William Sound, which has an unknown number of Dusky Canada Geese whose productivity is also unknown.

11.5.1.1 Breeding Habitat & Recruitment

In the absence of tidal influence on the Copper River Delta since 1984, the area has become less saline, and an accelerated rate of succession by salt-intolerant species has led to invasion of the breeding habitat by shrubs and trees, resulting in primarily closed habitat. Associated with this succession has been the advance onto the nesting grounds of a greater suite of predators, or at least of predators in greater numbers and with greater effectiveness. Greater depredation on eggs, nesting adults, and goslings has caused a gradual decline in goose productivity. Since 1978, years with the peak production that typified the earlier period have been lacking.

Studies of nest success have indicated that rates of depredation are similar in the various types of habitat used by nesting geese; that is, nests in each habitat type are predated in proportion to the number of nests located there (Campbell 1990, Crouse *et al*. 1996). Thus, even though the predatory agents largely responsible for nest loss have varied over time, nests seem to be similarly susceptible to destruction regardless of the habitat type in which they are found.

A current trend of particular interest is the tendency for eagles to establish new nest sites in maturing cottonwood and spruce trees on the nesting grounds of the outer delta (Grand pers. comrn). Given the tendency of adult eagles to remain within 2,000 yd (2 km) of their nest sites during nesting, and a nesting chronology that indicates egg laying in mid- to late April through early May (Bowman pers. comm.) overlapping with duskys, these efficient predators will become increasingly established and active on the high-density nesting areas of the geese. Their depredation on nesting female geese as well as on goose eggs has the potential to greatly elevate their effect on duskys.

Natural changes continuing on the Copper River Delta will lead to changes in habitat that will affect the types of depredation on nesting geese and their eggs. Colonization of the area by beaver in the mid-1980s caused the creation of many ponds and extensive areas of wet habitat, and thus may have deflected mammalian predators from many dusky nest sites (Campbell *et al*. 1988, Campbell and Rothe 1989). Nevertheless, avian depredation increased concurrent with lower depredation from mammals (Campbell *et al.* 1988), leading to little change in the net rate of nest loss. Undoubtedly some habitat changes will occur on the Copper River Delta that are unpredictable, and will cause similarly unpredictable chains of events that may affect goose productivity.

11.5.1.2 Depredation on Geese & Eggs

Predators of geese and goose eggs on the Copper River Delta during the reproductive season include brown bears, coyotes, wolves, red fox, river otter, mink, bald eagles, northern harriers, short-eared owls, glaucous-winged gulls, herring gulls, mew gulls, parasitic jaegers, ravens, northwestern crows, and possibly magpies. Depredation occurs on nesting adults, eggs, goslings and molting geese, but the known level of intensity varies widely throughout the season in association with the stage of reproduction and vulnerability of the birds. The amount of depredation has increased over the years. During initial studies when adult geese and goslings were being banded in the 1950s, several workers reported the presence of predators. Olson (1953, 1954a) noted the potential for nest depredation was high, because bears could travel along the slough banks and go from nest to nest. However, the effect of predation was apparently minimal (e.g. Elkins 1952). Both Courtright (in Olson 1954a, 1954b) and Trainer (1959) found low rates of nest depredation. Hanson (1962), referring to those studies, noted that there were so few predators after incubation was completed that juvenile mortality was considered negligible.

11.5.1.3 Depredation on Adults

Little depredation of geese is known to occur in early spring. During arrival on the Copper River Delta, flocks of foraging geese are very sensitive to bald eagles that frequently circle overhead, indicating at least the possibility of depredation attempts at that time. As geese disperse, become territorial, begin to lay eggs and become associated with nests, the adults become more susceptible to depredation. Geese may be vulnerable to predators while distracted during territorial disputes, or they may attempt to defend nests from predators, and be killed in the process. Based on remains of adult geese and their nests, bald eagles are a more important predator than previously thought.

Occasional remains of flightless duskys (based on stage of wing molt in carcass remains) have been found at mink and red fox dens (Bromley 1976). Brown bears have been observed from a distance engaging in behavior suspected to be running down and consuming molting geese and their young (Bromley 1976, Timm in Bromley 1976). Trainer (1959) found fresh remains of an adult-sized goose in the stomach of an adult coyote taken near Copper (Alaganik) Slough on the Copper River Delta. Wolves on the Copper River Delta are known to prey upon molting geese on occasion (Stephenson and van Ballenberghe 1995).

11.5.1.4 Depredation on Eggs

Generally not vulnerable when attended by geese (Bromley 1976), unattended eggs are vulnerable to depredation by avian, bear, and canid predators.

From 1987-98, the rates of nest loss to predators have remained high (Campbell and Rothe 1989, 1990; Campbell *et al*. 1992; Crouse et al. 1996, Grand and Anthony 1998, Youkey *et al.* 1998), and have included occasional depredation by wolves (Stephenson and Van Ballenberghe 1995). However, loss to mammalian predators has declined since 1990, perhaps partially in response to widely flooded areas and wetter habitat as a result of beaver activity (Campbell 1992b).

11.5.2 Hunting Mortality

In the mid- to late- 1900's, harvest by waterfowl hunters exceeded recruitment; hunting mortality was unsustainable and the population declined to a low of 7,500 in 1985. Hunting regulations have been restrictive since 1984, and the dusky quota is limited to 200 birds per hunting season. Hunting mortality is now low, and the population has recovered. A recent study found "average annual survival rates ... very high (76-85% ... and current hunting seasons are not adversely affecting population trends" (Pacific Flyway Council 1997, p. 18). Hunting, now strictly regulated, is no longer a factor limiting the population. In fact, the low Dusky population and continuing restrictive regulations for this subspecies are more likely to limit (or complicate) hunting opportunity for other subspecies of Canada Geese, which have increased dramatically in the past 30 years.

A limited harvest of Dusky Canadas occurs during migration in Alaska and British Columbia. In Alaska, bag limits are less restrictive than in Washington, but the beginning of the hunting season is delayed to allow most Duskies to migrate out of the area.

There is good evidence that hunting mortality can be a limiting factor for the dusky population. Duskys are known to be vulnerable to hunting and may be heavily harvested (Hanson 1962, Chapman *et al*. 1969, Jarvis and Cornely 1988). They frequent small fields which provide better access to hunters, they approach fields at lower altitudes with less circling before landing, they feed with smaller numbers of other geese than do other races of Canada geese (Have1 and Jarvis 1988), and they are known to be 2.7 to 3.0 times more vulnerable to hunting than are Taverner's Canada geese (Simpson and Jarvis 1979, Jarvis and Cornely 1988).

In 1984, hunting restrictions began to give extra protection to duskys. Seasons were delayed in Alaska to allow the departure of duskys before the hunting season, and bag limits and seasons were reduced in both Washington and Oregon (Pacific Flyway Council 1997). In 1985, hunting was limited to a quota of 300 duskys; this was reduced to 200 in 1995, with the Canada goose season to be closed in western Oregon and southwestern Washington when the quota was reached. A quota of 250 is in effect today (Table 11-2). In the recent past, a greater emphasis on hunter education has resulted in the reduction of dusky mortality. Hunters participating in the hunt must complete a home study course and pass an examination to be able to hunt geese in the special hunt areas. Harvest rates on duskys are now very low, less than 5% of the total goose harvest (Figure 11-5). It is unlikely that hunting limits the population.

Table 11-2. Composition and estimated total harvest of Canada geese in western Oregon and southwestern Washington, as determined from reporting station bag check information, 1984-1996 (PFC 1997).

Oregon

Year	*Dusky*	*Cackler*	*Taverner*	*Lesser*	*Western*	*Vancouver*	*Aleutian*	*Unknown*	*Total*
1984	603	0	641	0	0	0	0	21	*1,265*
1985	157	8	1,156	257	95	2	0	0	*1,675*
1986	134	19	1,157	103	0	0	0	127	*1,540*
1987	118	54	2,524	235	258	3	0	1	*3,193*
1988	142	26	3,067	273	415	3	0	0	*3,926*
1989	79	16	2,563	346	1,623	5	2	0	*4,634*
1990	177	18	2,684	572	1,846	6	0	0	*5,303*
1991	121	42	2,287	378	1,091	9	0	0	*3,928*
1992	147	36	2,294	422	1,333	8	0	0	*4,240*
1993	188	72	2,699	748	1,348	41	0	4	*5,100*
1994	142	1,220	2,669	447	1,415	9	1	8	*5,911*
1995	83	1,758	1,885	462	598	10	1	2	*4,799*
1996	87	2,503	1,773	809	1,110	9	1	0	*6,292*
Washington									
1984	37	0	63	0	20	0	0	0	*120*
1985	66	11	113	116	67	0	0	25	*398*
1986	36	8	172	51	241	0	0	0	*508*
1987	45	7	478	225	224	4	1	34	*1,018*
1988	43	17	617	136	763	0	1	6	*1,583*
1989	52	37	455	92	391	9	0	0	*1,036*
1990	65	28	555	165	383	20	0	3	*1,219*
1991	88	39	675	295	483	14	4	11	*1,609*
1992	91	84	1,340	270	722	25	2	0	*2,534*
1993	90	93	944	299	697	8	3	1	*2,135*
1994	77	422	1,011	246	703	31	3	2	*2,495*
1995	57	320	787	134	516	12	6	1	*1,833*
1996	35	1,018	1,724	222	967	9	0	2	*3,977*

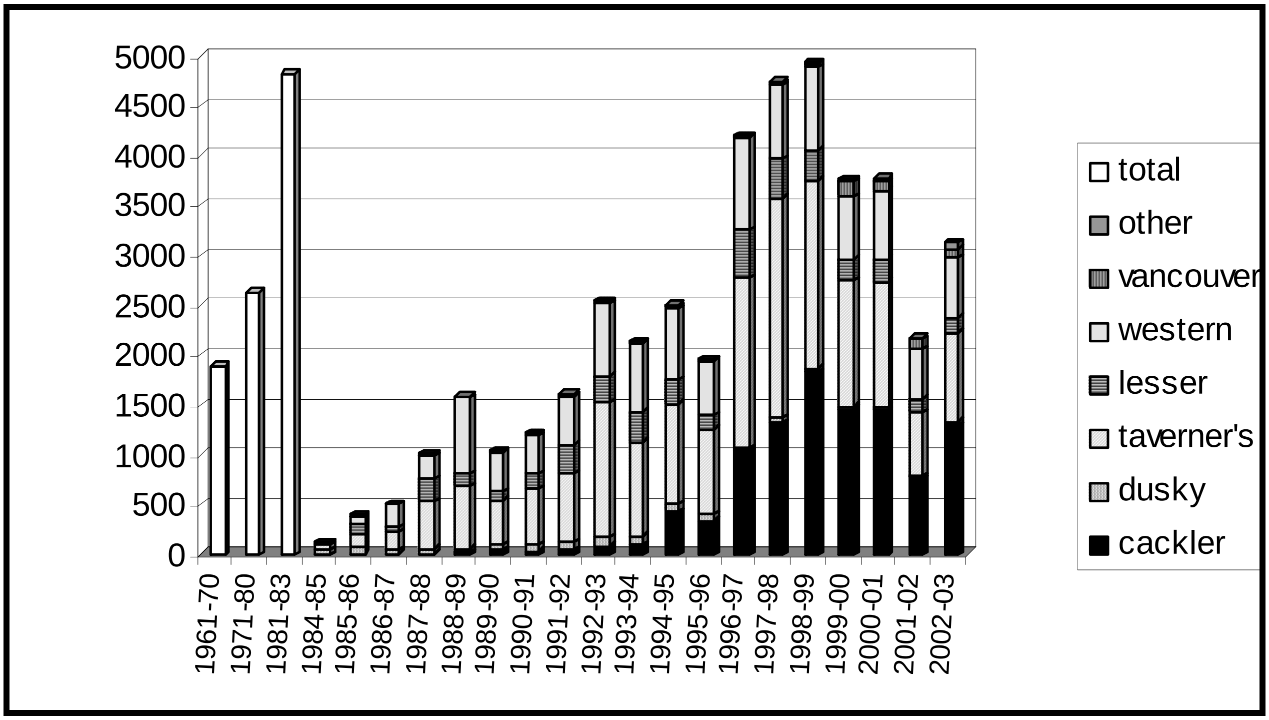

Figure 11-5. Southwest Washington Canada goose harvest summary, showing the geese harvested by year and by species. In recent years, dusky harvest has been a very low proportion of the total.

11.5.3 Wintering Habitat

The Lower Columbia River floodplain is one of only two primary wintering areas (Willamette Valley being the other). Wintering habitat is probably not limiting at present, although uses of private lands are changing. Number of acres of winter habitat under active management in 1998 on these public lands for goose foraging was about 17,458 acres (Pacific Flyway Council 1998). The amount of winter habitat under management has increased somewhat since that time, with additions to the Shillapoo Wildlife Area, Ridgefield Refuge and Clark County conservation holdings. Even with these additions, if populations of dusky geese were to increase significantly, or if private lands now providing forage were converted to other uses, wintering habitat could become a limiting factor.

As an example, conversion of bottomland pastures along the Columbia River to hybrid cottonwood plantations already threatens feeding sites. Bottomland pastures are also being converted for housing and development. Loss of wintering habitat and lack of sanctuary on refuges moves geese to remaining agricultural lands, increasing depredations (Pacific Flyway Council 1997, p. 20). It is not likely that public land now managed for geese can provide the entire amount of winter habitat required.

Prior to the establishment of federal refuges, hunt clubs strongly influenced the distribution and harvest of wintering duskys in Oregon (Hansen 1962, Chapman *et al*. 1969). Chapman *et al*. (1969) report that in the absence of a public refuge program, the harvest could have been much greater had the hunt clubs not provided refuge and regulation of the take. However, by the late 1960s, refuges had assumed this role (Hansen 1968, Chapman *et al*. 1969).

Duskys arrive in the lower Columbia River and Willamette Valley prior to the arrival of the most abundant races, Cackling and Taverner's Canada geese (Jarvis and Bromley 1998, Simpson and Jarvis 1979). The relative abundance of duskys is consistently highest at William L. Finley National Wildlife Refuge, particularly after the hunting season is over. Sheaffer (1993) studied subflock behavior based on 947 individually marked duskys, and concluded that those wintering at the southern and northern extremes of their wintering range had the highest wintering site fidelity. Over 65% of these geese were not observed outside of their respective wintering areas. The marked geese formed 9-10 clusters of 191-206 groups averaging 2.8 birds per group each of the three years of study, and groups had the same affiliations during both harvest and non-harvest periods (Sheaffer 1993).

Duskys typically select smaller fields for feeding than do other sympatric races of wintering Canada geese (Simpson and Jarvis 1979, Havel and Jarvis 1988). Based on research at Sauvie Island, Havel and Jarvis (1988) concluded that duskys are segregated during commuting flights but mixed during feeding, select fields with fewer geese to feed in, and approach lower and circle less prior to landing than do other subspecies. These characteristics result in higher vulnerability of duskys to harvest (Simpson and Jarvis 1979, Havel and Jarvis 1988, Jarvis and Cornely 1988).

Habitat losses in the wintering area are taking place at an alarming rate. Crop depredations have caused many farmers to select crops that geese do not eat and reduce the total winter habitat base. Conversions to berry crops are common in the Woodland Bottoms. Other farmlands have been leased to produce hybrid cottonwood for the pulp industry. Land is being converted to industrial development and housing. Securing adequate dusky habitat in the future will be important to the long-term vitality of the subspecies.

11.5.4 Sanctuary

Recreational use, such as hunting, dog training, bird watching, hiking and jogging, is allowed on some wildlife refuges and management areas. These activities reduce the area's usefulness to dusky Canada geese for foraging and sanctuary. Disturbances on managed wildlife areas also moves geese from public lands to private agricultural lands, where they along with other Canada goose subspecies, may cause depredations and be exposed to hunting pressure.

While there is curently a number of refugia for dusky geese where hunting is not allowed, there is also an increasing recreational pressure on these areas and indeed all areas, which may become a limiting factor in the future. In addition it is not possible to provide a refugium for one subspecies of goose without others using it too, which in the end contributes to the overall increase in wintering goose numbers.

The mixing of subspecies in winter, the scarcity of public hunting opportunities, and hunter confusion between subspecies complicate hunting opportunity to protect Dusky Canada Geese. Protective regulations and quotas to protect dusky Canadas increase the difficulty and expense of managing hunting programs both for this and other Canada Goose populations, and reduce hunting opportunities for the more abundant subspecies.

11.6 Inventory & Assessment of Existing Management Plans

- Pacific Flyway Council. 1988 Guidelines for management of the dusky Canada goose.

The management plan for the Dusky Canada goose was developed by USFWS, ODFW, WDFW, OSU, and Pacific Flyway representatives. This group developed harvest, nest survey, management and research tasks with the goal of improving the declining dusky population. If these tasks are funded, then the population of dusky geese will reach a level where special protection is not needed. Funding has been limited recently and many projects are not being implemented as planned.

- Pacific Flyway Council. 1998. Pacific Flyway management plan for NW Oregon/SW Washington Canada goose agricultural depredation control

This plan is a list of strategies and tasks to reduce the agricultural depredation committed by geese on private property. The plan was developed by WDFW, ODFW, USFWS, APHIS-WS, OSU, and the Oregon and Washington Farm Bureaus. The funding for this plan is inconsistent and recent reductions have caused landowners to potentially suffer more crop damage. Assistance from agencies to landowners has also declined by lack of funding.

- Wildlife Area Plan for Vancouver Shillapoo Lake Wildlife Areas

This plan outlines land management practices that will enhance goose habitat and provide a secure resting area for wintering geese. The plan was written by the Wildlife Area manager for WDFW with involvement of a Citizens Advisory Group and review by other biologists. Funding has limited implementation of all the appropriate land practices that could enhance goose habitat and improve population status.

11.7 Inventory & Assessment of Existing Restoration & Conservation Plans

- Annual Hunting Regulations, WDFW

The WDFW Wildlife Commission has adopted rules that reduce the harvest of dusky geese by implementing a daily and annual quota for dusky harvest. Hunters are also required to pass a home study course before hunting geese in the area where duskys winter. This program has reduced dusky harvest from 25% of all geese harvested to less than 5%. Funding limitations may restrict the number of hunt days or length of season.

- Nest Searches on Copper River Delta, coordinated by USFS

Nest searches are conducted every other year on the Copper River Delta to monitor population trends in the breeding grounds. The effort is coordinated by the USFS at Cordova, Alaska, and assistance is provided by ODFW, WDFW, ADFG, DU and other interested volunteers. Data is used to evaluate harvest quotas and harvest management.

- Banding Operations on dusky Canada geese

Dusky Canada Geese are leg banded and neck collared on a biannual basis. The project is designed to provide data for estimates of the wintering population in Oregon and Washington. The project is coordinated by ADFG and participants include WDFW, USFS, and ODFW. Funding is needed to purchase collars and charter a helicopter to conduct the capture.

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Volume III, Chapter 12
Caspian Tern

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12.0 Caspian Tern *(Sterna caspia)*

12.1 Introduction

Despite recent population increases, the Caspian tern (*Sterna caspia*) is of conservation concern in the Pacific Northwest because of the concentration of breeding terns at relatively few sites and fisheries conflicts at the Columbia River estuary, where 2/3 of the Pacific Coast and 1/4 of the North American population occurs. Although not listed at the national level, the species currently is listed as threatened or endangered in three states or provinces and is considered of special concern in ten more. The Caspian tern still occupies most of its historic range and has expanded slightly into new areas.

Historically, the Caspian tern suffered from harvest for the millinery trade, egging, human disturbance, habitat loss at interior wetlands, and, more recently, from contaminants. Historic population numbers are unknown but appear to have been substantially reduced early in the century. Relatively accurate population data for the Caspian tern in North America were unavailable until the late 1970s, when concerns over coastal habitat modification and offshore oil development prompted national multi-species surveys of colonial nesting waterbirds. Estimates of the US breeding population were roughly 9,454 pairs in the mid-1970s to early 1980s and 20,948 pairs in the late 1980s to late 1990s. Since the late 1970s, the population has increased in four of five major breeding regions in North America, and the continental population is estimated to be a minimum of 32,000 to 34,000 pairs, distributed differentially among regions: Pacific Coast/Western (interior) (45%), Central Canada (28%), Great Lakes (19%), Gulf Coast (7%), and Atlantic Coast (<1%).

Continent-wide population increases were fueled initially by the reduction or elimination of some historic pressures (e.g., hunting for millinery trade) but more recently by changes in breeding habitat and prey resources. Occupation of relatively stable artificial habitats (e.g., dredge spoil islands) has greatly concentrated the tern population leaving it more vulnerable to stochastic events, such as disease outbreaks, severe storms, disruption by predators or human disturbance, and oil spills. Caspian tern population increases in the Pacific region from the mid-1980s to 2001, primarily in the Columbia River estuary, may largely reflect the crucial juxtaposition of stable human-created habitats in conjunction with a predictable food supply. Human exploitation of native fish communities leading to dominance of small fish species favored by foraging terns appears to be a significant factor in tern increases in the Great Lakes and central Canada.

Conservation efforts will be most effective if focused on multiple fronts, including monitoring tern populations, resolving management conflicts with other species by addressing root causes, reducing risks to the tern population by distributing breeding colonies among a greater number of sites, filling gaps in knowledge of biology and threats on migration and the wintering grounds, and educating the public about the value of colonial waterbirds and possible effects of human actions on Caspian terns.

12.2 Life History & Habitat Requirements

12.2.1 Life History

12.2.1.1 Diet

Caspian terns are piscivorous in nature (Harrison 1984), requiring about 0.4 lbs (165 grams, 1/3 of their body weight) of fish per day during the nesting season. Diet analyses in 1997 showed that juvenile salmonids constituted 75% of the food consumed by the Rice Island colony (Roby *et al.* 1998). During the peak of the smolt migration, which coincides with the peak of nesting activity in May, the diet of Caspian terns on Rice Island was 98% salmonid smolts. Roby *et al.* (1998) estimated that the Caspian tern colony nesting on Rice Island consumed 6.6 to 24.7 million salmonid smolts in the estuary. Salmonid consumption rates are unknown for Threemile Canyon and Crescent Island, but they may be similar to rates found at Rice Island.

Roby *et al.* (1998) estimated that avian predators consumed 10-30% of the total estuarine salmonid smolt population in 1997; this means that between 100,000 and 600,000 listed smolts are being consumed. The large majority of salmonids being consumed by Caspian terns are hatchery fish (Independent Multidisciplinary Science Team 1998); many are from hatcheries constructed to mitigate the impacts of dam construction.

Breeding Caspian terns eat almost exclusively fish and rarely take crayfish, insects, and earthworms (Parkin 1998, Cuthbert and Wires 1999, P. Spiering pers. obs.). Globally, Caspian terns catch a variety of fish species with shallow plunge dives (Cuthbert and Wires 1999). The sizes of fish caught and diet composition are largely determined by geography and annual and seasonal prey availability, but most fish are between 2-10 in (5-25 cm) (Cuthbert and Wires 1999, Thompson *et al.* 2002, Roby *et al.* 2002).

In Oregon, concern over salmon conservation has motivated an intensive study of Caspian tern diets in the region (USACE 2001; Collis *et al.* 2001a, 2002; Roby *et al.* 2002). During 1999 and 2000, the diet of terns nesting on Rice Island in the Columbia River estuary was 77-90% juvenile salmonids, including coho salmon (*Oncorhynchus kisutch*), chinook salmon (*O. tshawytscha*), and steelhead (O. *mykiss*) (Roby *et al.* 2002). From 1999-2001, diet on East Sand Island, closer to the mouth of the Columbia River than Rice Island, was primarily non-salmonids, including anchovy (*Engraulis mordax*), herring (*Clupea pallasii*), shiner perch (*Cymatogaster aggregata*), sand lance (*Ammodytes hexapterus*), sculpins (*Cottidae*), smelt (*Osmeridae*), and flatfish; the yearly proportion of salmonids in the diet ranged from 33-47% (Roby *et al.* 2002). In 2000, diet on Threemile Canyon Island in the mid-Columbia River was 81% salmonids, with the remainder bass (*Micropterus spp.*), yellow perch (*Perca flavesceens*), and suckers (*Catostomus spp.*) (Collis *et al.* 2002). While salmonids comprised 65% of the diet of terns nesting on an experimental barge in Commencement Bay in May 2001 (Collis *et al.* in press), on the other hand, salmon were very uncommon diet items farther west on the outer coast in Grays Harbor, Washington (Smith and Mudd 1978, Penland 1981).

12.2.1.2 Reproduction

Most individuals do not breed until at least 3 years old, and usually wait until 4-5 years old. Pacific coast birds averaged 8.6 years. The average number of Caspian tern eggs per clutch is 2-3. The number of clutches per year is one or less. Caspian terns are seasonally monogamous. Caspian terns nest on bare open ground, on islands, on flat sand, or gravel or shell beaches.

12.2.1.3 Nesting

The terns arrive in April and nesting starts at the end of the month (Roby *et al.* 1998). Caspian terns form nesting colonies of a few hundred to thousands of pairs. To avoid predators, terns construct their nests on islands (Harrison 1984), and prefer barren sand. Clutch size is usually two eggs (Harrison 1984). First-time breeders often (58%) breed away from the natal colony. Caspian terns fledge at 30-35 days, but are partially dependent on the parents for 5-8 months.

Caspian terns forage 6.2-7.5 miles (10-12 km) from the colony and may forage up to 30 km from the colony. They defend a small territory around the nest site, about 1.6-4.9 ft (0.5-1.5m) in diameter. Caspian terns have high site fidelity to their summer range (IBIS 2003).

12.2.1.4 Nests & Nest Spacing

Caspian terns nest either in single-species colonies or in multi-species assemblages with other ground nesting waterbirds (gulls, skimmers, other terns, cormorants, and pelicans) (Cuthbert and Wires 1999). Colony sizes, varying widely among locations and years, typically range from tens to hundreds of pairs. Terns rarely breed as single pairs or small groups (2-3 pairs) or in colonies >1,000 pairs (Cuthbert and Wires 1999, Wires and Cuthbert 2000). Nests typically are densely packed at distances of 0.4-1.5 m as determined by territorial defense of a breeding pair (Cuthbert and Wires 1999). At large colonies in the Columbia River estuary, nesting density has varied from 0.25- 0.78 nest/m ^{2} depending on local habitat availability (Roby *et al.* 2002).

Nest sites often are on the highest point of low-lying islands, presumably for unobstructed views and to avoid flooding. Proximity to other terns, though, may override elevation in the selection process (Cuthbert and Wires 1999), and tern nests often extend to near the water’s edge in single-species colonies or often cluster on the edge of colonies of gulls or pelicans that initiated nesting prior to the terns (D. Shuford pers. obs.).

Nest substrates vary from sand, sand-gravel, spongy marshy soil, or dead or decaying vegetation to hard soil, shell banks, limestone, or bedrock. Of experimental nest substrates in Ontario, terns preferred sand over pea-gravel and crushed stone and all of these over pre-existing hard packed ground (Quinn and Sirdevan 1998). Nests range from simple depressions or hollows in a bare substrate to nests lined (or built up elaborately) with debris, such as shells, crayfish chelipeds, dried grasses and weed stems, wood, chips of salt crust, or pebbles (Bent 1921, Cuthbert and Wires 1999). Adult terns may raise rim heights of nests by >1 in (3 cm) in areas subject to immediate flooding and may move small chicks >100 m to alternate scrapes if the original nest is disturbed (Cuthbert and Wires 1999).

12.2.1.5 Migration

Spring migrants first arrive at breeding sites between mid-March to mid-May depending on latitude, elevation, and coastal or interior location (Cuthbert and Wires 1999). Migratory terns regularly move along major water features, such as the Columbia River (Cuthbert and Wires 1999). On the coast, Caspian terns first appear in March with a peak in April, later inland.

The timing of southward migration varies with region (Cuthbert and Wires 1999), but fall movement has been noted as early as late June along the Pacific Coast (Gilligan *et al.* 1994). More typically, the peak of fall migration occurs between mid-July and mid-September (Cuthbert and Wires 1999) with stragglers leaving by the end of November (Gilligan *et al.* 1994, Peterjohn 2001). Oregon breeders depart colonies in late June and July.

Most Caspian terns congregate for migration at traditional foraging locations along marine coasts and major rivers or freshwater lakes about a month after young have fledged (Cuthbert and Wires 1999). Terns migrate singly or in groups that range from only a parent and young to rare flocks of thousands (Gilligan *et al.* 1994, Stevenson and Anderson 1994).

Caspian terns winter in southern California, Gulf Coast and southeastern US coast, Mexico, and the West Indies. Washington birds migrate 1,584 miles (2,550 km) to Mexico.

12.2.1.6 Mortality

Caspian terns in the West Coast population are reported to live up to 27 years, over half of the fledglings reach their fourth year, and individual birds have a breeding life expectancy of nearly 9 years (Gills and Mewaldt 1983). Maximum life span is greater than 20 years. The oldest known wild bird is 29 years and 6 months. The greatest mortality occurs during the first 6 months of life. Once a bird reached maturity, it will likely survive a long time.

12.2.2 Habitat Requirements

12.2.2.1 Breeding

Caspian terns typically locate their colonies close to a source of abundant fish in relatively shallow estuarine or inshore marine habitats or in inland freshwater lakes, rivers, marshes, sloughs, reservoirs, irrigation canals, and (low-salinity) saline lakes (Cuthbert and Wires 1999).

Aerial surveys of terns breeding on Rice Island in 1998 determined that 50% of all terns seen off the colony were within 5 miles (8 km) of the island, 75% within 9.3 miles (15 km), and 90% within 13 miles (21 km) (Collis *et al.* 1999). Monitoring the movements of Caspian terns breeding at East Sand Island in 2001 found 76% of all off-colony detections were within the estuary; the remainder were in the vicinity of the nearshore Oregon coast (6%), Willapa Bay (16%), or Grays Harbor (2%, Collis *et al.* 2001b).

12.2.2.2 Nesting

Caspian tern colonies typically form at sites isolated from ground predators and human disturbance and within reach of abundant prey resources. Nesting sites typically are on sandy, earthen, or rocky islands or reefs, sandy beaches, and inland on floating tule-mat islands (formerly in Klamath Basin) or, rarely, peninsulas in lakes (Bent 1921, Cuthbert and Wires 1999). Although coastal birds may breed on natural estuarine, salt marsh, or barrier islands, they increasingly nest on human-created habitats, such as dredge spoil islands, salt pond levees, islands created for salt marsh restoration, or islands created to enhance nesting sites for endangered species such as the California Least tern (*Sterna antillarum browni*). In South San Francisco Bay, Caspian terns prefer to nest on long continuous or interrupted levees or long islands free of vegetation, large rubble, or debris (Rigney and Rigney 1981). Caspian terns have also been attracted to nest on experimental sand-covered barges in Commencement Bay, Washington (Collis *et al.* in press) and on rafts in the Great Lakes (Lampman *et al.* 1996); the latter were used as transitional nesting sites before attracting the terns to artificial islands designed for use by multiple species of colonial nesters (Quinn *et al.* 1996, Pekarik *et al.* 1997). Terns in Puget Sound have also nested on the roof of a flat-topped building, among the metallic rust debris of a floating barge, and on broken sand bags securing black plastic covering contaminated soil in the Commencement Bay area (Collis *et al.* in press, Thompson *et al.* 2002). In 2002, a new colony formed in San Francisco Bay on an insular portion of a dilapidated pier along the waterfront of the city of San Francisco (D. Singer, J. Yakich in lit.). Nesting islands in interior wetlands are usually in large freshwater or saline lakes, reservoirs, or rivers, and sometimes on islands created for nesting waterfowl or colonial waterbirds at refuge impoundments. In California, Caspian tern colonies have formed on intact or broken levees of agricultural evaporation ponds, sewage ponds, floodwater storage basins, and flooded agricultural fields.

12.2.2.3 Foraging

Where they co-occurred in a California estuary, Caspian terns fed mostly over main channels and the Forster's tern in shallow water covering mudflats (Baltz *et al.* 1979). Although prey resources typically are close at hand, some terns at a San Francisco Bay colony regularly flew 18 miles (29 km), and occasionally up to 38 miles (62 km), to forage at freshwater reservoirs (Gill 1976); birds at the small colony at hypersaline Mono Lake (devoid of fish) likewise must fly at least 9.3-12.4 miles (15-20 km) to forage at freshwater reservoirs (D. Shuford pers. obs.). In central Washington, Caspian terns may fly 28-37 miles (45-60 km) from the nesting colony to forage in the Columbia River, as evidenced by the recovery at the Potholes colony of passive integrated transponder tags from salmonids released or reared in that river (Ryan *et al.* 2001, 2002). Caspian terns breeding in the Columbia River estuary appear to feed primarily in the estuary (Collis *et al.* 1999, Collis *et al.* 2001b).

12.3 Population & Distribution

12.3.1 Population

Estimates of the size of the breeding population of the Caspian tern in the United States were roughly 9,454 pairs (18,908 adults) in the mid-1970s to early 1980s and 20,948 pairs in the late1980s to late 1990s (Spendelow and Patton 1988, Wires and Cuthbert 2000) (Table 12-1). During both periods, numbers of breeding Caspian terns were highest in the Pacific states and substantially smaller in the Great Lakes and Gulf Coast; numbers on the Atlantic Coast have always been very small (Table 12-1). Wires and Cuthbert (2000) also estimated during the latter period there were 32,000 to 34,000 breeding pairs in North America split among five more-or-less disjunct regions: Pacific Coast/Western (interior) (45%), Central Canada (28%), Great Lakes (19%), Gulf Coast (7%), and Atlantic Coast (<1%). The proportion of the continental population in various regions should be interpreted cautiously given that (1) totals are summed from surveys taken in multiple years and with varying methods and (2) regional and local populations can change greatly over short time periods, as described below. Kushlan *et al.* (2002) estimated the North America breeding population to be about 66,000 to 70,000 adults (not pairs) but did not document the source of this estimate or the reason for the difference between their estimate and that of Wires and Cuthbert (2000). Both of these are likely minimum estimates given the great uncertainty in the size of the large nesting population in Manitoba and hence Central Canada.

Table 12-1. Estimates of the Caspian Tern breeding population in the United States, by region, from 1976-1982 and 1996-1998.

	1976–82 [a]		1996–98[b]	
	Estimated Pairs	**% Population**	**Estimated Pairs**	**% Population**
Pacific Coast	6,218	65.8	14,534	6.4
Great Lakes	1,682	17.8	3,979	19.0
Gulf Coast	1,513	16.0	2,303	11.0
Atlantic Coast	41	0.4	122	0.6
Total	*9,454*		*2,038*	

[a] Data from Spendelow and Patton (1988) with numbers of adults divided by two to roughly estimate 2038 pairs. As some of the original data were raw counts of adults, these likely underestimated numbers of pairs given some adults usually are away from the colony at any given time.
[b] Data from Wires and Cuthbert (2000) with slight modifications. Numbers of pairs for each region were derived by separately adding the low and high estimates for each state to obtain range for the region then taking the mid-point of the range as the best estimate.

By comparison to other North American terns, the size of the Caspian tern population is not especially large. Of nine other temperate or arctic species of Sterna tern breeding in North America (exclusive of Hawaii) for which continental population estimates are available (none available for Arctic tern [*S. paradisaea*]), five have smaller and four have larger populations than the Caspian tern (Kushlan *et al.* 2002) (Table 12-2). Of those species with a relatively widespread coastal and interior breeding distribution in North America, only the Forster's tern (*S. forsteri*) has a smaller population than the Caspian tern.

Table 12-2. Population size and conservation status categories, from the *North American Waterbird Conservation Plan* (Kushlan *et al.* 2002), of 11 species of terns of the genus *Stern*a breeding in temperate and arctic regions of North America (exclusive of Hawaii).

Common Name	**Scientific Name**	**Population Size (adult breeders *not* pairs)**	**Conservation Status Category**
Gull-billed tern	*S. nilotia*	6,000 -8,000	High
Caspian tern	*S. aspia*	66,000 -70,000	Low
Royal tern	*S. maxima*	100,000 -150,000	Moderate
Elegant tern	*S. elegans*	34,000 -60,000	Moderate
Sandwich tern	*S. sandvicensis*	75,000-100,000	Not currently at risk
Roseae tern	*S. dougalli*	16,000	High
Common tern	*S. hirundo*	30,000	Low
Arctic tern	*S. paradisaea*	Insufficient information	High
Forster's tern	*S. forsteri*	47,000 -51,500	Moderate
Least tern	*S. antillarum*	60,000 -10,000	High
Aleutian tern	*S. aleutia*	14,594	High

12.3.2 *Distribution*

Caspian terns are highly migratory and exhibit cosmopolitan distribution (Harrison 1983; Harrison 1984). Caspian terns world-wide are expanding in range and numbers. Nesting has been reported from Baja California to the Bering Sea, from the Gulf Coast of Texas to Lake Athabaska and from the Florida panhandle to Labrador as well as in Australia, New Zealand, South Africa, Asia and Europe. The West Coast population winters in Southern California and Baja California and returns north to nest (Harrison 1983; Harrison 1984).

Dredging the navigational channel created several estuary islands on which piscivorous birds are now nesting. There were no terns in the estuary before 1984 when about 1,000 pairs apparently moved from Willapa Bay to nest on East Sand Island. Those birds moved to Rice Island in 1987. The combined total of the reestablished East Sand Island colony and the Rice Island colony has since expanded to approximately 10,000 pairs (the largest colony in North America) (Caspian Tern Working Group 1999).

Roby *et al.* (1998) report that two colonies have become established in the Columbia River above Bonneville Dam at ThreeMile Canyon and Crescent Island. Impounding the Columbia River behind John Day and McNary Dams created these islands. Populations have fluctuated in the past at these two sites with up to 1,000 pairs in residence at each. Populations estimated from aerial photographs in 1997 were 571 at ThreeMile Canyon and 990 pairs at Crescent Island (Roby *et al.* 1998).

12.3.3 *Breeding*

In North America, the Caspian tern breeds at widely scattered sites across the continent (Figure 12-1). In outlining patterns of regional distribution, this report follows Wires and Cuthbert's (2000) descriptions of five more-or-less disjunct breeding regions (see Figure 12-2), recognizing that future advances in knowledge may warrant adjustment of regional boundaries, as greater clarity is needed. For additional details see Cuthbert and Wires (1999), Wires and Cuthbert (2000), and pertinent sections of this report, on which the following summaries are based:

- Pacific Coast/Western (interior) Region—a very rare and recent breeder in coastal Alaska and southwestern British Columbia; a locally uncommon to abundant breeder along the coast of Washington, Oregon, and California; a locally uncommon to common breeder on the west coast of Baja California, Sinaloa, Mexico, and in the interior of Washington, Oregon, California, southern Idaho, Montana, Wyoming, western Nevada, and northern Utah.
- Central Canada—a locally rare to uncommon breeder in the Northwest Territories, Alberta, central Saskatchewan, and a locally uncommon to abundant breeder in south-central Manitoba.
- Great Lakes—an uncommon to abundant breeder on Lake Michigan, Lake Ontario, and Lake Huron.
- Atlantic Coast—a locally rare to uncommon breeder in Labrador, Newfoundland, southeastern Québec, Virginia, North Carolina and formerly, New Jersey, South Carolina, and Florida.
- Gulf Coast—a locally fairly common breeder at scattered sites from coastal Texas to Tampa Bay.

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Figure 12-1. Seasonal distribution of the Caspian tern in North, Central, and South America. The species winters locally within the dashed line. Adapted with permission from Figure 1 in Cuthbert and Wires (1999).

Figure 12-2. Outlines of five more-or-less distinct breeding regions of the Caspian tern in North America, after Wires and Cuthbert (2000). Regional boundaries may need refinement after further study.

Although recorded year-round in breeding areas on the southern Pacific Coast, Gulf Coast, and southern Atlantic Coast, it is unclear if individuals remain in these areas all year or if there is replacement by, or mixing with, birds from other breeding populations. Still, most Caspian terns in North America are highly migratory. Juveniles in fall migrate to wintering areas where they remain through their first full year; subadult (second year) birds may remain to summer on the winter grounds or return to breeding areas, whereas almost all third year and older birds migrate to and from breeding and wintering areas seasonally (Ludwig 1965, Gill and Mewaldt 1983, L'Arrivée and Blokpoel 1988). Migration generally occurs from August through October in fall and in April and May in spring. Despite the protracted period of migration in fall, individual birds may migrate fairly rapidly (L'Arrivée and Blokpoel 1988).

Caspian terns breeding on the Pacific Coast of Washington and California appear to migrate along the coast to reach wintering areas on the west coast of Mexico and Guatemala (Gill and Mewaldt 1983). Average distances traveled to the wintering grounds from major colonies at Grays Harbor, Washington, were 1,585 miles (2,550 km). On average, terns from Grays Harbor wintered farthest north and those from San Diego farthest south, suggesting there may be some segregation on the wintering grounds dependent on natal origin. Gill and Mewaldt (1983) reported that some newly fledged birds disperse north in late summer before migrating south; in two cases, hatching year birds were recovered 497 and 932 miles (800 and 1,500 km) north of their natal colonies two months following banding (Gill and Mewaldt 1979). Most resightings during the post-breeding period of Caspian terns banded at colonies in the Columbia River estuary are from the coasts of Oregon, Washington, and British Columbia and east to up-river tern colonies in the mid-Columbia River (Collis *et al.* 2000, 2001b). Later resightings have been from along the Pacific Coast south to Manzanillo, Mexico. Collectively, these data suggest that terns may disperse northward along the coast before heading south to overwinter.

Although migrants from some colonies in the interior of Oregon apparently follow the Columbia River to the Pacific (Gilligan *et al.* 1994), it is unclear if all or even most birds in the western interior pursue such a trajectory. Of four recoveries on the wintering grounds from birds banded in the interior of California, Idaho, and Nevada, two were from the west coast of Mexico along the Gulf of California and two from the central interior of Mexico (Gill and Mewaldt 1983), suggesting that terns from the interior of the western United States may take a direct overland route to reach wintering areas rather than moving diagonally to the Pacific Coast of the United States before continuing south.

12.3.4 Winter

In the Americas, the Caspian tern winters primarily on the Pacific Coast from southern California south through west Mexico and (locally) Central America; inland in the Central Volcanic Belt and Atlantic (Gulf) Slope of Mexico; along the southern Atlantic Coast of the United States, the Gulf Coast of the United States and Mexico, (locally) along the Caribbean/Atlantic coast of Central America and northern South America; and locally in the West Indies (Figure 12-1). Details of regional distribution are provided below.

12.3.4.1 Pacific Coast

Along and near the Pacific Coast, the Caspian tern winters mainly from southern California south through Baja California, the Gulf of California, and west Mexico to Guatemala (Howell and Webb 1995, BirdSource 2001). Data for Pacific Coast terns suggests there is some segregation on the wintering grounds dependent on natal origin, but sample sizes are too small to quantify how much mixing occurs (Gill and Mewaldt 1983).

Recent Christmas Bird Count (CBC) data (1991-2000) (BirdSource 2001) show the northern limit of the regular winter range in California to be at Morro Bay, San Luis Obispo County, on the southern coast (range = 3-23 birds/year, median = 9), though a few individuals now winter disjunctly on the northern coast at Humboldt Bay (range = 1-8, median = 3.5; combined data for two CBCs). The Caspian tern formerly wintered regularly on the California coast only as far north as Pt. Migu, Ventura County (Garrett and Dunn 1981). In winter, the species is casual inland in central and southern California away from the immediate coast (e.g., San Joaquin Valley) except at the Salton Sea, where numbers of wintering birds (range = 18-413, median = 27; combined data for two CBCs) may in some years rival or exceed those at sites on the southern California coast (range = 55-221, median = 139; combined data for various CBCs). Highest winter numbers at the Salton Sea from 1995-1997 (413, 197, 109) preceded peak breeding numbers there in 1996-1998 (Molina 2001).

12.3.5 Summer (Nonbreeding)

Small numbers of Caspian terns oversummer throughout most of the wintering range (Ludwig 1965, Gill and Mewaldt 1983, Hilty and Brown 1986, L'Arrivée and Blokpoel 1988, Stiles and Skutch 1989, Howell and Webb 1995, Raffaele *et al.* 1998). Others may occur in mid-summer within the general breeding range, but away from known colonies (Gill and Mewaldt 1983, Bayer 1984), or at areas along migratory pathways outside the breeding range (Zeranski and Baptist 1990, Sibley 1993). Although some birds at known migrant areas in summer may be failed adult breeders or wandering subadults, most birds on the wintering grounds at that season are young birds. Immature Caspian terns (age 6-18 months) apparently spend all four seasons in the adult wintering range, as do some sub-adults (age 18-30 months) (Gill and Mewaldt 1983, L'Arrivée and Blokpoel 1988).

Table 12-3. Counts of Caspian terns on Christmas Bird Counts in Canada and the continental United States, 1991-2000.[a]

State	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000
Ontario	1	1	0	0	0	0	0	0	0	0
Washington	0	0	1	0	0	0	1	0	0	0
California	161	154	83	108	508	297	265	265	245	184
Arizona	37	26	3	41	22	4	9	0	2	2
Indiana	0	0	0	0	0	0	0	1	0	0
Ohio	0	0	1	0	0	0	0	0	0	0
Texas	926	913	1,013	1,096	834	925	1,411	1,685	1,647	686
Louisiana	638	422	523	319	201	364	577	408	313	257
Mississippi	128	59	83	100	88	106	147	94	86	75
Alabama	83	38	98	19	112	20	70	89	48	73
Florida	585	590	645	532	543	629	635	906	1135	715
North Carolina	14	15	17	11	3	2	4	1	1	0
South Carolina	119	47	63	106	109	16	120	16	27	2
Georgia	27	40	29	16	14	4	35	8	13	6
Total	*2,719*	*2,305*	*2,676*	*2,348*	*2,434*	*2,367*	*3,274*	*3,473*	*3,517*	*2,018*[b]

[a] Numbers are raw counts summed over all CBCs on which the species was recorded in a particular state in a given year. Numbers are not adjusted to account for the number of count s conducted or for party hours or party miles. Data from BirdSource (2001).

[b] One Caspian Tern was also recorded on a CBC in Hawaii in 2000.

12.4 Status & Abundance Trends

12.4.1 Status

Caspian terns 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.

Some wetland breeding habitat used by Caspian terns in the United States is provided limited protection by the Clean Water Act (Section 404) and the Food Security Act (Swampbuster Provision, 1985). These measures as historically enforced are insufficient to prevent net losses of wetland habitat. Despite permit requirements, a review of wetlands lost to dredge and fill materials found almost 500,000 hectares lost in the conterminous United States between 1985–95 (Dahl *et al.* 1997). Although many Caspian tern colonies are located on public lands, future ownership and management of the largest breeding concentration in the Columbia River (East Sand Island) is uncertain. Most countries in the wintering range have no legal mechanism to protect Caspian terns or their habitats, and enforcement and effectiveness of existing regulations are variable.

Early colony size estimates in the Pacific Northwest were of 500 pairs mixed with gulls and cormorants as far north as Klamath Lakes in Oregon (Harrison 1984). Since the early 1900s, the population has shifted from small colonies nesting in interior California and Southern Oregon to large colonies nesting on human-created habitats along the coast (Gill and Mewaldt 1983). The current population in the Columbia River Basin is part of a dramatic northward and coast-wide expansion in the range and overall increase in the numbers of terns in western North America. Table 12-4 lists current known nesting sites.

Table 12-4. A summary of available data on Caspian Tern numbers at breeding colonies in Washington, 1997-1999 (from Caspian Tern Working Group 1999).

Location	1997	1998	1999	Supplemental Information	Source
Commencement Bay	~100	—	80-100	1995—“hundreds”	Pers. Comm. M. Tirhi
Banks Lake	—	—	—	~15/year, limited habitat	Pers. Comm. R. Friesz
Sprague Lake	—	—	—	~20/year, limited habitat	Pers. Comm. R. Friesz
Potholes Reservoir	259	—	—	150-270 pairs/year	Pers. Comm. R. Friesz
Crescent Island	990 ad	575 ad	890 ad	—	Pers. Comm. D. Roby

Several estuary islands on which piscivorous birds now nest were created by dredging in the navigational channel. There were no terns nesting in the estuary before 1984 when about 1,000 pairs apparently moved from Willapa Bay to nest on East Sand Island. Those birds moved to Rice Island in 1987. The combined total of the re-established East Sand Island colony and the Rice Island colony has since expanded to approximately 10,000 pairs (the largest colony in North America).

Roby *et al.* (1998) report that two colonies have become established in the Columbia River above Bonneville Dam at Threemile Canyon and Crescent Island. These islands were created by impounding the Columbia River behind John Day and McNary Dams. Populations have fluctuated in the past at these two sites with up to 1,000 pairs in residence at each. Populations estimated from aerial photographs in 1997 were 571 at Threemile Canyon and 990 pairs at Crescent Island (Roby *et al.* 1998). These colonies have not been studied as extensively as the colonies in the estuary, but limited food habitats studies and PIT tag collections indicate a diet similar to the diet of the terns in the estuary.

12.4.2 Trends

Data available for assessing population trends of the Caspian tern are from regional surveys and monitoring, Breeding Bird Surveys (BBS), CBCs, and anecdotal accounts. The latter dominated in most regions until the 1960s or later, after which broad-scale, quantitative surveys became more prevalent.

12.4.2.1 Trends from Regional Surveys

Although efforts to monitor and protect waterbirds at the regional level began in the early 1900s, national multi-species surveys of colonial nesting waterbirds were not conducted until 1976-1982 in response to concerns over coastal habitat modification and offshore oil development (Spendelow and Patton 1988). These surveys provide the first reliable estimates of the size and distribution of the Caspian tern's breeding population in the United States and thus form the baseline for assessing trends in ensuing decades.

Wires and Cuthbert (2000) reviewed trends in numbers and distribution of the Caspian tern in North America based mainly on a combination of anecdotal information and regional survey data. Their analysis provides the primary basis for the discussion below of population trends within the five more-or-less disjunct regions in which the species breeds in North America. This account will concentrate on the Pacific Coast/Western Region only.

12.4.2.2 Pacific Coast/Western (Interior) Region

The current regional population of about 13,000 pairs of breeding terns is the largest in North America, having more than doubled since 1980 (Wires and Cuthbert 2000). From 1992-2001, Caspian terns bred at a minimum of 44 sites in the region (Figure 12-3). In 2001, 84% of the regional population was on the coast and 16% in the interior (Table 12-5), nearly identical proportions to those in the late 1970s to early 1980s (Gill and Mewaldt 1983). The dynamic nature of this population is evidenced by dramatic shifts in its distribution and abundance over short periods of time (Gill and Mewaldt 1983, Wires and Cuthbert 2000). Since at least the late 1970s, about 99% of the regional population has been in Washington, Oregon, and California (Table 12-5), but the proportions in those respective states shifted from 50%, 4%, and 45% in 1979-1981 to 11%, 70%, and 18% in 2001.

Gill and Mewaldt (1983) reviewed trends in the Caspian tern population of the Pacific states through about 1981. The species was first documented breeding in the region at Lower Klamath Lake, Oregon, in the early 1900s (Finley 1907, Chapman 1908). The subsequent period of limited ornithological exploration coincided with great wetland loss, making it very difficult to establish a baseline on the terns' population size and distribution, let alone track population trends. Although Gill and Mewaldt (1983) reported that by 1930 no large colonies existed away from the Pacific Coast, historical data are so sparse it is unclear if interior colonies were few or many, small or large. For example, prior to 1945 only six breeding sites were known for California (five interior and San Francisco Bay), and data on population size of reported colonies was either limited or non-existent. It is clear, though, that with wetland loss and human habitat modification the Caspian tern increasingly concentrated on artificial habitats (e.g., salt ponds) on the coast and (secondarily) at reservoirs in the interior. By the 1950s, the species had expanded northward along the coast to Washington, and since the 1970s, small numbers have continued to expand north to Alaska and south to Baja California and Sinaloa in west Mexico.

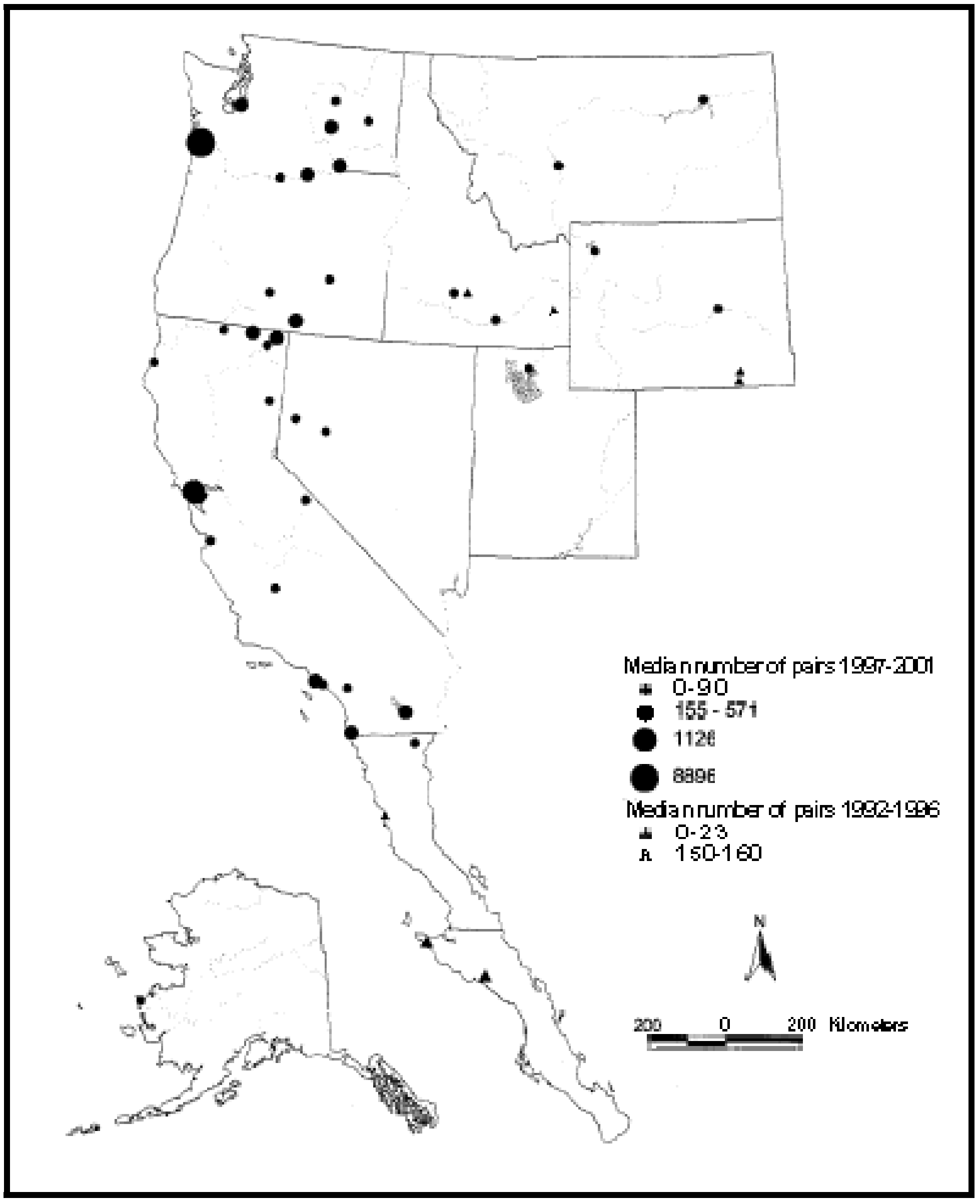

Figure 12-3. Distribution and relative size of Caspian tern colonies in the Pacific Region of western North America. Sites were mapped for 1992-1996 only if data were lacking for 1997-2001. The species has also bred at a number of other sites prior to 1992 and at some new sites in 2002.

The population of the Caspian terns in the Pacific states in the late 1970s to early 1980s was estimated to be about 5,780 pairs (84% coastal, 16% inland) (Table 12-5). Gill and Mewaldt (1983) indicated this represented an almost 74% increase since the early 1960s, but they did not report colony data or totals for the 1960s to compare to subsequent data or substantiate whether coverage was equal in both periods. Even if the size of this purported increase is valid, it might represent a rebound to, or below, the levels before the great loss of wetland habitat in the interior. Additional estimates for the Pacific region were about 14,900 pairs in the late 1990s (Wires and Cuthbert 2000) and 12,800-13,700 pairs in 2000–01 (see Table 12-5 for breeding pair estimates for individual colonies). Overall increases in the Pacific population since the 1960s appear to be in response to the terns' colonization of human-created nesting sites on the coast in close proximity to abundant fish resources (Gill and Mewaldt 1983, Wires and Cuthbert 2000), perhaps initially catalyzed by birds shifting coastward, as habitat was lost in the interior.

The regional increase since the early 1980s largely represents the great increase of the colony at the Columbia River estuary from 1984 to 2001 (Wires and Cuthbert 2000, Roby *et al.* 2002). Numerous anthropogenic and natural factors are thought to have contributed to this increase in tern numbers but the interactions among them are not well understood. Wires and Cuthbert (2000) conjectured that the increase might have been aided by the terns' exploitation of abundant and vulnerable hatchery-reared salmon. Collis *et al.* (2001a) speculated that the tern increase in the estuary was caused by the availability of hatchery-raised salmonids in combination with creation of dredge spoil islands, loss of breeding habitat elsewhere, and a build up of predators at former colonies outside the estuary. Clearly, the creation of Rice Island in 1963 substantially changed the characteristics and suitability of tern habitat in the upper estuary. Rice Island provided long-term stable nesting habitat, whereas historic habitat was ephemeral as spring river flows and tidal action created and eroded sand and gravel bars.

The magnitude and characteristics of Columbia River salmon outmigrations have also changed significantly from historic times, largely from over-harvest, hydroelectric development, mitigation measures to offset salmonid losses to dams, and various other factors. Taking into account the magnitude of current hatchery propagation and the transport of smolts (by barge or truck) to the lower river, the number of smolts in the estuary today is but a fraction of the number that occurred in the first half of this century (NMFS 2000, NWPPCC 2000). Since about the mid-1970s, the out-migration has predominately comprised hatchery-reared rather than wild smolts. Hatchery-reared yearling chinook salmon and steelhead, in some years, are more vulnerable to tern predation than their wild counterparts (Collis 2000a).

Table 12-5. Numbers of breeding pairs of Caspian terns at colonies in the Pacific Region (Washington, Oregon, California, Mexico, Idaho, Nevada, Montana, Wyoming), 1997 to 2001 and circa 1979-1981.[a]

	~1979[b]	1997	1998	1999	2000	2001
Washington						
Coastal Bays						
Commencement Bay, Pierce Co.	0	—	—	423	620[c]	388
Grays Harbor, Grays Harbor Co.	2,157	0	0	0	0	0
Willapa Bay, Pacific Co.	650	0	0	0	0	0
Mid-Columbia River						
Miller Rocks, Klickitat Co.	0	—	—	—	—	15
Crescent Island, Walla Walla Co.	0	614[c]	357[c]	552[c]	571	720
Columbia Basin Plateau						
Banks Lake, Grant Co.	—	—	—	—	10	23
Potholes Reservoir, Grant Co.	100	259	—	—	150	~250
Sprague Lake, Adams Co.	—0	—	—	~50	20	20
Oregon						
Columbia River Estuary						
East Sand Island, Clatsop Co.	0	0	0	1,400	8,513	8,896
Rice Island, Clatsop Co.	0	7,151	8,691	8,328	588	0
Miller Sands Spit, Clatsop Co.	0	0	17	0	0	0
Mid-Columbia River						
Threemile Canyon Island, Morrow Co.	210	354[c]	210[c]	238[c]	260	2

	~1979[b]	1997	1998	1999	2000	2001
Great Basin						
Malheur Lake, Harney Co.	—	65	25	30	192[c]	51[c]
Crump Lake, Warner Valley, Lake Co.	—	—	—	—	155[c]	—
Summer Lake, Lake Co.	—	—	—	38	16	0
California (Coast)						
Humboldt Bay	20[b]	—	—	—	—	~17[c]
San Francisco Bay	(1,500)[d]					
Little Island, Napa Co.	300	—	—	—	—	—
Knights Island, Solano Co.	0	400	~200	—	121[c]	43[c]
Brooks Island, Contra Costa Co.	0	~500	582	active	806[c]	512[c]
Hayward Regional Shoreline, Alameda Co.	0	1	1	1	1	1
Bair Island, San Mateo Co.	825	0	0	0	0	0
Ravenswood (Pond Rl), San Mateo Co.	0	0	(4 ad.)	0	1	1
Alameda NAS, Alameda Co.	0	285	267	1	0	0
Coyote Hills, Alameda Co.	0	30	22	0	0	—
Baumberg Tract, Alameda Co.	75	0	33	26	79	116
Turk Island, Alameda Co.	150	0	0	0	0	0
Drawbridge, Alameda Co.	150	0	0	0	0	0
Alviso (Pond A7), Santa Clara Co.	0	104	30	122	118	155
Central & Southern Coast						
Moss Landing salt ponds	105[c]	0	0	0	0	0
Elkhorn Slough, Monterey Co.	0	0	0	~30	~80	~65
Bolsa Chica, Orange Co.[e]	0	175	40	58	51	92
Los Angeles Harbor, Los Angeles Co.	0	25	146	250	336	160
South San Diego Bay NWR, San Diego Co.	409	320	198	261	380	350
California (Interior)						
Modoc Plateau/Great Basin						
Meiss Lake, Butte Valley WA, Siskiyou Co.	50	25[c]	16	27	19	0
Lower Klamath NWR	20	0	0	0	0	0
Clear Lake NWR, Modoc Co.	200	180[c]	68[c]	118	242c	201
Goose Lake, Modoc Co.	200	143[c]	—	310c	4	~240
Big Sage Reservoir, Modoc Co.	75	62[c]	—	0	48	0
Honey Lake WA, Lassen Co.	15	152	—	87	82	92
Mono Lake, Mono Co.	12	0	0	0	8	6
San Joaquin Valley, Tulare Basin, (All Kings Co.)						
Lemoore NAS sewer ponds	—	—	20[c]	0	—	—
Westlake Farms South Evaporation Basin, Kings Co.	—	0	3	0	0	0
Tulare Lakebed	—	0	20[c]	0	0	0
South Wilbur Flood Area		0	70	27	0	0
Tulare Lake Drainage District, North Evaporation Basin	—	0	0	0	0	1
Tulare Lake Drainage District, South Evaporation Basin	—	0	40	0	0	0
Coastal Slope, Southern California						
Lake Elsinore, Riverside Co.	—	—	—	14	—	—
Colorado Desert						

	~1979[b]	1997	1998	1999	2000	2001
Salton Sea, Imperial Co.	0	1,200	800	211	207	327
Mexico						
Baja California						
Cerro Prieto geothermal ponds, Mexicali Vy.	—	30	34	—	0	0
Idaho						
Snake River Plateau						
Morman Reservoir, Camas Co.	—	—	—	—	—	~2
Magic Reservoir, Camas and Blane Cos.	20	—	—	—	—	0
Blackfoot Reservoir, Caribou Co.	5	—	—	—	—	0
Minidoka NWR, Cassia Co.	—	—	—	—	1	0
Deer Flat NWR (Snake River Is.), Owyhee Co.		—	—	—	—	0
Bear Lake NWR, Bear Lake Co.	—	—	—	—	—	0
Nevada						
Great Basin						
Carson Sink, Churchill Co.	—	0	—	685	0	0
Anaho Island NWR, Pyramid Lake	6	1	5	0	0	0
Stillwater Point Reservoir, Stillwater NWR	5	0	0	0	0	0
Montana						
Canyon Lake Ferry Reservoir, Lewis and Clark Co.	—	5	0	2	7	35
Fort Peck Reservoir, Charles M. Russell NWR, Valley Co.	—	?	?	?	?	~25
Wyoming						
Molly Island, Yellowstone Lake, Yellowstone National Park	21	4	5	4	0	3
Soda Lake Islands, Natrona Co.	—	0	0	0	7	12
Pacific Region Totals[f]	*5,780*	*12,085*	*11,900*	*13,293*	*13,693*	*12,821*

[a] To enable estimation of the total numbers of breeding pairs in the entire region, we adjusted some raw counts or estimates. When a range was given for numbers of nests or pairs we report the mid-point (e.g., 800-850 pairs reported as 825 pairs) and for breeding adults we use the mid-point as the basis for estimating numbers of pairs. Counts or estimates of breeding adults were multiplied by 0.62 to approximately estimate numbers of breeding pairs on the basis of the average ratio of nests to adults at sites on the California coast (0.625, Carter et al. 1992, p. 1-45) and the California interior (0.61, D. Shufordun publ. data). Dashes (--) indicate that no survey was conducted or no data were available, zeroes (0) that a survey was conducted but no evidence of nesting was observed, and question marks (?) that nesting was strongly suspected but no solid data were available. All data presented are from published sources, unpublished reports, unpublished data, and personal and written communications as cited in regional accounts.

[b] Data for 1979-1981 from Gill and Mewaldt (1983) with the following modifications: (1) Humboldt Bay - numbers for this site for 1979 included although S. Harris (pers. comm.) knew of no breeding there after 1969, (2) Moss Landing –the report of 160-180 pairs is actually 160-180 breeding adults (Sowls et al. 1980, Harvey 1982), which we adjusted to 105 pairs (see above), (3) Mono Lake -we substituted 12 pairs as the mid-point of 10-15 pairs reported bJdehl(1986), (4) Pyramid Lake -excluded data for 1951-1965 as 6 pairs estimated in 1979 (W. Henry pers, comm.), (5) Columbia River (Threemile Canyon Is.) -instead of 200 pairs we used the 210 in 1978 reported by Thompson and Tabor (1981; also 184 pairs in 1977), (6) Molly Island, Yellowstone Lake - we added 21 pairs for 1979 (A. Cerovski pers. comn.), and (7) for consistency with treatment of recent data, we took the mid-point of the ranges for Hartson Reservoir (Honey Lake, WA) and Willapa Bay (see above).

[c] Counts of adults were converted to an estimate of breeding pairs by multiplying raw adults by the 0.62 described above.

[d] The number 1,500 is a total for San Francisco Bay in 1981 reported by Gill and Mewaldt (1983). A lack of data for all individual colonies required estimation of breeding pairs at Little Island.

[e] All counts from Bolsa Chica are of total nest attempts (on the basis of marked nests), which likely overestimates nesting pairs because of pairs that renest after initial failures.

[f] Totals are likely underestimates because of a lack of surveys at some sites in particular years or during the whole time period (e.g., most sites in Mexico).

In 1986, Caspian terns established the colony on Rice Island, which experienced rapid growth through the 1990s. Its initial growth appears to have been fueled by movement of terns from the large colonies at Grays Harbor and Willapa Bay, Washington. Thereafter, its continued growth and success can be attributed to the stability of the human-created nesting habitat, the reliable food supply of hatchery-reared salmon, the vulnerability of some hatchery smolts to tern predation, and the apparent immigration of terns hazed from other colonies (e.g. Everett Navel Base in 1996). These factors underscore the significance of human alterations of the environment to the growth of the Pacific population, especially in the Columbia River estuary. The success of the terns (e.g., 1.40 young/pair in 2001) following their relocation to East Sand Island, where salmonids represented only 33% of the diet (Roby *et al.* 2002), suggests that, at least in some years, the estuary could support a large and productive tern colony independent of significant alterations of nesting habitat or the attendant prey base.

12.4.2.3 Breeding Bird Survey Trends

Wires and Cuthbert (2000) cited Price *et al.* (1995) for large increases in Caspian tern numbers on BBS routes since the mid-1960s. Up-to-date and revised analyses of BBS data by Sauer *et al.* (2001), though, found a significant positive trend survey-wide only for the period 1966–79 but not for 1980–2000 or 1966–2000.

12.4.2.4 Christmas Bird Count Trends

Although rigorous analyses of CBC data for Caspian terns apparently have not been conducted, there does not appear to have been a unidirectional trend in wintering numbers in the United States over the last decade (Table 12-3). Raw CBC data suggest relatively stable numbers of wintering Caspian terns in the United States from 1991–96, substantially higher numbers from 1997–99, then a decline to the lowest numbers of the decade in 2000.

12.4.3 Productivity

Productivity levels for various North American colonies range from 0.6-1.6 young fledged per nest (Cuthbert and Wires 1999). Productivity of Caspian terns breeding at the large colonies in the Columbia River estuary has been closely monitored from 1997–2001 (Roby *et al.* 2002). Young fledged per nesting pair at Rice Island was 0.06 in 1997, 0.45 in 1998, 0.55 in 1999, and 0.15 in 2000. The proximate cause of most nest failure was predation on eggs or chicks by Glaucous-winged Gulls (*Larus glaucescens*), Western Gulls *(L. occidentalis*), and their hybrids. Disturbance from research activities in 1997 and management actions implemented to relocate the Rice Island tern colony in 1999 and 2000 may have also affected productivity levels at this site. Young fledged per nesting pair at East Sand Island has been consistently higher than at Rice Island: 1.20 in 1999, 0.57 in 2000, and 1.40 in 2001 (Roby *et al.* 2002).

Estimates of productivity at other Caspian tern colonies in the Pacific Coast region are limited. Kirven (1969) calculated an average of 1.1 young fledged per nesting pair at San Diego Bay, California, in 1967. Additional measures of colony breeding success were made in San Francisco by Ohlendorf *et al.* (1985). Two subcolonies ranged from about 0.69-0.82 young per nest and in Puget Sound by Shugart and Tirhi (2001) (0.40 chicks per pair). Anecdotal accounts and personal observations (C. Collins, K. Molina, D. Bell, G. Ivey, D. Shuford, C. Trost, and J. Parkin) suggest that most other colonies in the region in most years have experienced “good” productivity of about one young fledged per breeding pair. There are, however, accounts of colonies suffering total reproductive failure in a given year because of mammalian predators (Tulare, Elkhorn Slough, Threemile Canyon Island) or weather-related phenomena (Malheur, Bolsa Chica); reproductive success has also been greatly reduced by contaminants (Elkhorn Slough).

On the basis of a survivorship analysis of banded terns, Gill and Mewaldt (1983) estimated that the Pacific Coast population of Caspian terns needed to produce at least 0.64 young per pair per year to sustain the annual intrinsic growth rate of 2.7% observed between 1960 and 1980. Their analysis was limited, however, by the model assumption of no immigration or emigration from natal colonies, when in fact 58% of their breeders did not return to their natal colony.

12.5 Environmental Conditions

12.5.1 Habitat Distribution

Caspian terns are highly migratory and exhibit cosmopolitan distribution (Harrison 1983; Harrison 1984). Caspian terns world-wide are expanding in range and numbers. The West Coast population winters in Southern California and Baja California and returns north to nest (Harrison 1983; Harrison 1984).

Breeding populations are adaptable and able to exploit new habitats when bare sand and abundant prey are found in areas of low disturbance. Dredging the navigational channel created several estuary islands on which piscivorous birds are now nesting. There were no terns in the estuary before 1984 when about 1,000 pairs apparently moved from Willapa Bay to nest on East Sand Island and then moved to Rice Island in 1987. The combined total of the reestablished East Sand Island colony and the Rice Island colony has since expanded to approximately 10,000 pairs (the largest colony in North America) (Caspian Tern Working Group 1999) on about 10 acres of sand.

Roby *et al.* (1998) report that two colonies have become established in the Columbia River above Bonneville Dam at Threemile Canyon and Crescent Island. Populations have fluctuated at these two sites with up to 1,000 pairs in residence at each. Populations estimated from aerial photographs in 1997 were 571 at Threemile Canyon and 990 pairs at Crescent Island (Roby *et al.* 1998).

12.5.2 Habitat Status

Through the Interactive Biodiversity Information System (IBIS), the Northwest Habitat Institute (NHI) (2003) identified key habitat types (Table 12-6) and habitat elements (Table 12-7) with which Caspian tern is associated.

The most serious long-term threat to Caspian terns is the loss or deterioration of quality breeding habitat (i.e., insular, sparsely-vegetated islands). Although Cuthbert and Wires (1999) did not cite habitat loss as an important threat to Caspian terns in North America, it is estimated that 54% of wetland habitat has been lost in the conterminous United States (Dahl *et al*. 1997), including specific wetland losses impacting Caspian terns (e.g., Klamath Basin, Oregon-California; Bent 1921). Still, the species' breeding range and population size have increased in the face of wetland losses.

Although the reasons for population growth are complex and multifaceted, the creation of artificial breeding sites and alteration of fisheries by humans appear to be two important factors influencing the tern's population growth (Wires and Cuthbert 2000). Caspian terns clearly have the capacity to opportunistically respond to shifts in habitat and prey resources.

Table 12-6. Habitat types with which Caspian tern is associated (NHI 2003).

Habitat type	Association	Activities	Confidence	Comments
Open water (lakes, rivers, & streams)	Closely associated	Feeds & breeds	High	Nests on sandbars and dredge spoil islands within rivers.
Herbaceous wetlands	Closely associated	Feeds	High	None noted
Coastaldunes & beaches	Closely associated	Other (see comments)	High	Roosting/resting.
Coastal headlands & islets	Generally associated	Other (see comments)	High	Roosting/resting.
Bays and estuaries	Closely associated	Feeds	High	None noted
Marine nearshore	Closely associated	Feeds	High	None noted

Table 12-7. Habitat elements with which Caspian tern is associated (NHI 2003).

Habitat Element	Description	Comments
4.3	Ephemeral pools	Feeding.
4.4	Sand bars	Nesting and loafing.
4.5	Gravel bars	Nesting and loafing.
4.8	Islands	Nesting and loafing.
8.28	Hatchery facilities and fish	Good as a food source, but terns may be killed at aquaculture facilities.

Despite the persistence of large colonies for decades on dredge spoil islands, islands created by water impoundments, and salt dikes (McNair 2000, Wires and Cuthbert 2000, Collis *et al.* 2002), vegetation succession has and may continue to render many sites unsuitable for breeding terns.

Caspian terns are less tolerant of vegetation succession than the longer-legged, shorter-winged gulls that frequently share their breeding islands. Encroachment of vegetation and/or displacement by gulls were considered factors contributing to the decline of some large tern colonies, among them Sand Island and East Sand Island (Penland 1981, Neuman and Blokpoel 1997, J. Albertson pers. comm., S. Harris pers. comm.).

In the short term, vegetation succession may pose the greatest threat to colonies, particularly in the Pacific Region; in the long term, coastal colonies across the continent may be severely affected by sea level rises from global warming (Titus 1991). High water levels (not associated with global warming) have inundated nesting islands in the Great Lakes (Neuman and Blokpoel 1997), and tidal action has eroded and flooded breeding sites on salt pond levees in San Francisco Bay (Ryan 2000) and on islands in Grays Harbor and Willapa Bay. Overall, at least five historic nesting sites on the Pacific Coast have been lost to natural processes, such as vegetative succession, erosion, or inundation.

In Oregon and Washington, management actions have destroyed habitat or discouraged nesting at the largest and most recent coastal colonies, resulting in the loss of three additional breeding sites (Bird 1994, Collis *et al.* 2001a). Habitat modification (wooden stakes and monofilament lines) and hazing (e.g., walking through potential breeding sites to discourage colony establishment) were used to prevent nesting at Everett Naval Station, Washington, to reduce bird strike hazards to aircraft. These actions eliminated a nesting site that had 2,600 breeding adults the previous year (Smith *et al.* 1997, J. Flavin in lit.). In 2001, hazing and habitat modification were implemented to prevent nesting at the contaminated ASARCO Superfund clean-up site in Ruston, Washington. This site had at least 423 pairs in 2000 (Collis *et al.* 2001b, Shugart and Tirhi 2001). In 2001, as many as 388 breeding pairs moved to a barge provided as experimental nesting habitat. However, the barge was removed because of a breakdown of interagency coordination (Collis *et al.* in press). From 1999–2001, habitat modification (i.e., fencing, flagging, and winter wheat planting) and early season hazing (in 1999) were implemented on Rice Island to reduce fisheries conflicts in the Columbia River estuary (USACE 2001). These actions occurred concurrently with efforts to attract terns to nest at East Sand Island. Rice Island had previously been the largest colony in North America (Wires and Cuthbert 2000).

Future losses or degradation of habitat may also occur, such as increasing salinity, changing water priorities, and drought.

12.6 Factors Affecting Population Status

The factors limiting Caspian tern population growth are unknown or poorly understood. As with other seabirds, Caspian terns are long-lived, exhibit delayed maturation before breeding, and have low fecundity (clutch size, breeding frequency, and breeding success) (Weimerskirch 2002). This suggests that adult survival is likely one of the more important demographic parameters of Caspian terns. Both Gill and Mewaldt (1983) and Ludwig (1965), though, found that annual survivorship was lowest for terns in the interval between fledging and first breeding. The evolution of extended post-fledging parental care suggests that post-fledging survival may also be a factor in population regulation. Given that the North American population is currently increasing, it does not appear the number of Caspian terns is being unduly limited by any factor or combination of factors.

12.6.1 Overutilization

Historically, humans severely harmed Caspian tern colonies by collecting hundreds of eggs and adults for food and feathers (Finley 1907, Bent 1921, Ludwig 1965, and Lock 1993). In addition to the mortality and direct loss of eggs, it is likely that these activities resulted in undocumented colony failures and abandonment. Caspian terns are also vulnerable to direct persecution by people killing adults and young on the wing or at the colony (Penland 1976, Koonz 1982).

12.6.2 Disease & Predation

Caspian terns sometimes die in outbreaks of Newcastle disease and botulism, but these diseases do not appear to be a threat to the survival of the species (Campbell and Key 1996, Klinger 1997, K. Molina pers. comm.). The internal and external parasites known to infect Caspian terns are also not perceived as threats (Cuthbert and Wires 1999). Disease, though, may pose a threat to highly concentrated tern populations (see Concentration Risk below).

Caspian tern colonies are always vulnerable to predators, but there are no specific predator threats to the species at large. Persistent bald eagle activity at the Caspian tern colony on Rice Island in the Columbia River estuary caused significant egg and chick losses when gulls capitalized on the eagle-induced panic flights (see Burger and Gochfeld 1991, Collis *et al.* 2000). Bald eagle activity and gull nest predation have been suggested as factors in the abandonment of some coastal Washington colonies late in their history (e.g., Sand Island, Grays Harbor; Everett, Puget Sound; Penland 1976, Bird 1994).

Caspian tern colonies can also suffer from the introduction of predators by people that perceive a conflict between fish-eating birds and commercial or sport fisheries (Buchal 1998). In the Columbia River, researchers have removed raccoons and opossums (*Didelphis viginianus*) that were thought to be released by someone intent on destroying Caspian tern nests at Rice Island (Collis *et al.* 1999). Large Caspian tern colonies maintained by management of near-shore islands are perhaps the most likely to be threatened by predators in the long run. Long-established colonies may be most vulnerable to loss if there are no alternative sites nearby to relocate to when predation forces a colony move. Some colonies may need persistent predator monitoring and control to maintain them as long-term colony sites (Kress 2000).

12.6.3 Availability of Suitable Nesting Habitat

To be suitable, nesting habitat must be bare ground or in the earliest stages of vegetative succession, high enough above river or tide levels to avoid flooding of nests, eggs and young; large enough to accommodate a colony (critical mass of birds); and free of disturbance from predators (including humans). This combination of conditions is somewhat uncommon; therefore, available nesting habitat is limited.

Nesting habitat must consist of either bare or sparsely vegetated ground: In the short-term, vegetation succession may pose the greatest threat to colonies. The most serious long-term threat to Caspian terns is the loss or deterioration of quality breeding habitat (i.e., sparsely-vegetated islands). With reduction of flooding and peak flows in the Lower Columbia River subbasin, nesting sites have become less likely to be scoured by floods. In the absence of flooding, dredge spoil islands provide secure, stable nesting locations in the estuary, but nesting habitats are more susceptible to rapid vegetation succession, and need to be managed (scarified to set back succession) to allow continued tern nesting over time.

A number of known, historically active tern colonies have been lost in the Pacific Northwest along coasts and in interior marshes. The dense tule marsh at Lower Klamath, for example, where terns and other colonial water birds nested on matted sedges, was destroyed by the Klamath Basin water development project for agriculture, lack of water for wildlife, and by refuge management practices that open up marsh to make it more attractive for waterfowl; at the Everett Naval Base, the Defense Department, apparently without a permit, destroyed a nesting site used by a large colony of approximately 3,000 terns; and the relatively large colonies (1500-3590 pairs) on natural islands in Willapa Bay and Gray's Harbor were abandoned as sites became susceptible to flooding due to natural erosion, and disturbance and predation caused by Bald Eagles increased.

Potential nesting sites and methods of managing colonies along the Oregon and Washington coast, as well as elsewhere in the western region, are detailed in Seto et al. (2003). Gray's Harbor and an "unnamed island" in Umpqua estuary were among the better possibilities for re-establishing colonies. However, these and most other sites where colonies might be established are relatively small. Few good sites were identified; the capacity (numbers of terns that can be accommodated) was low at most sites, and those potential sites that were identified would require continual management, i.e., retardation of vegetative succession. Throughout North America, many if not most of the now-existing tern colonies nest at man-made sites, on dredge spoil islands or on islands in reservoirs.

Since nesting habitat was historically ephemeral, Caspian Terns have evolved a flexibility in their choice of nesting sites, and are able to occupy or move from a site when conditions change, such as through vegetative succession, erosion, and flooding probability.

Regulation of the hydro-system has made island habitat in the river, including dredge spoil islands, less erodible, less prone to flooding, and more secure as nesting habitat for colonial birds.

A nesting location must be large enough to accommodate a colony, as Caspian Terns are colonial.

A nesting site must be free from disturbance by predators, including humans. Access by mink, coyotes, etc. is sufficient to cause abandonment of a nesting site. Terns derive a degree of protection against avian predators by their colonial nature, and reproductive success tends to be higher at larger colonies, because of defense of the colony against gull predation. Therefore a colony must achieve some critical mass, i.e., be at least moderately sized, to be successful. Intentional human harassment, including management, at nesting sites is a potential limiting factor at any colony, and especially so at East Sand Island, the largest colony in North America.

12.6.4 Food Supply

Without a large predictable food suppy Caspian terns cannot nest successfully. The diet of Caspian Terns consists wholly of small fish. Salmonid smolts, in the Columbia River predominantly hatchery fish, provide a significant part (1/3) of their energy needs. At the mouth of the estuary, at East Sand Island, marine fish species comprise a larger portion of the diet. Hatchery salmonid smolts are more vulnerable to tern predation than wild smolts . Collis et al.2001, shows that hatchery yearling Chinook appear to be three times more vulnerable to predation than wild stock in 1998 – the only year for which they have data. Hatchery Steelhead appeared to be nearly twice as vulnerable to predation in 1997, but were not more vulnerable in 1998. At East Sand Island, which is close to the mouth of the Columbia River, the terns forage both in the mouth of the estuary as well as along the outer coast. This may affect the abundance and availability of marine fish, and therefore the proportion of marine fish in the diet.

12.6.5 Pesticides & Other Contaminants

In general, levels of organochlorines are declining, and current levels are not likely to threaten most Caspian tern colonies in North America though individual colonies may be affected or threatened (Henny *et al.* 1982, Cuthbert and Wires 1999, J. Buck pers. comm.). The effects of pesticides and other environmentally toxic compounds on Caspian terns have best been evaluated in the Great Lakes region, especially at the industrially-impacted colonies of Green Bay, Lake Michigan, and Saginaw Bay, Lake Huron (Cuthbert and Wires 1999). Eggs from Green Bay and Saginaw Bay had the highest polychlorinated biphenyls (PCBs) levels of eggs analyzed in the Great Lakes (Ewins *et al.* 1994). Grasman *et al.* (1996, 1998) found organochlorine compounds, especially PCBs, associated with the suppression of the immune system in prefledging Caspian tern chicks. This is coincident with the findings of low natal philopatry in areas of high PCB contamination (Struger and Wesloh 1985). These high PCB concentrations are thought to be lowering the reproductive success and juvenile survivorship of Caspian terns (Grasman *et al.* 1998).

Impacts of organochlorine pollutants, especially DDE (a breakdown product of DDT), have been documented on the Pacific Coast. Ohlendorf *et al.* (1985) found high chick mortality in San Diego associated with high DDE levels in eggshells. High DDE levels were also found in egg shells in the San Francisco Bay area (Ohlendorf *et al.* 1985, 1988). In 1995, residual DDE and other pollutants re-suspended by record flooding were also considered to be responsible for a reproductive collapse of a Caspian tern colony in Elkhorn Slough, California (Parkin 1998).

Ludwig *et al.* (1993) described a similar failure in the Great Lakes also caused by re-suspension of contaminants by floodwaters.

These accounts underscore that despite pollutants such as DDE and PCBs being better regulated today, individual Caspian tern colonies continue to be threatened by them long after they have been banned. Caspian terns are well suited as sentinel species (Grasman 1998), and hence their colonies should be monitored on a regular basis if they are associated with sources of contaminants, such as manufacturing in the Great Lakes or channel deepening on the Columbia River. In general there are ongoing concerns for the potential risk to waterbirds of reproductive impairment or immunotoxicity from selenium, boron, mercury, DDE, PCBs, and trans-nonachlor (Ohlendorf 1985, 1988; Setmire *et al.* 1990, 1993; Grasman 1996, 1998; Bruehler and de Peyster 1999).

12.6.6 Human Disturbance

Human disturbance is a well-known cause of reproductive failure in a wide range of seabirds (Carney and Sydeman 1999, Nisbet 2000, Carney and Sydeman 2000). Caspian tern colonies are especially vulnerable during the early courtship and incubation stages (Cuthbert and Wires 1999). Human visitors that approach Caspian terns during these stages typically cause panic flights of the entire colony. Such human disturbances can lead to permanent nest or colony abandonment (Cuthbert and Wires 1999). Most of the well-documented cases of human impact are from research activities, underscoring the vulnerability of Caspian terns.

The impacts of human disturbance are often magnified by the response of predators or the terns themselves. Egg losses may result from adults damaging or kicking their eggs out of the nest when abruptly fleeing human disturbance (Cuthbert and Wires 1999). Similarly, chicks may flee nest sites by swimming and get lost, drown, or die of exposure (Quinn *et al.* 1996). Fleeing chicks may also be attacked and often killed by neighboring adults (G. Shugart in lit.). The impact of a colony disturbance can be greatly increased when nearby gulls act as egg and chick predators (Penland 1982, Quinn 1984). Although a panic flight of a colony reacting to disturbance may last only a few seconds, gulls at Rice Island stole hundreds of eggs and young chicks per day during these brief disturbances (Collis *et al.* 2000). The Rice Island colony appeared most vulnerable to gull predation during the early chick stage, when small chicks (5-10 days old) ran from the nest but were still easily consumed in a single bite by gulls on the wing (D. Craig pers. obs.). Chicks are also particularly vulnerable to humans entering a colony at this stage as evidenced by chick mortality (about 30% died) following a 1-hour banding effort in Grays Harbor (Penland 1981). In subsequent years, chick mortality due to researcher disturbance was avoided by selecting the banding date to be at a stage when most chicks had just hatched and by restricting banding to 20-minute periods (WDFW pers. comm.). In 1998, 72 chicks died at Rice Island from heat exhaustion when too many chicks became crowded together in a holding pen during a midday banding effort (D. Craig pers. obs.). Since 2000, banding activities on the Columbia River have been conducted at either dawn or dusk, and groups of about eight nearly fledged chicks have been held in pheasant crates to minimize crowding (D. Craig pers. obs.). Although researchers often document their impact, the majority of human intrusions and disturbances by the general public are undocumented and their effects unmeasured.

12.6.7 Introduced Species

There are no apparent threats to Caspian terns directly associated with introduced species. Introduced plants such as tansy ragwort (*Senecio jacobaea*), common evening primrose (*Oenothera biennis*), and European beach grass (*Ammophila arenaria*) may be accelerating the degradation of quality breeding habitat by advancing vegetation succession at a rate faster than that of native plants of the Columbia River (D. Craig pers. obs.). The introduction of non-native mammalian predators has been documented at several colonies, particularly those in conflict with human interests.

12.6.8 Population Size & Isolation

Although limited information is available on the size of historic populations, numbers of Caspian terns have increased markedly in North America in the last 30 years, when relatively good population data have been gathered (Wires and Cuthbert 2000). The species still occupies most of its former range and has expanded into new areas. The continent-wide breeding population numbers at least 32,000 to 34,000 pairs. The current population size itself does not warrant conservation concern. Although there are insufficient data regarding the mixing of Caspian terns among regions in the breeding or non-breeding seasons, isolation of populations is not an apparent conservation threat.

12.6.9 Concentration Risk

Natural and human-caused events have reduced or eliminated habitat at many colonies. In the Pacific Coast region, eight of 15 historic colonies have been lost or abandoned in the last 20 years. This has apparently led to terns concentrating on few remaining suitable sites (e.g., Rice Island, Oregon) or colonizing new sites in conflict with human interests (e.g., ASARCO, Ruston, Washington). Shipping traffic on the Columbia River leaves large breeding aggregations of terns, such as those at East Sand Island, especially vulnerable to oil spills or other spilling or shipping accidents. The large breeding concentration in the Columbia River estuary is also more vulnerable to stochastic events (e.g., storms, predators, and human disturbance) and disease (e.g., Newcastle and botulism) than a comparable population dispersed among many smaller colonies (Klinger 1997, Roby *et al.* 2002, K. Molina pers. comm.). Natural and human disturbances that cause panic flights at larger colonies may result in significant chick mortalities, as the probability of chicks becoming lost and then killed by adults increases with colony size (Penland 1976, D. Craig pers. obs.). Roby *et al.* (2002) suggested that in years with poor ocean conditions near large concentrations like East Sand Island there is an increased likelihood of terns being reliant on juvenile salmon. Large concentrations of Caspian terns are also more likely to engender conflict with fisheries interests and hence may be subjected to organized eradication efforts through introduced predators (e.g., pigs) (Buchal 1998).

12.7 Inventory & Assessment of Existing Management Plans

Seto, N., J. Dillon, W.D. Shuford, and T. Zimmerman. 2003. A Review of Caspian tern (*Sterna caspia*) Nesting Habitat: A Feasibility Assessment of Management Opportunities in the US Fish and Wildlife Service Pacific Region. US Department of the Interior, Fish and Wildlife Service, Portland, Oregon.

This document is a comprehensive review of management options and strategies that will direct Caspian tern management in Oregon and Washington. It describes the conflicts of Caspian tern management and helps to direct solutions in Washington and Oregon.

Shuford, W.D., and D.P Craig 2002. Status Assessment and Conservation Recommendation for the Caspian tern in North America. US Department of the Interior, fish and Wildlife Service, Portland Oregon.

This document is a comprehensive review of management options and strategies that will direct Caspian tern management in Oregon and Washington. It describes the conflicts of Caspian tern management and helps to direct solutions in Washington and Oregon.

Collis, K., D.D. Roby, C.W. Thompson, D.E. Lyons, and M. Tirhi. 2002. Barges as temporary breeding sites for Caspian terns: Assessing potential sites for colony restoration. Wildlife Society Bulletin 30: 1140-1149.

This document describes a unique treatment of a management solution, the use of temporary barges for nesting. It could be valuable in helping to direct innovative solutions for providing nesting area for Caspian terns to draw them into areas where nesting can be tolerated in Washington and Oregon.

Roby, D.D., K. Collis, and D.E. Lyons. 2003. Conservation and management for fish-eating birds and endangered salmon. Proceedings of the Third International Partners in Flight Conference, Asilomar, California.

This is a report on managing Caspian terns in Oregon and Washington. It describes the many conflicts of Caspian tern management and helps to direct solutions in Washington and Oregon.

Caspian tern Interim Management Plan FY 2003-2004 and Pile Dike Excluder Maintenance to Discourage Cormorant Use Lower Columbia River Oregon Interim Environmental Assessment: March 26, 2003.

This is the governing document for current tern management on the Lower Columbia River.

12.8 Inventory & Assessment of Existing Restoration & Conservation Plans

The USFWS anticipates releasing the final Caspian Tern EIS by January 2005 and issuing a signed Record of Decision by February 2005. This is a comprehensive review of management options and strategies that will direct Caspian tern management in Oregon and Washington. If respected and implemented, it will be instrumental in ensuring the secure future of Caspian terns in Washington and Oregon.

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Volume III, Chapter 13
Columbian White-tailed Deer

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13.0 Columbian White-tailed Deer (*Odocoileus virginianus leucurus)*

13.1 Introduction

The Columbian white-tailed deer (*Odocoileus virginianus leucurus*) is one of 38 recognized subspecies of *O. virginianus*. Columbian white-tailed deer is one of the largest terrestrial mammals associated with the Columbia River estuary (NPPC 2002). Columbian white-tailed deer are prevalent in the upper estuary and along the river corridor. Low-lying mainland areas and islands in and along the Columbia River from about Skamakowa, Washington, to Port Westward, Oregon are the preferred habitats of the Columbian white-tailed deer.

The Columbian white-tailed deer, a subspecies of the white-tailed deer, is on the Endangered Species List. It is classified as endangered in Washington and Oregon. This deer once ranged from Puget Sound to southern Oregon, where it lived in floodplain and riverside habitat. The conversion of much of its homeland to agriculture and unrestricted hunting reduced its numbers to a just a few hundred in the early 20th century. Habitat conversion and losses coupled with the low prdocutivity of the population are the most important threats now to the population. It now lives in a few scattered populations, and its numbers have climbed to approximately 300-500 in the lower Columbia and over 5,000 in the Roseburg area. Recovery goals outline the need to secure additional habitat for population re-introduction (USFWS 1976).

13.2 Life History & Habitat Requirements

13.2.1 Life History

13.2.1.1 Diet

Recent studies have indicated the importance of grasses and forbs in the diet of white-tailed deer in North America (Allen 1968, Coblentz 1970, Segelquist *et al.* 1972, Sotala and Kirkpatrick 1973, Harder and Peterle 1974, McCaffery *et al.* 1974, Anthony and Smith 1977). Gavin *et al.* (1984) concluded that water foxtail provided forage of high quality and that Columbian whitetails preferred to graze rather than browse.

Whitetails on the Columbian White-Tailed Deer National Wildlife Refuge (CWTDNWR) were observed grazing on forbs and grasses almost exclusively during the early and mid-1970s. Suring (1974) and Suring and Vohs (1979) reported that grazing was detected in 99% of their nearly 18,000 observations of deer feeding. Stomach contents from 32 whitetails collected from all seasons during 1972–77 consisted of grasses (59%), forbs (16%) and browse (25%) (B.B. Davitt personal communication: 1981). Essentially, all browse consumed was nonwoody (such as blackberry leaves). Dublin (1980) concluded that Columbian whitetails on the refuge selected for browse in every season except spring and selected for forbs in all seasons, but selected against grass (relative to its availability) in autumn, winter, and spring. It is possible that at least part of this paradox in describing the food habits of these deer is due to a change in vegetation height, productivity, and availability on the refuge between the early and late 1970s (Gavin 1984).

13.2.1.2 Reproduction

Rutting activity begins the first week of November and probably reaches a peak during the second week. By the end of the month, reproductive behavior by males decreases noticeably, although some deer are apparently capable of breeding as late as March. This conclusion is based on an observation of twins born sometime in late September–early October in 1974, assuming the gestation period of Columbian whitetails approximates that of eastern white-tailed deer (210 days) (USFWS 1976).

Observations (spring 1975) indicated that the peak of fawning was the second week in June. This correlates well with the observed rutting period of the second week of November, and corroborates a gestation of about 7 months. Available data indicate that nearly all adult females become pregnant and give birth to one or two fawns. However, recruitment into the population, based on fawn:doe ratios of marked females in November, is relatively low. Fawns comprised 21-33% of the November population from 1972–77 (no data for 1973), while yearling and adult males comprised 18-21%. There was no evidence that female fawns were fecund (USFWS 1976).

13.2.1.3 Home Range

Severinghaus and Cheatum (1956) summarized the literature published prior to 1956 and generalized that seasonal range of an individual whitetail rarely exceeded one mile in diameter; this has been supported generally by studies conducted since that time. Home ranges of whitetails in Texas (Thomas *et al.* 1964, Michael 1965, Alexander 1968) and the Southeast (Marshall and Whittington 1968, Byford 1969, Kammermeyer and Marchinton 1976) seemed most similar to those of Columbian whitetails on the CWTDNWR with respect to size and temporal stability (Gavin *et al.* 1984).

Gavin *et al.* (1984) estimated the home range size among sex-age classes of whitetails on the CWTDNWR mainland and found that the mean home range size for females was 391.6 acres; for males, the mean area of home ranges was 475 acres. Home ranges of males tended to increase in size as males became older, but there was no significant trend with age among females. There was no apparent relationship between the geographic location or size of a marked female's (≥ 3.5 years old) home range and her success in recruiting fawns.

In Oregon, Smith (1985) found that the average home range size was 109 acres for does and 116.3 acres for bucks in the Roseburg area.

Observation of marked deer on the CWTDNWR mainland indicated that individual whitetails had the same home ranges in successive years (Gavin 1979). The average home range was 391.5 acres for does and 474.7 for bucks. The area traveled by a deer in any 24-hour period, however, was considerably smaller than these averages. No movement by marked deer off the refuge was ever observed (Gavin 1984).

13.2.1.4 Migration

The Columbian white-tailed deer is a non- migratory species that exists in the historic floodplain areas of the lower Columbia River from Longview, Washington, to the mouth of the Columbia River.

13.2.1.5 Mortality

On the CWTDNWR, Gavin *et al.* (1984) found that the oldest ages at death observed for females and males coupled with the tertiary sex ratio of 3 females:1 male strongly indicated that annual mortality among yearling and adult males was about twice as high as for yearling and adult females (Figure 13-1). Estimation of mortality rates for marked males and females also indicated a higher rate of mortality for males. Gavin *et al.* (1984) summarized the relative importance of proximate causes of mortality of whitetails on the CWTDNWR in Table 13-1.

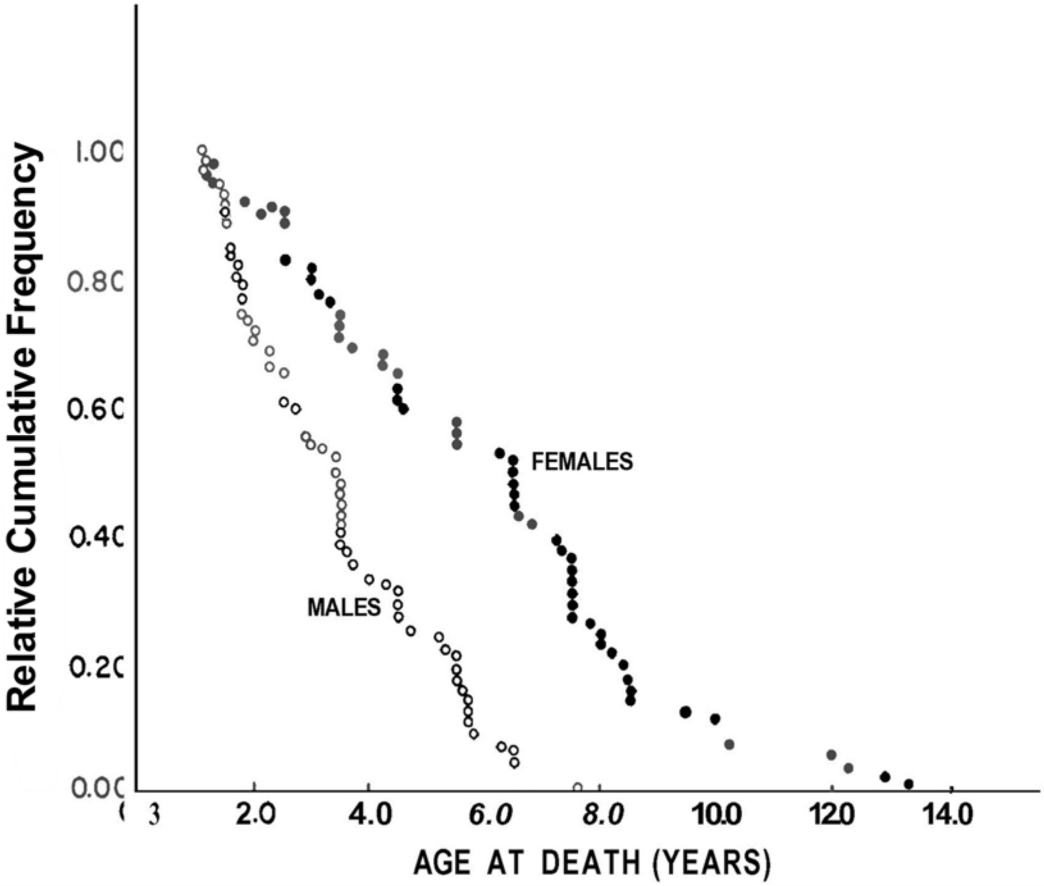

Figure 13-1. Relative cumulative frequency of ages at death for males and females > 1 year old at death on the CWTDNWR (Gavin *et al.* 1984).

Table 13-1. Proximate causes of mortality of white-tailed deer on the CWTDNWR mainland during June 1974-December 1977 (Gavin *et al.* 1984).

Cause of death	# dead			# with symptoms of necrobacillosis
	Males	**Females**	**Fawns**	
Automobile	5	3	0	1
Capture attempt	5	4	1	5
Coyote	0	0	9	0
Dog	5	0	0	4
Drowning	0	2	1	2
Fence	1	0	1	0
Necrobacillosis (bacterial)	1	1	2	4
Nutritional stress	9	3	0	6
Poached	6	2	0	4
Unknown	26	28	40	23
Total	*58*	*43*	*54*	*49*

Gavin *et al.* (1984) further concluded that the temporal distribution of natural mortalities of yearlings and adults in this population suggested that mortality of these age classes resulted ultimately from the activities and stress associated with reproduction (Table 13-2). For males, natural mortality was heaviest after peak activity in rutting (November). At least 22 yearling and adult males died during November–January (1974-76), nearly twice as many male deaths as any other 3-month period.

Table 13-2. Temporal distribution of natural mortalities (plus road kills) of white-tailed deer on CWTDNWR mainland, June 1974–December 1977 (Gavin *et al.* 1984).

	Nov-Jan	Feb-Apr	May-July[a]	Aug-Oct
Fawns	6	3	6	8
Adult and yearling males	22	13	6	9
Adult and yearling females	6	4	9	15
Total	*34*	*20*	*21*	*32*

[a] No fawn mortahties were found in May

Because white-tailed fawns are not fully weaned until at least 3 months old (Moen 1973:144) and energy requirements of the female are greatest at the peak of lactation (Moen 1973:362), Columbian whitetail females that were successful in raising fawns were probably under greatest physiological stress in late summer-early autumn. Females whose fawns failed to survive after birth would have been under peak stress at late gestation or parturition in late May or June.

Klein and Olson (1960) found higher natural mortality among males than females in a lightly hunted population of black-tailed deer in Alaska and associated this with an unbalanced sex ratio favoring females. Flook (1970), who studied differential sex ratios in elk in Canada, concluded that nonhunting mortality of males older than yearlings contributed to ratios heavy to females.

Mortality rates of white-tailed deer fawns tend to be high in unhunted populations. Cook *et al.* (1971) found a mean annual mortality rate of 71.5% from birth to 3 months of age during a 2-year study of radio-marked fawns in Texas (Gavin *et al.* 1984). White (1973:467) estimated a 60% mortality rate of fawns during summer on the Aransas National Wildlife Refuge in Texas. Hunted populations of whitetails generally have fawn mortality rates less than half as high as those from the unhunted populations (O'Pezio 1978).

Fawn mortality at the CWTDNWR in the early 1990s was believed to be limiting the population growth. Studies were undertaken in 1996 and 1997 where fawns were radio-collared soon after birth and followed until collar loss or death. In 1996, all 12 collared fawns died within 45 days of capture. Coyote control the next spring removed nine coyotes from the refuge and that year, three of the 17 radio-collared fawns were killed by coyotes (USFWS 1997). Public concern over the trapping of coyotes has stopped the program and fawn recruitment is now low (Miller, pers. comm.).

13.2.2 Habitat Requirements

O'Neil *et al.* (2001) found that the Oregon habitat type with which Columbian white-tailed deer is most closely associated is westside oak within 200 meters of a stream or river (Table 13-3).

Table 13-3. Habitat types with which Columbian white-tailed deer is associated (O'Neil *et al.* 2001).

Habitat Type	Association	Activities	Confidence	Comments
Westside lowlands conifer-hardwood forest	Generally associated	Feeds & breeds	High	None noted
Westside grasslands	Generally associated	Feeds & breeds	High	None noted
Agriculture, pastures, & mixed environs	Generally associated	Feeds & breeds	High	None noted
Westside oak & dry douglas-fir forest & woodlands	Closely associated	Feeds & breeds	High	Strong association with oak within 200m of a stream or river.
Urban & mixed environs	Generally associated	Feeds & breeds	High	None noted
Herbaceous wetlands	Generally associated	Feeds	High	None noted
Westside riparian-wetlands	Generally associated	Feeds & breeds	High	None noted
Southwest Oregon mixed conifer-hardwood forest	Generally associated	Feeds & breeds	High	None noted

The relative preference of vegetative communities by Columbian whitetails on the CWTDNWR mainland was documented by Suring (1974) and Suring and Vohs (1979). The study identified 12 different plant communities and concluded that Columbian whitetail use was greatest in communities that provided both forage and cover (USFWS 1976). Plant communities on the refuge that provided cover taller than 70 cm in the vicinity of forage were used more frequently than communities that provided cover or forage alone (Suring and Vohs 1979). Deer showed a preference for the park forest community—especially in the fall, winter, and spring. The primary activity in the park forest was feeding (66% of the observations), followed by resting (22%), and movement (12%) (USFWS 1976). Other plant communities that received high whitetail deer use were open canopy forest, sparse rush, and dense thistle. Open canopy forest or dense thistle (*Cirsium arvense*) communities were preferred to closed canopy forest or improved pasture, for example (Gavin *et al.* 1984). Higher percentages of deer were observed resting and moving in the forest than in non-forest communities. The high frequency of resting behavior in forest communities is probably related to the thermal protection of woody cover.

O'Neil *et al.* (2001) found that Columbian white-tailed deer are generally associated with certain forest structural conditions for feeding and breeding (Table 13-4).

In general, browse is chosen in summer, fall, and winter while forbs are most heavily utilized in spring, summer, and early fall. Grasses are not preferred at any time of the year but are eaten in proportion to their availability only in the early spring (Dublin 1980). Heavy use of forbs occurs as they emerge in the spring and throughout the summer. O'Neil *et al.* (2001) summarized Columbian white-tailed deer associations with grass/scrubland structural conditions inTable 13-5.

Table 13-4. Forest structural conditions with which Columbian white-tailed deer is associated (O'Neil *et al.* 2001).

Structural Condition	Activity	Association	Confidence	Comments
Grass/forb-open	Feeds	Generally associated	High	Predominately feeds in this structural condition.
Small tree-single story-closed	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition. Forested structural conditions also used for cover.
Medium tree-single story-open	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition. Forested structural conditions also used for cover.
Medium tree-single story-moderate	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition. Forested structural conditions also used for cover.
Medium tree-single story-closed	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition. Forested structural conditions also used for cover.
Small tree-multi-story-open	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition. Forested structural conditions also used for cover.
Small tree-multi-story-moderate	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition. Forested structural conditions also used for cover.
Small tree-multi-story-closed	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition. Forested structural conditions also used for cover.
Grass/forb-closed	Feeds	Generally associated	High	Predominately feeds in this structural condition.
Medium tree-multi-story-open	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition. Forested structural conditions also used for cover.
Medium tree-multi-story-moderate	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition. Forested structural conditions also used for cover.
Medium tree-multi-story-closed	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition. Forested structural conditions also used for cover.
Shrub/seedling-open	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition. Forested structural conditions also used for cover.
Shrub/seedling-closed	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition. Forested structural conditions also used for cover.
Sapling/pole-open	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition. Forested structural conditions also used for cover.
Sapling/pole-moderate	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition. Forested structural conditions also used for cover.
Sapling/pole-closed	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition. Forested structural conditions also used for cover.
Small tree-single story-open	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition. Forested structural conditions also used for cover.
Small tree-single story-moderate	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition. Forested structural conditions also used for cover.

Table 13-5. Shrub/grassland structural conditions with which Columbian white-tailed deer is associated (O'Neil *et al.* 2001).

Structural Condition	Activity	Association	Confidence	Comments
Grass/forb-open	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition.
Medium shrub-open shrub overstory-mature	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition. Also used for cover.
Medium shrub-open shrub overstory-old	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition. Also used for cover.
Medium shrub-closed shrub overstory-seedling/young	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition. Also used for cover.
Medium shrub-closed shrub overstory-mature	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition. Also used for cover.
Medium shrub-closed shrub overstory-old	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition. Also used for cover.
Tall shrub-open shrub overstory-seedling/young	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition. Also used for cover.
Tall shrub-open shrub overstory-mature	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition. Also used for cover.
Tall shrub-open shrub overstory-old	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition. Also used for cover.
Tall shrub-closed shrub overstory-seedling/young	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition. Also used for cover.
Tall shrub-closed shrub overstory-mature	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition. Also used for cover.
Grass/forb-closed	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition.
Tall shrub-closed shrub overstory-old	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition. Also used for cover.
Low shrub-open shrub overstory-seedling/young	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition.
Low shrub-open shrub overstory-mature	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition.
Low shrub-open shrub overstory-old	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition.
Low shrub-closed shrub overstory-seedling/young	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition.
Low shrub-closed shrub overstory-mature	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition.
Low shrub-closed shrub overstory-old	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition.
Medium shrub-open shrub overstory-seedling/young	Feeds & breeds	Generally associated	High	May feed & breed in this structural condition. Also used for cover.

13.3 Population & Distribution

13.3.1 Population

Population declines led to the classification in 1967 of Columbian white-tailed deer as endangered under the ESA (32 FR 4001). The subspecies was automatically included in the lists of threatened and endangered species when the ESA was authorized in 1973 (16 US C. 1531 *et seq.*). Prior to 1977, only the Columbia River population was listed as endangered since the Douglas County population was considered by Oregon to be a black-tailed deer (*Odocoileus hemionus columbiana*) or a hybrid between the black-tailed deer and the whitetail. In 1978, Oregon recognized the white-tailed deer population in Douglas County as the Columbian white-tailed deer and prohibited hunting the species in that county (ODFW 1995) (WDOT 2001).

Today, these are the only two populations of any consequence west of the Cascades—the one along the lower Columbia River and the second near Roseburg, Oregon (NPPC 2002).

Approximately 300-400 whitetails live in four major and one minor subpopulations along the lower river in Oregon and Washington from Wallace Island (RM 50) downstream to Karlson Island (RM 32). Each subpopulation is separated from the next by a main river channel or patches of unfavorable habitat that limit consistent interchange. The largest subpopulation occurs on the Washington mainland near Cathlamet. Establishment of the CWTDNWR in 1972 secured about 4,800 acres of this prime habitat. The refuge population on the Washington mainland has been declining since 1977.

13.3.1.1 Washington

The population on the 1,952-acre mainland portion of the CWTDNWR was estimated at 200-230 during the winter of 1972–73 (Suring 1974). This was an average density of 65.6-75.4 deer/mi^2. Gavin (1979) conservatively estimated the population in the Novembers of 1974, 1975, 1976, and 1977 to be 214, 180, 164, and 202, using a mark-recapture technique (Schnabel). The November–December population during 1978, 1979, and 1980 was estimated at 212, 191, and 159, respectively (Columbian White-tailed Deer Recovery Team 1982). Population estimates for Columbian whitetails on off-refuge islands in the Columbia River near the refuge include 50-75 for Puget Island, 70-80 for Wallace Island-Westport, and 8-12 for Karlson Island (Columbian White-tailed Deer Recovery Team 1982).

Today, lower Columbian whitetails comprise five herds: Tenasillahe Island reserve, the CWTDNWR (mainland), Puget Island, Westport, and Wallace Island. Table 13-6 summarizes herd composition and population survey information for the mainland, Puget Island, Tenasillahe Island, and Westport herds.

The Columbian whitetail throve under the protection of the refuges and in 1995, was even a candidate for downlisting. However, in February of 1996, both Tenasillahe Island and the mainland deer refuge experienced severe flooding. At the height of the floods, when 2 to 3 feet of water inundated the refuge mainland and 6 inches to a foot covered Tenasillahe Island, more than 75% of the deer population was estimated to have left the refuge seeking higher ground. Deer populations before the flood were estimated at between 115 and 120 on the mainland and more than 200 on Tenasillahe Island. After the flood, biologists estimated a population of 60 deer on the mainland unit and 100 deer on 2,000-acre Tenasillahe Island in the Columbia River. The unaccounted-for deer are presumed to have died (USFWS 1996).

Table 13-6. Deer counts and fawn:doe ratios by subpopulation: 1985–2002*

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Mainland

Year	Deer Count	Fawn:Doe Ratio
1985	208	62
1986	216	43
1987	227	34
1988	173	14
1989	158	29
1990	146	30
1991	118	21
1992	117	28
1993	73	11
1994	49	1
1995	47	14
1996	37	16
1997	52	61
1998	53	43
1999	26	15
2000	44	34
2001	38	49
2002	44	25

Puget Island

Year	Deer Count	Fawn:Doe Ratio
1985	143	40
1986	129	40
1987	122	59
1988	135	53
1989	135	40
1990	104	55
1991	84	38
1992	100	58
1993	82	48
1994	88	55
1995	133	47
1996		
1997		
1998		
1999	60	45
2000	58	70
2001	48	49
2002	39	40

Tenasillahe

Year	Deer Count	Fawn:Doe Ratio
1985		
1986	13	27
1987	22	38
1988	25	48
1989	27	43
1990	35	67
1991	41	55
1992	55	67
1993	66	47
1994	71	52
1995	76	53
1996		
1997		
1998		
1999	38	6
2000	32	8
2001	30	18
2002	11	0

Westport

Year	Deer Count	Fawn:Doe Ratio
1985	54	40
1986	53	40
1987	66	57
1988	53	66
1989	57	29
1990	61	56
1991	56	30
1992	69	58
1993	54	42
1994	100	57
1995	91	23
1996		
1997		
1998		
1999	45	11
2000	28	23
2001	36	39
2002	36	29

* Claskanie Flats also had deer but only in 2002; 11 deer were counted with a fawn:doe ratio of 84.

Several recent attempts to re-introduce Columbian white-tailed deer have taken place in the Lower Columbia region. Twelve deer were transplanted to Fisher and Hump Islands in the spring of 2003 (Miller, pers. comm.).

13.3.1.2 Oregon

From 1928–52, whitetails found northeast of Roseburg, Oregon, in an area containing approximately 30.5 mi^2 were considered by ODFW part of the refuge population (Gavin 1984). Hunting was prohibited. Crews (1939) estimated the number of whitetails in this high-density area at 200-300 in 1938. This refuge was dissolved in 1952, and hunting was resumed. In 1970, ODFW estimated that 450-500 whitetails existed in the old refuge area, at a density of 14.8-16.3 $deer/mi^2$ (Smith 1985). In a detailed study of Roseburg whitetails, Smith (1985) estimated the density in his 6,783-acre study area along the North Umpqua River to be 59-70 $deer/mi^2$ in 1979–80. He used a mark-recapture technique similar to that used by Gavin (1979) for the Columbia River population.

In 1970, ODFW estimated 450-500 deer and by 1983, the number had increased to about 2,500 (Smith 1985). The population is now estimated to be 5,900-7,900 deer (ODFW 1999). The range also has expanded to an area of approximately 308 mi^2 (ODFW 1995). Approximately 9,586 acres of suitable Douglas County habitat are considered secure on federal, county, and private lands. (For delisting, habitat is considered secure if it is protected by legally binding measures or law from adverse human activities for the foreseeable future.) The current total population is estimated as approximately six times the population size required for downlisting, which greatly reduces the risk to the population. The Douglas County population has met the objectives in the recovery plan, and greatly exceeded the habitat objectives (USFWS 1999).

Deer have been re-introduced into the lower Columbia River in Oregon at Crimms Island and Lord Island near Longview, Washington. Deer from the Crimms Island project have become established in the Willow Grove area near Longview. The deer at Lord Island were released in early 2003 (Miller, pers. comm.).

13.3.2 Distribution

Columbian white-tailed deer were formerly distributed throughout the bottomlands and prairie woodlands of the lower Columbia, Cowlitz, Willamette, and Umpqua River basins in Oregon and southern Washington (Bailey 1936). Early accounts suggested this deer was locally common, particularly in riparian areas along the major rivers (Gavin 1978). The number of deer declined rapidly with the arrival and settlement of pioneers in the fertile river valleys. Conversion of brushy riparian land to agriculture, urbanization, uncontrolled sport and commercial hunting, and perhaps other factors apparently caused the extirpation of this deer over most of its range by the early 1900s (Gavin 1984). Only a small herd of 200-400 animals survived in Clatsop, Columbia, Cowlitz, and Wahkiakum Counties in Washington, along with a disjunct population of unknown size in Douglas County, Oregon. The two populations are geographically separated by about 200 miles of unsuitable or discontinuous habitat (USFWS 1999).

Today, there are only two whitetail populations of any consequence west of the Cascade Mountains. One is located along the lower Columbia River on the CWTDNWR. The other—the Roseburg herd—is located in Douglas County, Oregon (Figure 13-2) (Gavin 1984). Recently, deer have been transplanted into islands in the Columbia River near Longview, Washington. A few scattered deer also exist in the Willow Grove area near Longview. These scattered populations may form the nucleus for populations to become established in the future (P. Miller, pers.comm.).

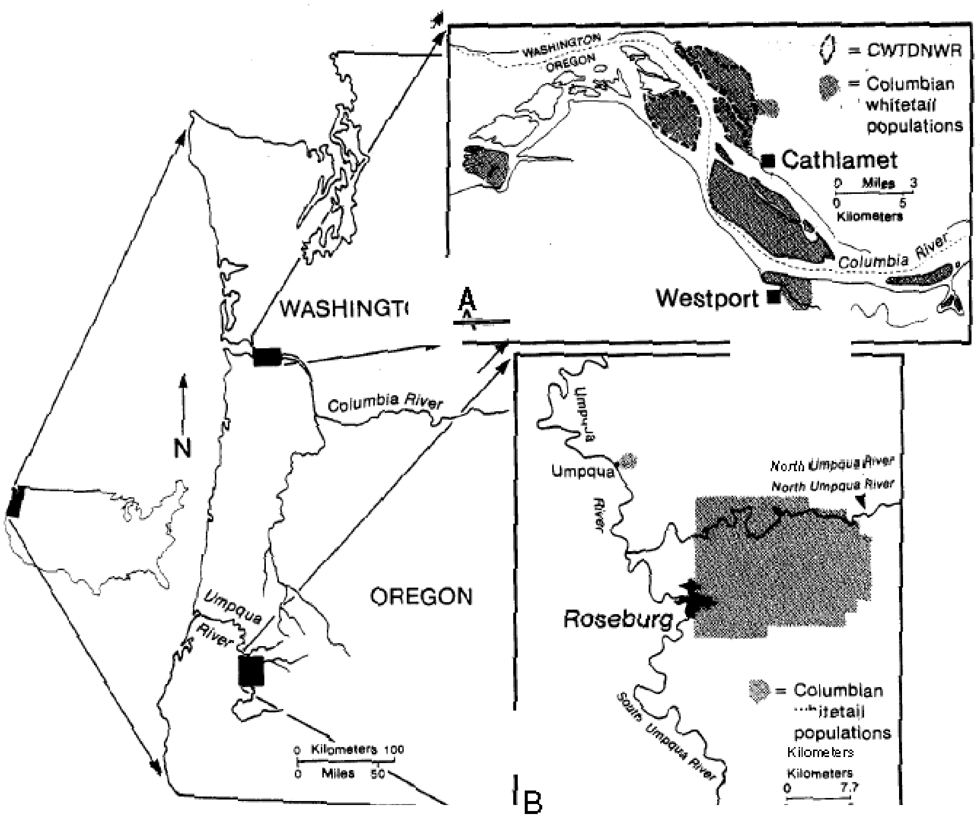


Figure 13-2. Distribution of white-tailed deer in western Washington and Oregon, 1983.

13.4 Status & Abundance Trends

13.4.1 Status

Population declines led to classification of Columbian white-tailed deer as endangered in 1967 under the ESA (32 FR 4001). The subspecies was automatically included in the lists of threatened and endangered species when the ESA was authorized in 1973 (16 US C. 1531 *et seq.*). Prior to 1977, only the Columbia River population was listed as endangered since, as mentioned above, the Douglas County population was considered a black-tailed deer or a hybrid. In 1999, USFWS proposed to delist the Columbian white-tailed deer in the entire range. Public concern over delisting caused USFWS to withdraw the proposal. The Roseburg population recently has been proposed for delisting but this has not been adopted by USFWS. The lower Columbia population is not proposed for any listing change at this time (David, pers. comm.).

13.4.2 Trends & Productivity

The lower Columbia River population exhibits a long-term decline. Populations in all the major areas were affected by flooding in 1996 and conversion of pastures and woodlots to homes. Deer counts from the mainland refuge have declined from a mean of 159 from 1985–93 to a mean of 43 from 1994–2002 (Table 13-6).

13.5 Habitat

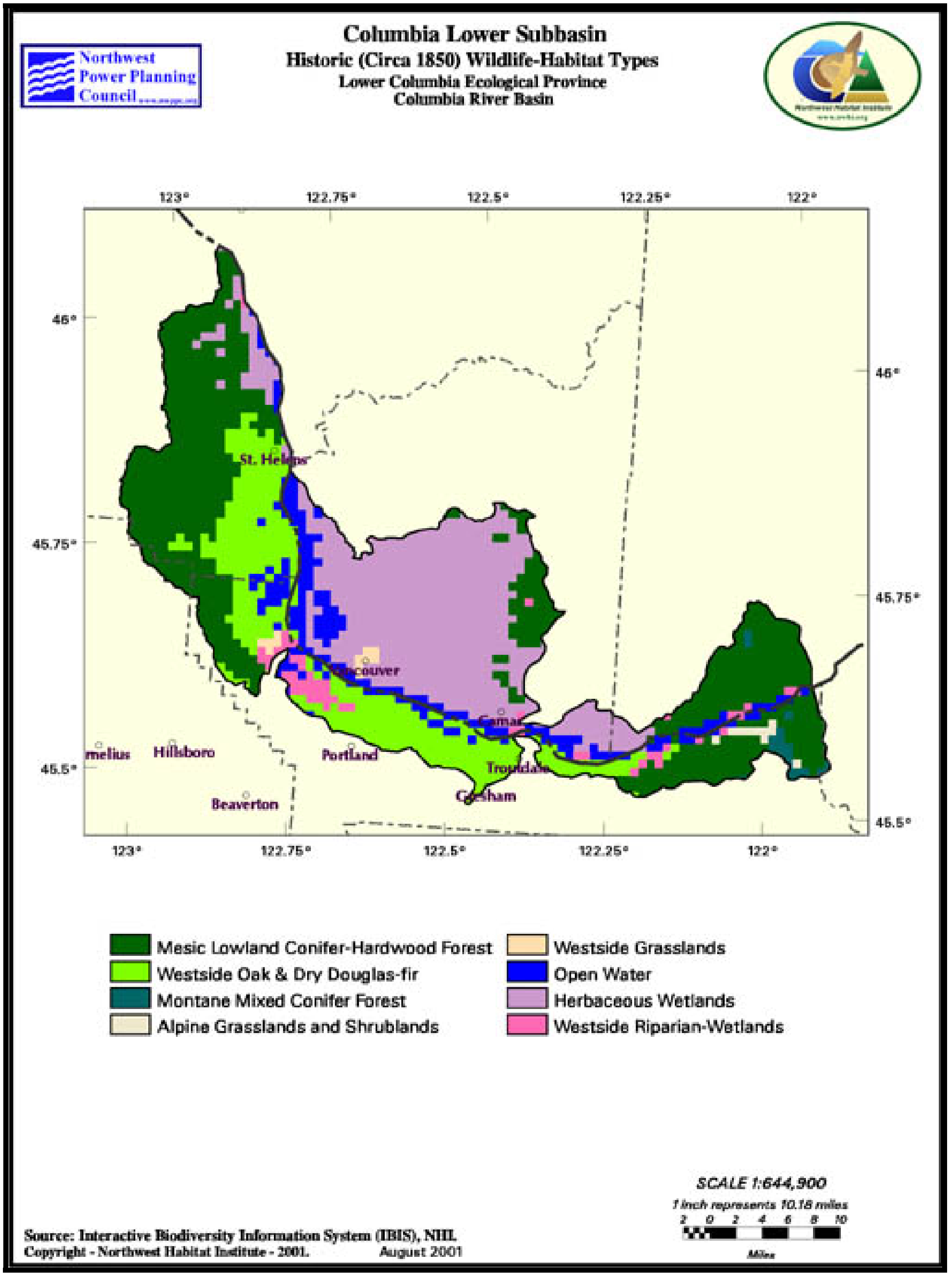
Columbian white-tailed deer were formerly distributed throughout the bottomlands and prairie woodlands of the lower Columbia, Cowlitz, Willamette, and Umpqua River basins in Oregon and southern Washington (Bailey 1936). This discussion focuses on habitat in the region of the lower Columbia River in Washington.

Extensive losses of habitat have occurred in the lower Columbia and estuary provinces as a result of dredging, filling, diking, and channelization. Figure 13-3 illustrates the status of historic habitat types of the lower Columbia subbasin (IBIS 2003).

The floodplain and lowlands likely were much more heavily forested, with hardwood and perhaps some coniferous riparian species. There were many more lakes, ponds, sloughs, overflow channels, backwaters and wetlands. Openings were likely associated with the wet areas, accreting lands or lands having recently experienced a scouring flood. These openings would have been dynamic in location; they would not have remained stationary in the landscape. Uplands were likely characterized by a coniferous forest. Fish and wildlife were much more abundant and diverse.

Based on the available information and excluding the Columbia River itself, it may not be unreasonable to speculate that the composition of the landscape types at the time of the Lewis and Clark expedition was in the range of:

- 60-70% forest (hardwood, conifer and mixed forest);
- 15-25% openings (meadows, accreting lands, recently scoured lands); and
- 15-25% water and wetlands (lakes, ponds, sloughs, wetlands, streams) (NPPC 2002).



Columbia Lower Subbasin
Historic (Circa 1850) Wildlife-Habitat Types
Lower Columbia Ecological Province
Columbia River Basin

Source: Interactive Biodiversity Information System (IBIS), NHI.
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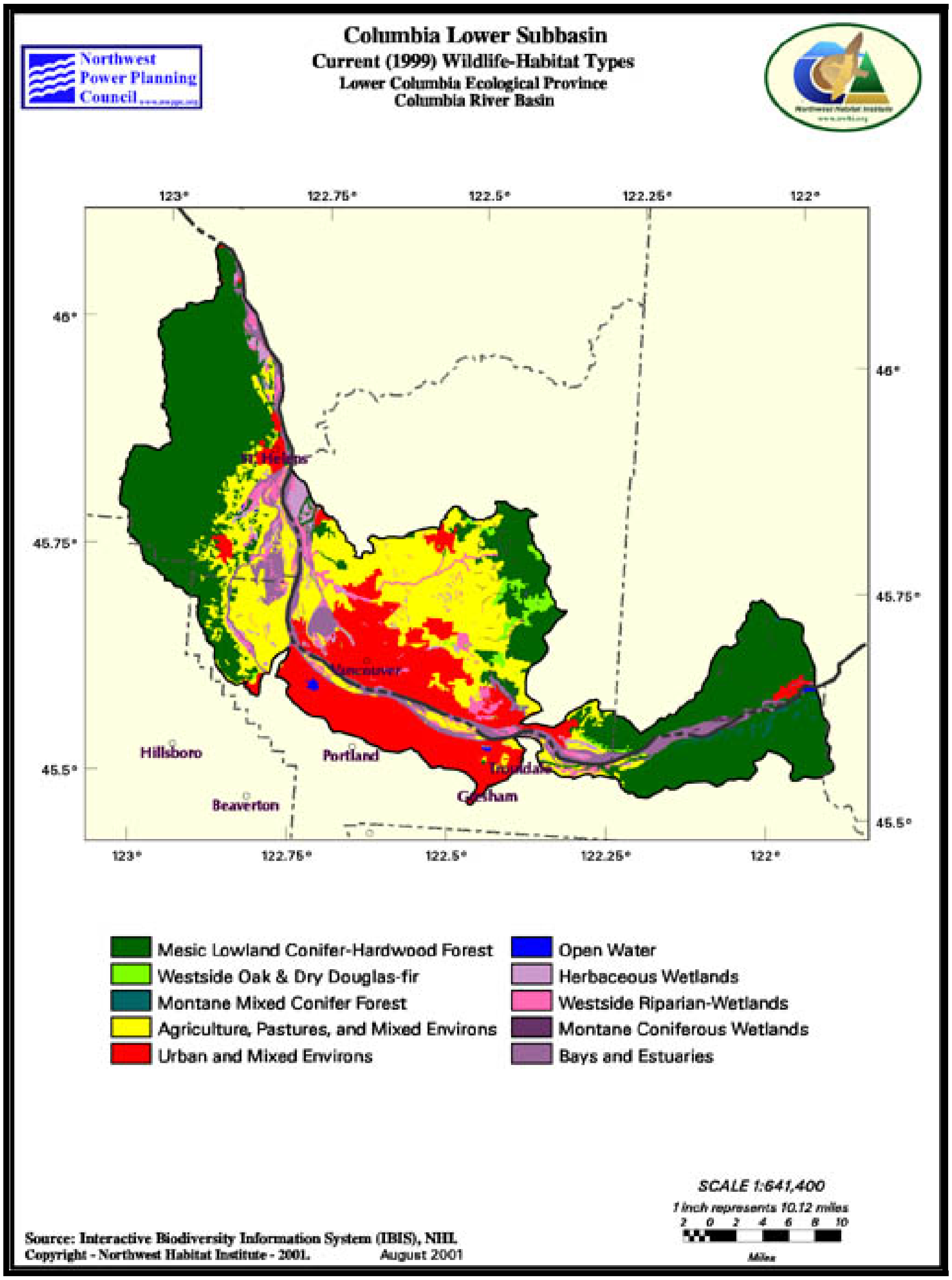
Figure 13-3. Status of historic wildlife habitat types in the Lower Columbia subbasin (IBIS 2003).

Figure 13-4 illustrates the status of current habitat types of the Lower Columbia subbasin (IBIS 2003). Estimates from 1870–1970 indicate that 20,000 acres of tidal swamps (with woody vegetation; 78% of estuary littoral area), 10,000 acres of tidal marshes (with nonwoody vegetation) and 3,000 acres of tidal flats have been lost (NPPC 2002).



Columbia Lower Subbasin
Current (1999) Wildlife-Habitat Types
Lower Columbia Ecological Province
Columbia River Basin

Source: Interactive Biodiversity Information System (IBIS), NHI.
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Figure 13-4. Status of current wildlife habitat types of the Lower Columbia subbasin (IBIS 2003).

The Oregon Natural Heritage Program's 1992 inventory determined that the primary tree species on the lower Columbia River floodplain are cottonwood, ash and Pacific willow, and Sitka spruce would have been found in the lower river.

On the Washington side of the Columbia River, the amount of woody cover has changed significantly (Suring 1974). In 1939, 70% of the land now contained in the mainland portion of the CWTDNWR was classified as wooded. In 1972 however, only 17% was classified as wooded; the remainder had been converted to pasture. This reduction of year-round riparian cover is typical on Columbia River and Willamette Valley floodplains. Over most of the region, the acreage left in woodlots is far less than 17% (Gavin 1984).

Native vegetation of the Columbia River tidelands consists of dense, tall shrub or tree community containing Sitka spruce, red-osier dogwood, black cottonwood, red alder and willow (Franklin and Dyrness 1973). Most of the bottomlands have been cleared of trees and brush, and seeded to grasses and forbs that provide feed for beef and dairy cattle. Plants commonly found in the pastures include fescue, orchardgrass, clover, bluegrass, velvetgrass, creeping buttercup and ryegrass. Reed canary grass and water foxtail are common invaders on wet sites. Blackberries, rushes, sedges, roses, American elder and snowberries are common plants utilized as food or cover by deer (Gavin 1984).

Davison considers the tidal spruce community to be the historical habitat of Columbian white-tailed deer for forbs and grass in open pastures, as observed by Suring (1974) and Gavin (1979), and may actually be an adaptation to available habitat rather than on actual feeding preference (USFWS 1976).

13.6 Factors Affecting Population Status

The USFWS (1976) reported that the integrity of the Columbia River population of Columbia white-tailed deer and their habitat is threatened by a variety of factors, including both natural and man-caused phenomena including:

- Degradation of riparian habitats through logging and brush removal (Crews 1939; Scheffer 1940; Gavin 1978),
- Recent interest in development of riparian zones for beef production, cottonwood and alder harvest and for marina development,
- Automobile collisions,
- Poaching,
- Entanglement in barbed wire fences,
- Competition with livestock,
- Introduction of feral swine on Wallace Island in 1980,
- Major flooding,
- The inundation of over 1,400 acres for nearly 1.5 years due to a dike failure,
- High tides which are a limiting factor on undiked islands of the lower river,
- Disease (foot rot) and parasites (stomach worms), two threats common to the Columbia River population,
- The potential threat of black-tailed deer to Columbian white-tailed deer by direct competition for available food sources and by hybridization, and
- Presence of Roosevelt elk on the mainland portion of the CWTDNWR.

13.6.1 Availability of habitat

Columbian White-tailed Deer are resident in suitable habitat and show little tendency to wander outside the home range. Preferred habitat in the lower Columbia Subbasin is limiting. Extensive losses of habitat have occurred in the lower Columbia and estuary provinces as a results of dredging, filling, diking, and channelization. The floodplain and lowlands likely were much more heavily forested and historically there were many more lakes, ponds, sloughs, overflow channels, backwaters and wetlands. Between 1850 and 1999, 20,000 acres of tidal swamps (with woody vegetation, 10,000 acres of tidal marshes (with non-woody vegetation), and 3,000 acres of tidal flats have been lost along the lower Columbia River (BPA unpub. data).

Perhaps the greatest single man-caused threat to Columbia White-tailed Deer is the continued degradation of riparian habitats through logging and brush removal. On the mainland, in 1939 70% of the refuge was wooded; in 1972 it was 17% wooded, and there has been little change between 1972 and 2004 (A. Clark, pers. obs.). On private land, most bottomlands have been cleared of trees and brush, and seeded to grasses and forbs that provide feed for beef and dairy cattle. On Puget Island, which is in private ownership, woodlots are being cleared for agriculture and housing, and have decreased from 43% of the island in 1938 to 1% in 1977. On private land in 2004, degradation of riparian habitats through logging and brush removal is continuing.

Columbian White-tailed Deer use pastures less than other habitat types for feeding and reproduction, and there is more pasture and less woody vegetation than is needed by deer for cover and for naturally-occurring forage at the refuge. It is recognized that re-establishment of woody vegetation is necessary on the refuge, but is difficult to achieve in practice (A. Clark, pers. comm. October 2003), due to browsing pressure from deer and elk.

On the mainland portion of the federal refuge, white-tailed deer are not more common there now than they were in the 1970's, and numbers seem stable at moderate densities (40+/sq mi). For more than three decades, the national wildlife refuge has been managed for deer, without population increase or natural range expansion. It appears as if the long-term carrying capacity of the mainland portion of the refuge has been reached in relation to the types of habitats provided within refuge boundaries, namely a relative abundance of pasture and a relative scarcity of woody cover and browse.

13.6.2 Lack of Continuity Between Suitable Habitats

Preferred secure habitat is non-contiguous along the lower Columbia River. By themselves, deer cannot easily pioneer new habitat, because suitable non-occupied habitat is for the most part unavailable adjacent to occupied habitat. Therefore, deer have to be artificially reintroduced for range expansion to occur. However, some deer have moved from island sites to which they had been relocated in 2003, to the Washington mainland at Willow Grove near Longview. Other unoccupied habitat exists along the lower Columbia River, e.g., Vancouver bottomlands, to which deer could be relocated.

Lastly, there is limited suitable habitat to which deer can escape, and survive, when uncommon flooding events occur.

13.6.3 Occasional Low Productivity:

Fawn:doe ratios have been variable, low in some years, high in others. The data on fawn:doe ratios vis-a-vis predation are inconclusive. The ratio on the refuge mainland was high in 1997 with coyote control, but also high in 1984-85 and 2001 with no coyote control. The fawn:doe ratio has been consistently higher on Puget I. (private land where it is possible that coyotes are better controlled). Fawn production on Tenasillahe I. was satisfactory in the 80's and 90's, but has been low in recent years.

13.6.4 Disease

The incidence of parasites (liver flukes, stomach worms, etc.) suggests overcrowding and habitat at carrying capacity, but the incidence is considered moderate. Necrobacillosis (foot-rot), found in 1/3 of 155 carcasses examined, has been called probably a major debilitating factor contributing to mortality of adult deer. However, Columbian white-tailed deer are adapted to marsh habitat, and may not be adversely affected by the observed levels of disease and parasites.

13.7 Inventory & Assessment of Existing Management Plans

- Columbian White-tailed Deer Recovery Plan.

The Recovery Plan for Columbian whitetails was written by the recovery team, which is composed of members of the USFWS, ODFW, WDFW, and OSU faculty. The plan outlines steps toward creation of three stable, secure, viable subpopulations of Columbian white-tailed deer so that delisting may proceed. Plan components include the need for new habitat acquisitions, transplanting of existing populations to create new populations, enforcement of hunting riles and management of publicly owned lands.

13.8 Inventory & Assessment of Existing Restoration & Conservation Plans

- Hunting rules and WAC of Washington

The WDFW Commission adopted rules to protect the Columbian white-tailed deer by closing hunting in the areas where Columbian white-tailed deer exist. These regulations substantially reduced the legal harvest.

- WAC of Washington

The Columbian white-tailed deer is listed as an endangered species and hunting is prohibited. Enforcement reduces illegal harvest.

- Population re-introduction

Recent attempts to re-introduce Columbian white-tailed deer in the lower Columbia region include transplanting 12 deer to Fisher and Hump Islands in spring 2003(Miller, pers. comm.). Deer were re-introduced into the lower Columbia River in Oregon at Crimms and Lord Islands near Longview, Washington. Deer from the Crimms Island project have become established in the Willow Grove area. The deer at Lord Island were released in early 2003 (Miller, pers. comm.).

- Refuge management

USFWS manages public lands near Cathlamet that are critical to the existing population of Columbian white-tailed deer. The lands are managed to provide food, water, and cover for a resident population of Columbian white-tailed deer.

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Volume III, Chapter 14
Sandhill Crane

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14.0 Sandhill Crane *(Grus Canadensis)*

14.1 Introduction

The sandhill crane (*Grus canadensis*) is one of 15 species within the family *Gruidae*, one of the world's most imperiled avian families. Habitat destruction and hunting have severely reduced several species of cranes; 47% now are listed as either endangered or threatened, with several at risk of extinction (Ellis et al. 1996). For crane species in general, all but two occur in Africa, Australia, or Eurasia. The sandhill and whooping (*G. americana*) cranes are the only family members in North America; however, common cranes (*G. grus*) have strayed into Canada and the United States on rare occasions.

The sandhill crane has been listed as an endangered species by Washington since 1981. The species is represented in Washington by a small number of greater sandhills that breed in Klickitat and Yakima Counties, about 23,000 lesser sandhills that stop in eastern Washington during migration, and 3,000-4,000 sandhills (Canadians and possibly some lessers and greaters) that stop on lower Columbia River bottomlands. Up to 1,000 sandhills have wintered on lower Columbia bottomlands in recent years, but most cranes seen in Washington winter in California.

The greater sandhill cranes that breed in Washington are part of the Central Valley Population, so called because they winter in California's Central Valley. Other members of this population nest in Oregon, California, Nevada, and interior British Columbia. The lesser sandhill cranes are of the Pacific Flyway Population that stop during migration on the way to breeding grounds in Alaska or wintering areas in California. The Canadian sandhill cranes have not been defined as a population, and recent studies of the Mid-Continent Population suggest that they may not differ genetically from greaters. Some breed along the coast of central British Columbia and winter in Washington, while some stop during migration en route to wintering areas in California. Further studies are needed to clarify their status and distribution.[1]

The historical distribution of breeding cranes in Washington was poorly documented, but the few historical accounts mention breeding in south central, northeastern, and southeastern regions, and the southern Puget Sound basin. Crane numbers had been severely reduced due to widespread habitat destruction concurrent with human settlement, and perhaps more importantly, unregulated hunting which continued until passage of the federal Migratory Bird Treaty Act in

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[1] The majority of this chapter is taken from C.D. Littlefield and G. L. Ivey, Washington State Recovery Plan for the Sandhill Crane, WDFW, 2002.

1916. The species was extirpated as a breeder from the state after 1941 when the last nest was documented at Signal Peak, Yakima County, in south central Washington. Some 31 years later, cranes were again found summering in 1972 in the Glenwood Valley on Conboy Lake National Wildlife Refuge, Klickitat County, but it was not until 1979 that nesting was confirmed. A total of 19 territorial pairs was documented in 2000: 16 at Conboy Lake National Wildlife Refuge, and one each on Yakama Indian Nation lands, Yakima County; Panakanic Valley, Klickitat County; and on WDNR lands along Deer Creek, Yakima County. The total summer population in Washington in 2000 was 53 birds. No nests produced chicks to fledging age in 2001, probably due to factors relating to drought conditions; the total summer population was 50.

Factors affecting breeding greater sandhills in Washington include predation, incompatible grazing and haying practices, water availability and management, and habitat loss. Crane habitat on the lower Columbia bottomlands between Vancouver and Woodland is threatened with industrial development, conversion of agricultural lands to cottonwood plantations, tree nurseries, or other incompatible uses, and crane use is affected by disturbance by hunters and other recreationists.

The goal of the recovery plan is to restore a healthy breeding population of cranes and to maintain the flocks that winter or stop in Washington. To reach this goal, this plan calls for expansion of the breeding range of greaters into former breeding areas in eastern Washington and protection of habitat for crane wintering and staging during migration. The plan identifies recovery objectives that must be reached, and outlines strategies to use in meeting them before down-listing of the species to threatened or sensitive can occur.

The sandhill crane will be considered for down-listing from state endangered to state threatened status when the state’s overall breeding population reaches at least 65 territorial pairs with an average annual recruitment rate of >8 %, and effective water management control is established at Conboy Lake National Wildlife Refuge. The sandhill crane will be considered for down-listing to state sensitive when the state’s breeding population reaches at least 130 territorial pairs with an average annual recruitment rate of >8 %, and habitat used by cranes at the major staging sites in eastern Washington is protected through management agreements or easements. Also, for down-listing to sensitive, enough habitat needed to maintain 2,000 migrant and 500 wintering cranes should be secured and managed for cranes on the lower Columbia River bottomlands in Washington. Recovery objectives may need to be updated as better information is available about habitat needs.

14.2 Description

Sandhill cranes are large, stately, and symbolic of the remote, isolated wetlands they depend on. The sexes are similar in appearance with a bare red forehead, lores, and crown, and feathered whitish cheeks, ear coverts, chin, and upper throat. Pale slate gray, ashy gray, and brownish-gray characterize the body, wing, and tail feathers. The body and wing feathers are frequently stained with rust, particularly in summer and autumn. This reddish-brown coloration is from ferric oxide, not pigmentation (Taverner 1929). Sandhills smear mud onto their feathers using their beaks; if this occurs in iron-rich soils, the rust coloration results. The purpose of this behavior is unknown. Cranes have 10 primaries and 16 secondaries, with the innermost secondary coverts and tertials elongate, ornamental, and drooping over the tail. The bare red crown of adults is covered with black hairlike bristles, and extends from the base of the bill above the eyes to the back of the head. This red papillose skin is connected to muscles and, when the bird is territorial or involved in aggressive encounters, the crown area can be expanded and the red coloration intensified (Grooms 1992, Tacha et al. 1992). The adult iris is orangish or reddish; the bill is dull slate to partially olive gray, stout, elongate, with a perforated internasal septum. The legs and toes are blackish. The foot is anisodactyl, with three toes forward and one elevated hind toe (hallux) (Tacha et al. 1992). Cranes fly with neck and legs extended except in cold weather; on cold mornings birds occasionally will fly with legs retracted into their belly feathers (Walkinshaw 1953).

Fledged young and immatures have some juvenile body and wing feathers tipped with tawny and ocher during their first autumn and sometimes into early winter. The head and upper neck are cinnamon, with the crown and nape covered with tawny feathers (Johnsgard 1983). The other body feathers are similar to adults, and all feathers are identical by spring. The iris is gray brown to reddish brown until winter.

Sandhill cranes are large birds, standing about 4 feet tall, and often weighing over 10 pounds (Tacha et al. 1992). Adult male greater sandhill cranes from Lincoln County, Wyoming, averaged 11.75 pounds and weighed up to 14.6 pounds. (Lockman et al. 1987). Females averaged 10.6 pounds, and weighed up to 12.5 pounds. Ninety-five percent of all females weighed < 12 lbs, 0 oz (5,450 g) and had culmens <4.4 in (108 mm), whereas all males had weights >12 lbs, 8 oz (5,674 g) and culmens >4.5 in (110 mm) (Lockman et al. 1987). Greater sandhill cranes are the largest of the six subspecies, lessers are the smallest, and Canadians are intermediate (Figure 14-1).

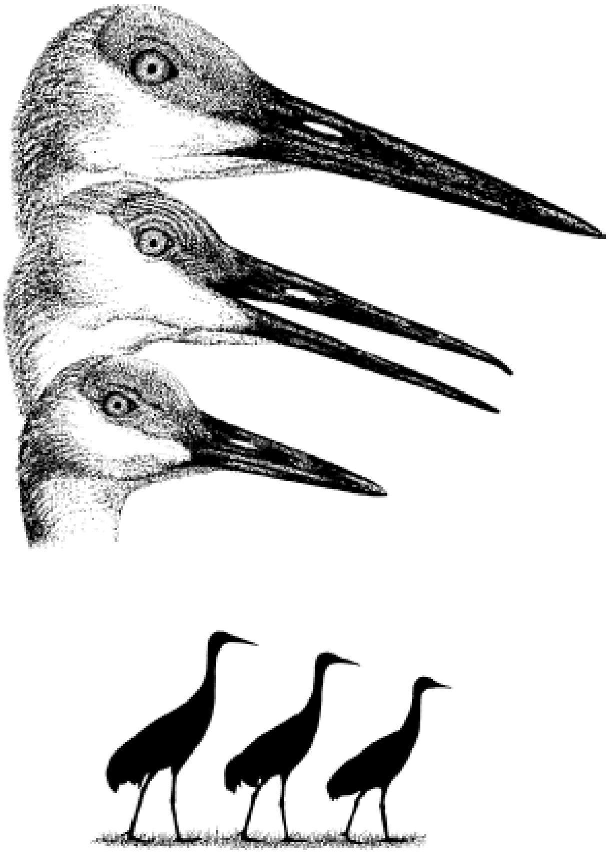

Figure 14-1. Relative size of three sandhill crane subspecies: greater (top and left); Canadian (middle); and lesser (bottom and right).

Although with training and experience, greaters and lessers are easy to distinguish from each other, the presence of the Canadian subspecies confounds identification, especially between Canadians and greaters. When birds are captured, anatomical measurements can be taken to verify subspecies identity (Table 14-1).

Table 14-1. Anatomical dimensions (mm) of greater, Canadian, and lesser sandhill crane subspecies.

Subspecies	Sex (n)	Exposed culmen[a]	Tarsus[b]	Longest toe
Greater Sandhill	M (11)	131.8 ±5.0	236.5 ±8.3	87.2 ±2.5
(G.c. tabida)	F (10)	120.4 ±2.9	228.3 ±6.3	84.5 ±2.6
Canadian Sandhill	M (51)	119.7 ±5.9	230.6 ±9.5[c]	86.4 ±3.2
(G.c. rowani)	F (33)	114.1 ±3.9	217.0 ±7.6	83.3 ±3.7
Lesser Sandhill	M (31)	97.3 ±3.9[c]	187.5 ±14.4	75.4 ±3.2
(G.c. canadensis)	F (17)	92.0 ±5.2	179.2 ±10.8	73.4 ±4.8

[a] Exposed culmen = the length between the tip of the bill and the edge of the feathering at its base

[b] Tarsus = lower leg bone

[c] Sample size was 1 less than indicated

Source: Johnson and Stewart, 1973

Adult calls are rattling, loud, and resonating (Johnsgard 1983), whereas full grown young have a shrill *peeeer* (Walkinshaw 1949). The call of the sandhill crane has been described by some as the voice of the Pleistocene. Sandhills have an extraordinarily long trachea (≈48 in) coiled within their chest that apparently improves the harmonics of their vocalizations (Grooms 1992).

14.3 Distribution

14.3.1 North America

Of the six subspecies of sandhill cranes found in North America, the Cuban, Florida, and Mississippi are nonmigratory, and the lesser, greater, and Canadian are migratory. Distinct populations are recognized for both lesser and greater sandhill cranes.

- *Lessers* are divided into two populations: the Mid-Continent Population breeds in western and northern Alaska, northern Canada, and Siberia, and winters in the southwestern United States and northern Mexico; the Pacific Flyway Population breeds in south-central and southwestern Alaska, and winters mostly in California's Central Valley.
- *Greaters* are divided into five populations: the Eastern, Prairie, Rocky Mountain, Lower Colorado River Valley, and Central Valley (Meine and Archibald 1996) (Figure 14-2). The greater sandhill cranes that breed in Washington are members of the Central Valley Population—greater sandhills that winter in the Central Valley of California. This population is divided into two segments because of their current disjunct distribution:
    - The southern segment breeds in south-central Washington, eastern and central Oregon, northeastern California, and northwestern Nevada with a pair in Douglas County, Nevada (American Birds 45:1142, North American Birds 53:414) marking the southernmost known pair for the Central Valley Population.
    - The northern segment of the Central Valley Population breeds in British Columbia, and is widely distributed and much less concentrated than cranes in the southern segment; their exact range is unknown.

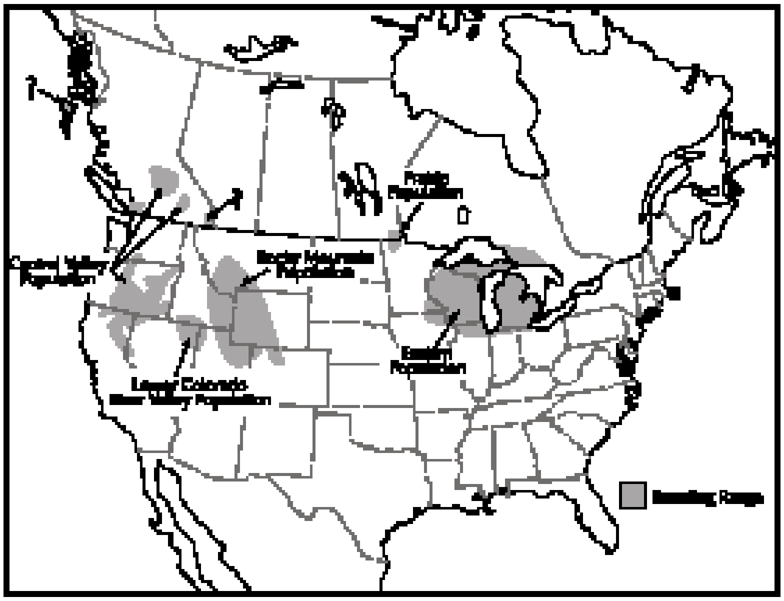

Figure 14-2. Breeding distribution of greater sandhill cranes in the United States (from Tacha *et al.* 1992, Cooper 1996, Meine and Archibald 1996, Ivey and Herziger, in prep.).

- *Canadians* are thought to breed along the Pacific Coast in British Columbia, although little is known about their distribution in the Pacific Flyway (Cooper 1996). Some Canadians winter at Ridgefield National Wildlife Refuge (NWR) in southwest Washington (Clark County) and adjacent Sauvie Island in Oregon (Multnomah and Columbia Counties) (Ivey et al. in prep.). Six Canadian sandhills marked in 2001–02 on Sauvie Island and Ridgefield NWR returned to their summer range along the coast of British Columbia and southeastern Alaska; five of the six used offshore islands (Ivey et al. in prep.).

14.3.2 Washington

The greater sandhill crane is the only subspecies that nests in Washington. The only known breeding sites are: Conboy Lake NWR and Panakanic Valley, Klickitat County; Polo Field/Signal Peak on Yakama Indian Nation lands, Yakima County; and Deer Creek on WDNR lands in Yakima County (Engler and Brady 2000) (Figure 14-3). All pairs in the Glenwood Valley are listed here as on Conboy Lake NWR because all territories are at least partially within the boundaries of the refuge (Engler and Brady 2000). From 1995–97, a pair was on territory 12 mi (19 km) south of Fort Simcoe in an area known as the Camas Patch; this site apparently no longer provides suitable habitat (J. Engler, personal communication). Additionally, a few summer records of sandhill cranes from dispersed localities have not been confirmed as breeding (Table 14-2).

A few migrant greater sandhill cranes stage in Washington as they move to or from breeding areas in British Columbia, but most apparently over-fly the state. Little evidence is apparent that significant numbers of British Columbia greaters stop in Washington. In eastern Washington, a flock was documented as containing about 20 greaters near Othello in 2000 (R. Hill, personal communication), and 200-300 stop annually in spring near Waukon, Spokane County (M. Rule, personal communication). Migrants also have been noted from Grant and Klickitat Counties, and the subspecies also likely occurs in Douglas County (Field Notes 50:989). A few greaters may stop in Adams, Lincoln, and Okanogan Counties, particularly during inclement weather, but accounts are lacking (R. Friesz and M. Murphy, personal communications); there are multiple sightings of lesser or unidentified sandhill cranes there. In western Washington, some greaters may stage at Ridgefield NWR, but their occurrence there has not been confirmed. Most migrants in the western portion of the state have been presumed to be lessers, but a recent study suggests they may be Canadians (Ivey at al. in prep.).

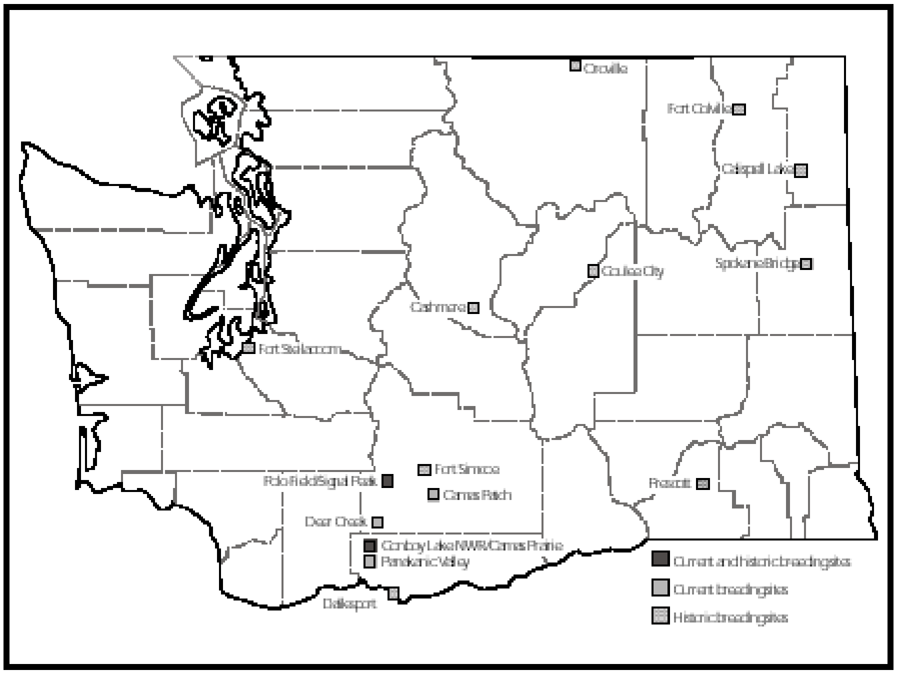

Figure 14-3. Known past and current breeding distribution of greater sandhill cranes in Washington (from Dice 1918, Jewett *et al.* 1953, and Engler and Brady 2000).

Table 14-2. Recent breeding-season sightings of greater sandhill cranes in Washington that were not confirmed as breeding (likely subadults).

Location	County	Date	Number	Source
Sequim[1]	Clallam	10 June 1980	3	American Birds 34:923
Wenas Lake	Yakima	20 June 1981	3	J.Smith/B. Lamb (WDFW files)
Anatone	Asotin	Jul–Aug 1981	1	Canyon Birders Audubon
Field Spring State Park	Asotin	26 Jul 1981	4	Canyon Birders Audubon
Glenoma	Lewis	14 Jul 1981	1	Tahoma Audubon
Lower Columbia River[1]	? [not noted]	June 1982	2	American Birds 36:1009
Nile[1]	Yakima	9 June 1982	1	American Birds 36:999
Ellensburg	Kittitas	3 May 1989	1	Paulson (1989)
Columbia NWR	Grant	5 May 1987	1	American Birds 41:464
Ridgefield NWR	Clark	6 June 1996	1	Field Notes 50:989
Atkins Lake[1]	Douglas	9 June 1996	1	Field Notes 50:989
Ridgefield NWR	Clark	15 June 1997	10	Field Notes 51:1045
near Prosser	Benton	April 1999	1	D. Friesz
N. Whidbey Island	Island	4 June 1999	4	*fide* Randy Hill
Tiger Meadow, 7 mi	Pend Oreille	summer-early	1	J. McGowan, USFS

[1] Probably greater sandhill cranes but not confirmed.

Most of the estimated 21,000-23,000 cranes that occur during migration in eastern Washington are lesser sandhill cranes (Littlefield and Thompson 1982). Also, this subspecies was believed to migrate through the western portion of the state, staging at Ridgefield NWR (Kramer et al. 1983). However, Pogson and Lindstedt (1991) estimated 900 or so Canadian sandhills in this area in the early 1980s; during winter 2001–02 and spring 2002, all of the cranes observed at Ridgefield NWR and Sauvie Island Wildlife Area were Canadians, while no lessers were observed (Ivey et al. in prep.).

14.4 Natural History

14.4.1 Reproduction

Sandhill cranes have a life history strategy that involves a low reproductive rate but high investment in the pair bond and in defending the breeding territory. This contrasts to some birds, like California quail, that live short lives but lay large clutches (12-16 eggs) and, given favorable conditions, can raise large broods that more than compensate for high mortality rates. Cranes usually take three or more years to mature, may nest for several years before successfully hatching eggs, and still may not be successful in raising a chick. When successful, cranes rarely raise more than one young. Sandhills compensate for this low production with a relatively long life of up to 30 years or more (C. Littlefield and G. Ivey, unpubl. data).

14.4.1.1 Chronology

In February, greater sandhill cranes begin migrating north from the California Central Valley to their breeding territories. At Conboy Lake NWR, birds usually arrive between late February and mid-March. Pairs generally arrive first, accompanied with chicks from the previous year, whereas 2-3 year old birds (subadults) generally arrive a few weeks later (J. Engler, personal communication). Pairs usually return to the same territory, and generally remain on or near the site for a month or more before beginning nesting activities, usually in mid-April. Yearling young are driven away when pairs get ready to nest. At Conboy Lake NWR in 2000, the first nest was noted on April 11 and the first hatching occurred around May 11; the latest hatch date was July 4. At Malheur NWR, the earliest known clutch was on March 25, but peak of nest initiation is usually around April 21 (C. Littlefield, unpubl. data), suggesting clutches are deposited about the same time in Washington as at other Central Valley Population nesting localities. Thus, the incubation season extends from late March into early July; the brooding season is generally from late April into late August, occasionally extending to early September.

14.4.1.2 Pair Bonding

Greater sandhill cranes generally form lifelong pair bonds and are monogamous. Sandhills in a growing population may pair and defend a territory at 2 years of age, but the chance of nesting success probably improves as the birds mature. Birds usually defer first breeding until > 3 years of age (Drewien et al. 1995), with most nesting for the first time at age 4. A 3-year old crane from Conboy Lake NWR was found paired and on territory at Camas Prairie on the Mt. Hood National Forest in Oregon in 2000, approximately 59 mi (94 km) south of Conboy. The pair was acting broody, suggesting they had a chick (G. Ivey, personal observation). Nesting of the pair was confirmed on May 20, 2001, when two chicks were observed; both are believed to have fledged (M. Gould, personal communication to J. Engler). Sandhills have been known to delay breeding until 5 years, but on rare occasions have bred at 2. For example, at Conboy Lake NWR, two 2-year old male color-banded siblings displaced a territorial pair, divided the territory, and nested within 981 ft (300 m) of each other (Engler and

Anderson 1998). At Modoc NWR in northeastern California, a color-banded 2-year old female successfully nested (Radke and Radke 1986). Most successful reproduction (>75%) in the Mid-Continent Population occurs in birds > 8 years old (Tacha et al. 1992).

14.4.1.3 Territories

Sandhill cranes defend exclusive nesting territories, and foraging habitats rarely overlap. They are highly philopatric, usually returning annually to the same breeding territory. Nine pairs observed at Conboy Lake NWR over 2-6 nesting seasons had territories (including both nesting and foraging sites) that averaged 345 ac (140 ha), and ranged from 143-540 ac (58-218 ha) (J. Engler, personal communication). At Grays Lake NWR, Idaho, five territories ranged from 25-57.5 ac (10-23 ha) and averaged 42.5 ac (17 ha) (Drewien 1973), whereas at Malheur NWR, size varied depending on pair density, ranging from 3-170 ac (1.2-68 ha) and averaging 62.5 ac (25 ha) (Littlefield and Ryder 1968). At a high-density Malheur NWR site, eight territories averaged 22.5 ac (9 ha) (C. Littlefield, unpubl. data).

14.4.1.4 Nest Building, Eggs, & Incubation

Both pair members participate in nest building. Nests are composed of vegetation from the surrounding wetland left from the previous growing season. Cranes collect nesting material and pile it into a mound, usually in shallow water. The clutch is usually two eggs, but occasionally only a single egg is laid, and on rare occasions, three. At Malheur NWR, for 974 completed clutches, 84 (8.6%) contained one egg, 886 (91%) two eggs, 3 (0.3%) three eggs, and 1 (0.1%) contained four eggs; mean clutch size was 1.9 (Littlefield 1995a). In California, average size for 42 clutches was 1.9 (Littlefield 1995b). Eggs are sub-elliptical to long oval, and vary in color from brownish-buff to light olive, irregularly marked with darker brown, reddish-brown, or pale gray (Tacha et al. 1992, Littlefield 1995c). The incubation period is normally 30 days, but the second egg frequently hatches at 29; however, the incubation period may extend to 33 days for fertile eggs and 43 for infertile or addled (Littlefield and Holloway 1987).

14.4.1.5 Brood Rearing & Fledging

Since a crane pair initiates incubation shortly after the first egg is laid, there is a 24–48 hour difference in hatching times between eggs. Soon after the second chick dries and gains sufficient strength to swim and walk, the adults lead the chicks from the nest to feed in nearby moist meadows or subirrigated ecotones. Both parents tend the young and the birds remain as a close family unit through the brooding period. Young chicks are brooded by the female at night, but once they attain sufficient size, they spend the night roosting in shallow water with their parents. The fledging period lasts from 66–75 days; however, after birds fledge, it takes a few weeks for chicks to become strong fliers. After fledging, cranes maintain their family association as young remain with their parents in migration and winter, usually returning together to breeding grounds the following spring. Two siblings banded at Conboy Lake NWR in 1996 were observed together in California the following winter (Engler and Brady 2000). Eleven of 16 chicks color-marked at Conboy Lake NWR since 1996 have fledged and successfully migrated (Engler and Brady 2000).

14.4.1.6 Nesting Success & Recruitment

Nest success can vary considerably between years due to weather, water and habitat conditions, and predation pressure. At Conboy Lake NWR, nest success since 1995 has been 67% (n = 69) (Engler and Brady 2000). In 2000, 7 of 13 nests (54%) hatched young. The pair at the Polo Field on Yakama Nation lands hatched two eggs and fledged one chick in 1997 (Stepniewski 1999, R. Leach, personal communication), but the pair at the Camas Patch was not reproductively successful through 1997, apparently because of early drying and many cattle (R. Leach, personal communication). Outside of Conboy Lake NWR, other Washington sites have rarely been monitored for nest success.

Generally, nesting success rates in the Pacific states are less than those reported elsewhere within the subspecies' breeding range. Nesting success ranged from 77-78.9% in Michigan (Hoffman 1979, Walkinshaw 1981), 78% in Idaho (Drewien 1973), and 84% in Wisconsin (Bennett 1978), whereas in south-central Oregon, success was 29.8% at Sycan Marsh (Stern et al. 1987), and at Malheur NWR, 44% were successful from 1966–74 (Littlefield 1976a), and 54% from 1976–89 (Littlefield 1995a). In total, for 1,702 clutches assessed at Malheur NWR (1966–98), 978 (57%) hatched at least one egg. Elsewhere, 56 nests in northeastern California had an average success rate of 37.5% in 1988, and in another study on privately-owned lands at scattered locations in eastern Oregon, 69.8% of 63 clutches successfully hatched in 1976 and 1986 (Littlefield 1999b).

Reproductive success for this long-lived species is usually low. However, recruitment (% of fledged young in the population; calculated using known breeding pairs and counts of fledged young) in Washington has averaged 10% (range 0-27.3 %) from 1990–2001 (Engler and McFall 2001). Recruitment rates for about 50 breeding pairs at Klamath Marsh in Oregon were 8% in 1993 and 2% in 1994 (Drew et al. 1994). At Malheur NWR, recruitment for the period 1970–89 averaged 6.7% and nesting pairs were declining (Littlefield 1995a) and from 1990–98, recruitment averaged 5.8% (G. Ivey, unpubl. data). Low recruitment (4.5%) was reported for cranes breeding at Sycan Marsh, Oregon (Stern et al. 1987) and for the entire Central Valley Population (5.6-6.1%). These recruitment rates are among the lowest recorded for North American cranes (Drewien et al. 1995). For example, the number of greater sandhill cranes nesting in the Great Lakes region (Eastern Population) has been increasing, and recruitment rates have averaged 12-12.7% (Lovvorn and Kirkpatrick 1982a). Recruitment for the Rocky Mountain Population ranged from 9.4-12% in the early 1970s, and the population was increasing; however, since 1986, recruitment has declined (ranging from 3.4-6.5%) and the population is stable or slightly decreasing (Drewien et al. 1995). In the past, an 8-10% annual recruitment rate was considered necessary for population maintenance (Littlefield and Ryder 1968). Recent data suggests that with improved and active management, possibly coupled with a reduction in illegal kills, stability may be maintained with an annual recruitment rate of 7-9%, but a higher rate is needed for a population increase.

14.4.2 Longevity & Mortality

Greater sandhill cranes can reach an age of at least 30 years in the wild (C. Littlefield and G. Ivey, unpubl. data). If young survive the brooding period, mortality rates decline dramatically once they develop sufficient flying skills. The mean life expectancy for Florida sandhills that reached independence was 7 years (Tacha et al. 1992). In an eastern population of greater sandhills, annual survival rate (all post-juvenile age classes combined) was 0.874 for males and 0.858 for females (Tacha et al. 1992). Primary causes of sandhill crane mortality are predation of young (occasional in adults) and collisions with powerlines. Other sources of fatality include entanglements in fences, diseases, and illegal shooting.

14.4.2.1 Chick Mortality

Predation is the primary cause of chick mortality, but intraspecific aggression (fratricide, infanticide), drowning, starvation, parasites, and accidents such as fence entanglements and road-kills contribute to losses. Coyotes are thought to be the primary predator of crane chicks at Conboy Lake NWR (Engler and Brady 2000). To assess chick mortality, several radio-telemetry studies have been completed at different locations within the Central Valley Population's breeding range. At Modoc NWR in 1990 and 1992 during a period of predator management, four of 28 (14%) monitored chicks were killed by minks, three (11%) by coyotes, one (4%) each were lost to infection and starvation, and seven (26%) were lost to unknown causes (including tag loss) (DesRoberts 1997). For 10 transmitter-equipped chicks at Klamath Marsh NWR in 1993 and 1994, three were lost to undetermined predators, two to coyotes, two lost transmitters, one died of exposure, and two were found dead but the causative agent could not be determined (Drew et al. 1994). Eighteen chicks were radio-marked at Sycan Marsh in 1984, and total mortality was 44%. Predation accounted for 83% of the mortalities and all predation except one was attributed to coyotes; one was attributed to an unidentified raptor. Fratricide accounted for the other explicable death, whereas two others apparently died but were not recovered, and 10 (56%) fledged (Stern et al. 1984).

A telemetry study at Malheur NWR in 1983 and 1984 (a period without predator control) showed that from a sample of 39 transmitter-equipped chicks, in 1983 13 were lost to predators, one died from a parasitic gapeworm infection, one drowned, contact was lost with four, and three died from unknown causes; in 1984, four were lost to predators and 10 transmitters malfunctioned, but eight of these chicks were known to have died before fledging. Of 17 chicks where predator identity was known, coyotes took 13 (77%), great horned owls two (12%), raccoon one (6%), and domestic dog one (6%) (Littlefield and Lindstedt 1992). In a more extensive telemetry study conducted on Malheur NWR when predators (particularly coyotes) were being or had recently been intensively managed for 8 years (1986–93), from 1991–98, 219 chicks were transmitter-equipped (G. Ivey unpubl. data). Fates of 41 chicks were undetermined and 27 of 178 (15%) fledged. Of the known fates, predators were responsible for 109 (61%), intraspecific causes 11 (6%), parasitic gapeworms 10 (6%), drowning nine (5%), starvation four (2%), unknown deaths three (1%), abandoned one (<1%), fence entanglement one (<1%), vehicle one (<1%), hay-swather one (<1%), and study-related mortality one (<1%). Of the 109 killed by predators, 29% were lost to minks, 21% to coyotes, 17% to great horned owls, 13% to unidentified predators, 9% to golden eagles, 8% to unidentified raptors, 0.5% to a northern harrier, and 0.5% to a raccoon. Between 1970–98 at Malheur NWR, during years when predator control was practiced, chick mortality was 84.4% compared with 91.1% in years when predators were not controlled (G. Ivey and C. Littlefield, unpubl. data).

14.4.2.2 Adult Predation

Few predators are capable of taking adult or subadult greater sandhill cranes. There are, however, several records of cranes being attacked by golden eagles (Ellis et al. 1999) or coyotes (Littlefield 1986), and there are records of bobcats killing cranes in other regions. Bald eagles are known predators of lesser sandhill cranes (Herter 1982, Littlefield 1999a), but greaters usually pay little attention to the species (C. Littlefield, personal observation). However, two subadult bald eagles were noted stooping at an adult crane after a nest exchange at Conboy Lake NWR in 1998 (J. Engler, personal communication), and migrant and wintering cranes at Ridgefield NWR and Sauvie Island took flight from approaching bald eagles (G. Ivey, personal observation). Certainly both black and grizzly bears, as well as gray wolves and mountain lions would be capable of killing adult cranes.

14.4.2.3 Powerline Collisions

Young fledglings are prone to collisions with utility wires, particularly on windy days. Even in adulthood, utility wires pose a threat, and collisions are considered a major mortality factor, particularly at staging areas and on the wintering grounds. At a staging site in southwestern Colorado, 15% of 597 powerline mortalities were sandhill cranes (Brown and Drewien 1995). For the Central Valley Population, the critical mortality period is winter. Persistent winter fog in California, coupled with an extensive network of utility lines, frequently kills cranes—usually in the early morning as birds leave roost sites and fly to nearby grainfields to feed (Littlefield 1999a) (R. Schlorff, personal communication). On the breeding grounds, territorial adults have been found dead beneath utility wires (T. Melanson, personal communication; C. Littlefield and G. Ivey, personal observation). One crane died after colliding with utility wires at Conboy Lake NWR in 1984 (Paulson 1989) and two migrant lesser sandhills were found dead under a powerline in Douglas County in 1981.

14.4.2.4 Fences

To a lesser extent, collisions and entanglements with barbed-wire fences have resulted in crane deaths. Unlike collisions with utility wires, most known fence mortalities have occurred on the breeding grounds; at least six victims have been found in southeastern Oregon (C. Littlefield and G. Ivey, unpubl. data). Of 135 deaths of color-marked greater sandhill cranes in the Rocky Mountain Population, Drewien et al. (in prep.) reported 8 (4.5%) died from fence collisions or entanglements.

14.4.2.5 Disease

Within the Central Valley Population, little information is available on diseases; however, avian cholera has resulted in mortality in California’s San Joaquin County (S. Lindstedt, personal communication), and botulism killed at least one crane in Oregon (G. Ivey, personal observation), whereas aspergillosis, salmonella, and avian tuberculosis have killed sandhill cranes elsewhere in the United States. All of these diseases occur in the west, and cranes in the Pacific states certainly would be susceptible should an outbreak occur (Littlefield 1999a).

14.4.2.6 Illegal Shooting

While cranes were frequently shot illegally in the late 1960s and early 1970s, increased public awareness and interest, in addition to increased enforcement, have apparently resulted in this mortality factor being greatly reduced. For example, several cranes were known to have been shot in the Central Valley in 1969–72, but none is known to have been killed from 1991–93 (C. Littlefield, personal observation).

14.4.2.7 Other Factors

Elsewhere, other lethal factors have included aflatoxicosis, lead poisoning, and catastrophic/environmental mortalities (Windingsted 1988). For example, 90 sandhill cranes were killed by lightning in Nebraska in April 1978 and about 600 were killed in an Oklahoma hailstorm in October 1979 (in Windingsted 1988), and more than 1,000 lesser sandhill cranes died from hail in eastern New Mexico in October 1960 (Merrill 1961). Most unusual was a 4-

year old male greater sandhill crane at Grays Lake NWR, Idaho, that was killed by a male whooping crane during a breeding territory dispute (Drewien et al. in prep.).

14.4.3 Migration & Dispersal

Individual greater sandhill cranes consistently return to the same nesting territories and wintering sites as long as habitat conditions remain suitable (Tacha et al. 1992, Drewien et al. 1999). Distances from natal site to first breeding site have not been reported. Males are believed to be more philopatric than females; that is, males typically establish a breeding territory closer to their natal site than do females, as is typical in many territorial birds (Tacha et al. 1992, Greenwood 1980).

14.4.3.1 Spring Migration

Except during inclement weather, adult greater sandhills usually do not linger along the migration corridor as they migrate north to breeding sites, whereas subadults spend some time at traditional spring staging areas. Annual spring use varies, but traditional sites for the Central Valley Population have been identified in California at Davis Creek and Surprise Valley (Modoc County), and Grass Lake and Lower Klamath NWR (Siskiyou County). Flocks have been seen at these sites in May and well into June (Littlefield et al. 1994). In Oregon, known staging areas include Malheur NWR, Diamond Valley, and Silvies River Floodplain (Harney County), Williamson River Delta and Klamath Marsh (Klamath County), Warner Basin (Lake County), and near Fox (Grant County). In eastern Washington, small numbers of greater sandhill cranes stage near Waukon and, to a lesser extent, Othello, along with flocks of lesser sandhills (Figure 14-4). In western Washington, a few greaters may migrate through the Puget Trough region, but there are no recent records.

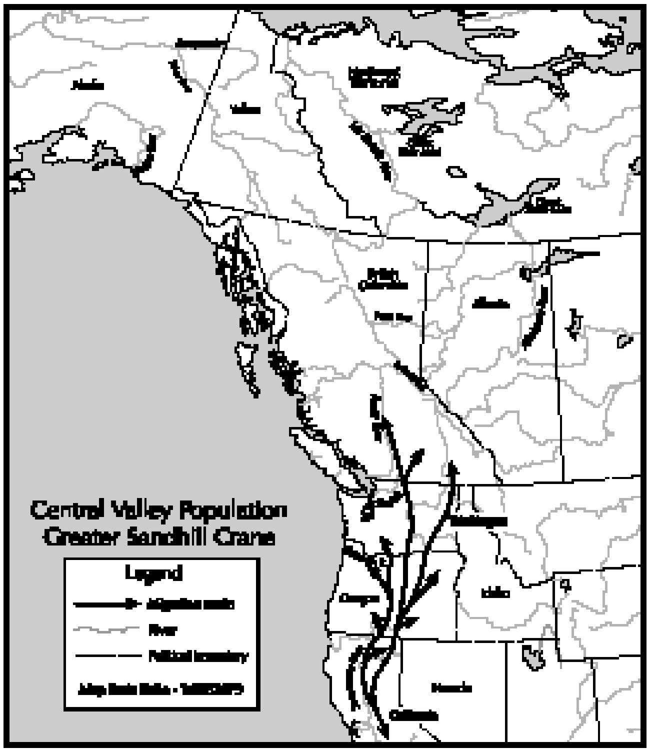

Figure 14-4. Migration routes of Central Valley Population of greater sandhill cranes (based on Littlefield and Thompson 1979, Campbell et al. 1990, and Pacific Flyway Council 1997).

The lesser and Canadian subspecies migrate through the state primarily from February–April. The Pacific Flyway lesser sandhills follow an inland route east of the Cascades en route to breeding grounds in Alaska at Cook Inlet, Bristol Bay, and the Alaska Peninsula (Figure 14-5). It is uncertain if any lessers migrate through western Washington, but no Canadian sandhill cranes have been identified using eastern Washington staging areas. Canadian sandhills migrate through western Washington apparently en route to scattered breeding sites along the coast of British Columbia and southeast Alaska (Figure 14-6). Canadians marked at Ridgefield NWR and Sauvie Island Wildlife Area flew out to the coastline, possibly following the Columbia River, and flew along the coast northward to Cape Flattery and the British Columbia and Alaska coasts (Ivey et al. in prep.). The number of sandhills recorded at Elma, Olympia, Montesano, and around Puget Sound suggest they often do not follow the Columbia, and occasionally travel through the Puget Trough.

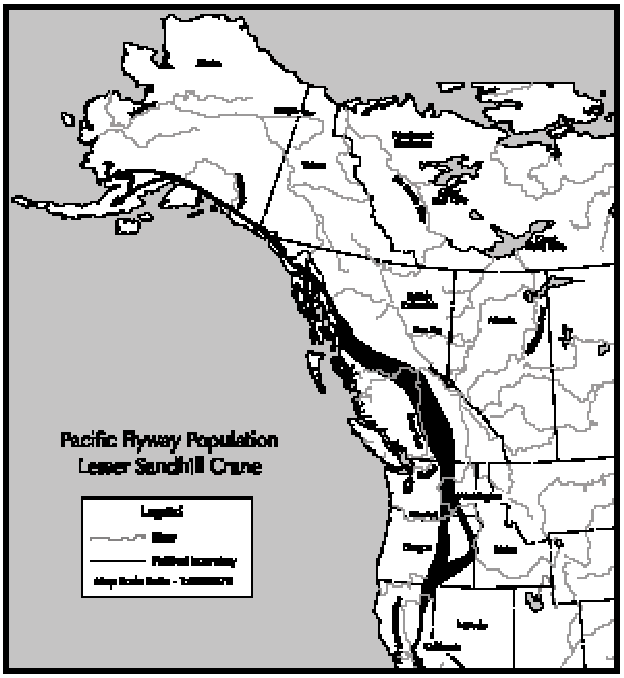

Figure 14-5. Migration route of the Pacific Flyway lesser sandhill cranes (based on Littlefield and Thompson 1982, Alaska Dept. Fish and Game 2001, Ivey *et al.* in prep.).

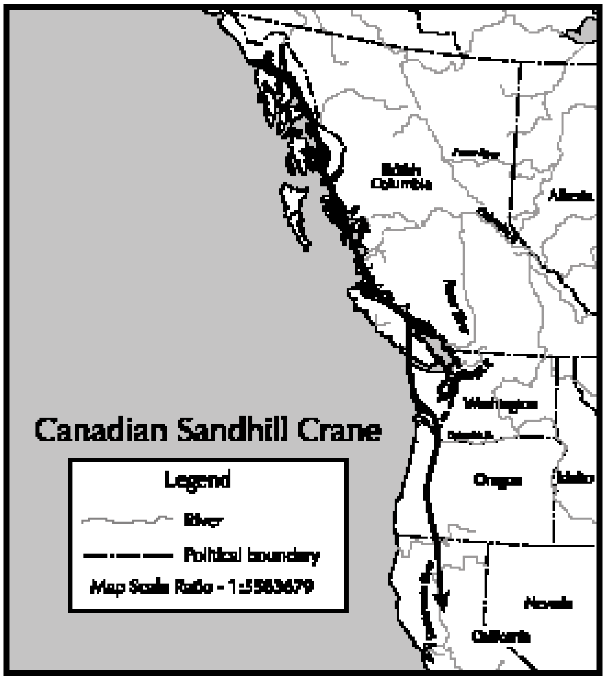

Figure 14-6. Migration route of Canadian sandhill cranes (based on Ivey *et al*. and records).

14.4.3.2 Autumn Migration

Migration from Conboy Lake NWR usually occurs between late September and mid-October (Engler and Anderson 1998). On September 29, 1998, two color-banded juveniles from Conboy were noted at Lower Klamath NWR, indicating the staging area for cranes which breed in Washington. Numbers at Lower Klamath NWR have increased from a peak of 425 on October 24, 1985 to 1,385 on October 28, 1998; in 2000, the peak was 1,188 on October 6 (J. Beckstrand, personal communication). Cranes begin staging in late August and peak numbers are present in mid- to late October. The increased use at Lower Klamath NWR perhaps reflects an increasing number of breeding pairs within the Cascades in Oregon and to some extent, Washington.

Other than the pre-migration aggregation of the local breeders and subadults at Glenwood Valley, there are no certain autumn records of greater sandhills for eastern Washington. Large flocks of lessers may contain some greaters, however, because greaters that breed in interior of British Columbia presumably migrate through the state.

The Canadian and lesser subspecies migrate through the state primarily in late September and October using the same general routes and staging areas as in the spring. Birds using the western portion of the state migrate south through the Willamette Valley, with some birds staging at Camas Swale in Lane County, Oregon, before moving south to California. Table 14-3 summarizes autumn counts of sandhill cranes at Sauvie Island and Ridgefield NWR. Although these data suggest an increasing trend, this may be due to refined survey efforts in recent years. The annual survey is affected by timing and water levels at the traditional roost sites (J. Engler, personal communication). Past efforts to visually differentiate between the three subspecies during these surveys were not very successful. In October 1973, 327 “large cranes” were recorded at Sauvie Island, along with 1,100 lessers, but a bird that was illegally killed there was identified as a Canadian (Littlefield and Thompson 1979). Also, during a recent effort to capture and mark cranes for a satellite telemetry study, no lessers were observed in late November 2001 or during March–April 2002 (G. Ivey, personal observation). The question of the status of lesser sandhill cranes in this region needs further study.

Table 14-3. Numbers of sandhill cranes recorded at Ridgefield NWR and Sauvie Island staging area in autumn, 1991–2000.

Date	Ridgefield National Wildlife Refuge[a]	Sauvie Island[b]	Total[c]
October 2, 1991	866	2,368	3,234
October 7, 1992	331	887	1,218
September 30, 1993	441	2,592	2,632
October 6, 1994	415	1,920	2,335
September 27, 1995	835	1,271	2,107
October 11, 1995	1,222	2,640	3,615
October 9, 1996	1,175	2,440	3,216
October 7, 1997	1,321	1,895	3,862
October 8, 1998	992	3,281	4,273
October 12, 1999	1,417	1,629	3,046
October 12, 2000	1,729	2,265	3,994
October 9, 2001	2,209	1,875	4,084

[a] Includes birds on Vancouver bottoms and Woodland area; [b] Includes nearby Oregon sites; [c] Numbers peak in the first half of October in most years. Apparent increases or changes in numbers may be due to refinements in survey effort or timing. Source: USFWS, unpubl. data

In the migration corridor to the east, Malheur NWR was the most important traditional autumn staging area for greaters in the Pacific states until the 1980s (Littlefield 1986). Cranes have arrived there as early as August 5 (1977), but birds believed to be from British Columbia generally do not appear until mid-September (Littlefield 1992). Peak numbers were usually present by mid-October, but if mild autumn weather persisted, and grain was abundant, the peak was delayed until early November. Autumn migration out of Malheur usually began in October, but cranes were seen departing as early as August 23 (1968). Normally the majority migrated between 1–15 November. Occasionally a few lingered into December, but normally all had migrated by the end of November; latest departures were December 10, 1947; December 20, 1951; December 31, 1961; December 11, 1965; and December 15, 1977. The mean departure date for 36 years was November 16. The greatest number ever recorded at Malheur was 3,408 on October 25, 1979 (Littlefield 1986).

14.4.3.3 Winter

The only wintering area for sandhill cranes in Washington is the lower Columbia bottomlands near Vancouver, Ridgefield, and Woodland. All cranes observed wintering at Ridgefield NWR and Sauvie Island Wildlife Area, Oregon, in late November 2001 and February 2002 were Canadian sandhills, and based on observations of marked birds, wintering cranes regularly move back and forth between these areas (Ivey et al. in prep.). Though not known to be a historical wintering area, an average of few hundred, but up to 1,000 cranes have wintered in the area during the last seven or eight years (J. Engler, personal communication).

Some cranes appear in the Central Valley in mid- to late September, but most arrive between mid-October and late November. The two principal wintering locations for greaters are the rice-growing regions of the Sacramento Valley and the corn-growing areas of the San Joaquin-Sacramento Delta. The distribution of wintering Canadian sandhills has not been described.

14.4.4 Foraging & Food

Sandhill cranes forage by probing, surface gleaning, and occasionally by spearing. Generally, the species can be categorized as an opportunistic omnivore (Armbruster 1987), feeding on a variety of food items including roots, bulbs, grains, berries, snails, earthworms, insects, amphibians, lizards, snakes, mice, and greens (Ridgway 1895, Barrows 1912, Bent 1926, Gabrielson and Jewett 1940, Brown 1942). Sandhill cranes also have been noted consuming eggs and young birds (Harvey et al. 1968, Littlefield 1976b, Reynolds 1985). In spring, cranes primarily eat macroinvertebrates, with insects (particularly scarab beetle larvae) being of most importance (Davis and Vohs 1993). Another dominant food, at least in portions of its breeding range, is earthworms. These food items are important sources for protein and calcium, nutrients needed for daily maintenance requirements (Reinecke and Krapu 1986). Such food items are essential, particularly on breeding grounds. The diet of greater sandhill cranes at Conboy Lake NWR has not been assessed, but may include Oregon spotted frogs (Rana pretiosa); eight territorial pairs nest or forage regularly at seven sites which are considered to be core areas for spotted frog breeding. The behavior of cranes foraging in pastures before nesting suggested that they were eating worms and beetles (J. Engler, personal communication).

In autumn and winter, sandhills feed on waste grains to help meet their high energy demands during migration and for survival through the winter period. Migrational staging sites are important for conditioning cranes for migration (Krapu et al. 1985; Krapu and Johnson 1990). Principal grains consumed are milo, corn, wheat, oats, barley, and rice (Swarth 1919, Wood 1921, McLean 1930, Tanner 1941, Munro 1950, Madsen 1967, Stephen 1967, Guthery 1972, Drewien and Bizeau 1974, Hoffman 1976, Crete and Toepfer 1978, Fritzell et al. 1979, Lewis 1979, Tebbel and Ankney 1979 Buller 1981, Iverson 1981, Perkins and Brown 1981, Lovvorn and Kirkpatrick 1982b, Iverson et al.1985, Littlefield 1986, Reinecke and Krapu 1986, Walker and Schemnitz 1987, Sugden and Clark 1988, Sugden et al. 1988). Cranes using the Ridgefield/ Sauvie Island area have been observed feeding on corn, barley, green grasses, and chufa (nutsedge) tubers (*Cyperus esculentus*) (G. Ivey, personal observation).

Littlefield (1986) described an autumn staging area at Malheur NWR where most feeding was in barley fields, but in some years oat, rye, and wheat fields were used when available. Though cranes showed no special preference between oat, rye, and barley, they did prefer wheat. Malheur NWR feeding fields ranged in size from 25-345 ac (10 to 138 ha), and birds concentrated in harvested areas (Littlefield 1986). In landscapes dominated by deep organic soils, grit may be a limiting factor, especially for cranes feeding predominately on waste grains (Littlefield and Ivey 2000).

Agriculture in the Sacramento Valley of California, where at least some of the Conboy Lake NWR cranes winter, is dominated by rice. Newly-planted winter wheat was second in importance, but use was of short duration; once seedlings emerged, cranes generally abandoned wheat fields. Though few corn hectares were present, waste corn accrued 7.5% of total use; waste corn, which is rich in carbohydrates, became increasingly important immediately before cranes migrated in February. Finally, 3.9% of the use was on cattle-grazed grasslands; these grasslands, however, were little used before the onset of winter rains. Few cranes were noted on other agricultural crops.

14.5 Habitat Requirements

14.5.1 Breeding

14.5.1.1 Territories

Primary components of a breeding territory are the nest site, roosting area, feeding area, and to some degree, isolation (Armbruster 1987). In the west, greater sandhill cranes occupy breeding territories in wetlands adjacent to riverine systems, closed drainage basins at the base of desert mountain ranges, and isolated mountain meadows. With a few exceptions, most pairs select sites rather isolated from human activity (Cooper 1996; G. Ivey, personal observation; Littlefield et al. 1994).

At Conboy Lake NWR, breeding territories include dry grass uplands, partially timbered uplands, emergent marshes, and wet meadows (Engler and Brady 2000). This prairie-like valley beneath the southeastern slope of Mt. Adams lies at an elevation of only 1,820 ft (555 m) but the influence from surrounding mountains makes the climate harsh. Valley topography is mostly level in this 9 mile-long (14 km) wetland basin. Historically, the water level in Conboy Lake remained high later into the season, and portions held more or less permanent water. Ditching and agricultural development in the early 1900s have speeded annual drying. Water now gradually recedes during early summer as Camas Ditch empties into Outlet Creek. Surrounding timbered uplands are predominately forested with ponderosa pine, Douglas fir, grand fir, and lodgepole pine, with some stands of Oregon white oak (H. Cole, personal communication; USFWS 1983).

14.5.1.2 Nesting Habitat

Generally, sandhill cranes require wetlands for nesting, will use a wide range of wetland classes and vegetation types, and occasionally will use uplands. Within the greater sandhill cranes' breeding range, nesting habitat varies from open meadows to deep water bogs and marshes (Armbruster 1987). At Conboy Lake NWR, 55% comprises wet meadows; where cranes nest, the vegetation includes reed canarygrass, rushes, sedges, and spikerushes. Portions of the lakebed are dominated by reed canarygrass (J. Engler, personal communication) (Paveglio and Kilbride 2000), but most areas are a mixture of canarygrass and native species. The prevalence of canarygrass and natives varies with weather and hydrology, but canarygrass often appears dominant because it is tall and later-growing than native species (J. Engler, personal communication). Some areas contain bulrushes and cattails, but these are less than 5% of the refuge area. Native grasses include redtop and foxtail barley.

Peripheral areas of these meadows (11%) are slightly to heavily encroached upon by lodgepole pine, Douglas' spirea, and willow which crane pairs use for both nesting substrate and cover. Approximately half of the crane pairs nest in areas with some trees and shrubs, but heavy encroachment by these species may preclude nesting cranes.

On Yakama Indian Nation lands, one pair nests in a meadow covering approximately 195 ac (79 ha) that is vegetated with willows, sedges, tufted hairgrass, and timber oatgrass. It is situated between stands of lodgepole pine, Douglas fir, and grand fir, with smaller amounts of ponderosa pine and western larch (Leach 1995). Portions of the meadow have standing water in spring and summer. A pair on WDNR land uses a small meadow.

Several studies have reported on nest habitat for crane pairs in California, Oregon, and British Columbia. In some areas, pairs nest in open, exposed meadows, whereas other nest preference sites are in dense, coarse emergents. Nesting habitat varies from open shallow-flooded meadows, to coarse emergents, seasonally flooded meadows (sedge/rush/grass), hardstem bulrush, and open water with little or no vegetative cover ((Littlefield 1995a, Littlefield 2001, Drew et al. 1994). Fifteen crane sites in the central-interior region of British Columbia were in sedge-dominated wetlands surrounded by coniferous forests with many bays and points of land; pairs have also been found nesting in heavily vegetated bulrush marshes surrounded by rangelands (Cooper 1996).

14.5.1.3 Nest Vegetation

Greater sandhill cranes will use a variety of vegetation types for nesting. At Conboy Lake NWR, nesting habitat has been characterized during occasional post-breeding season visits to nest locations (E. Anderson, personal communication). In 1996, nest vegetation was occularly estimated at five sites: nest composition ranged from predominantly reed canarygrass to entirely spirea.

In Oregon, crane nesting was studied at Malheur NWR in most years from 1966–98. In an early study of 111 nests (1966–67), broad-fruited burreed surrounded 61 nests (54%), hardstem bulrush 28 (25%), common cattail 11 (9.7%), and meadows 11 (9.7%)—90.3% of nests were in coarse emergents with few in open meadows. An additional 1,018 nests were assessed in 1969–89); as with the 1966–67 study, burreed and hardstem bulrush were used most extensively, with 76.8% (n = 782 nests). There was less use of cattail, rushes, grasses, sedges, and forbs. Nests among shrubs were a rarity (n = 4). Nest placement at 727 sites was in vegetation with a mean height of 37.3 cm (14.5 in) (range = 0-205 cm; 0-80 in). Distance from

515 nest sites to the nearest feeding meadow averaged 131 ft (40 m; range 0-1,132 ft or 0-345 m) (Littlefield 2001).

Elsewhere in eastern Oregon, 54 nests on privately owned wetlands in Harney County were primarily on open, cattle-grazed meadows (40 of 52; 77%). Eight (15%) were in burreed, two (4%) in hardstem bulrush, one (2%) on a non-vegetated island, and one (2%) in flooded greasewood. Vegetation height ranged from 0-50 cm (0-20 in). On privately-owned lands in the Blue Mountains, seven of nine nests were in meadows, one in a beaver pond among a stand of beaked sedge, and another in a small saltgrass basin (Littlefield 1999b).

14.5.1.4 Water Depths at Nest Sites

Water depth data were not available for Washington nests, as sites have not been visited while birds were incubating. At 881 nests at Malheur NWR, water depth averaged 25.8 cm (10 in) (range = 0-105 cm; 0-41 in) and 34 were on dry sites (Littlefield 2001); at 54 nests on privately-owned wetlands in the Great Basin portion of Harney County, water depth ranged from 0-23.6 cm (9.2 in); and on privately-owned lands in the Blue Mountains, depths were 8.5-15 cm (3.3 - 5.9 in) (Littlefield 1999b). At Sycan Marsh, nests situated in hardstem bulrush were in 40-60 cm (15.6-23.4 in) (mean = 50.3 cm; 19.6 in) of water, whereas for nests in wet and dry meadow habitats, depths ranged from 0-30 cm (0-11.7 in) (Stern et al. 1987). At Klamath Marsh NWR in 1993, water depths at nest sites averaged 13.1 cm (5.1 in) in meadows, compared to 41 cm (16 in) in bulrush; average depth at all sites was 24.8 cm (9.7 in), and for 13 nests assessed in 1994, depths averaged 18.4 cm (7.2 in) and ranged from 2-36.2 cm (0.8-14.1 in) (Drew et al. 1994).

14.5.1.5 Roost Sites

Once young fledge, families join with unsuccessful pairs, yearlings, and subadults at communal roosting sites until migrating south. Cranes usually roost by standing in open water where little emergent vegetation is present.

14.5.2 Wintering & Staging Areas

14.5.2.1 Foraging Habitats

Cranes feed in a variety of habitats; security from disturbance and tradition are key factors in selection of areas during migration and winter. Birds generally concentrate in agricultural regions with extensive areas of small grain crops. However, associated wetlands are still used for some feeding, as well as for nighttime roosting and midday loafing (Littlefield and Ivey 2000). Cranes usually leave roosting locations in the early morning and fly to nearby grain fields, where they feed until mid-morning. In midday, birds occasionally feed in pastures, alfalfa fields, along canals, ditches, and dikes, or use shorelines and pond, lake, and other wetland shallows where they may obtain essential amino acids and minerals not present in grains (Reinecke and Krapu 1979). In mid-afternoon, most return to grainfields where they feed until early evening before returning to roost sites (Littlefield and Ivey 2000). At Ridgefield NWR, sandhill cranes use areas with agricultural crops, pasturelands, hayfields, and wetlands (Littlefield 1999a).

14.5.2.2 Night Roosts & Loafing Areas

Sandhill cranes migrating and staging within the lower Columbia River roost on the Ridgefield NWR and on Sauvie Island, Oregon. Those using the refuge roost primarily on

Campbell Lake, a large shallow lake connected to the Columbia River by a slough. Water levels in the lake rise and fall with the river levels. Depending on the year and season, extensive mudflats and bars are exposed providing considerable roosting habitat. Roosting also occurs in the shallow waters of the lake. During high water events, cranes are known to abandon this roost. Vegetation of the lake is primarily aquatic submergents, but low to tall emergents line the lake edges. Cranes also roost in small numbers on shallow managed units of Bachelor Island, the River 'S,' and Carty units when water levels are low and/or management practices have reduced the emergent vegetative cover and provided shallow mudflats. Cranes also have been observed roosting on Post Office Lake and a few small seasonal pools created by Campbell Slough backwaters. These latter sites are open with low vegetation, but not available every year. Post Office Lake lies adjacent to a dead-end county road and use is probably limited by traffic. None of these roost sites, other than Campbell Lake, is consistently suitable because their water and vegetative condition fluctuates annually (J. Engler, personal communication). Cranes also roost on nearby Sauvie Island, particularly at Sturgeon Lake.

14.5.3 Population Status

14.5.3.1 Past

North America

Historically, greater sandhill cranes occupied a larger range than they do today. In colonial times, the subspecies commonly occurred east to the Atlantic seaboard, at least in migration, but by the early 1800s, their numbers had been greatly reduced. Numbers declined dramatically between 1870–1915, as increasing human populations hunted birds, drained wetlands, and built over nesting habitat (Walkinshaw 1949). Similar to eastern North America, western populations decreased in the late 1800s and early 1900s. Cranes were extirpated from Arizona by 1910 (Bailey 1928) and from Washington by 1942. By the early 1940s, cranes were only nesting sparingly in Nevada, Utah, Idaho, Montana, and Wyoming. Walkinshaw (1949) estimated only 1,339 to 1,836 greaters left in the United States in 1944. Little is known about the historic range of lesser and Canadian sandhill cranes.

Washington

As in the rest of the United States, the historic distribution of sandhill crane subspecies in Washington is clouded and somewhat confusing. Most early 20th century ornithologists were reluctant to accept subspecies crane accounts without specimen evidence. This reluctance has resulted in gaps concerning the true historic subspecific status for sandhills throughout the state. Greaters did occur in western Washington, at least as migrants, as one was collected by Suckley in present-day Pierce County in October 1853 (Baird et al. 1860). This specimen in at the US National Museum in Washington, DC is the only historical greater sandhill crane specimen for the state (Jewett et al. 1953).

The historical status of breeding greater sandhill cranes in Washington also was poorly documented. Although the evidence of breeding in western Washington is meager, they apparently nested in at least small numbers. Though there may have been some confusion on subspecific identity, George Suckley in the 1850s reported for spring and summer: "In the vicinity of Fort Steilacoom, only stragglers remain to breed," and James Cooper observed:

> *. . . a common summer resident arriving at the Straits of Juan de Fuca in large flocks in April and then dispersing in pairs over the interior prairies to build their nest, which are placed amid tall ferns on the highest and most open ground, where they can*

> *see the approach of danger. They frequent, at this season, the mountains to a height of 6,000 feet above sea level* (Suckley and Cooper 1860:227-228).

Dawson and Bowles (1909) listed the greater as a “not common summer resident both sides of the Cascades” (p. 620) and suggested that sandhill cranes are found “in mountain meadows of both the Cascade and Olympic Mountains, and upon the lesser prairies which dot the western forest. . . .” (p.621).

Among other locales, breeding cranes apparently occurred at Camas Prairie and Dallesport—both in Klickitat County) (Jewett et. al 1953). It is doubtful breeding cranes historically occupied the Columbia Plateau lowlands, as high summer temperatures and early seasonal drying would have perhaps precluded successful reproduction (R. Friesz, personal communication). The last historical nesting record was in 1941 near Signal Peak, on Yakama Indian Nation lands (Jewett et al. 1953) on a small brush-covered island at an elevation of 4,500 feet. This site is apparently the same location where a crane pair re-established in 1991 (Leach 1995).

Historical migration accounts are limited because of the lack of specimen evidence. Bent (1926) listed earliest spring arrival dates for Camas in Clark County as March 26, 1923, and Dallesport as April 27, 1924 (Jewett et al. 1953).

Yocom and Hansen (1958) described spring crane migrations in 1950 and 1951, noting that flocks of cranes were observed leaving the state by flying up the Okanogan River and the Columbia River valleys.

14.5.3.2 Present

North America

After their near extermination in the 19th and early 20th centuries, it has been a slow recovery process for the greater sandhill crane. Even with complete protection after 1916, crane numbers did not begin to rebound until the mid-1940s (Peterjohn 1989). Populations began to increase primarily due to: 1) development of efficient predator control methods for the livestock industry in the west, 2) protection from market hunting with enactment of the Migratory Bird Treaty Act in 1916, 3) development of flood-irrigated meadows for cattle forage which increased available habitat. However, since cranes have traditionally been considered a game species by some, hunting seasons have been proposed and initiated, supposedly to relieve agricultural crop depredation complaints. Greaters of the Rocky Mountain Population, for example, have been hunted since 1981; of 135 recoveries of color-banded birds, 96 were killed by hunters (Drewien et al. in prep.). This, coupled with a continually increasing human population, will perhaps threaten crane populations far into the future.

In the mid-1980s, the Central Valley Population of large sandhills was estimated to total 6,000-6,800; this included at least 839 Canadian sandhills (Pogson and Lindstedt 1991). The Pacific Flyway Population of lesser sandhill cranes is thought to be approximately 23,000 birds (Kramer et al. 1983).

Washington

After 1941, some 31 years lapsed before summering greater sandhill cranes were again found in Washington. The subspecies’ return apparently began in 1972 when two appeared at Conboy Lake NWR in September, remaining into late November. In 1996, eight out of ten pairs were known to nest at Conboy, two pairs were known on Yakama Indian Nation lands, and a pair was found in Panakanic Valley on private lands (nesting was confirmed in 1997). By 2000, the state’s known greater sandhill crane population was 53 birds, consisting of 19 pairs (15 known nesting), 9 subadults, and 6 fledged young (Engler and Brady 2000). No chicks were known to survive to fledging in 2001, probably due to factors related to drought conditions; only about 20-25% of the wetlands typically available at Conboy Lake were present (Engler and McFall 2001).

For the period 1990–2001, Washington’s breeding population fledged 30 chicks, with successful reproduction in all years except 1993, 1994, and perhaps 2001 (The greatest number was six in 2000, while five chicks fledged annually during the three previous years.)

Table 14-4. Greater sandhill crane pairs, productivity, and total population estimate in Washington (1990–2000)[1] .

Year	No. Breeding Pairs			Total Breeding Adults	Subadults (known)	# Young Fledged	Recruitment[3](%)	WA Population Estimate
	Conboy Lake NWR	YIN[2]	Private & WDNR					
1990	3	—	—	6	—	1	14.3	7
1991	3	(1)[4]	—	8	—	1	11.1	9
1992	3	(1)[4]	—	8	—	3	27.3	11
1993	3	(1)[4]	—	8	—	0	0	8
1994	3	1	—	8	—	0	0	8
1995	7 (2)	1 (1)	—	22	0	1	4.3	23
1996	8 (2)	2	(1)	26	0	3	10.3	29
1997	12	2	1	30	4	5	14.3	39
1998	14	(2)	(1)	34	5	5	12.8	44
1999	13 (1)	1 (1)	2	36	4	5	12.2	45
2000	13 (3)	1	1 (1)	38	9	6	13.6	53
2001[5]	14 (2)	(1)	1 (2)	40	10	0	0	50

[1] Data includes confirmed nesting pairs, unconfirmed pairs, and subadults. Data in parenthesis represent territorial pairs without confirmed nesting data; 1990–94 data is based on incidental observations (from Engler and Brady 2000). Systematic surveys of breeding cranes began in 1995.

[2] YIN = Yakama Indian Nation lands.

[3] Recruitment = no. fledged young / no. of breeding adults + fledged young x 100 (excludes subadults).

[4] Leach (1995).

[5] Drought conditions in 2001 negatively affected production; 1 pair was assumed to be present on the YIN which was not surveyed (Engler and McFall 2001).

Other Central Valley Population Range

Beginning in the mid-1940s, Central Valley Population greater sandhill crane pairs began to increase as efficient predator control methods were devised for livestock protection; indirectly this had a positive impact on cranes, as reproductive success increased (Littlefield 1976a). The beginning of crane recovery corresponded closely with the introduction of Compound 1080 (sodium fluoroacetate), a poison used extensively for coyote control throughout much of the western United States between 1944–72 (Littlefield 1995d). Also, several large deep-water marshes, formerly unsuitable for crane nesting, were drained, developed, and irrigated for livestock forage. This meadow development provided new habitat for breeding pairs (Littlefield and Thompson 1979). In recent years, wildlife management programs that historically dealt almost exclusively with hunted species have been broadened to include non-game species, including sandhill cranes. These three factors—plus protected status—have resulted in an increase and subsequent re-occupation of breeding range left vacant for several decades.

14.6 Habitat Status

14.6.1 Breeding Habitat

Sandhill crane breeding habitat is somewhat limited in Washington, when compared with the large wetland complexes found in southeastern and south-central Oregon and northeastern California. However, Glenwood Valley has potential for becoming a more important summer crane use-area. On private and federal lands, habitat is available to accommodate an increasing and expanding population (D. Anderson, personal communication); however, currently there are limitations on quality of habitat. Wetlands in Glenwood Valley are comparable to other mountainous locations where many cranes breed. Sycan Marsh, Oregon, and Grays Lake NWR, Idaho, are similar areas which support high densities of breeding cranes. There are approximately 15,000 ac (6,070 ha) of potential crane habitat in the Glenwood Valley, but this includes about 5,000 ac (3,035 ha) of private irrigated pastures near Glenwood where land use practices reduce the suitability to cranes. Since Conboy Lake NWR was established in 1964, 5,814 ac (2,353 ha) have been acquired by the USFWS (H. Cole, personal communication) and an additional 3,522 ac (1,409 ha) are proposed for acquisition (USFWS 1983). If Conboy Lake NWR were managed specifically for cranes, it could perhaps accommodate 50 to 75 pairs (C. Littlefield and Steve Thompson, memo to Refuge Manager, Lower Columbia River Complex, Vancouver, Washington, dated December 26, 1984). This number is not likely to be realized, however, given the current conditions and water issues in the valley. Breeding pairs have increased from one in 1984 to 16 in 2000 and if favorable management practices and environmental conditions continue, crane pairs should continue to increase and eventually disperse onto nearby sites.

Outside the valley, there is generally no immediate threat to the wetlands where cranes presently breed other than summer livestock grazing on both tribal and privately-owned lands (D. Anderson, H. Cole, and R. Leach, personal communications). Potential threats include drainage, trespass grazing, and property sales and subsequent development. No cranes were observed by helicopter at the Camas Patch site on June 9, 2000, and the area was dry and being grazed and may no longer be suitable breeding habitat (Engler and Brady 2000). The Polo Field site on Yakama Indian Nation lands is located within a grazing unit, but cattle generally do not reach the site until after July 15; a 20-meter no-entry, no-logging buffer zone surrounds the meadow, but there were about four log-truck trips per day on a nearby closed road in 1994 (Leach 1995).

Other potential greater sandhill crane breeding habitat that appears to be suitable includes: 1) Colville Tribal lands (Okanogan County), particularly at Moses Meadows (M. Murphy, personal communication); 2) isolated meadows near the Pend Oreille River (Pend Oreille County) (D. Friesz and S. Zender, personal communications); 3) large hardstem bulrush marshes on Turnbull NWR (M. Rule, personal communication); and 4) a series of high Cascade meadows 10-12 mi (16-19 km) north of Mt. Adams in the Two Lakes area (Yakima County); a single crane was observed at the latter site several years ago, but there was no evidence of nesting (H. Cole, personal communication). Cranes have also been sighted, and may nest at Trout Lake Natural Area Preserve, a 920-acre (327 ha) wetlands complex in Klickitat County recently acquired by WDNR. Several other summer crane records since 1980 may have been subadults seeking a territory (see Table 14-2). The most recent was a bird that summered on Tiger Meadow in Pend Oreille County in 2001.

On Colville Tribal lands in Okanogan County, no summer cranes have been found (M. Murphy, personal communication), but there are isolated remote wetlands with limited human access where cranes might nest (M. Monda, personal communication). Other than possible disturbance from livestock grazing and logging, meadow habitat within the 1,417,000 ac (566,800 ha) reservation seems to be well protected. There are also apparently favorable and secure meadows in the Pend Oreille Valley, particularly at Cusick Flat (Pend Oreille County); however, there have been very few recent summer crane records for Pend Oreille, Ferry, or Stevens counties (S. Zender, personal communication). Additionally, potential nesting habitat exists at Turnbull NWR. The refuge contains a number of semi-permanent and permanent wetlands in depressions, some which are suitable for crane territories, but most are surrounded by steep banks and basalt cliffs and not suitable for crane territories (Monda and Ratti 1988). Northeast of Turnbull NWR, most of the suitable wetlands around Spokane have been lost because of residential housing, powerline corridors, gravel mines, and encroachment by forest (McAllister 1995).

The high mountain wetlands of the Cascade Range would perhaps provide substantial habitat for breeding sandhill cranes, and isolated sedge meadows occur in the Okanogan Highlands as well (J. Ball, personal communication); however, snow frequently lingers well into June. Thus, in most years there might be insufficient time for cranes to successfully reproduce. However, if global climate change lengthens summers, these wetlands may eventually become suitable. Crane pairs have been expanding and successfully reproducing in mountainous situations at more southerly latitudes in Oregon and northeastern California.

Several sites were previously used by breeding cranes but are no longer suitable habitat. The nesting site near Calispell may have been inundated behind Calispell Dam; Matt Monda personal communication) reported that waterfowl studies have been in progress for a number of years, but there have been no reports of cranes in this area. At Oroville where summer cranes were last reported in 1922, the area presently consists of orchards and grain farms with some wetlands; however, during 40 years of waterfowl surveys, summering cranes have not been observed in this region (M. Monda, personal communication). Further south in the Columbia River Plateau region, if habitat ever existed, it would have perhaps been lost when the upper Grand Coulee was flooded by the filling of Banks Lake, an equalizing reservoir between Coulee Dam and Coulee City, in the spring of 1951 (Yocom and Hansen 1960).

14.6.2 Wintering & Staging Habitat

14.6.2.1 Lower Columbia Bottomlands

The lower Columbia bottomlands staging area is the only sandhill crane use-area in the United States adjacent to a major metropolitan area, and habitat will continue to be threatened. About 4,000 cranes stop during migration, and up to 1,000 winter in the area. Few, if any, alternate migrational stopover sites are available between northern California and southeastern Alaska for birds which migrate west of the Cascade Range. Habitat in the area needs to be protected if this crane flock is to continue to survive. A total of 7,518 ac (3,044 h) are owned by wildlife agencies and protected from development, but several thousand acres of habitat have no conservation status. USFWS owns 5,150 ac at Ridgefield NWR, and WDFW owns 2,371 ac at Shillapoo Wildlife Area. In addition, 416 ac adjacent to Vancouver Lake are owned by Vancouver/Clark County Parks.

Presently, about 70% of Shillapoo Wildlife Area is used as pasture or agriculture lands (35% each). Pheasant releases at two sites result in high hunter use of some agricultural fields and pheasant season coincides with the fall peak of crane migration. Crane use of otherwise suitable habitat is reduced by the presence of hunters during upland bird and waterfowl seasons, and people training dogs at other times. Future plans include restoration of Shillapoo Lake (900 ac) that would flood agriculture fields and pastures and restore native wetland vegetation. About 950 ac of pasture and cropland will remain for geese and cranes. The effects on cranes of this change are not clear. Some seasonal foraging area for migrants may be lost, but roosting sites and native foods may increase.

The Port of Vancouver owns the 1,100 acre-Columbia Gateway property. It is agricultural, woodland, and wetland, and perhaps 75% receives a high level of use by cranes. The Port has prepared a master plan calling for development of >700 acres for industry and port facilities (Port of Vancouver 1998). The development would use fill, including dredged material from deepening of the Columbia River navigation channel by the ACOE, to raise the area above seasonal flooding.

Other habitat losses in this region are anticipated. Former row-crop agricultural land on Sauvie Island has recently been converted to tree nurseries (M. Stern, personal communication). Additional agricultural lands on Sauvie and Woodland bottoms have been planted to cottonwood plantations. Other uses that have been responsible for incremental losses of crane habitat include tulip production, berry crops, smaller industrial developments, residential development, and public recreational development.

14.6.2.2 Eastern Washington

Sandhill cranes use agricultural fields and wetlands for staging at several locations in eastern Washington, including the Columbia NWR (23,200 ac) and Potholes Reservoir Wildlife Area (32,500 ac). Cranes have staged on the Waterville Plateau in the Mansfield/St. Andrews area for many years (R. Friesz, personal communication).

14.6.2.3 Breeding Habitat

Crane breeding habitat in Oregon and California is under threat from development and incompatible management practices. Habitat is threatened by late irrigation, the presence of cattle on meadows until late spring, draining of wetlands, pivot irrigation replacing flood-irrigated meadows, houses and alfalfa fields encroaching on historic territories, and loss of irrigation rights (Littlefield and Thompson 1979, Littlefield 1989, Ivey and Herziger 2000, 2001).

14.6.2.4 Staging & Wintering Habitat

On the wintering grounds in the Central Valley, agricultural lands traditionally used by cranes are being lost to urban expansion, as well as conversion to incompatible crops such as vineyards and orchards (Littlefield and Ivey 2000).

14.7 Conservation Status

The sandhill crane was first granted federal legal protection under the Migratory Bird Act of 1916. Presently, the species, its nests, and its eggs are protected from unlawful direct persecution in Canada and the United States under the Migratory Birds Convention Act of 1994 which prohibits the killing, capturing, injuring, taking, or disturbing of migratory birds, or damaging, destroying, removing, or disturbing of nests. It also prescribes protected areas for migratory birds and nests, and for the control and management of those areas. The Central Valley population is not subject to legal harvest during hunting seasons, as are several other sandhill crane populations (Tacha et al. 1992).

14.7.1 Washington

The Washington Department of Game (the predecessor to WDFW) listed the sandhill crane as endangered in 1981 (WAC 232-12-014; see also WAC 232-12-297, Appendix A). Bettinger and Milner (2000) reported that sandhill cranes were in jeopardy in Washington because of their limited distribution, low numbers, poor breeding success and chick survival (in general throughout their range), and loss of shallow marshes and wet meadows for feeding and nesting. The Revised Code of Washington (RCW) prohibits the sale, possession, exchange, buying, transport, or shipping of articles made from an endangered species. Though all Washington sandhill subspecies are included under this classification, major emphasis has been placed on greater sandhill cranes.

Sandhill cranes are also listed on the WDFW's Priority Habitats and Species List. Crane habitats—breeding areas, regular large concentrations, and migration staging areas—are also listed. Crane habitat is not explicitly protected by state law, but as habitat of a state endangered species, it would be protected by ordinance in many counties. Under the state GMA, counties are required to identify critical areas and can also select species of local significance. Many counties have adopted the state's list of endangered, threatened, and sensitive species, and require review and mitigation before issuing permits for projects that would impact habitat.

Under the Washington Forest Practices Act, sandhill cranes and their habitat also are protected. In particular, timber harvest, road construction, aerial application of pesticides, and site preparation are restricted within 1/4 mile (0.4 km) of a known active nesting area.

On tribal lands, the Yakama Indian Nation has listed the greater sandhill crane as a sensitive species in the Yakama Indian Reservation Forest Management Plan (Bureau of Indian Affairs 1993), and it is considered a species of cultural importance (R. Leach, personal communication). In habitat management guidelines written by the wildlife program of the tribe (Leach et al. 1992), recommendations are to survey for cranes when activities are planned near large wet meadows, and if they are found breeding, a 1/2 mile (0.8 km) no-entry buffer around the meadows should be designated during the breeding season (March–October), and road construction should be avoided within 1/2 mile (0.8 km) of the meadow.

14.8 Factors Affecting Continued Existence

14.8.1 Breeding Areas

14.8.1.1 Predation

A major mortality factor that confronts cranes on the breeding grounds is predation on eggs and chicks. An abundance of predators can reduce crane reproductive success; for example, at Malheur NWR in both 1973 and 1974, only two young fledged from 235 pairs (Littlefield 1976a). Though other predators prey on crane eggs and chicks, common ravens, minks, raccoons, and especially coyotes are the most destructive, and under certain conditions can be highly detrimental to sandhill crane productivity. Coyotes are thought to be the primary predator of crane chicks at Conboy Lake NWR (Engler and Brady 2000). High predation rates are particularly evident at large breeding locales such as Malheur NWR and Sycan Marsh, Oregon; reasons for this are unclear but may reflect relatively recent changes in the balance of predator and prey populations in the region. The ban on the use of Compound 1080 may have contributed to an increase in coyotes and ravens, the principal nest predators, and these higher numbers have been responsible for low annual recruitment in some areas. Why this effect would be more pronounced on the large wetland complexes is uncertain, but these sites generally support relatively high densities of nesting waterfowl, thus perhaps predator populations occur in greater densities than on smaller wetlands. Additionally, many of the smaller areas are privately-owned and local efforts to control coyotes may effectively reduce predation (Littlefield et al. 1994).

14.8.1.2 Grazing & Haying

In spring, sandhill cranes generally prefer to forage in open, flooded meadows. Frequently these sites are the result of mowing and livestock grazing practices which can be detrimental to nesting and fledging. Though meadows are generally good foraging sites for cranes, late June and July meadow mowing can kill crane chicks as they hide in dense vegetation and remain motionless, waiting for the threat to pass (Littlefield and Ivey 1994). In addition, meadows are often dried in June for hay harvest, and early drying can result in the unavailability of invertebrate foods, sometimes contributing to chick starvation. Winter livestock grazing of wetlands generally removes residual cover, leaving crane nests exposed to predators in April and May. At Malheur NWR, nest success in the absence of predator control was significantly lower in wetlands winter grazed by cattle than in wetlands not grazed (Littlefield and Paullin 1990). Spring grazing can also be detrimental to nesting success; grazing from April 10–July 15 can prevent nesting attempts, and in some cases, cause nest abandonment (Littlefield 1989). Cattle have trampled crane chicks in northeast California (R. Johnstone, personal communication) and Idaho (R. Drewien, personal communication).

Management of lands for cranes could be improved by excluding livestock from crane habitat during the spring breeding season, delaying hay harvest and grazing until after 10 August, and limiting human disturbance to nesting cranes.

14.8.1.3 Water Availability

Because cranes are dependent on wetlands, they are vulnerable to changes in hydrology. Water rights are an issue in some areas, and loss of irrigation rights could eliminate existing habitat for cranes (Ivey and Herziger 2000). Irrigation timing is also important, as cranes should have water applied to their territories by mid-March to prepare for April nesting; water should be maintained through the brooding period (early August). Historical sandhill crane pairs were absent from some sites surveyed in Oregon and California where irrigation was delayed (Ivey and Herziger 2000, 2001). Early drying of wetlands and irrigated fields can lead to increased chick mortality.

14.8.1.4 Habitat Loss

Most crane pair territories in Washington are on protected lands, primarily those managed by the USFWS, but also by the Yakama Indian Nation and the WDNR. However, in the other Pacific states, cranes nest mostly on unprotected, privately owned wetlands. During surveys in 1999 and 2000, 63% of 1,616 pairs found in California and Oregon were on private lands (Ivey and Herziger 2000, 2001). Such a large percentage of pairs using private land is reason for concern because harmful management practices such as late irrigation and the presence of cattle on meadows until late spring could eliminate crane pairs. Loss of habitat has also displaced breeding pairs (Littlefield and Thompson 1979, Littlefield 1989, Ivey and Herziger 2000, 2001).

At Conboy Lake NWR, development of wetland impoundments could displace cranes and reduce the amount of available crane habitat; however, if carefully planned, impoundments may enhance habitat conditions for breeding cranes. Therefore, a habitat development plan for

Conboy Lake NWR should carefully consider the locations of any new impoundments in the context of enhancing crane breeding habitat.

14.8.2 Staging & Wintering Areas

14.8.2.1 Availability of Winter Foraging Habitat

Numbers of cranes at Ridgefield have increased, which indicates either increased production on the breeding grounds or a redistribution of cranes using the Lower Columbia River habitats.

It seems likely that the amount of migration/wintering habitat is limited for Canadian Sandhill Cranes. Migrating and wintering sandhills forage in cropland on waste grain in agricultural fields, pastures and in wet meadows and shallow marshes. The amount of protected habitat in the lower Columbia River subbasin in Washington consists of a total of 7518 acres (3044 ha), of which 5150 acres are within Ridgefield NWR, and 2371 acres at Shillapoo Wildlife Area (Washington Dept. Fish and Wildlife). About 70% of Shillapoo WA is in pasture or agricultural fields; future plans include restoration of Shillapoo Lake to benefit waterfowl, which may reduce crane foraging habitat, but could increase potential crane roosting habitat depending on levels of disturbance. About 950 acres of pasture and cropland would remain for geese and crane foraging. Whether these acreages are bioenergetically sufficient to sustain the numerical crane objectives is unclear. Wintering populationsof Canada geese also use the areas for feeding. In addition, 416 acres adjacent to Vancouver Lake are owned by Vancouver/Clark County Parks, and the Port of Vancouver owns 1,011 acres (Columbia Gateway), a portion of which will be included as habitat mitigation for cranes.

Thousands of additional acres have no conservation status, and habitat losses are anticipated. Potential or actual crane habitat on the lower Columbia River bottomlands between Vancouver and Woodland is threatened with industrial and residential development. Aditionally, agricultural lands have been planted to cottonwood plantations and other crops (berries, tree nurseries, tulips, small industrial, residential, etc.), that are incompatible with cranes. In Washington, counties require that potential impacts to the habitats of endangered species be reviewed. Mittigation may be required concurrent to development, and so some level of protection may result.

The availability and loss of foraging habitat (including waste grain fields) is probably the most important specific limiting factor factor for migrating and wintering sandhill cranes. This would include conversion of grain crops managed for waterfowl to other waterfowl forage, such as in moist soil management.

Changes in Farming Practices

In the Vancouver to Woodland bottomlands, the availability of corn may be affected by the status of the local dairy industry. The number of dairy farms in the area has been declining. Planting of crops on state wildlife area lands depends on lessees because public funding is rarely available. On the Shillapoo Wildlife Area, WDFW plans to restore Shillapoo Lake, which will flood some pasture and agricultural fields, but 950 ac (385 ha) of agricultural/pasture lands will remain for geese and cranes. Corn planted on Ridgefield NWR (100 acres in recent years) has helped compensate for losses on state and private lands.

Farming practices after harvest frequently determine the amount of waste seed available for wintering sandhill cranes. For example, in the northern Central Valley in the early 1990s, 71.4% of crane feeding use was in harvested rice fields, of which 59.3% was in unaltered rice stubble, 16.2% in flooded stubble, and 14.4% in burned stubble (Littlefield 1993a). Autumn-tilled rice stubble had infrequent use (3.3%), as did burned-flooded (5.6%) and tilled-flooded (0.3%). Thus, practices on harvested grainfields can have a serious impact on food availability.

Waterfowl Enhancement & Mitigation Practices

Programs intended to improve habitat for waterfowl can have negative effects on sandhill crane foraging habitat. Flooded grainfields are generally avoided by cranes, except for infrequent use for roosting and loafing. Dissimilar to ducks and geese, feeding cranes visually surface-glean seeds, and are highly inefficient in finding small unexposed seeds; generally, it is only a short time before cranes abandon a grain field after flooding. As most grain types have declined in the northern Central Valley, rice production has been maintained, though not at the levels planted in the early 1980s. However, the Agricultural Waterfowl Incentive Program is designed to enhance waterfowl habitat by providing seeds, tubers, graze and invertebrates. In 1998, 49 landowners participated to create 38,949 ac (15,769 ha) of waterfowl habitat, a 75% increase from the proceeding year. Enrolled landowners were predominantly rice producers in the northern Central Valley, with only one elsewhere (Garrison 1999). Much of this flooding is in addition to the 60,021 ac (24,300 ha) already being flooded before the program was initiated; thousands of acres have been lost to cranes as foraging sites, and additional fields are expected to be lost in the future. Should this program continue to gain momentum, it will have a negative impact on the remaining winter food resources available to cranes wintering in the Central Valley (Littlefield 1999a).

Wetland mitigation projects often focus on creating habitat that may not be suitable for sandhill crane use. Mitigation and other wetland projects in crane wintering and staging areas should be planned to provide sandhill crane foraging and loafing habitats in addition to waterfowl and other wetland goals. Proposals to mitigate wetland filling associated with Port of Vancouver development are focused on open-water habitat for ducks and geese, although a portion of this mitigation effort will also be directed towards providing crane forage habitat.

14.8.2.2 Availability of Roosting Habitat

For roosting, secure habitat is needed in the form of shallow lakes, wet meadows, and/or occasionally agricultural fields. Some of the considerations in the subsection above on foraging habitat also apply to roosting habitat. Roosting cranes are extremely vulnerable to disturbance.

In southwestern Washington and northeastern Oregon, activities to reduce Canada goose depredation of crops with hazing, propane cannons, extended hunts, dogs, field flags, and other scaring devices, have also effectively reduced usable wintering/migration habitat on private lands (E. Anderson, personal communication).

Recreational use, such as hunting, dog training, bird watching, hiking and jogging, is allowed on some wildlife refuges and management areas such as the Shillapoo Wildlife Area, Sauvie Island Wildlife Area and at the Ridgefield National Wildlife Refuge (Littlefied and Ivey 2002). These activities reduce the area's usefulness to sandhill cranes for foraging and sanctuary. Disturbances on managed wildlife areas also move cranes from public lands to private agricultural lands, where they may cause depredations.

While there are currently a number of refugia for wintering cranes, there is also an increasing recreational pressure on these areas and indeed all areas, which may become a limiting factor in the future. A bike-path/jogging trail recently built on Port of Vancouver property threatens a crane use area. Disturbance at daytime and night-time roosts is particularly problematic.

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Volume III, Chapter 15
Red-eyed Vireo

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15.0 Red-eyed Vireo (*Vireo olivaceus*)

15.1 Introduction

Over the past several years, songbirds and the reasons for declines in their populations have been a focal point of interest. Many species of neotropical songbirds birds have experienced population declines due to losses and fragmentation of breeding, wintering, and migratory habitats. These long-distance migrants tend to be more vulnerable to habitat loss and fragmentation than resident birds or those that migrate only short distances within North America.

At least 49 neotropical bird species are highly associated with riparian forest and shrub habitats. Many are generalists that also occur as breeders in other habitat types. Other riparian-associated bird species are tied to unique features, but most are insectivores and likely dependant upon the high insect productivity that riparian areas produce (Sibley 2001; Yong et al. 1998). It is sometimes useful to choose an index species to represent a habitat used by many other species.

The red-eyed vireo (*Vireo olivaceus*) is strongly associated with riparian and wet, deciduous habitats throughout its North American range. It is positively associated with forested habitats in riparian areas, making it a good species index of this habitat (Altman 2001; Sauer *et al*. 2003).

15.2 Life History & Habitat Requirements

15.2.1 Life History

15.2.1.1 Diet

Vireos are primarily insectivorous, with 85% of their diet composed of insects and only 15% of vegetable material. During fall migration, generally August to October, they eat mostly fruits and berries, eating fruit exclusively on wintering grounds. A third of its total food is composed of caterpillars and moths, mainly the former. Beetles, hymenoptera bugs and flies rank next to Lepidopteron in importance as food items for the Red-eyed Vireo (Bent 1965; Sibley 2001).

They are primarily insectivores on their breeding grounds, and this enables them to take advantage of the high insect productivity that occurs in riparian areas. Generally, there is a positive relationship in, the greater the structural layering and complexity of the habitat, the greater the insect productivity, and the greater the diversity of bird species. Many studies have reported higher species richness, abundance, or diversity in riparian zones than adjacent habitats, particularly at lower elevations (Stauffer and Best 1980; Sibley 2001).

15.2.1.2 Reproduction

The red-eyed vireo has been one of the most abundant neotropical birds in North America. The red-eyed vireos breeding range extends from British Columbia to Nova Scotia, north through parts of the Northwest Territories, and throughout most of the lower United States. Its numbers seem to have declined recently, possibly as a result of the destruction of wintering habitat, loss and fragmentation of northern breeding grounds, and loss of critical habitat along migratory routes. Its principal habitat, broad-leaved forests, often supports one breeding pair per acre.

15.2.1.3 Nesting

Pair formation and nest construction may begin within a few days of arrival at the breeding site (Loather et al. 1999). Egg dates have been reported from British Columbia, and range between 10 May and 16 August; the peak period of activity there was between 7 and 23 June (Campbell et al. in press). The incubation period is about 11 days and young fledge 8-10 days after hatching. The young often associate with their parents for up to 3 weeks following fledging (Loather et al. 1999). Red-eyed vireos typically lay only one clutch with 4 or 5 eggs. Re-nesting may occur, however, following nest failure or nest parasitism by Brown-headed Cowbirds (Sibley 2001; Loather et al. 1999).

Courtship begins in May, with the peak of egg laying in the first half of June. The nest is a thin-walled pendant cup of bark strips and plant fibers, decorated with lichen and attached to a forked twig, usually containing 3 or 4 white eggs, sparsely marked with dark brown. The incubation period is 12-14 days. Usually the nest is built from 5-35 feet above the ground, although nests as low as 2 feet and as high as 60 are reported (Bent 1965; Ehrlich et al. 1988). The sexes share in incubation and both develop brood patches (Pyle 1997; Sibley 2001). Occasionally a pair may raise two broods in a season, but this unusual (Pyle 1997; Bent 1965).

The red-eyed vireo typically lays 3 to 4 eggs, but is often parasitized by the Brown-headed Cowbird. red-eyed vireos haven’t developed effective responses to nest parasitism by the Brown-headed Cowbird. They are considered an “acceptor species” as they rarely recognize the cowbird egg as an intruder (Pyle 1997, Sibley 2001). The host bird incubates and cares for the young interlopers, commonly to the detriment of its own young. Often the young cowbird will push the young of the host out of the nest causing failure of the host’s nesting. This parasitism may compromise productivity, especially in areas where habitat modification creates openings close to the riparian zone (Sibley 2001).

15.2.1.4 Migration

Songbirds are nocturnal, or powered migrants, and tend to migrate in a couple of different patterns. It is thought that powered migrants are much less affected by topography because of their night travel, and therefore show little concentration at particular landforms. (Corral 1989). Unlike the larger, diurnal migrants that depend upon updrafts for “soaring” migration, powered migrants must generate all the energy themselves for the long- distance water crossings thus, adding to the importance of stopover habitat during migration (Kerlinger 1995). For the most part, powered migrants rely on food supply and prevailing winds to determine their specific migration pattern for the season, thus spring migration does not always follow the fall migration pattern. In general, however, North American powered migrants are pushed east in fall by prevailing winds and do concentrate on the Atlantic Coast as they move to wintering areas (Corral 1989).

The red-eyed vireo is known in Central America as a transient, journeying between its breeding range in North America and its winter home in South America. September is the month when these vireos pass southward through the Isthmus of Panama in the greatest numbers, but stragglers have been recorded in Costa Rica as late as October 28, and November 10 (Bent 1965; Pyle 1997; Sibley 2001), and are regularly documented into late November, along the Caribbean Coast of Costa Rica (Renan 1995; Ralph et al.1999). The northward migratory passage to breeding grounds begins in late March and peaks in April. An occasional straggler or small flock may be seen in early May passing through Central America (Bent 1965).

15.2.2 Habitat Requirements

The habitat requirements of neotropical bird migrants are extremely diverse. Within a single species, the habitat and food preferences on breeding grounds, is often different than wintering areas (Petit et al. 1993). Initial findings define the Washington breeding population of red-eyed vireos preferred habitat as: “tall, some what extensive, closed canopy forests of cottonwood, maple, or alder; deciduous trees (cottonwood, alder, maple, and ash; optimum cottonwood gallery forest) >15 m., high mean canopy closure (>60%), deciduous shrubby or young trees in understory (>10% cover). red-eyed vireo forages in understory more than Warbling Vireo; forest stand sizes should be larger than 50 acres (20 hectares) in size, and riparian corridor widths should be >50 m. (164 feet) in width, as they are more common in stand interiors, yet will tolerate some tree removal and canopy opening (Bushman and Therres 1988).

The Washington red-eyed vireo populations are likely dependant on riparian areas for necessary food requirements, but use the cottonwood stands or other broadleaf trees for nesting and singing. A tall tree perch allows a singing male to take advantage of the height, enabling their vocalizations to carry further in hopes of attracting a mate. They also sing to claim and define territories from other breeding males in the area (Sibley 2001).

Partners in Flight have 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 >50 ft (15 m), mean canopy closure >60%, young (recruitment) sapling trees >10% cover in the understory, riparian woodland >164 ft (50 m) wide (Altman 2001). Red-eyed vireos are closely associated with riparian woodlands and black cottonwood (*Populus trichocarpa*) stands and may use mixed deciduous stands (Altman 2001).

15.3 Population & Distribution

15.3.1 Population

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 (Bent 1965). This species is one of the most abundant in the northeastern United States, but is much less common in Washington due to habitat limitations. In Washington they are strongly associated with the tall, somewhat extensive, closed canopy forests of cottonwood, maple, or alder in the Puget Lowlands (C. Chappell pers. comm. 1998).

15.3.2 Distribution

Little is known about the size of the breeding population in Washington. Their patchy distribution correlates with the distribution of large black cottonwood groves, which are usually limited to riparian areas. Their associated habitat is most abundant in northeastern river valleys, especially the Sanpoil, Kettle, Columbia, Colville, and Pend Oreille Rivers (Sauer et al. 2003). Areas of relatively high red-eyed vireo density in Washington are the Skagit River (Whatnot/Skagit Counties), the Fort Lewis area (Pierce County), and major river valleys of the northeastern part of the state.

In Washington they are locally common in riparian growth with cottonwood stands (especially along the Nooksack and Skagit Rivers and at Fort Lewis); along the Columbia River in Clark, Skamania and Klickitat Counties; and in eastern Washington along major rivers. They are more widespread in northeastern Washington and southeastern Washington than elsewhere in the state (Sauer et al. 2003). See Figure 15-1 for Washington breeding distribution of red-eyed vireo from 1987-1995.

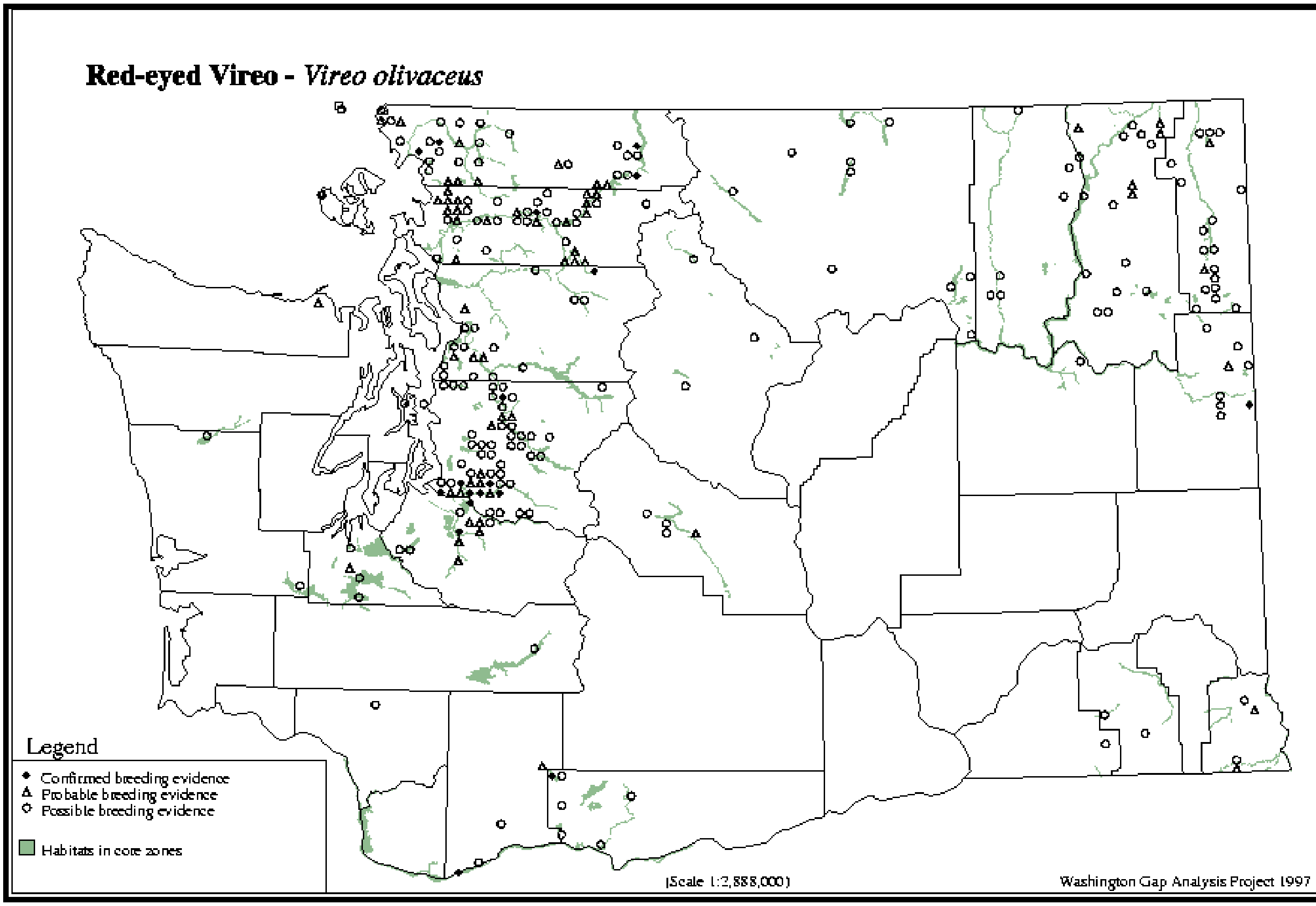


Figure 15-1. Breeding bird atlas data (1987–95) and species distribution for red-eyed vireo.

15.4 Status & Abundance Trends

15.4.1 Status

Red-eyed vireo populations were once considered stable, but have seen localized declines across North America in the last 10 years (Saurer et al. 2003). Red-eyed vireo populations are protected throughout their breeding range by the: Migratory Bird Treaty Act (1918) in the US, the Migratory Bird Convention Act (1916) in Canada, and the Convention for the Protection of Migratory Birds and Game Mammals (1936) in Mexico.

Unanswered questions regarding habitat requirements and population constraints need to be addressed in order to provide adequate management recommendations and appropriate conservation measures, aimed at stabilizing and reversing population declines.

15.4.2 Trends

In Washington, Breeding Bird Survey (BBS) data show a long-term decline, this represents an annual population decline in Washington of 2.6%, although the change is not statistically significant, largely because of scanty data (Sauer et al. 2003). Because BBS dates back only about 30 years, population declines in Washington resulting from habitat lost before the surveys began would not be accounted for. The overall abundance trend in North America for this species is both declining and increasing, appearing to be localized and likely tied into habitat changes at breeding areas. Figure 15-2 shows red-eyed vireo North American breeding trends from 1966-2002.

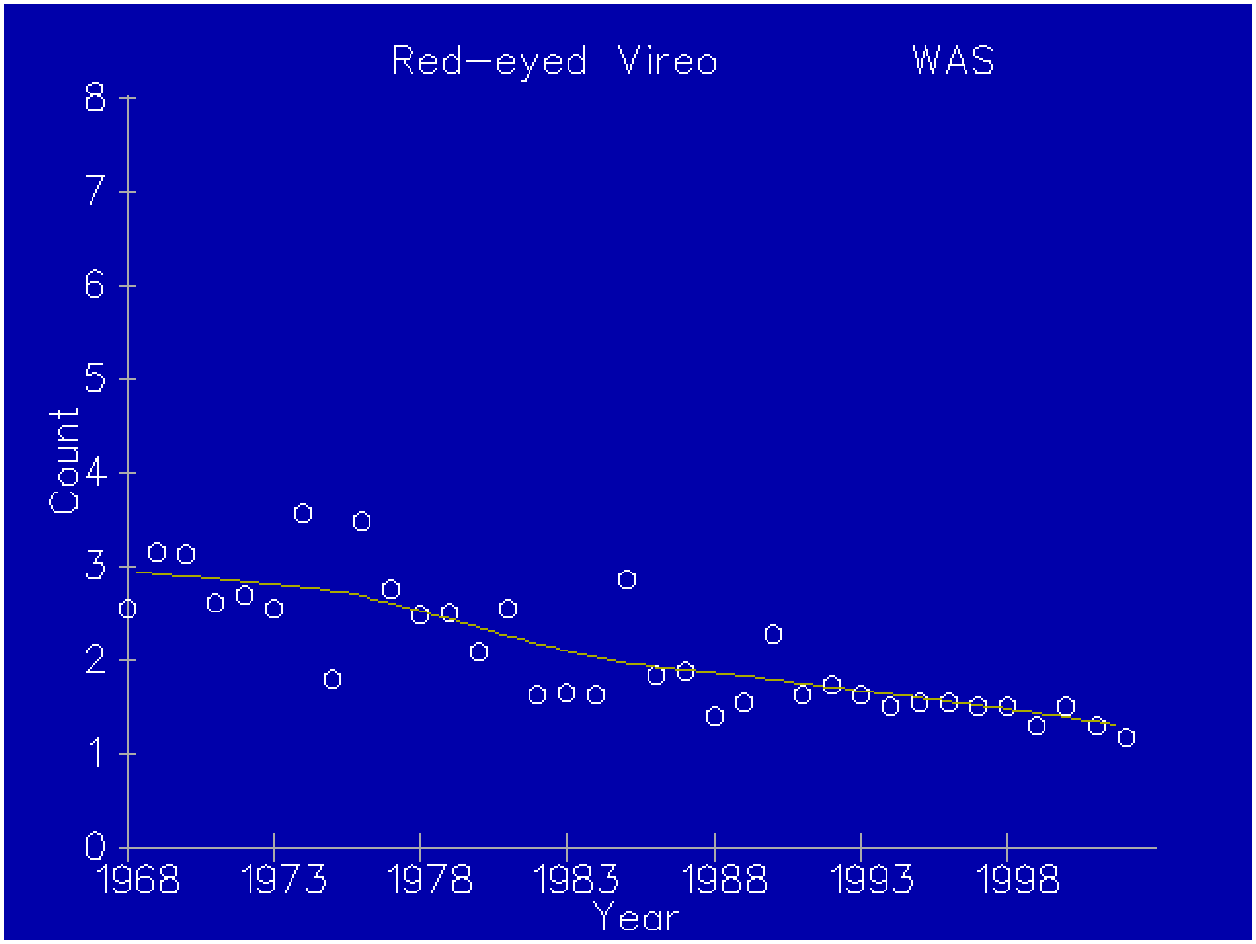


Figure 15-2. Red-eyed vireo North American Breeding Bird Survey trend results (Sauer *et al.* 2003).

15.4.3 Productivity

The red-eyed vireo typically lays 3 to 4 eggs, but it is often parasitized by the brown-headed cowbird. The host bird incubates and cares for the young interlopers, commonly to the detriment of its own young. Often the young cowbird will push the young of the host out of the nest causing failure of the host's nesting. This parasitism may compromise productivity, especially in areas where habitat modification creates openings close to the riparian zone.

Although little data is available on annual survival rate of populations in Washington, the average adult annual survival is 55%-75%, with a mean longevity of 2-4 years from hatching (Sauer et *al*.2003).

15.5 Environmental Conditions

15.5.1 Habitat Distribution

The red-eyed vireo is one of the most abundant species in northeastern United States, but is much less common in Washington due to limited habitat. The patchy distribution in Washington for the red-eyed vireo species correlates with the distribution of large black cottonwood groves, which are usually limited to riparian areas. The species is locally common in riparian growth with cottonwood stands in western Washington (especially along the Nooksack and Skagit Rivers and at Fort Lewis), along the Columbia River in Clark, Skamania, and Klickitat Counties, and in eastern Washington along major rivers. It is more widespread in northeastern Washington and southeastern Washington than elsewhere in the state. Habitat is most abundant in northeastern river valleys, especially the Sanpoil, Kettle, Columbia, Colville, and Pend Oreille Rivers.

15.5.2 Habitat Status

The status of historic habitat conditions is largely unknown. However, the Northwest Habitat Institute (2001) mapped historic riparian/wetland habitat in the Lower Columbia subbasin and the current riparian/wetland habitat in the lower Columbia subbasin, see Figure 15-3 and Figure 15-4 on following pages. It is difficult to determine if these are accurate representations. The numbers available from the Northwest Habitat Institute (2003) indicate that no riparian habitat loss has occurred in the Columbia River subbasin since 1850. The number of acres of west-side riparian wetlands in the Columbia River subbasin and Columbia River Estuary represented in Figure 15-3 and Figure 15-4, respectively 1850 and 1999 are as follows:

Columbia Estuary: 1850 (14,186 acres)   Lower Columbia River: 1850 (12,982 acres)
1999 (20,064 acres)   1999 (16,086 acres)

In interpreting this data, it should be noted that west-side riparian habitats are represented on a large, ecological landscape level, but on a local level, the relevance of the plant communities making up these riparian areas cannot be ignored. Even if there is currently more west-side riparian acreage, the historic riparian vegetation most likely contained much more native vegetation, and thus, were probably more functional on a local, and landscape level.

A study on neotropical songbird use of native and non-native riparian areas in the mid-Columbia River Basin during fall migration confirmed species richness and abundance was significantly greater in areas dominated by native shrub vegetation. The riparian sites consisted of similar vegetation features aside from the dominant shrub layer, which was either a native willow species (*Salix spp.)*, or the non-native Russian Olive (*Elaegnus angustifolia*). In addition to greater neotropical songbird abundance and species richness, riparian areas with dominant native shrub (willow spp.) vegetation also had greater invertebrate abundance. Non-native, Russian Olive dominated riparian areas, had greater abundance of resident or "non-migratory" songbirds; no significant difference was found in species richness of invertebrates, although non-native sites primarily contained demapterans (earwigs), while native sites contained mostly homopterans (aphids and hoppers) (Hudson *et al*. 1999). These results demonstrate the importance of natively vegetated riparian areas, and how plant species on a local level, can change the functions, and thus species use of that habitat; indicating the importance of conserving riparian areas dominated by native vegetation, and the importance in restoring non-native dominant riparian areas.

15.6 Factors Affecting Population Status

15.6.1 Key Factors Inhibiting Populations & Ecological Processes

15.6.1.1 Habitat Loss and Degradation

Neotropical migrants tend to be more vulnerable to habitat loss and fragmentation than resident birds, or those that migrate only short distances within North America. Habitat loss due to hydrological diversions and control of natural flooding regimes (e.g., dams) has resulted in an overall reduction and /or conversion of riparian habitat for red-eyed vireos. Habitat losses are also caused by inundation from impoundments, cutting and spraying for ease of access to watercourses, gravel mining, forest management, etc.

The status of historic habitat conditions is largely unknown.

15.6.1.2 Habitat Degradation

Habitat degradation from loss of vertical stratification in riparian vegetation can be caused from: lack of recruitment of young cottonwoods, ash, willows, and other sub-canopy species; stream bank stabilization (e.g., riprap) which narrows stream channel, reduces the flood zone, and reduces extent of riparian vegetation; invasion of exotic species such as reed canary grass and blackberry; overgrazing which can reduce under story cover; and reductions in riparian corridor widths which may decrease suitability of the habitat and may increase encroachment of nest predators and nest parasites to the interior of the stand (Marzluff 2001; Hutto 1998; Sibley 2001).

Certain cycles/timing periods in a songbird life are more critical than others, and the habitat uses during that time, also rank in importance. Migratory habitat is critical in fulfilling the feeding and energy renewals of migrating birds. It is thought these brief stops for feeding and energy renewal are critical, can affect population trends, and are important in conservation efforts (Hutto 1998). Amongst the age classes, immature birds seem to suffer the most from degradation or loss of migration habitats (stopover areas). This is because the juveniles migrate south after the adults, and have less experience at foraging for food, selecting habitat, competing against adults, and dealing with predators. These migration habitats are essential to birds for fat accumulation, in order to make flights of long distances without stopping (Yong et al. 1998). Without sufficient fat stores energy depletion and/or exhaustion can cause mortality during long flights or inhospitable habitats. The common observation of grounded birds far at sea reflects these phenomena and may become more common as humans further impinge on the habitats where migrants obtain these energy stores (Sibley 2001; Yong *et al.* 1998).

15.6.1.3 Human Disturbance

Hostile landscapes, particularly those close to agricultural and residential areas, may have 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 especially in high-use recreation areas, may have an impact on red-eyed vireos (Marzluff 2001).

15.6.1.4 Pesticides/Herbicides

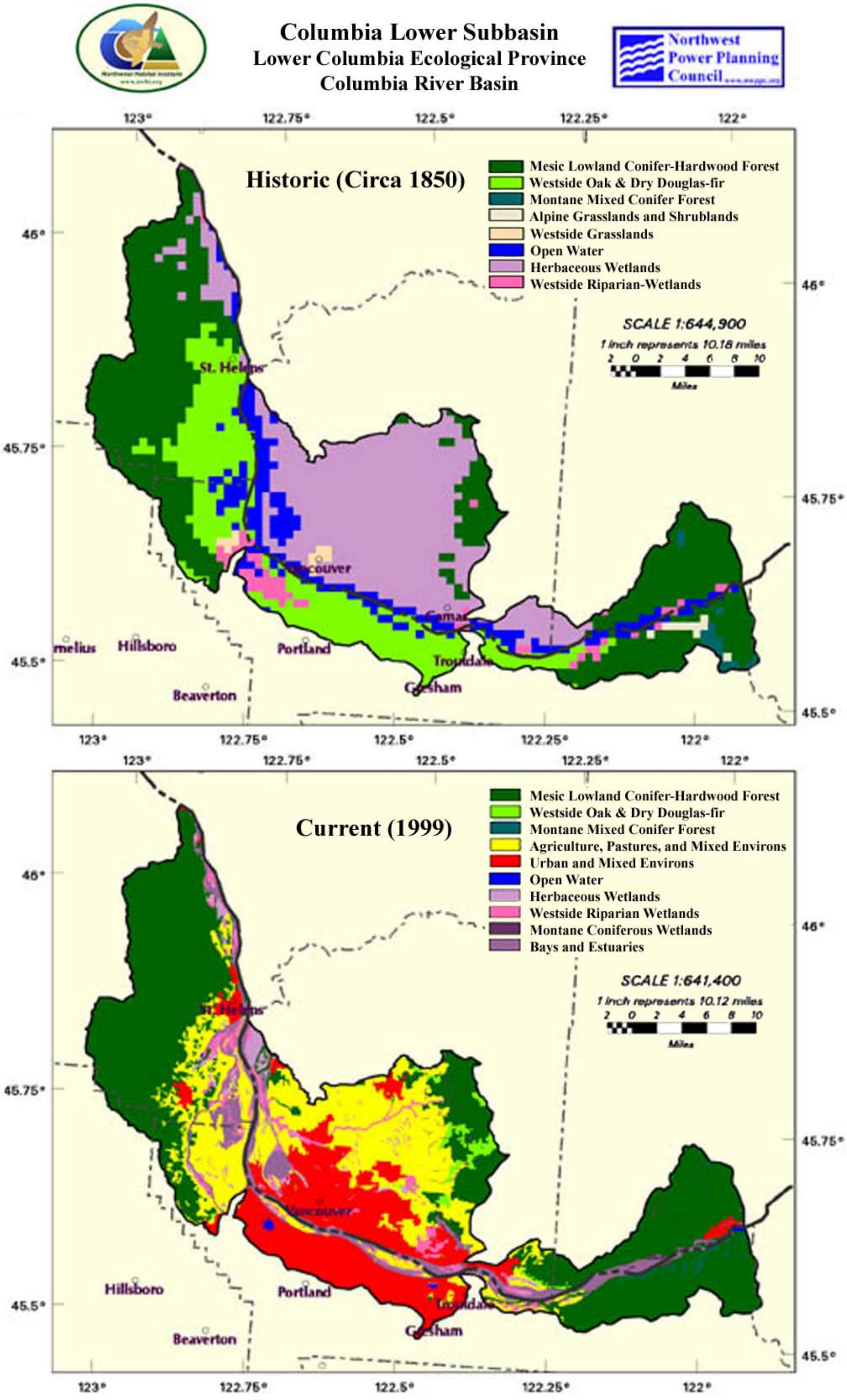
Increased use of pesticide and herbicides associated with agricultural and forestry practices may reduce insect food base. Washington State Forestry rules (Forest and Fish) allow spraying of herbicides during important timing periods, like fall migration, when abundant food sources are necessary to gather adequate fat stores (Sibley 2001; Alltman 2001).

14.6.1.3 Nest Depredation and Brood Parasitism

Nest parasitism from Brown-headed Cowbirds is increasingly becoming an issue in songbird populations. Fragmentation of habitats, resulting in reduced patch size and increased edge, is correlated with higher cowbird brood parasitism (Marzluff 2001). The Brown-headed Cowbird is an obligate nest brood parasite that does not build a nest, but instead lays eggs in the nests of other species. Often the young cowbird will push the young of the host out of the nest, causing failure of the host’s nesting (Sibley 2001; Ehrlich et al.1988). Cowbirds have been shown to affect red-eyed vireo productivity at localized breeding areas (Ehrlich et al. 1988), this parasitism may compromise productivity especially in areas where habitat modification (forest fragmentation) creates openings close to the riparian zone (Sibley 2001; Burton 1995; Marzluff 2001).

15.7 Inventory & Assessment of Existing Management and Conservation Plans

Westside Lowlands and Valleys Bird Conservation Plan (Partners in Flight 2001) is the only existing comprehensive plan for management of habitats for neotropical migrant birds in Washington and Oregon. It establishes biological objectives for the species in the lowlands of western Oregon and western Washington. These include providing habitats that meet the following definition: mean canopy tree height >50 ft (15 m), mean canopy closure >60%, young (recruitment) sapling trees >10% cover in the under story, riparian woodland >164 ft (50 m) wide (Altman 2001). Red-eyed vireos are closely associated with riparian woodlands and black cottonwood (*Populus trichocarpa*) stands and may use mixed deciduous stands. It is very detailed and would go a long way towards preserving and enhancing the critical habitats needed for the protection of the red-eyed vireo in Washington. Currently, no active restoration is taking place towards conserving neotropical migrants in the Lower Columbia River.



Source: Interactive Biodiversity Information System (IBIS), NHI.
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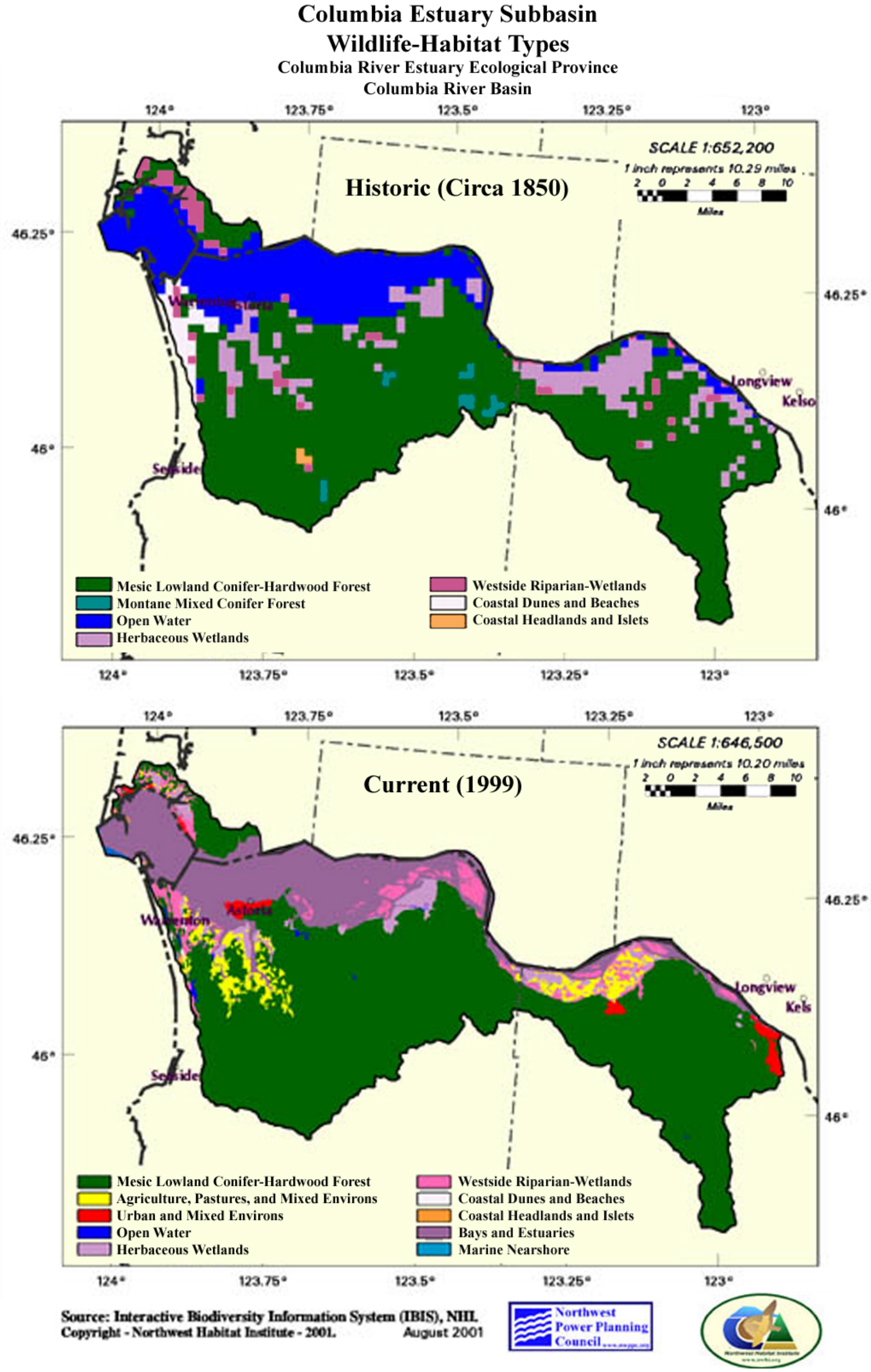
Figure 15-3. Historical (circa 1850) and current (1999) wildlife habitat types in the Columbia Lower Subbasin (IBIS 2003).



Source: Interactive Biodiversity Information System (IBIS), NHI.
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Northwest Power Planning Council

Figure 15-4. Historical (circa 1850) and current (1999) wildlife habitat types in the Columbia Estuary Subbasin (IBIS 2003).

15.8 Conservation Implications

Conserving viable populations of migratory species and their associated habitats may seem impossible when we consider that only 7% to 8% of available lands in the United States have been set aside as nature preserves, wilderness, refuges, sanctuaries, and parks. It is apparent that the reversal of these declines will also depend on the management, conservation, or enhancement of the other 92-93% of the land in the United States. This land consists of privately owned, or is managed for multiple uses by states, counties, cities, or federal natural resource agencies such as U.S. Forest Service and Bureau of Land Management (Finch and Stangel 1993). Private, state, and federal land owners are realizing the necessity for multiple- land use management, and that, managing for single resources, such as wood products, livestock, minerals, or single species, such as game species, endangered species, and charismatic species, is costly, time-consuming, and potentially in conflict with sustaining other resources and species (Finch and Stangel 1993). Identifying critical habitat, inventorying habitat remaining, and monitoring habitat changes, both locally and at a landscape level, will become crucial to the future management and protection of fish and wildlife, including but not limited to ESA salmon, game birds/mammals, and non-game species, like neotropical songbirds.

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Volume III, Chapter 16
Yellow Warbler

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16.0 Yellow Warbler (*Dendroica petechi*a)

16.1 Introduction

Over the past several years, songbirds and the reasons for declines in their populations have been a focal point of interest. Many species of neotropical songbirds birds have experienced population declines due to losses and fragmentation of breeding, wintering, and migratory habitats. These long-distance migrants tend to be more vulnerable to habitat loss and fragmentation than resident birds or those that migrate only short distances within North America.

At least 49 neotropical bird species are highly associated with riparian forest and shrub habitats. Many are generalists that also occur as breeders in other habitat types. Other riparian-associated bird species are tied to unique features, but most are insectivores and likely dependant upon the high insect productivity that riparian areas produce (Sibley 2001; Yong et al. 1998). It is sometimes useful to choose an index species to represent a habitat used by many other species.

The yellow warbler (*Dendroica petechi*a) is strongly associated with riparian and wet, deciduous habitats throughout its North American range. It is positively associated with sub-canopy/shrub habitats in riparian areas, making it a good species index of this habitat (Altman 2001; Sauer *et al*. 2003).

16.2 Life History and Habitat Requirements

16.2.1 Life History

16.2.1.1 Diet

Yellow warblers capture and consume a variety of insect species. Yellow warblers consume insects and occasionally wild berries, especially when migrating. Food is generally obtained by gleaning from sub-canopy vegetation, although the species also sallies and hovers to a much lesser extent (Loather *et al*. 1999; Sibley 2001).

They are primarily insectivores on their breeding grounds, and this enables them to take advantage of the high insect productivity that occurs in riparian areas. Generally, there is a positive relationship in, the greater the structural layering and complexity of the habitat, the greater the insect productivity, and the greater the diversity of bird species. Many studies have reported higher species richness, abundance, or diversity in riparian zones than adjacent habitats, particularly at lower elevations (Stauffer and Best 1980; Sibley 2001).

16.2.1.2 Reproduction

The yellow warbler is a common species associated riparian habitat throughout its breeding range. Locally common, it can be found along rivers and creeks in the Columbia Basin, but is declining in some areas (Sauer et *al*. 2001). Little is known about yellow warbler breeding behavior in Washington, although substantial information is available from other parts of its range.

Yellow warblers have developed effective responses to nest parasitism by the brown-headed cowbird (*Molothrus ater*). The brown-headed cowbird is an obligate nest brood parasite that does not build a nest and instead lays eggs in the nests of other species. When cowbird eggs are recognized in the nest the yellow warbler female will often build a new nest directly on top of the original. In some cases, particularly early in the incubation phase, the female yellow warbler will bury the cowbird egg within the nest. Some nests are completely abandoned after a cowbird egg is laid (Lowther *et al.* 1999). Up to 40% of yellow warbler nests in some studies have been parasitized (Lowther *et al.* 1999).

Nesting

Pair formation and nest construction may begin within a few days of arrival at the breeding site (Loather et al. 1999). Egg dates have been reported from British Columbia, and range between 10 May and 16 August; the peak period of activity there was between 7 and 23 June (Campbell et al. in press). The incubation period is about 11 days and young fledge 8-10 days after hatching. The young often associate with their parents for up to 3 weeks following fledging (Loather et al. 1999). Yellow warblers typically lay only one clutch with 4 or 5 eggs. Re-nesting may occur, however, following nest failure or nest parasitism by Brown-headed Cowbirds (Sibley 2001; Loather et al. 1999).

Migration

Songbirds are nocturnal, or powered migrants, and tend to migrate in a couple of different patterns. It is thought that powered migrants are much less affected by topography because of their night travel, and therefore show little concentration at particular landforms. (Corral 1989). Unlike the larger, diurnal migrants that depend upon updrafts for “soaring” migration, powered migrants must generate all the energy themselves for the long- distance water crossings thus, adding to the importance of stopover habitat during migration (Kerlinger 1995). For the most part, they rely on food supply and prevailing winds to determine their specific migration pattern for the season, thus spring migration does not always follow the fall migration pattern. In general, however, North American powered migrants are pushed east in fall by prevailing winds and do concentrate on the Atlantic Coast as they move to wintering areas (Corral 1989).

Western populations overwinter primarily in Mexico and northern Central America.

Spring migrants begin to arrive in the Columbia River Basin in April; dates of 2 April and 10 April have been reported from Oregon and British Columbia, respectively (Gilligan et al. 1994, Campbell et al. in press). Average arrival dates are somewhat later, the average for south-central British Columbia being 11 May (Campbell et al. in press). The peak of spring migration in Washington and the Columbia Basin is in late May (Gilligan et al. 1994). Southward migration begins in late July, and peaks in late August to early September; very few migrants remain in the region in October (Lowther et al. 1999).

Mortality

Little has been published on annual survival rates. Roberts (1971) estimated annual survival rates of adults at 0.526 ±0.077 SE, although Lowther *et al.* (1999) felt this value underestimated survival because it did not account for dispersal. The oldest yellow warbler on record lived to be nearly 9 years old (Klimkiewicz *et al.* 1983).

16.2.2 Habitat Requirements

The habitat requirements of neotropical migrants are extremely diverse. Within a single species, the habitat and food preferences on breeding grounds, is often different than wintering areas (Petit et al. 1995). The yellow warbler is a common breeder in riparian habitats with hardwood trees throughout the state, generally found at lower elevations. Associated with riparian habitats, they prefer the presence of nearby water. Their habitat suitability index strongly associates them with a dense deciduous shrub layer 1.5-4 m. (5-13.3 feet), with edge, and small patch size (heterogeneity). Other suitability index associations include % of deciduous shrub canopy comprised of hydrophytic shrubs (wetlands dominated by shrubs had the highest average of breeding densities of 2males/ha) and deciduous tree basal area (abundance is positively associated). Negative associations are closed canopy and cottonwood proximity. Some nests have been found in cottonwood, but more often in shrubs with an average nest height of 0.9-2.4 m., maximum being 9-12 m. (Schroeder 1982).

Partners in Flight have established biological objectives for this species in the lowlands of western Oregon and western Washington. These include providing habitats that meet the following definition: >70% cover in shrub layer (<3 m) and subcanopy layer (>3 m and below the canopy foliage) with subcanopy layer contributing >40% of the total; shrub layer cover 30-60% (includes shrubs and small saplings); and a shrub layer height >2 m. At the landscape level, the biological objectives for habitat included high degree of deciduous riparian heterogeneity within or among wetland, shrub, and woodland patches; and a low percentage of agricultural land use (Altman 2001).

16.3 Population and Distribution

16.3.1 Population

Washington breeders represent the western subspecies *D. p. morcomi* (AOU 1998). Little is known about population size, although it is locally common where habitat exists.

16.3.2 Distribution

In the Wood-Warbler Family, Parulidae, the yellow warbler is a common species which breeds across much of the North American continent, from Alaska to Newfoundland, south to western South Carolina and northern Georgia, and west through parts of the southwest to the Pacific coast (AOU 1998). Their wintering range extends from western Mexico south to the Amazon lowlands in Brazil Brazil (AOU 1998). Neither breeding, nor winter ranges appear to have changed (Loather et al. 1999). It is one of two widespread species in the Wood-warbler family exhibiting vast geographic variation, each species containing 10 or more sub-species occurring north of Mexico. Browning (1994) recognized 43 subspecies of the yellow warbler; two of these are known to occur in Washington. One of them, Dendroica petechia brewsteri, is found in western Washington (Sibley 2000).

Little is known about the size of the breeding population in Washington State. Yellow warblers are most abundant in riparian areas in the lowlands of eastern Washington, but also occur in west side riparian zones and the lowlands of the western Olympic Peninsula. Where high rainfall limits hardwood riparian habitat, they are less common (Sharpe 1993). Locally common where riparian and wet, deciduous habitat exists, the yellow warbler can be found in the riparian areas along the Columbia River, and most riverine systems.

Core zones of distribution in Washington are the forested zones below the sub alpine fir and mountain hemlock zones, plus steppe zones other than the central arid steppe and canyon grassland zones, which are peripheral. There are no Breeding Bird Atlas records at the probable or confirmed level from sub alpine habitats in the Cascades, but Sharpe (1993) reports them nesting at 4000 feet in the Olympics. Numbers decline in the center of the Columbia Basin, but this species can be found commonly along most rivers and creeks at the margins of the Basin. A local breeding population exists in the Potholes area (Sauer et al. 2003). See Figure 16-1 for Washington breeding distribution of yellow warbler from 1987-1995.

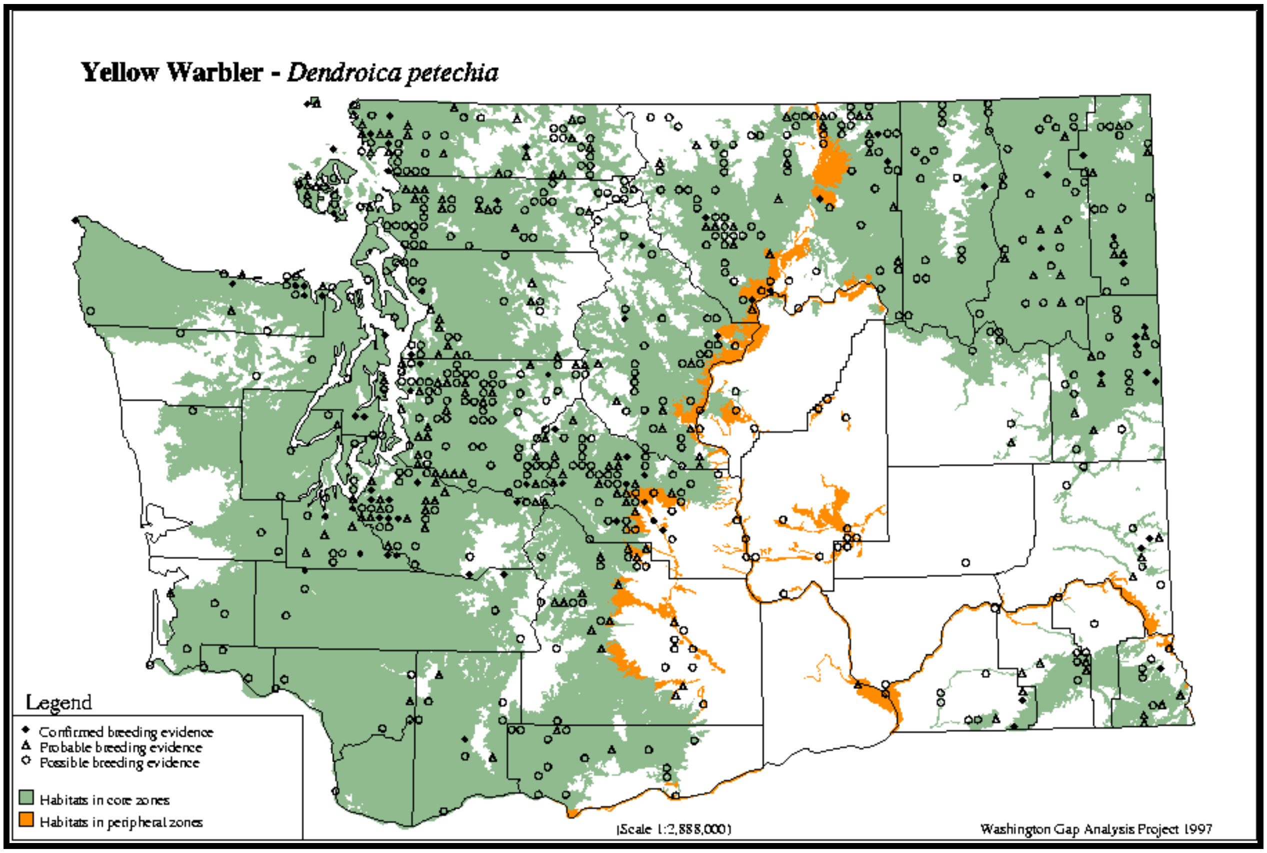


Figure 16-1. Breeding bird atlas data (1987–95) and species distribution for yellow warbler.

16.4 Status and Abundance Trends

16.4.1 Status

The yellow warbler is one of the more common warblers in North America (Loather et al. 1999). Yellow warbler populations are protected throughout their breeding range by the: Migratory Bird Treaty Act (1918) in the US, the Migratory Bird Convention Act (1916) in Canada, and the Convention for the Protection of Migratory Birds and Game Mammals (1936) in Mexico.

Information from Breeding Bird Surveys indicates that the overall populations are declining (Petit et al. 1993; Saurer et al. 2003). Some subspecies, particularly in southwestern North America, have been heavily impacted by degradation or destruction of riparian habitats (Loather et al. 1999). Unanswered questions regarding habitat requirements and population constraints need to be addressed in order to provide adequate management recommendations and appropriate conservation measures, aimed at stabilizing and reversing population declines.

16.4.2 Trends

Washington populations appear relatively stable from 1980 up to present, but show a significant decline of 6.9% in the population from 1968 to 1979 (Saurer et al. 2003). Because the Breeding Bird Survey 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. Data results of Washinton population trends presented below in Figure 16-2 are highly significant (p<0.1).

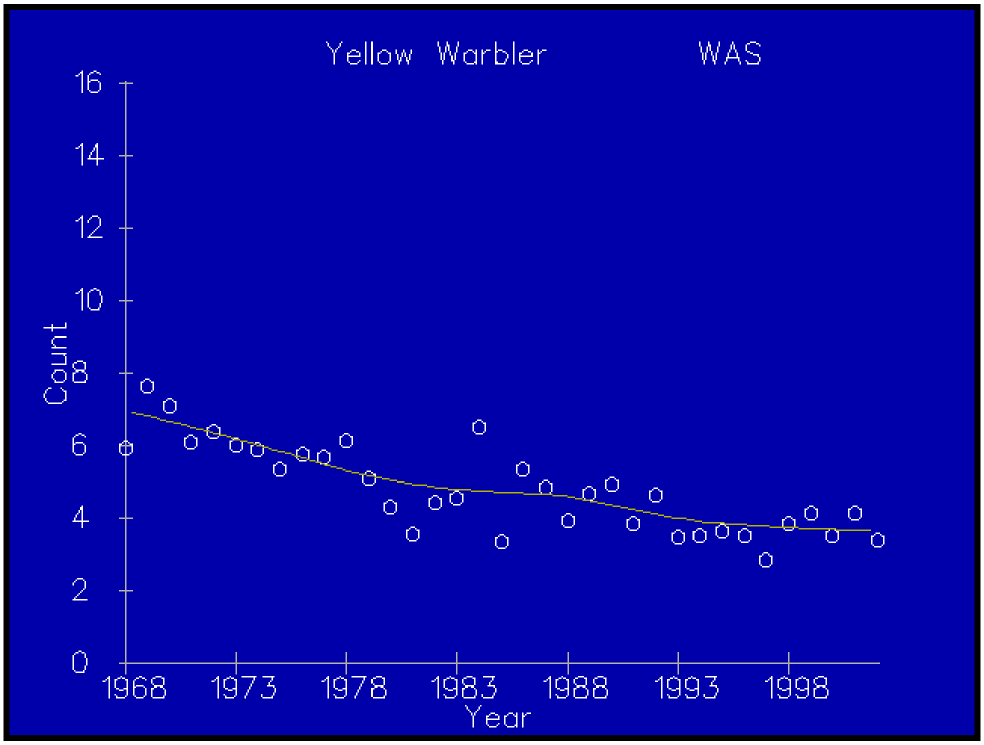


Figure 16-2. Breeding Bird Survey data for Washington State show a significant population decline of 2.9% per year (p < .1) from 1966–91 (Peterjohn 1995).

16.4.2.1 Productivity

Little has been published on annual survival rates. Roberts (1971) estimated annual survival rates of adults at 0.526 ±0.077 SE, although Lowther et al. (1999) felt this value underestimated survival because it did not account for dispersal. Results of research on breeding activities indicate variable rates of hatching and fledging. Two studies cited by Lowther et al. (1999) had hatching rates of 56% and 67%. Of the eggs that hatched, 62% and 81% fledged; this represented 35% and 54%, respectively, of all eggs laid. Two other studies found that 42% and 72% of nests fledged at least one young (Lowther et al. 1999); the latter study was from British Columbia (Campbell et al. in press). This data shows that site variability is common, local conditions often affecting or contributing to productivity and survivorship. The oldest yellow warbler on record lived to be nearly 9 years old (Klimkiewicz et al. 1983).

16.5 Environmental Conditions

16.5.1 Habitat Distribution

The yellow warbler breeds across much of the North American continent, from Alaska to Newfoundland, south to western South Carolina and northern Georgia, and west through parts of the southwest to the Pacific coast (AOU 1998). Browning (1994) recognized 43 subspecies; two of these occur in Washington, and one of them, *D.p. brewsteri*, is found in western Washington. This species is a long-distance migrant and has a winter range extending from western Mexico south to the Amazon lowlands in Brazil (AOU 1998). Neither the breeding nor winter ranges appear to have changed (Lowther *et al.* 1999).

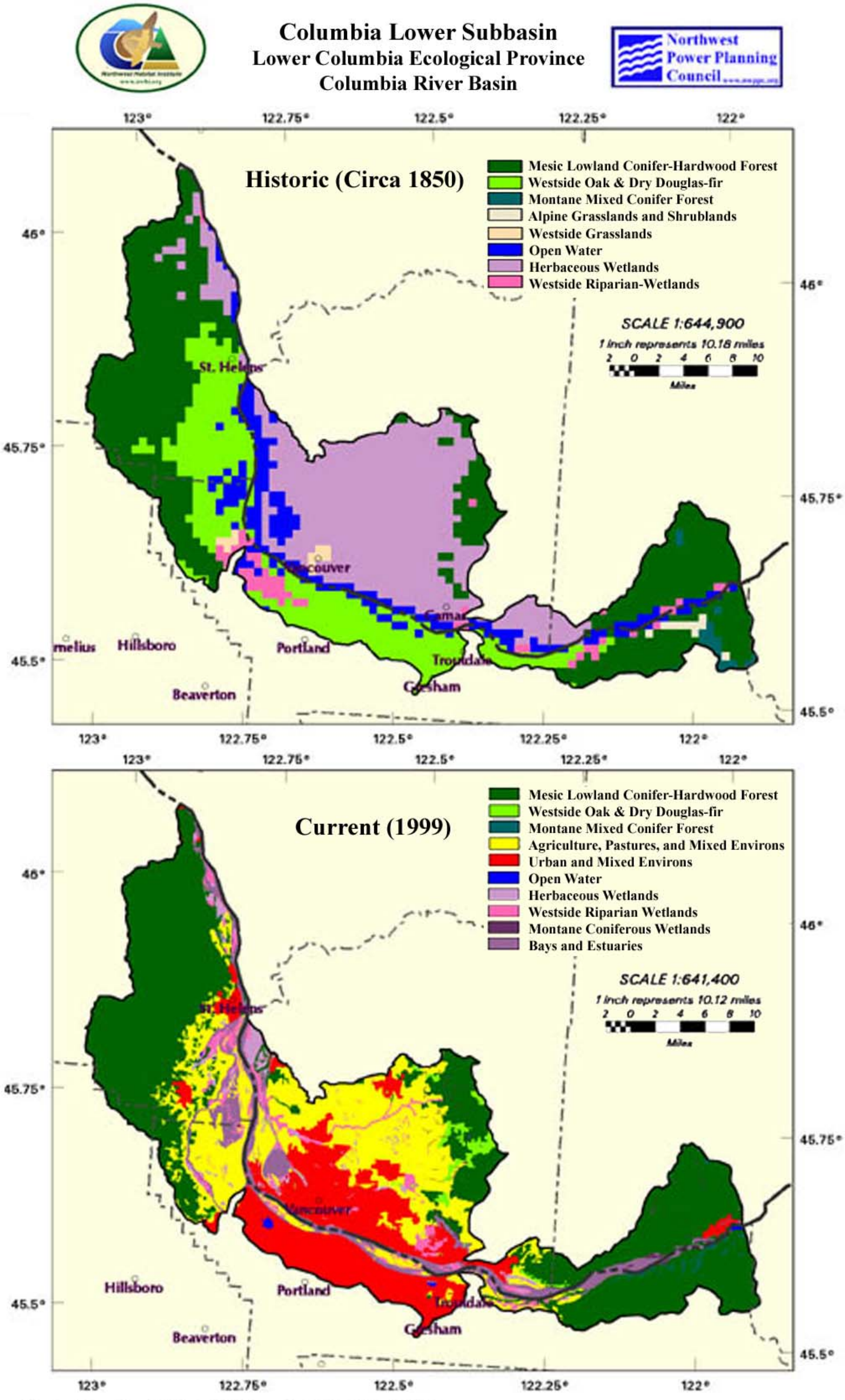
Breeding yellow warblers are closely associated with riparian hardwood trees, specifically willows, alders, or cottonwood. They are most abundant in riparian areas in the lowlands of eastern Washington, but also occur in west-side riparian zones, in the lowlands of the western Olympic Peninsula, where high rainfall limits hardwood riparian habitat. Yellow warblers are less common (Sharpe 1992). There are no BBA records at the probable or confirmed level from subalpine habitats in the Cascades, but Sharpe (1993) reports them nesting at 4000 feet in the Olympics. Numbers decline in the center of the Columbia Basin, but this species can be found commonly along most rivers and creeks at the margins of the Basin. A local breeding population exists in the Potholes area.

16.5.2 Habitat Status

The status of historic habitat conditions is largely unknown. However, the Northwest Habitat Institute (2001) mapped historic riparian/wetland habitat in the Lower Columbia subbasin and the current riparian/wetland habitat in the lower Columbia subbasin, see Figure 15-16-3 and Figure 16-4 on following pages. It is difficult to determine if these are accurate representations. The numbers available from the Northwest Habitat Institute (2001) indicate that no riparian habitat loss has occurred in the Columbia River subbasin since 1850. The number of acres of west-side riparian wetlands in the Columbia River subbasin and Columbia River Estuary represented in Figure 15-16-3 and Figure 16-4.

Columbia Estuary: 1850 (14,186 acres)
1999 (20,064 acres)

Lower Columbia River: 1850 (12,982 acres)
1999 (16,086 acres)



Source: Interactive Biodiversity Information System (IBIS), NHI.
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Figure 15-16-3. Historical (circa 1850) and current (1999) wildlife habitat types in the Columbia Lower Subbasin (IBIS 2003).

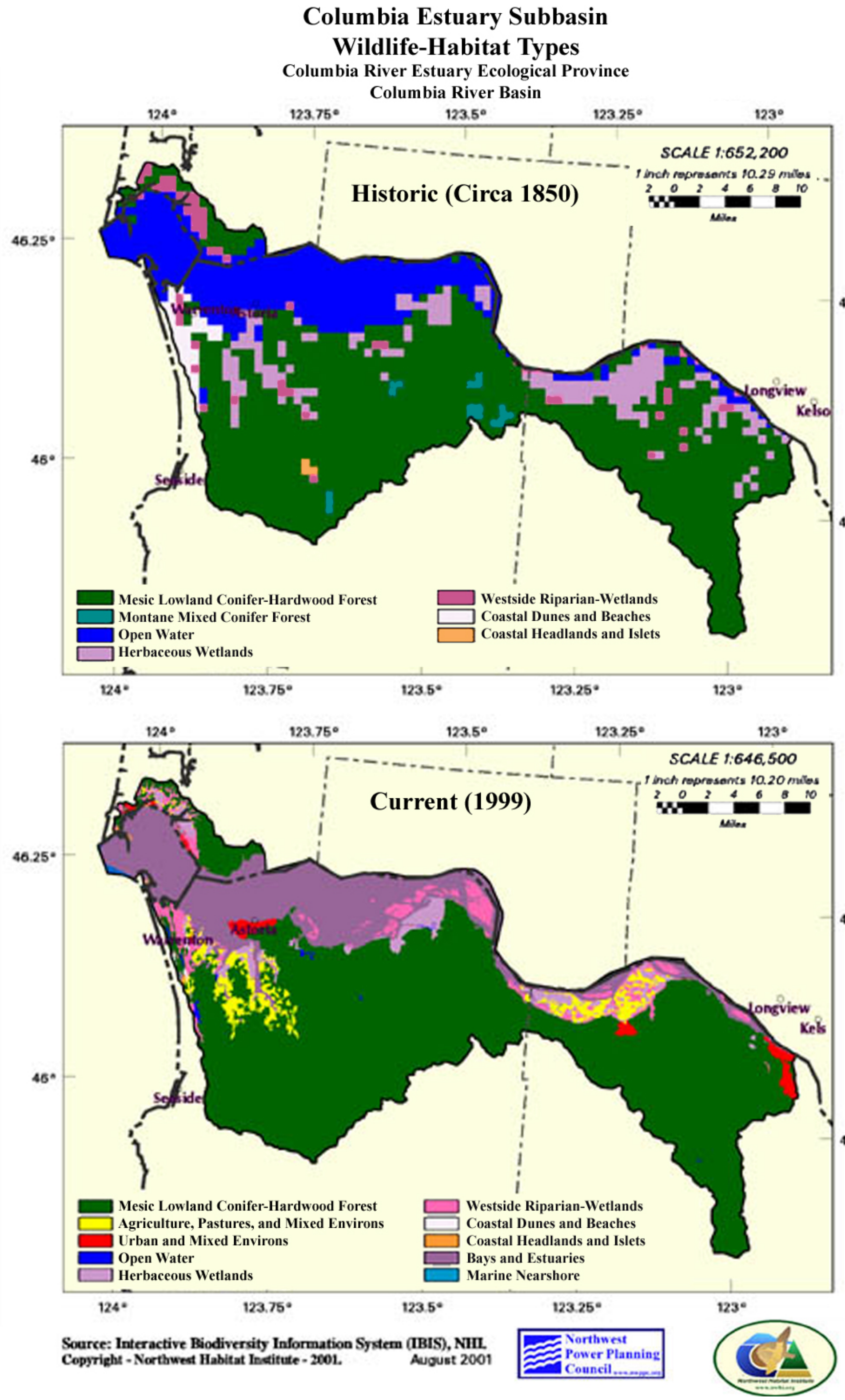

Figure 16-4. Historical (circa 1850) and current (1999) wildlife habitat types in the Columbia Estuary Subbasin (IBIS 2003).

In interpreting this data, it should be noted that west-side riparian habitats are represented on a large, ecological landacape level, but on a local level, the relevance of the plant communities making up these riparian areas cannot be ignored. Even if there is currently more west-side riparian acrerage, the historic riparian vegetation most likely contained much more native vegetation, and thus, were probably more functional on a local, and landscape level.

A study on neotropical songbird use of native and non-native riparian areas in the mid-Columbia River Basin during fall migration confirmed species richness and abundance was significantly greater in areas dominated by native shrub vegetation. The riparian sites consisted of similar vegetation features aside from the dominant shrub layer, which was either a native willow species (*Salix spp.)*, or the non-native Russian Olive (*Elaegnus angustifolia*). In addition to greater neotropical songbird abundance and species richness, riparian areas with dominant native shrub (willow spp.) vegetation also had greater invertebrate abundance. Non-native, Russian Olive dominated riparian areas, had greater abundance of resident or "non-migratory" songbirds; no significant difference was found in species richness of invertebrates, although non-native sites primarily contained demapterans (earwigs), while native sites contained mostly homopterans (aphids and hoppers) (Hudson *et al*. 1999). These results demonstrate the importance of natively vegetated riparian areas, and how plant species on a local level, can change the functions, and thus species use of that habitat; indicating the importance of conserving riparian areas dominated by native vegetation, and the importance in restoring non-native dominant riparian areas.

16.6 Factors Affecting Population Status

16.6.1 Key Factors Inhibiting Populations and Ecological Processes

16.6.1.1 Habitat loss

Neotropical migrants tend to be more vulnerable to habitat loss and fragmentation than resident birds, or those that migrate only short distances within North America. Habitat loss due to hydrological diversions and control of natural flooding regimes (e.g., dams) has resulted in an overall reduction and /or conversion of riparian habitat for yellow warblers. Habitat losses are also caused by inundation from impoundments, cutting and spraying for ease of access to watercourses, gravel mining, forest management, etc. The status of historic habitat conditions is largely unknown.

16.6.1.2 Habitat degradation

Habitat degradation from loss of vertical stratification in riparian vegetation can be caused from: lack of recruitment of young cottonwoods, ash, willows, and other sub-canopy species; stream bank stabilization (e.g., riprap) which narrows stream channel, reduces the flood zone, and reduces extent of riparian vegetation; invasion of exotic species such as reed canary grass and blackberry; overgrazing which can reduce under story cover; and reductions in riparian corridor widths which may decrease suitability of the habitat and may increase encroachment of nest predators and nest parasites to the interior of the stand (Marzluff 2001; Hutto 1998; Sibley 2001).

Certain cycles/timing periods in a songbird life are more critical than others, and the habitat uses during that time, also rank in importance. Migratory habitat is critical in fulfilling the feeding and energy renewals of migrating birds. It is thought these brief stops for feeding and energy renewal are critical, can affect population trends, and are important in conservation efforts (Hutto 1998). Amongst the age classes, immature birds seem to suffer the most from degradation or loss of migration habitats (stopover areas). This is because the juveniles migrate south after the adults, and have less experience at foraging for food, selecting habitat, competing against adults, and dealing with predators. These migration habitats are essential to birds for fat accumulation, in order to make flights of long distances without stopping (Yong et al. 1998). Without sufficient fat stores energy depletion and/or exhaustion can cause mortality during long flights or inhospitable habitats. The common observation of grounded birds far at sea reflects these phenomena and may become more common as humans further impinge on the habitats where migrants obtain these energy stores (Sibley 2001;Yong *et al*. 1998).

16.6.1.3 Human Disturbance

Hostile landscapes, particularly those close to agricultural and residential areas, may have 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 especially in high-use recreation areas, may have an impact on yellow warblers (Marzluff 2001).

16.6.1.4 Nest Depredation and and Brood Parasitsm

Nest parasitism from Brown-headed Cowbirds is increasingly becoming an issue in songbird populations. Fragmentation of habitats, resulting in reduced patch size and increased edge, is correlated with higher cowbird brood parasitism (Marzluff 2001). In temperate North America the yellow warbler is one of the principal victims of the cowbird. A cowbird lays only one egg per foster nest, but she may lay eggs in four or five nests in a short time, thus jeopardizing many broods. If the female yellow warbler discovers a cowbird egg in her nest, she usually covers the alien egg with a new foundation and lays another clutch. Occasionally a nest is found with up to six layers, each containing one cowbird egg. This parasitism may compromise productivity especially in areas where habitat modification (forest fragmentation) creates openings close to the riparian zone (Sibley 2001; Burton 1995; Marzluff 2001).

16.6.1.5 Pesticides and Herbicides

Increased use of pesticide and herbicides associated with agricultural and forestry practices may reduce insect food base. Washington State Forestry rules (Forest and Fish) allow spraying of herbicides during important timing periods, like fall migration, when abundant food sources are necessary to gather adequate fat stores (Sibley 2001; Alltman 2001).

16.7 Inventory and Assessment of Existing Management Plans

Westside Lowlands and Valleys Bird Conservation Plan (Partners in Flight 2001) is the only existing comprehensive plan for management of habitats for neotropical migrant birds in Washington and Oregon. It establishes biological objectives for the species in the lowlands of western Oregon and western Washington. These include providing habitats that meet the following definition: >70% cover in shrub layer (<3 m) and sub canopy layer (>3 m and below the canopy foliage) with sub canopy layer contributing >40% of the total; shrub layer cover 30-60% (includes shrubs and small saplings); and a shrub layer height >2 m. At the landscape level, the biological objectives for habitat included high degree of deciduous riparian heterogeneity within or among wetland, shrub, and woodland patches; and a low percentage of agricultural land use (Altman 2001). It is very detailed, and if followed, would go a long way towards preserving and enhancing the critical habitats needed for the protection of the yellow warbler in

Washington. Currently, no active restoration is taking place towards conserving neotropical migrants in the Lower Columbia River.

16.8 Conservation Implications

Conserving viable populations of migratory species and their associated habitats may seem impossible when we consider that only 7% to 8% of available lands in the United States have been set aside as nature preserves, wilderness, refuges, sanctuaries, and parks. It is apparent that the reversal of these declines will also depend on the management, conservation, or enhancement of the other 92-93% of the land in the United States. This land consists of privately owned, or is managed for multiple uses by states, counties, cities, or federal natural resource agencies such as U.S. Forest Service and Bureau of Land Management (Finch and Stangel 1993). Private, state, and federal land owners are realizing the necessity for multiple- land use management, and that, managing for single resources, such as wood products, livestock, minerals, or single species, such as game species, endangered species, and charismatic species, is costly, time-consuming, and potentially in conflict with sustaining other resources and species (Finch and Stangel 1993). Identifying critical habitat, inventorying habitat remaining, and monitoring habitat changes, both locally and at a landscape level, will become crucial to the future management and protection of fish and wildlife, including but not limited to ESA salmon, game birds/mammals, and non-game species, like neotropical songbirds.

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Volume III, Chapter 17
Steller Sea Lions

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17.0 Steller Sea Lion *(Eumetopias jubatus)*

17.1 Introduction

The Steller sea lion (*Eumetopia jubatus*) is the largest of the eared or otariid seals in northwest waters and are present year-round. Both sexes occur in Washington and Oregon waters, with adult males (to 2,200 lbs) being considerably larger than females (to 700 lbs). Breeding rookeries are located along the Oregon and British Columbia coasts. No breeding rookeries are found in Washington. Coloration varies from tawny through yellowish brown to dark brown. Vocalizations from adults can be described as a deep growling sound (Jeffries et al. 2000).

Steller (or northern) sea lions range along the North Pacific from California to Russia and Japan (Loughlin et al. 1984). For management purposes the Steller sea lion population is divided into two distinct population segments or stocks that are designated as the Western U.S. and Eastern U.S. Steller Sea Lion Stocks (Angliss and Lodge 2002). Steller sea lions west of 144° W. longitude (near Cape Suckling, AK) covering western Alaska to Russia and Japan form a distinct population segment and are identified as the Western U.S. Stock by National Marine Fisheries Service (NMFS). Steller sea lions east of 144° W. longitude covering S.E. Alaska and south into British Columbia, Washington, Oregon, and California form another distinct population segment and are identified as the Eastern U.S. Stock of Steller Sea Lions (Federal Register Vol. 62 No. 86:24345-24355). Steller sea lions in Washington and Oregon waters are considered part of the Eastern U.S. Stock (Angliss and Lodge 2002).

Steller sea lions are protected under the Endangered Species Act (ESA) and Marine Mammal Protection Act (MMPA). The Western U.S. stock is listed as “Endangered” under the ESA, and t the Eastern U.S. Stock is listed as “Threatened” under the ESA. Steller sea lions are listed as a state “Endangered” species by the states of Washington and Oregon.

17.2 Life History & Habitat Requirements

17.2.1 Life History

17.2.1.1 Diet

Steller sea lions are an opportunistic predator that feeds primarily on fish and squid, with prey varying by season, area and water depth (Fiscus and Baines 1966; Antonellis and Fiscus

1980, Sinclair and Zeppelin 2002). In the waters off California and Oregon, their diet consists of Pacific whiting, herring, salmon, lamprey, rockfish, flatfish and squid (Fiscus and Baines 1966, Beach *et al*. 1985; Riemer and Brown 1997). Prey identified from stomach and scats in British Columbia included Pacific whiting, herring, octopus, Pacific cod, rockfish, and salmon (Spalding 1964). In the Rogue River, lamprey made up 87% of the prey eaten at the surface (Jameson and Kenyon 1977). Scats from the Rogue Reef rookery show that Pacific whiting (in 62-100 percent of scats) and Pacific lamprey (in 2-83 percent of scats) were the two most common prey of Steller sea lions during the late May to July breeding season (Riemer and Brown 1997). Other commonly eaten prey also included Pacific herring, rockfish, flatfish and cephalopods. Salmon remains in the diet ranged from 6-33 percent (Table 17-1). Predation on salmon occurs but is not considered a primary prey of this species (NMFS 1992a). Although the South Jetty at the Columbia River is a traditional Steller sea lion haulout used by several hundred animals and is a location where scats could be collected to determine diet, no comprehensive study has been conducted to address Steller sea lion diet in the Columbia River.

Table 17-1. Summary of food habit studies for Steller sea lions at Rogue Reef, Oregon. Prey species indicated occurred in at least 10 percent of samples except for salmonids (Riemer and Brown 1987).

Source:	Reimer and Brown 1997
Location:	Rogue Reef, Oregon
Season and Year:	May 1986
Sample Size:	60
Type:	Scats
Prey species	% of Samples
Pacific whiting	73
Skate	38
Pacific lamprey	37
Salmon sp.	**33**
Pacific herring	33
Smelt	27
Octopus	18
Sculpin	18
Various Rockfish	17
Other species: Northern anchovy, Northern clingfish, rex sole, and sculpin	
Source:	Reimer and Brown 1996
Location:	Rogue Reef, Oregon
Season and Year:	June 1986
Sample Size:	18
Type:	Scats
Prey species	% of Samples
Pacific lamprey	83
Pacific whiting	67
Various Flatfish	17
Various Rockfish	11
Squid	11
Salmon sp.	**6**
Other species: Northern anchovy, Northern clingfish, rex sole, and sculpin	
Source:	Reimer and Brown 1997

Location: Rogue Reef, Oregon
Season and Year: May 1986
Sample Size: 60
Type: Scats

Prey species	% of Samples
Pacific whiting	73
Skate	38
Pacific lamprey	37
Salmon sp.	**33**
Pacific herring	33
Smelt	27
Octopus	18
Sculpin	18
Various Rockfish	17

Other species: Northern anchovy, Northern clingfish, rex sole, and sculpin

Source: Reimer and Brown 1996
Location: Rogue Reef, Oregon
Season and Year: June 1986
Sample Size: 18
Type: Scats

Prey species	% of Samples
Pacific lamprey	83
Pacific whiting	67
Various Flatfish	17
Various Rockfish	11
Squid	11
Salmon sp.	**6**

Other species: Northern anchovy, Northern clingfish, rex sole, and sculpin

Source: Reimer and Brown 1996
Location: Rogue Reef, Oregon
Season and Year: July 1990
Sample Size: 46
Type: Scats

Prey species	% of Samples
Pacific whiting	96
Pacific lamprey	26
Pacific herring	20
Salmon sp.	**17**

Other species: Jack mackerel, Pacific mackerel, spiny dogfish, Pacific hagfish, Pacific staghorn sculpin, surfperch, and squid

17.2.1.2 Reproduction

Breeding adult Steller sea lion as well as some juveniles, occupy breeding rookeries from late May to early July, with pregnant females usually arriving at the rookery several days before pups are born (Pitcher and Calkins 1981; Gisiner 1985). Pregnant females generally arrive 3 days before pups are born (Gentry 1970) and continue to maintain site fidelity to a rookery of choice (Sandegren 1970). Copulation usually occurs on the breeding territory within two weeks postpartum (Gentry 1970, Sandegren 1970) with the female usually only copulating with one male (Gisiner 1985). Females generally return to rookeries of the birth to pup and breed (NMFS 1992b).

17.2.1.3 Migration and Seasonal Movements

Steller sea lions are not known to migrate, but they do disperse widely outside of the breeding season with males typically dispersing away from their breeding rookeries (NMFS 1992b). During the fall and winter in Alaska, sea lions may occur at rookeries and haulout locations that are used in the summer and they may also be seen away from haulouts or rookeries (NMFS 1992b). Animals marked at rookeries in the Gulf of Alaska have been sighted in southeast Alaska and British Columbia, and animals marked in British Columbia have been seen in Alaska (Calkins and Pitcher 1982, Calkins 1986). Steller sea lions tagged as pups at Rogue Reef in Oregon have been resighted in northern California, Washington, British Columbia, and southeast Alaska (R. Brown, ODFW, unpubl. data).

17.2.2 Habitat Requirements

17.2.2.1 Breeding Habitat

There are four Steller sea lion haulout areas but no rookeries in Washington (Jeffries et al. 20000. The main breeding rookeries for Steller sea lions along the Oregon coast are located at Rogue and Orford Reefs, with relatively small breeding rookeries located at Sea Lion Caves and Three Arch Rocks (Brown 1988, NMFS 1992b). Eight additional haulout sites are used by Steller sea lions in Oregon. Pup numbers have increased with total combined counts at Orford and Rogue Reefs of 1,020 in 1996 and 1128 in 2002 (ODFW, unpubl. data). The total Steller sea lion population (including pups) in Oregon during the breeding season was estimated at about 5,076 animals (R. Brown, ODFW, pers. comm., ODFW, unpubl. data).

17.3 Population & Distribution

17.3.1 U.S. Population

The most recent population estimate for Steller sea lions in the two U.S. stocks are based on combining aerial and ground counts of pups and non-pups made throughout the Steller sea lion range. The estimate for the Western U.S. Stock was 34,595 (Angliss and Lodge 2002) and was based on non-pups counted in 2000 and pups counted in 1998 (Sease et al. 2001, Sease and Loughlin 1999). The estimate for the Eastern U.S. Stock was 31,028 (Angliss and Lodge 2002) and was based on aerial and ground counts of rookeries and haulout sites in California, Oregon, Washington, and British Columbia (Angliss and Lodge 2002).

17.3.2 Distribution

Steller sea lions occur year round in Washington and Oregon nearshore waters, and include both breeding and non-breeding animals. The main breeding rookeries for Steller sea lions along the Oregon coast are located at Rogue Reef and Orford Reef; with relatively small breeding rookeries located at Sea Lion Caves and Three Arch Rocks (Brown 1988, NMFS 1992b). Additional haulout locations in Oregon are located at Cape Arago, Cascade Head and South Jetty of the Columbia River. Although both adult male and female Steller sea lions are present in Washington, no breeding rookeries occur. Haulout locations are found along the outer Washington coast at Split Rock, Carroll Island, Bodelteh Island, Cape Alava and Tatoosh Island (Jeffries *et al*. 2000). Seasonal abundances range from 500-1,500 animals along the outer

Washington coast. Relatively small numbers of Steller sea lions occur at haulout locations in the inland waters of Washington, although 500-1,000 animals move through the Strait of Juan de Fuca and into British Columbia waters annually to feed on herring spawning in the Strait of Geogia north of Nanaimo (P.Olesiuk, Fisheries and Oceans-Canada, pers. comm.).

17.4 Status & Abundance Trends

17.4.1 Status

Steller sea lions are protected under the federal MMPA and ESA, as well as being designated as protected wildlife species by the states of Washington and Oregon (WAC 232-12-011; OAR 635-044-013). Washington and Oregon also list the Steller sea lion as a state "Threatened" species. The Eastern U.S. Stock of Steller sea lions that occur in Washington and Oregon are listed as "Threatened" under the ESA, and are therefore designated as "Depleted" under the MMPA as well (Angliss and Lodge 2002).

17.4.2 Trends

17.4.2.1 U.S. Eastern Stock

In recent years attention has been focused on the U.S. Western Stock due to a precipitous decline since the 1970s of about 85% (Braham et al., 1980, Merrick et al., 1987, Loughlin et al., 1992, Trites and Larkin 1996, Sease et al., 2001) resulting in an "Endangered" classification under the ESA. In the case of the U.S. Eastern Stock that includes animals at rookeries and haulout sites from California, Oregon, Washington, British Columbia, and southeast Alaska, population estimates have in general been increasing (Calkins et al., 1999, Olesiuk 2001, Brown et al., 2002). A number of smaller rookeries and haulout locations in California have declined substantially (Le Boeuf et al., 1991, Hastings and Sydeman 2002). The Eastern U.S. Stock of Steller sea lions that occur in Washington and Oregon are listed as "Threatened" under the ESA.

<u>Washington:</u> In Washington, Steller sea lions occur four major haulout sites along the Olympic Peninsula coast throughout the year. Counts of nonpups have been made during the breeding season during most years since the early 1990's (Figure 1). During that period, numbers of sea lions counted increased on average 9.2%/yr. These animals are assumed to be immature animals and nonbreeding adults associated with rookeries from other areas. Older records suggest the current population in Washington is reduced from historical levels with 2,000-3,000 Steller sea lions reported during August and September of 1914, 1915, and 1916 in the Carroll Island area (Kenyon and Scheffer 1959, Scheffer 1995) while the maximum observed during 60 complete surveys of Washington haulouts between 1980 and 2001 was 1,275 (Steven Jeffries, WDFW, unpubl. data).

<u>Oregon:</u> Along the Oregon coast, Steller sea lion occupy two rookeries at Rogue Reef and Orford Reef, as well as using eight additional haulout sites. The total number of non-pup sea lions counted during breeding season surveys at all of these sites has increased from 1,461 in 1977 to 3,648 in 2001 an annual rate of increase of about 3.7% (R. Brown, ODFW, unpubl. data). Although not nearly as well documented, pup numbers also appear to have increased. In 1996, 685 and 335 pups were counted at Rouge Reef and Orford Reef respectively, while in 2002, 746 and 382 pups were counted at the two sites. The total Steller sea lion population, including pups, associated with Oregon rookeries is estimated at about 5,076 animals based on the 4.5 pup multiplier previously discussed.

With the exception of rookeries in California, the Eastern U.S. Stock of Steller sea lions that includes animals in Washington and Oregon has increased at over 3% annually since the 1970s. The Steller sea lion population in Southeast Alaska, British Columbia, and Oregon has more than doubled in this time. The rookeries at Saint George Reef and Sugarloaf Island in northern California are near levels recorded early in the 20th century.

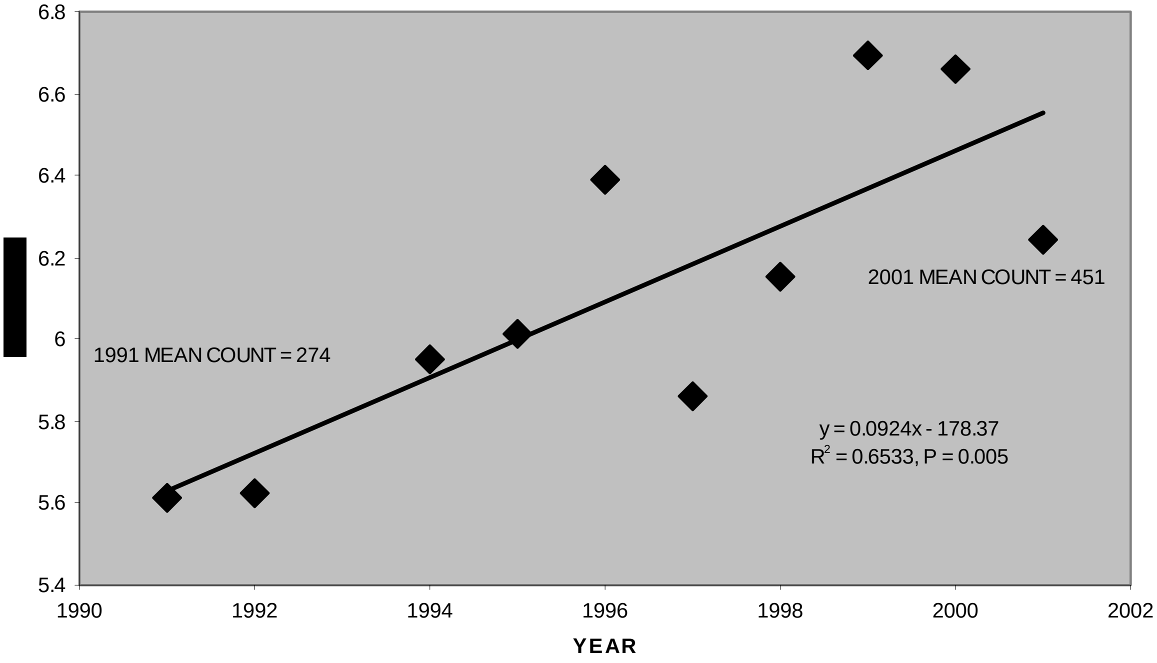

Figure 17-1. Trends in Steller sea lion abundance at Washington haulout sites from 1991 to 2001 (Steven Jeffries, WDFW, unpubl. data).

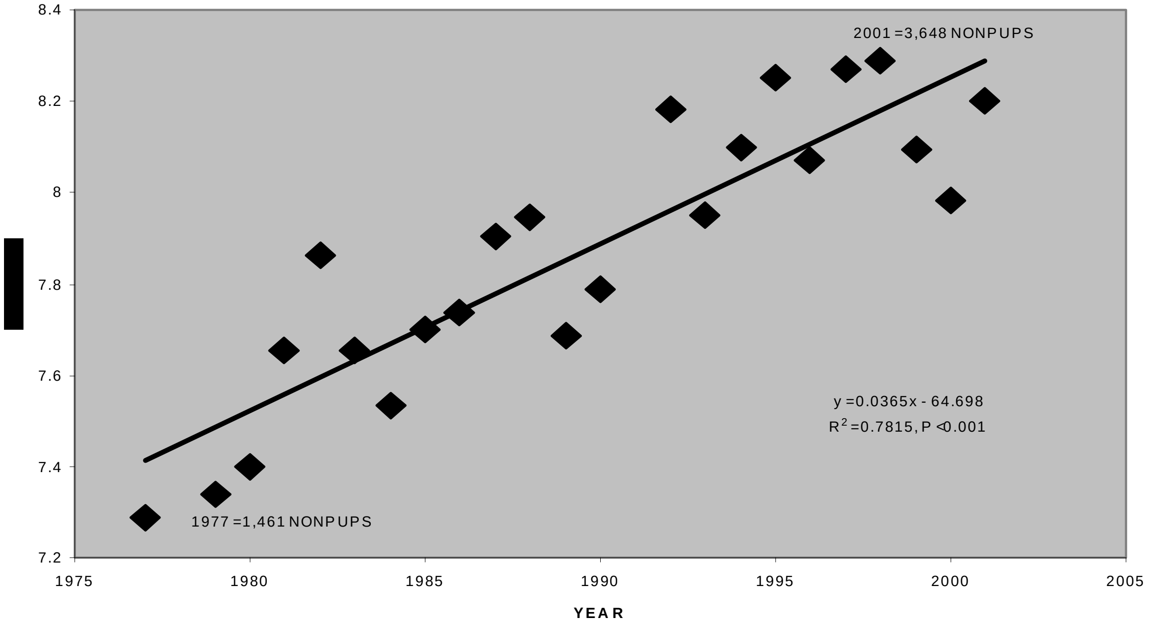

Figure 17-2. Trends in non-pup counts of Steller sea lions at Oregon rookery sites from 1977 to 2001 (R. Brown, ODFW, unpubl. data).

17.4.2.2 Steller Sea Lions in the Columbia River

Historical data on Steller sea lion abundance in Columbia River is limited, although some animals were reported upriver as far as the mouth of the Willamette River (Lyman et al. 2002). Numbers in Washington and Oregon including the Columbia River were believed to be substantially reduced due to extensive human caused mortality, in part stimulated by a bounty (Pearson and Verts 1970). Counts of Steller sea lions at the South Jetty of the Columbia River typically peak during the winter months with peak counts of 50-60 animals reported by Beach et al. (1985). Beach et al. (1985) also reported a small number of Steller sea lions sighted upriver past Tongue Point apparently feeding on eulachon in the winter. Recent surveys by WDFW and ODFW show a substantial increase in Steller sea lion abundance at the South Jetty with peak counts of 300-700 animals recorded (WDFW, unpubl. data, ODFW unpubl. data). In 2003, a small number of Steller sea lions were reported at Bonneville Dam (R. Stansell, USACOE, pers. comm.).

17.4.3 Environmental Conditions

17.4.3.1 Rookery and Hallout Sites

Steller sea lion rookeries are areas where animals congregate for pupping and breeding, and usually occur on beaches of relatively remote islands or reefs (NMFS 1992). Haulout sites are locations used by breeding, non-breeding, and subadult sea lions during the non-breeding season (NMFS 1992b). Similar to California sea lions, Steller sea lion haulout sites are typically associated with jetties, offshore rocks and islands, logbooms, marina docks, and navigation bouys (Jeffries et al. 2000). Although rookeries and haulout sites occur in many types of areas, locations are used on an annual basis and change little from year to year. In Washington, Steller sea lions use haulout sites along the Olympic Peninsula coast in the vicinity of Split Rock, Carroll Island, Cape Alava, and Tatoosh Island (Jeffries et al. 2000). On the Oregon coast, Steller sea lions use rookery sites at Rogue Reef and Orford Reef, and haulout at Rogue Reef, Orford Reef, Cape Arago, Sea Lion Caves, Cascade Head, Three Arch Rocks, and tip of the South Jetty at the mouth of the Columbia River.

17.4.3.2 Seasonal Use

Use of the South Jetty haulout site at the mouth of the Columbia River by Steller sea lions is associated with movements of animals in Oregon and Washington coastal waters associated with their May to July breeding season. At this time, Steller sea lion abundance is lowest as adults return to breeding rookeries at Rogue Reef and Orford Reef, as well as rookeries in British Columbia. Following the breeding season, Steller sea lion abundance increases at the South Jetty and is associated with seasonally abundant prey in the river and offshore such as eulachon, salmon, and Pacific whiting that are important prey (Beach et al. 1985, Fiscus and Baines 1966). Maximum abundances of Steller sea lions at the South Jetty typically have been recorded in fall and winter when seasonally abundant prey ie Pacific whiting, salmon, and eulachon, are present in or near the Columbia River. Although few Steller sea lions move upriver beyond Tongue Point (Beach et al. 1985), a few individuals were reported at Bonneville Dam in 2003 during spring chinook run (R. Stansell, USACOE, pers. comm.).

17.5 Factors Affecting Population Status

Drift gillnet fisheries in the Columbia River have the potential to take Steller sea lions incidental to the fishery (Beach et al. 1985). However, because few Steller sea lions move very far upriver in the Columbia, the likelihood of incidental takes are considered minimal.

New construction, repairs or alteration of jetty design may influence use by Steller sea lions, as the tip of the South Jetty is regularly used as a haulout location.

17.6 Inventory & Assessment of Existing Management and Conservation Plans

The MMPA as well as the Endangered Species Act protect Steller sea lions. Both species are also protected by state regulations (WAC 232-12-011 and OAR 635-044-013).

A recovery plan exists for Steller sea lions under the ESA (NMFS 1992b).

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Volume III, Chapter 18
Harbor Seals

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18.0 Habor Seals (*Phoca vitulina*)

18.1 Introduction

The harbor seal (*Phoca vitulina*) is a small, stocky phocid seal found throughout the temperate and arctic waters of the northern hemisphere, and has the widest distribution of any pinniped. In the Pacific Ocean, harbor seals inhabit coastal and estuarine waters from Baja California, north along the western coast of the continental U.S., British Columbia, and southeast Alaska, west through the Gulf of Alaska and Aleution Islands to Russia and Japan, and in the Bering Sea north to Cape Newenham and the Pribolof Islands (Carretta *et al*. 2002). Harbor seals generally considered non-migratory, with local movements associated with daily and seasonal variation in tides, weather, prey availability and reproduction (Scheffer and Slipp 1944; Fisher 1952; Bigg 1969a; Bigg 1973; Jeffries 1984; Jeffries 1985; Jeffries 1986). Harbor seals are considered a non-migratory species, breeding and feeding in the same general area throughtout the year (Scheffer and Slipp 1944; Bigg 1969a).

Harbor seals are the most common, widely distributed pinniped found in nearshore waters of Washington and Oregon, and use hundreds of sites to rest or haulout including intertidal sand bars and mudflats in estuaries, intertidal rocks and reefs, sandy, cobble, and rocky beaches, islands, logbooms, docks and floats. Group sizes typically range from small numbers of animals on some intertidal rocks to several thousand animals found seasonally in coastal estuaries. Males and females are similar in size (to 250 lbs) and coloration. Pelage patterns are typically a light base pelage with dark spots, although some individuals have a pelage that is reversed in coloration with dark base and light spots.

During the first half of the twentieth century, numbers of harbor seals were severely reduced in Washington and Oregon by a state-financed population control programs that considered harbor seals to be salmon predators in direct competition with commercial and sport fishermen. The Washington Department of Fisheries paid a bounty for seals and sea lions until 1960, and the Oregon Fish Commission maintained a Columbia River Seal Control Program which paid a seal hunter to kill and control seal numbers in the Columbia River until 1970 (Pearson and Verts 1970; Newby 1973). After the bounty and control ceased and with federal protective status established with passage of the Marine Mammal Protection Act (MMPA) in 1972, Washington and Oregon harbor seal populations began to recover.

As managed by National Marine Fisheries Service (NMFS) under the MMPA, harbor seals in Washington and Oregon have been separated into coastal and inland stocks because of differences in cranial morphology, pupping phenology, and genetics (Temte 1986; Lamont *et al*. 1996; Carretta *et al*. 2002). The Oregon/Washington Coast Stock includes all harbor seals from the California/Oregon border to Cape Flattery on the Olympic Peninsula of Washington (Carretta *et al*. 2002). Harbor seals in the Columbia River are part of the Oregon/Washington Coast Stock.

18.2 Life History & Habitat Requirements

18.2.1 Life History

18.2.1.1 Diet

Harbor seals are considered opportunistic feeders and eat a wide variety of fish, cephalopods, and crustaceans. In northwest waters, fish species commonly eaten by harbor seals include Pacific herring, northern anchovy, various salmon species, various codfish species, flatfish species, pricklebacks, greenlings, and sculpins (Scheffer and Slipp 1944; Bigg 1969a; Beach *et al*. 1985; Brown *et al*. 1989; Olesiuk 1993; NMFS 1997).

In general, the waters of the Columbia River estuary provide a variety of food for harbor seals with important prey items in their diet consisting of schooling fishes (Pacific whiting, smelts, herring), various flatfish, lamprey and salmonids (Beach *et al*. 1985; Reimer and Brown 1997; NMFS 1997). Important year-round prey of harbor seals in the Columbia River include longfin smelt, staghorn sculpin, Pacific tomcod, English sole, starry flounder, snake prickleback and Pacific herring. Seasonally important prey species include eulachon, Pacific herring, salmon, staghorn sculpin, Northern anchovy, and a variety of flatfish species (see Table 18-1 and Beach *et al*. 1985; Reimer and Brown 1997; Brown *et al*. 1989).

During winter months, eulachon make up the majority of prey consumed by harbor seals in the Columbia. Frequent foraging on oily prey like eulachon, northern anchovy and Pacific herring is considered important due to their seasonal abundance, high caloric content and energetic value to female harbor seals for blubber deposition needed during lactation (Jeffries 1984). Annual shifts in abundance of the regional harbor seal population from Grays Harbor, Willapa Bay, Tillamook Bay and Netarts Bay during the winter were correlated with the winter eulachon run when peak abundance of harbor seals occur in the Columbia (Jeffries 1984). Harbor seal abundance in the Columbia declines in spring and summer, when the population shifts to adjacent estuaries that provide protected haulout sites, relatively shallow feeding areas and preferred prey for females with pups.

Browne *et al*. (2002) reported that based on analysis of scats collected seasonally from haulout sites in the lower Columbia River near Astoria, harbor seals consumed adult and juvenile salmonids throughout the year, but with greatest frequency in the spring. Although identification of salmon species is difficult using otoliths and other skeletal remains found in harbor seals scats is difficult, Browne et al. (2002) also reported in their study that the most common age and species of salmon consumed by harbor seals were juvenile chinook salmon.

Browne *et al*. (2002) also point out the inherent difficulties of determining salmon species and age classes using otoliths and other skeletal structures and suggest genetic identification techniques may provide a valuable tool to better quantify importance of various salmon species in the diet of harbor seals.

Table 18-1. Summary of food habit studies for harbor seals in the Columbia River. Prey species indicated occurred in more than 10% of samples except for salmonids.

Source:	Beach *et al.* 1985
Season and Year:	Year-round 1980-82
Sample Size:	436
Type:	Scats
Prey species	% of Samples
Whitebait smelt	36
Northern anchovy	21
Pacific lamprey	14
Flatfish spp.	12
Gadids	12
Staghorn sculpin	11
Eulachon	10
Salmonids	**6**
Steelhead	<1
Other species: Pacific herring, Bay goby, Snake prickleback, Pacific whiting, Lingcod, Pile perch, Shiner perch, Kelp greenling	
Source:	Brown *et al.* 1989
Season and Year:	Winter 1986-88
Sample Size:	83
Type:	Gastrointestinal tracts
Prey species	% of Samples
Eulachon	100
Longfin smelt	14
Pacific lamprey	10
Source:	Reimer and Brown 1997
Season and Year:	Winter and Spring 1992-93
Sample Size:	51
Type:	Scats
Prey species	% of Samples
Eulachon	84
Pacific lamprey	20
Starry Flounder	12
Other species: Staghorn sculpin, Pacific herring, Whitebait smelt, Longfin smelt, Pacific sandlance, Pacific tomcod, Pacific whiting	

Source:	Reimer and Brown 1997
Season and Year:	Fall 1994
Sample Size:	36
Type:	Scats
Prey species	% of Samples
Northern anchovy	50
Pacific herring	44
Salmonids	**39**
Smelt spp.	25
Staghorn sculpin	19
Other species: Pacific whiting, Pacific lamprey, Rex sole, Whitebait smelt, Pacific sandlance, Peamouth, Surfperch, Shiner perch, Pacific mackeral	

Source:	Reimer and Brown 1997
Season and Year:	Spring 1995
Sample Size:	67
Type:	Scats
Prey species	% of Samples
Staghorn sculpin	49
Starry flounder	36
Pacific herring	28
Salmonids	**19**
Smelt spp.	18
Pacific lamprey	16
Snake Prickleback	15
Other species: Pacific sand lance, River lamprey, Shiner perch, Pile perch, Surfperch, Peamouth, Northern anchovy, Whitebait smelt, American shad, Pacific whiting, Threespine stickleback	

18.2.1.2 Reproduction

Harbor seals have an annual reproductive cycle with the birth season typically lasting up to two months (Scheffer and Slipp 1944; Bigg 1969b; Bigg and Fisher 1974). Females produce one pup per year, beginning at age four or five. Pups are precocious at birth, capable of swimming and following their mothers into the water immediately after birth. Pups typically remain with their mothers until weaning at 4-6 weeks of age and following weaning feed on their own (Scheffer and Slipp 1944; Bigg 1969a; Bigg 1973). Lactation lasts from two to six weeks followed by estrous, ovulation and mating, then blastocyst implantation up to three months later (Bigg 1969a; Bigg 1969b; Bigg 1973; Bigg and Fisher 1974; Tempte 1986). Experimental studies suggest that over the range of harbor seals each population maintains its unique reproductive timing through a specific response to photoperiod, the existence of an annual endogenous reproductive rhythm, genentic uniqueness and availability of abundant prey following weaning (Bigg 1969b; Bigg 1973; Bigg and Fisher 1974; Tempte 1986; Tempte *et al*. 1991; Lamont *et al.* 1996; Westlake and O’Corry-Crowe 2002).

Harbor seal pupping season varies by geographic area, with pups born along the Washington and Oregon coast including the Columbia River from mid-April through June (Jeffries 1984; Jeffries 1985; Huber *et al*. 2001). Coastal estuaries of Oregon and Washington (including Netarts Bay, Tillamook Bay, Willapa Bay and Grays Harbor) are important pupping areas that provide protected haulout and nursing areas for mothers and pups, as well as shallow feeding areas with abundant prey for weaned pups. Although harbor seal pups are born in the Columbia River, adjacent estuaries along the Oregon and Washington coast are used more extensively for pupping by the regional harbor seal population (Beach *et al*. 1985; Jeffries 1986).

18.2.1.3 Migration and Seasonal Movements

Harbor seals are considered non-migratory although movements in response to seasonally abundant prey, ie to the Columbia during winter eulachon runs and into adjacent esturaries along the Oregon and Washington coasts (Netarts Bay, Tillamook Bay, Willapa Bay and Grays Harbor) during pupping season, have been reported (Beach *et al*. 1985; Jeffries 1986).

18.2.2 Habitat Requirements

The harbor seal in the most common marine mammal found in nearshore waters of Washington and Oregon, and is especially numerous in bays and esturaries including the Columbia River. Preferred haulout locations are usually in areas where access to deepwater channels is maintained. Typically these locations include intertidal sandbars, mudflats, offshore rocks and reefs. Artificial haulout sites such as docks, floats and logbooms are regularly used in some locations (Scheffer and Slipp 1944; Bigg 1969a; Jeffries *et al*. 2000).

Historically, harbor seals were reported upriver at Celilo Falls by the Lewis and Clark Expedition (Cutright 1989; Moulton 1990), as well as being found at a number of upriver prehistoric archaeological sites (Lyman et al. 2002). This upriver movement in the Columbia River suggests harbor seals were following returning runs of salmon upriver and feeding in areas of concentration and restricted passage such as Celilo Falls.

18.3 Population & Distribution

18.3.1 Population

The harbor seal population in the north Pacific is estimated between 222,000 and 235,000 animals consisting of 28,000 in the California Stock, 25,000 in the Oregon/Washington Stock, 15,000 in the Washington Inland Stock, 75,000-88,000 in British Columbia, 37,000 in the Southeast Alaska Stock, 29,000 in the Gulf of Alaska Stock and 13,000 in the Bering Sea Stock (Carretta et al. 2002; Angliss and Lodge 2002).

18.3.2 Distribution

18.3.2.1 Winter and Spring (Non-Breeding Season)

Peak harbor seal abundances in the Columbia River occur during the winter and spring when a number of upriver haulout sites are used. Peak abundances and upriver movements in the winter and spring months are correllated with spawning runs of eulachon smelt and outmigration of salmonid smolts (Beach *et al*. 1985; Jeffries 1986; NMFS 1993).

18.3.2.2 Summer and Fall (Pupping and Molt Season)

Following the decline and disappearance of eulachon in the Columbia, harbor seals discontinue use of upriver haulout sites and move back downriver. At this time only haulout sites at Desdemona Sands, shoals north of Tongue Point, in Grays Bay and Cathlamet Bay were used by harbor seals (Beach *et al*. 1985; Jeffries 1986; WDFW unpubl. data; ODFW unpubl. data).

By mid-April, harbor seal pupping begins in the Oregon/Washington Coast Stock with important pupping and nursery areas located in estuaries adjacent to the Columbia River ie Netarts Bay, Tillamook Bay, Willapa Bay and Grays Harbor (Beach *et al*. 1985; Jeffries 1986). At this time harbor seal abundance in the Columbia River has decreased and coincides with movements and increases in harbor seal abundance in adjacent estuaries (Jeffries 1986). In the Columbia, abundance declines to annual lows with 800-1200 seals using haulout sites in the lower river at this time.

Harbor seal pupping season is followed by an annual molt cycle that occurs from earlyJuly through September. At this time, harbor seal counts remain high in adjacent estuaries. By late September, counts in all areas decrease to seasonal lows. Counts remain relatively low until winter increases begin in the Columbia River as seals move into the river to feed on eulachon (Jeffries 1986).

18.4 Status & Abundance Trends

18.4.1 Status

Harbor seals are protected under the federal Marine Mammal Protection Act (MMPA) as well as being designated as protected wildlife species by the states of Washington and Oregon (WAC 232-12-011; OAR 635-044-013). Harbor seals in the Columbia River are part of the Oregon/Washington Coast Stock as defined under MMPA regulations (Carretta *et al*. 2002). The Oregon/Washington Coast Stock of harbor seals is not considered as “depleted” under the MMPA or listed as “threatened” or “endangered” under the ESA (Carretta *et al*. 2002).

18.4.2 Trends

18.4.2.1 Oregon/Washington Coast Stock

Harbor seal numbers were severely reduced in the early 1900s by bounty hunters under state-financed control programs that considered harbor seals to be predators in direct competition with commercial and sport fishermen. After the bounty program ceased in 1960 and with federal and state protection, the harbor seal populations in Washington and Oregon began to recover.

Newby (1973) estimated that a total of 2,000–3,000 harbor seals resided in Washington in the early 1970s. In the late 1960s, Pearson and Verts (1970) conducted shorebased surveys of the Oregon coast including the Columbia River and estimated fewer than 500 harbor seals were present in Oregon with fewer than 100 present in the Columbia River. Beginning in late 1970s, systematic surveys of harbor seal populations in Washington and Oregon were initiated by various researchers, including biologist from WDFW, ODFW and NMFS (Brown and Mate 1983; Brown 1997; Jeffries 1984; Jeffries 1985; Jeffries *et al*. 2003).

Aerial surveys for harbor seals along the Washington and Oregon coast that included the Columbia River were conducted by WDFW, ODFW and NMFS, and were typically flown during the pupping season when maximum numbers were onshore. Data collected during surveys included date, time, location, a visual estimate of seal numbers, and photographs of all sites where more than 25 seals were hauled out. Total number of seals (including pups) present at each site was counted from slides (Jeffries 1984; Jeffries 1985; Brown 1997; Jeffries *et al.* 2003). Some proportion of the seals remain in the water and these seals were missed by the aerial surveys (Jeffries 1985; Huber *et al*. 2001). A correction factor was used to adjust counts of seals hauled out to estimate total numbers of seals (Jeffries 1985; Huber *et al*. 2001; Jeffries *et al*.

2003). Because a large proportion of harbor seals haul out onto land in discrete aggregations at specific times, a count of hauled out seals provides a precise measure of population trend.

Aerial surveys during the 1999 pupping season resulted in a mean count of 16,165 harbor seals in the Oregon/Washington Coast Stock. Using a correction factor of 1.53 to account for seals in the water and missed during surveys results in a population estimate of 24,732 harbor seals in the Oregon/Washington Coast Stock (Carretta *et al*. 2002; Jeffries *et al*. 2003; ODFW, unpubl. Data). Results of surveys from the late 1970s to 1999 indicate growth of the Oregon/Washington Coast Stock has slowed and may have reached equilibrium (Brown 1997; Carretta *et al*. 2002; Jeffries *et al*. 2003). The Oregon/Washington Coast Stock including the Columbia River has reached an apparent equilibrium indicating the harbor seal is most likely at or near carrying capacity (Brown 1997; Jeffries *et al*. 2003).

18.4.2.2 Harbor Seal Trends in the Columbia River

Following federal and state protection of harbor seal populations, numbers of seals present in the Columbia River increased during the 1970's and 1980's. During this period, the harbor seal population grew at 6-10 percent annually (NMFS 1997; Brown 1997; Jeffries *et al*. 2003). Based on analysis of regional pupping season counts (Jeffries *et al*. 2003; see Figure 18-1), harbor seal populations began to level off and reached equilibrium in the early 1990's. Since then, harbor seal abundance has changed little with counts of 800-1200 seals recorded during annual surveys.

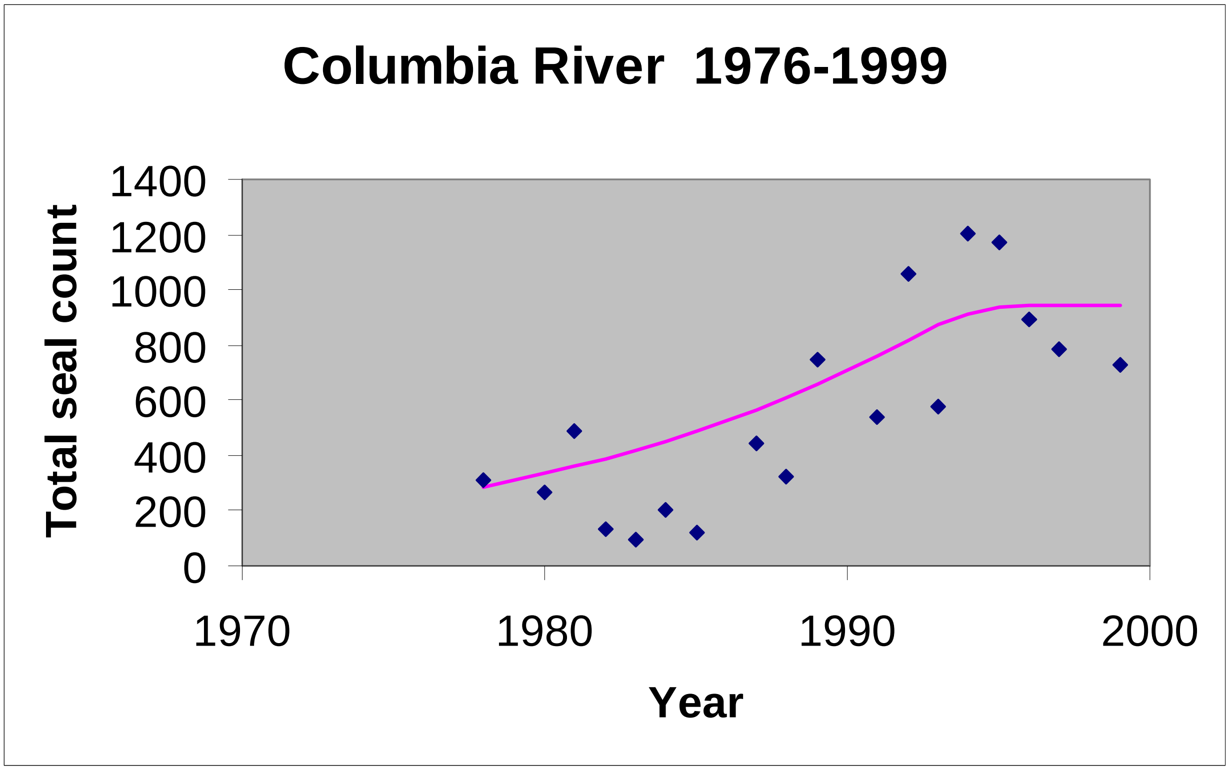

Figure 18-1. Generalized logistic trend line fit to annual harbor seal pupping season counts for the Columbia River from 1976-1999 (Beach et al. 1985; WDFW unpubl. data; Jeffries *et al*. 2003).

18.4.3 Environmental Conditions

18.4.3.1 Haulout Sites

Harbor seals occur throughout the year in the Columbia River with numerous haulout sites found on intertidal mudflats and sand bars in the lower river. Highest numbers of harbor seals use the haulout sites on Desdemona Sands, Taylor Sands and Miller Sands. During periods of peak abundance in the winter these sites may have between 500-2000 seals hauled out on them. Additional haulout sites used by smaller numbers of seals (10-300 seals) were identified at the South Jetty; in Baker Bay; in Grays Bay; in Cathlamet Bay, and below Woody Island. The farthest upriver haulout sites were located near Wallace Island and at the mouth of the Cowlitz River near Longview (Jeffries 1984; Jeffries 1986; Jeffries *et al*. 2000; WDFW unpubl. data; ODFW unpubl. data). Use of these upriver sites is correlated with upriver movement of harbor seals feeding on eulachon smelt runs (Jeffries 1985).

18.4.3.2 Seasonal Use

Use of haulout sites in the Columbia River varies seasonally with peak abundances of 2,000-2,500 seals occurring during winter months corresponding to annual eulachon smelt runs into the river. At this time the largest groups of harbor seals are present in the Columbia River with groups of 800-1200 seals regularly using haulouts near Desdemona Sands and north of Tongue Pt (WDFW unpubl. data; ODFW unpubl. data). During these periods of peak abundance in the winter, harbor seals use haulout sites in the lower Columbia River, as well as moving upstream to use haulout sites above Woody Island, near Wallace Island and at the mouth of the Cowlitz River (Jeffries 1984; Beach *et al*. 1985; Jeffries 1986).

Although pups are born in the Columbia River, relative abundance of harbor seals declines in late spring and summer during the pupping season corresponding to the movement of pregnant females into preferred pupping and nursery areas in adjacent estuaries (Netarts Bay, Tillamook Bay, Willapa Bay and Grays Harbor) (Jeffries 1986). However, during the pupping season the greatest number of haulout sites are used by harbor seals in the Columbia River as pregnant females and females with pups segregate into nursery areas. In the Columbia, pupping and nursery areas are located in Cathlamet Bay, Baker Bay and near lower Woody Island.

18.5 Factors Affecting Population Status

Drift gillnet fisheries in the Columbia River have taken harbor seals incidental to the fishery with highest incidental takes likely to occur during winter chinook salmon fisheries when harbor seal abundance is greatest (Beach *et al*. 1985; Matteson *et al*. 1993). In recent years with reduced gillnet seasons the level of incidental mortality of harbor seals in Columbia River has been considered minimal (Carretta *et al*. 2002).

Alteration of Columbia River shoals and sandbars used as haulout sites by harbor seals is possible due to dredging activities by USACOE. Dredging activities have the potential to destroy or remove existing haulout sites which would likely result in shifting of harbor seals to other haulout areas or use of new sites.

18.6 Inventory & Assessment of Existing Management & ConservationPlans

Harbor seals are protected by the federal Marine Mammal Protection Act and state regulations (WAC 232-12-011 and OAR 635-044-013).

No federal or state management plan exists for harbor seals.

No federal or state restoration or conservation plan exists for harbor seals.

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