Volume II, Chapter 2
Columbia River Estuary and Lower Mainstem Subbasins

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2.0 Columbia River Estuary and Lower Mainstem

This chapter describes physical processes, habitat, fish and wildlife species, and ecological relationships within the lower Columbia River mainstem (i.e. below Bonneville Dam) and estuary. A balanced and complete ecosystem-based approach was desired for this assessment, however, was not possible based on currently available data. Certain topics are discussed in far greater detail than others because of this difference in data availability. For example, the estuary is discussed in detail throughout the chapter, while specific discussions regarding the lower mainstem are not presented, simply because the data do not exist. In the same regard, considerable research has focused on salmonid species in the Columbia River while much less is known about the other species presented here.

Another necessary point of clarification is the use of the word *estuary*, which was not standardized across all previous research efforts. For our purposes, the Columbia River estuary was defined as the tidally influence portion of the Columbia River from the mouth to Bonneville Dam (rm 146) as well as the Columbia River plume. However, many other studies have defined the estuary differently. For example, some define the estuary upper boundary as the extent of salt water intrusion (typically Harrington Point at rm 23) while others define the upper boundary as the extent of river flow reversal (up to Oak Point at rm 53). Also, recent research suggests that the Columbia River plume environment should also be considered as part of the estuary. Thus, when presenting the work of others, *estuary* refers to the estuary boundaries described by the research and the reader is encouraged to review the original publication to alleviate any confusion as to which part of the estuary is being discussed. Where possible, clarification was provided to indicate if the information being presented applied to the tidal freshwater portion of the lower mainstem (i.e. rm 46-146), the lower portion of the river (rm 0-46), or the Columbia River plume.

The geographic area covered in this subbasin assessment and qualitative analysis includes the Columbia River estuary and the lower Columbia River up to Bonneville Dam (Figure 2-1, Figure 2-2, and Figure 2-3); the major tributaries are not included in this analysis as they have been designated as subbasins by the Northwest Power and Conservation Council (NPCC) and are addressed separately in this Technical Foundation. The description and analysis, however, focuses on the Columbia River estuary by default; far more research to date has focused on the estuary and not the tidal freshwater portion of the lower mainstem. Where possible, data specific to the lower Columbia River mainstem were included; elsewhere, assumptions where made as to whether the habitat conditions, habitat-forming processes, and species-habitat interactions in the estuary were also applicable to the lower Columbia River mainstem.

Figure 2-1. Large-scale map of the lower Columbia River mainstem and estuary, depicting major tributaries and population centers (R2 2003).

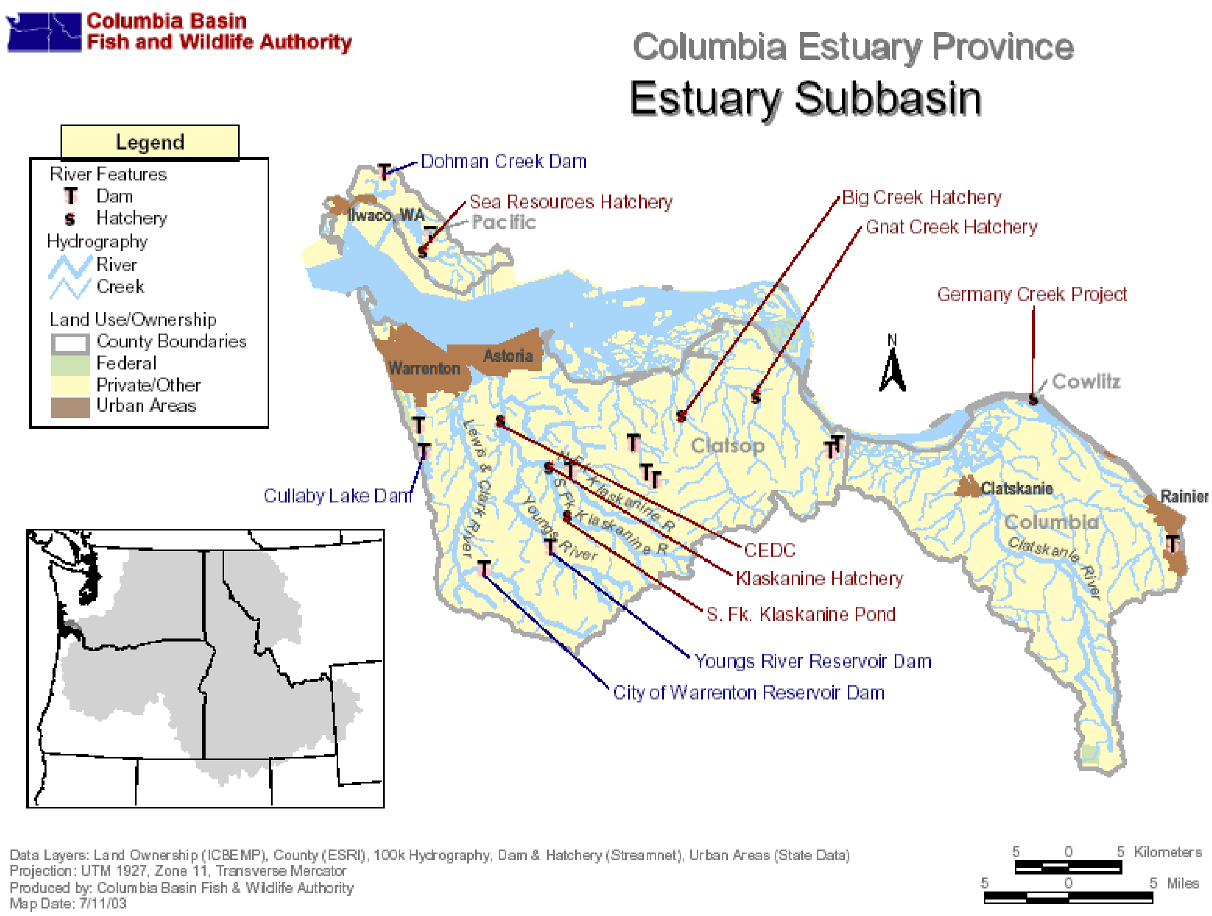


Figure 2-2. Boundaries of the Columbia Estuary Subbasin as defined by the Northwest Power and Conservation Council.

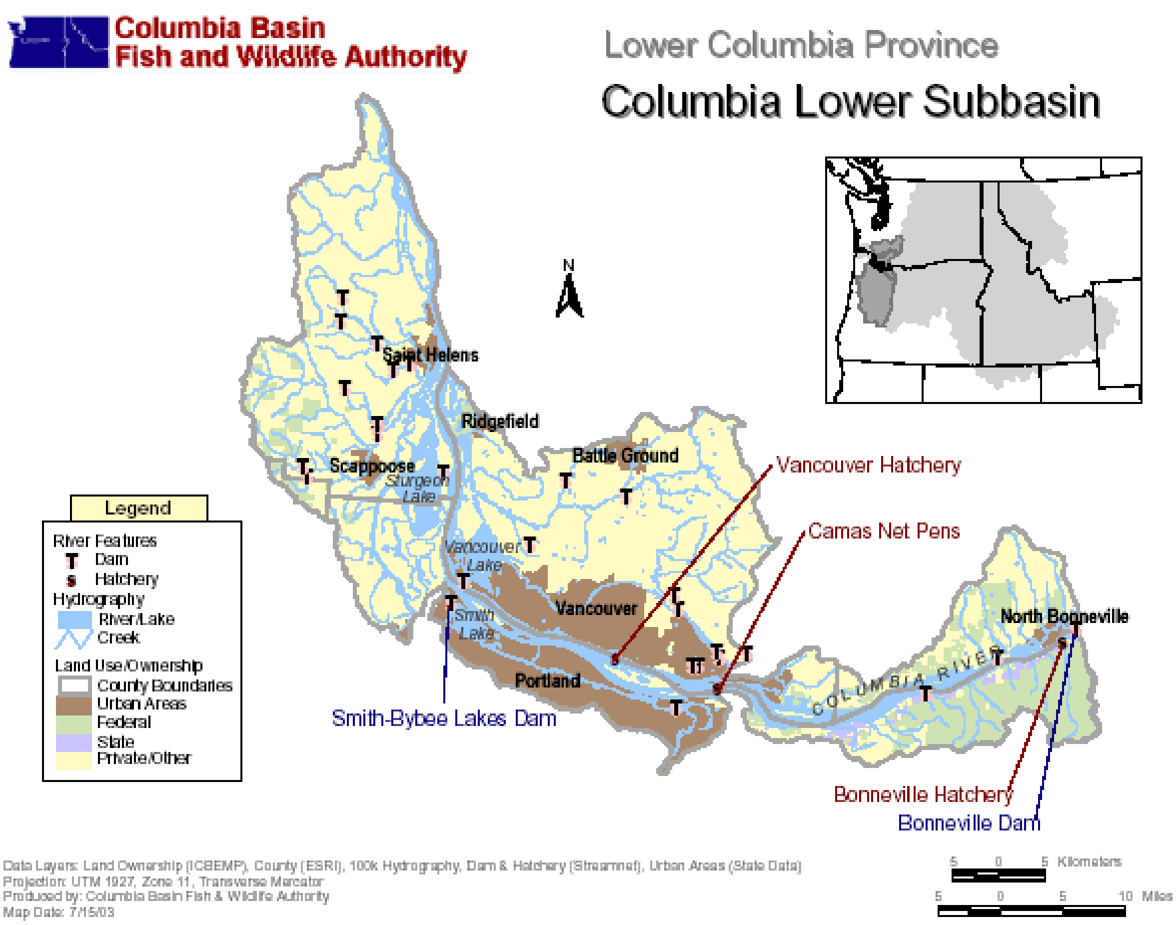


Figure 2-3. Boundaries of the Lower Columbia Subbasin as defined by the Northwest Power and Conservation Council.

This chapter is organized into the following sections: 2.1 Subbasin Description, 2.2 Focal Species, 2.3 Habitat, 2.4 Species/Habitat Interactions, 2.5 Ecological Relationships, 2.6 Knowledge Gaps, 2.7 Hypothesis Statements. Section 2.1 Subbasin Description provides the context for the subbasin assessment as well as an overview of the physical setting, fish and wildlife resources, and habitats in the lower Columbia River mainstem and estuary subbasin. Section 2.2 Focal Species describes the selection process for identifying focal species and provides a brief description of each species status and abundance trends as well as life history as it relates to the potential use of lower mainstem and estuary habitats. Section 2.3 Habitat discusses the physical processes that create habitats in the lower mainstem and estuary, identifies the natural and anthropogenic factors that have affected habitat change in the lower mainstem and estuary, and compares the historical and modern day acreage of specific habitat types. Section 2.4 Species/Habitat Interactions presents the association of focal species with lower mainstem and estuary habitats. Further, this section discusses potential relationships between lower mainstem and estuary habitat change and focal species, particularly salmonids. Section 2.5 Ecological Relationships briefly discusses potential ecological interactions among native and exotic species in the Columbia River estuary and lower mainstem . Section 2.6 Knowledge Gaps identifies and prioritizes critical areas where we lack adequate understanding of linkages between lower mainstem and estuary habitats and focal species; the section also acknowledges the on-going development of tools designed to describe physical and biological processes in the estuary. Finally, Section 2.7 Hypothesis Statements presents a series of hypotheses that are intended to summarize our current knowledge of estuary processes, habitat condition, and focal species; collectively, the hypotheses constitute the working hypothesis of the subbasin assessment as defined by the Northwest Power Planning Council (2001).

2.1 Subbasin Description

The subbasin description is divided into the following sections: 2.1.1 Purpose, 2.1.2 History, 2.1.3 Physical Setting, 2.1.4 Fish and Wildlife Resources, 2.1.5 Habitat Classification, 2.1.6 Estuary and Lower Mainstem Zones, 2.1.7 Major Land Uses, and 2.1.8 Areas of Biological Significance. Section 2.1.1 Purpose describes the purpose of this subbasin assessment in the context of the Northwest Power and Conservation Council (NPCC; formerly Northwest Power Planning Council) subbasin planning process and how the chapter integrates with the Washington Lower Columbia River Fish Recovery Plan Technical Foundation. Section 2.1.2 History provides a brief description of the rich history of the subbasins. Section 2.1.3 Physical Setting describes the general physical of the subbasins. Section 2.1.4 Fish and Wildlife Resources provides a species list of the fish and wildlife species known to occur in the subbasins. Section 2.1.5 Habitat Classification describes estuary and mainstem habitat types, the abundance of habitat classification systems available to describe habitat, the habitat classification systems utilized in this analysis, and the potential relationship among each habitat classification system. Section 2.1.6 Estuary and Lower Mainstem Zones describes geographic estuary and mainstem areas utilized to facilitate subsequent discussions of habitat change. Section 2.1.7 Major Land Uses identifies the variety of human activities that occur within the subbasins. Section 2.1.8 Areas of Biological Significance identifies areas that provide critical natural habitats and help maintain the delicate balance of the ecosystem.

2.1.1 Purpose

In the context of the NPCC subbasin planning process, this chapter is intended to serve as the Subbasin Assessment portion of the Columbia River Estuary and Lower Mainstem Subbasin Plan. As such, this subbasin assessment will provide an overview of the subbasins (Section 2.1), describe focal species (Section 2.2), environmental conditions (Section 2.3), and ecological relationships (Sections 2.4 and 2.5), identify limiting factors (Sections 2.3, 2.4, and 2.6), and provide a synthesis of the information (Section 2.7). This subbasin assessment will not include a complete inventory of existing activities in the lower Columbia River mainstem and estuary nor will it present a Management Plan for the subbasins; these are both future activities in the subbasin planning process. Thus, components of a Management Plan, such as biological objectives or a research, monitoring, and evaluation plan, will not be developed here. From the perspective of subbasin planning, the most important outcome of the subbasin assessment is the development of the working hypothesis (Section 2.7); all of the other information presented in the assessment provides a means to that end. The working hypothesis provides a metric of our current understanding of the subbasins and serves as the link between the subbasin assessment and the future management plan.

This chapter describes two of the eleven subbasins considered in the Washington Lower Columbia River Fish Recovery Plan Technical Foundation. To avoid repetition, references are used throughout this chapter if the topic has been discussed in more detail in the Technical Foundation. Primary reference to the Technical Foundation occurs in the abundance trends and life history description of focal species. Additionally, the Technical Foundation includes a detailed discussion of the salmonid limiting factors common across the subbasins.

2.1.2 History

By the early 1800s, approximately 50,000 Native Americans (primarily the Chinooks) inhabited villages scattered along the banks of the Columbia River (Cone and Ridlington 1996, Thompson 2001). Paleological records indicate that people in the region harvested Pacific salmon as early as 9,000 years ago (Lichatowich 1999). The Chinook peoples were skilled traders and the Columbia River served as a major trade route; tribes came from inland valleys and as far away as the Great Plains to trade for salmon and other valuable resources (Thompson 2001). Estimates indicate that the Chinookan peoples harvested almost 41 million pounds of salmon annually, much of which was traded to interior tribes (Cone and Ridlington 1996).

As early as 1543, European explorers ventured along the Oregon coast, but failed to find the mouth of the Columbia River. Finally, in 1792, Captain Robert Gray of the United States sailed across the bar at the mouth of the river and explored the vicinity of Astoria. Later, William Robert Broughton, a Spanish lieutenant, mapped and named many features of the lower Columbia River as far upriver as the Portland area (Miller 1958).

In 1803, Meriwether Lewis and William Clark began an expedition in St. Louis with the intent of finding a trade route across the continent to the Orient. By 1805, the expedition reached the lower Columbia River, making contact with the native people. After this expedition, European settlement in the region advanced rapidly; the Hudson Bay Company played a substantial role in establishing trade with the native people. In 1840, 'Oregon Fever' brought many settlers from the Mid-West; timber and fisheries became the driving forces behind European settlement of the region.

Earliest accounts of European exploitation of salmon date around 1830, when salmon were dried and salted for storage and distribution. The salmon industry began to realize its full potential when the first cannery began operating in Eagle Cliff, WA, in 1867; many other canneries began operating over the next decade and by 1883, there were 55 canneries on or near the Columbia River. Initially, chinook salmon were the primary catch, but fisheries began harvesting other salmon by the late 1800s; catch of all species peaked at 47 million pounds in 1911 (Cone and Ridlington 1996).

Introductions of exotic fish species had substantial impacts on early fisheries. For example, American shad were introduced to San Francisco in 1871; by 1903, Columbia River fisherman reported that shad had become so numerous they were a nuisance. Other species (i.e. warm-water fish such as bluegill, crappie, and bass) were becoming increasingly abundant in the lower reaches of many Columbia River tributaries and slough habitats of the lower mainstem Columbia River; these sloughs are ideal habitats for these warm water species (Fies 1971).

Concomitant to the growth of the fishing industry, the timber industry was experiencing a boom. Timber industry practices included the removal of stream debris, temporary construction of splash dams to store timber, and log drives that flushed timber through the system as freshet flows blasted the splash dams (Farnell 1980). Although efficient and inexpensive, such practices destroyed instream and riparian habitat. Log drive practices were eliminated by 1914, but other logging practices (such as the lack of riparian buffers) continued to negatively affect fish and wildlife habitat, including that of salmonids.

Early settlers maintained farms for subsistence; initially, commercial farming was not a major industry. By the late 1800s, a substantial amount of acreage in the subbasin had been cleared of trees, burned, and converted to agricultural land; much of this land conversion was occurring in the lower Columbia River floodplain and the interior valleys. Many of these floodplain areas remain in agricultural use today.

Since the late 1800s, the US Army Corps of Engineers has been responsible for maintaining navigation safety on the Columbia River. In 1878, Congress directed the Corps to maintain a 20-foot minimum channel depth, authorizing the Columbia River navigation channel project. To maintain this channel depth, periodic dredging was required in a few shallow reaches where controlling depths ranged from 12-15 feet (USACE 1999). At the mouth of the Columbia River, construction of the south jetty began in 1885; an extension to the original south jetty began in 1903 and was completed in 1914 (Sherwood et al. 1990). Meanwhile, construction of the north jetty began in 1913 and was completed in 1917 (Sherwood et al. 1990). Additionally, use of pile dikes to assist in channel depth maintenance began in the lower Columbia River in 1885 at St. Helens Bar; other early dikes included Martin Island Bar and Walker Island Bar in 1892-93. Over time, Congress continually authorized increases to the minimum navigation channel depth and width: 1899 – depth authorized to 25 ft; 1912 – depth authorized to 30 ft, width established at 300 ft; 1930 – depth authorized to 35 ft, width authorized to 500 ft, channel course was realigned in some reaches; 1936-1957 – periodic channel alignment adjustments; 1962 – depth authorized to 40 ft; 1999 – depth authorized to 43 ft. Most of the current pile dike system was during the periods 1917-1923 and 1933-1939; the existing system consists of 256 dikes totaling 240,000 linear feet (USACE 2001).

In the early 1930s, the Columbia River was slated for development of the next major federal hydropower project; Bonneville Dam began operation in the late 1930s, affecting salmonid access to spawning habitat above Bonneville Dam. With extensive hydroelectric development, the lower Columbia River was quickly viewed as a production zone for salmon. Mitigation for the loss of habitat caused by dams came in the Mitchell Act of 1948, which created a system of hatcheries on the Columbia River. Although the some of the first hatcheries where generally unsuccessful, hatcheries were viewed as the solution to overfishing, habitat loss, and hydroelectric development.

2.1.3 Physical Setting

The Columbia River estuary has formed over geologic time by the forces of glaciation, volcanism, hydrology, and erosion and accretion of sediments. Circulation of sediments and nutrients throughout the estuary are driven by river hydrology and coastal oceanography. Sea levels have risen since the late Pleistocene period, which has submerged river channels and caused deposition of coarse and fine sands (Marriott et al. 2001).

The Columbia River estuary and lower mainstem span over 2 ecological provinces as defined by the NPCC: Columbia River Estuary (river mouth, including nearshore waters and Columbia River plume, to rm 34) and the Lower Columbia River (rm 34 to Bonneville Dam). The historical (circa 1880) total surface area of the Columbia River estuary has been estimated from 160-186 square miles (Thomas 1983, Simenstad et al. 1984), with extensive sand beds and variable river flow. The current estuary surface area has been estimated as 101,750 acres, which is equivalent to 159 square miles (Marriott et al. 2002). The Willamette River is the largest tributary to the lower Columbia River. Major tributaries originating in the Cascades include the Sandy River in Oregon and the Washougal, Lewis, Kalama and Cowlitz Rivers in Washington. Major Coast Range tributaries include the Elochoman and Grays Rivers in Washington and the Lewis and Clark, Youngs and Clatskanie Rivers in Oregon. Numerous other minor tributaries drain small watersheds but do not have substantial influence on the Columbia River because of their small size (Marriott et al. 2002).

In the Columbia River, tidal impacts in water level have been observed as far upstream as Bonneville Dam (RM 146) during low flow, reversal of river flow has been measured as far upstream as Oak Point (RM 53), and intrusion of salt water is typically to Harrington Point (RM 23) at the minimum regulated monthly flow, although at lower daily flows saltwater intrusion can extend past Pillar Rock (RM 28) (Neal 1972). The lowest river flows generally occur during

September and October, when rainfall and snowmelt runoff are low. The highest flows occur from April to June, resulting from snowmelt runoff. High flows also occur between November and March, caused by heavy winter precipitation. The discharge at the mouth of the river ranges from 100,000 to 500,000 cfs, with an average of about 260,000 cfs. Historically, unregulated flows at the mouth ranged from 79,000 cfs to over 1 million cfs, with average flows about 273,000 cfs (Neal 1972, Marriott et al. 2002).

The estuarine shoreline in both Washington and Oregon consist primarily of rocky, forested cliffs or low elevation, gently sloping floodplain areas. The topography of the riverine portion of the two ecological provinces does not vary considerably (Marriott et al. 2001).

The climate conditions vary across the subbasins; in general, coastal areas receive more precipitation and experience cooler summer temperatures and warmer winter temperatures than inland areas. In the lower part of the subbasin, climate data has been collected in Astoria, Oregon, since 1953 (WRCC 2003). Total average annual precipitation is 68 inches, ranging from 1.04 inches in July to 10.79 inches in December. January is the coldest month in Astoria with an average maximum temperature of 48.2°F and an average minimum temperature of 36.5°F; August is the warmest month with an average maximum temperature of 68.7°F and an average minimum temperature of 52.8°F. In the middle part of the subbasin, climate conditions have been recorded at St. Helens, Oregon, since 1976 (WRCC 2003). Total average annual precipitation is 44 inches, ranging from 0.79 inches in July to 6.77 inches in December. January is the coldest month in St. Helens with an average maximum temperature of 46.9°F and an average minimum temperature of 33.5°F; August is the warmest month with an average maximum temperature of 82.7°F and an average minimum temperature of 55.6°F. In the upper part of the subbasin, climate conditions have been recorded at Bonneville Dam since 1948 (WRCC 2003). Total average annual precipitation is 77 inches, ranging from 0.90 inches in July to 12.91 inches in December. January is the coldest month at Bonneville with an average maximum temperature of 42.4°F and an average minimum temperature of 32.7°F; August is the warmest month with an average maximum temperature of 78.7°F and an average minimum temperature of 56.4°F.

2.1.4 Fish and Wildlife Resources

An abundance of fish and wildlife species are known to occur in the Columbia Estuary and Columbia Lower Subbasins, either as year-round residents, seasonal residents, or migratory visitors. Early species survey work in the estuary was performed for aquatic species (Gaumer et al. 1973, Bottom et al. 1984, Dawley et al. 1985), birds (Hazel 1984), mammals (Howerton et al. 1984), and marine mammals (Jeffries et al. 1984). More recently, Marriott et al. (2002) provided an excellent summary of the aquatic species, birds, mammals, reptiles, and amphibians found in the Columbia River estuary and lower mainstem. A species list adapted from Marriott et al. (2002) and IBIS (2003) has been included here to demonstrate the variety of species present in the subbasins (Table 2-1).

Table 2-1. List of fish and wildlife species known to occur in the Columbia Estuary and Columbia Lower Subbasins.

Species Group	Common Name	Scientific Name
FISH	Pacific lamprey	*Lampetra tridentata*
	River lamprey	*Lampetra ayresi*
	Spiny dogfish	*Squalus acanthias*
	Big skate	*Raja binoculata*
	Green sturgeon	*Acipenser medirostris*
	White sturgeon	*Acipenser transmontanus*
	American shad	*Alosa sapidissima*
	Pacific herring	*Clupea harengus pallasi*
	Northern anchovy	*Engraulis mordax*
	Chum salmon	*Oncorhynchus keta*
	Coho salmon	*Oncorhynchus kisutch*
	Sockeye salmon	*Oncorhynchus nerka*
	Chinook salmon	*Oncorhynchus tshawytscha*
	Steelhead	*Oncorhynchus mykiss*
	Cutthroat trout	*Oncorhynchus clarki clarki*
	Mountain whitefish	*Prosopium williamsoni*
	Whitebait smelt	*Allosmerus elongates*
	Surf smelt	*Hypomesus pretiosus*
	Night smelt	*Spirinchus starksi*
	Longfin smelt	*Spirinchus thaleichthys*
	Eulachon	*Thaleichthys pacificus*
	Common carp	*Cyprinus carpio*
	Peamouth	*Mylocheilus caurinus*
	Northern pikeminnow	*Ptychocheilus oregonensis*
	Largescale sucker	*Catostomus macrocheilus*
	Yellow bullhead	*Ictalurus natalis*
	Brown bullhead	*Ictalurus nebulosus*
	Channel catfish	*Ictalurus punctatus*
	Pacific hake	*Merluccius productus*
	Pacific tomcod	*Microgadus proximus*
	Walleye Pollock	*Theragra chalcogramma*
	Threespine stickleback	*Gasterosteus aculeatus*
	Bay pipefish	*Syngnathus leptorhynchus*
	Pumpkinseed	*Lepomis gibbosus*
	Warmouth	*Lepomis gulosus*
	Bluegill	*Lepomis macrochirus*
	Walleye	*Stizostedium vitreum*
	Smallmouth bass	*Micropterus dolomeiui*
	Largemouth bass	*Micropterus salmoides*
	White crappie	*Pomoxis annularis*
	Black crappie	*Pomoxis migromaculatus*
	Yellow perch	*Perca flavenscens*
	Redtail surfperch	*Amphistichus rhodoterus*
	Shiner perch	*Cymatogaster aggregata*

Species Group	Common Name	Scientific Name
	Striped seaperch	*Embiotoca lateralis*
FISH CONT.	Spotfin surfperch	*Hyperprosopon anale*
	Walleye surfperch	*Hyperprosopon argenteum*
	Silver surfperch	*Hyperprosopon ellipticum*
	White seaperch	*Phanerodon furcatus*
	Pile perch	*Rhacochilus vacca*
	Pacific sandfish	*Trichodon trichodon*
	Snake prickleback	*Lumpenus sagitta*
	Saddleback gunnel	*Pholis ornata*
	Pacific sand lance	*Ammodytes hexapterus*
	Bay goby	*Lepidogobius lepidus*
	Black rockfish	*Sebastes melanops*
	Kelp greenling	*Hexogrammus decagrammus*
	Lingcod	*Ophiodon elongatus*
	Padded sculpin	*Artedius fenestralis*
	Coastrange sculpin	*Cottus aleuticus*
	Prickly sculpin	*Cottus asper*
	Buffalo sculpin	*Enophyrs bison*
	Red Irish lord	*Hemilepidotus hemilepidotus*
	Pacific staghorn sculpin	*Leptocottus armatus*
	Cabezon	*Scorpaenichthys marmoratus*
	Warty poacher	*Ocella verrucosa*
	Tubenose poacher	*Pallasina barbata*
	Pricklebreast poacher	*Stellerina xyosterna*
	Slipskin snailfish	*Liparis fucencis*
	Showy snailfish	*Liparis pulchellus*
	Ringtail snailfish	*Liparis rutteri*
	Pacific sanddab	*Citharichthys sordidus*
	Speckled sanddab	*Citharichthys stigmaeus*
	Butter sole	*Isopsetta isolepis*
	English sole	*Parophrys vetulus*
	Starry flounder	*Platichthys stellatus*
	C-O sole	*Pleuronichthys coenosus*
	Sand sole	*Psettichthys melanostictus*
	Larval smelt	
	Larval flatfish	
	Other larval fish	
AMPHIBIANS	Northwestern Salamander	*Ambystoma gracile*
	Long-toed Salamander	*Ambystoma macrodactylum*
	Cope's Giant Salamander	*Dicamptodon copei*
	Pacific Giant Salamander	*Dicamptodon tenebrosus*
	Columbia Torrent Salamander	*Rhyacotriton kezeri*
	Cascade Torrent Salamander	*Rhyacotriton cascadae*
	Rough-skinned Newt	*Taricha granulosa*
	Dunn's Salamander	*Plethodon dunni*
	Larch Mountain Salamander	*Plethodon larselli*
	Van Dyke's Salamander	*Plethodon vandykei*

Species Group	Common Name	Scientific Name
	Western Red-backed Salamander	*Plethodon vehiculum*
AMPHIBIANS CONT.	Ensatina	*Ensatina eschscholtzii*
	Clouded Salamander	*Aneides ferreus*
	Oregon Slender Salamander	*Batrachoseps wrighti*
	Tailed Frog	*Ascaphus truei*
	Western Toad	*Bufo boreas*
	Pacific Chorus (Tree) Frog	*Pseudacris regilla*
	Red-legged Frog	*Rana aurora*
	Cascades Frog	*Rana cascadae*
	Oregon Spotted Frog	*Rana pretiosa*
	Columbia Spotted Frog	*Rana luteiventris*
	Bullfrog	*Rana catesbeiana*
BIRDS	Red-throated Loon	*Gavia stellata*
	Pacific Loon	*Gavia pacifica*
	Common Loon	*Gavia immer*
	Yellow-billed Loon	*Gavia adamsii*
	Pied-billed Grebe	*Podilymbus podiceps*
	Horned Grebe	*Podiceps auritus*
	Red-necked Grebe	*Podiceps grisegena*
	Eared Grebe	*Podiceps nigricollis*
	Western Grebe	*Aechmophorus occidentalis*
	Clark's Grebe	*Aechmophorus clarkii*
	Sooty Shearwater	*Puffinus griseus*
	Short-tailed Shearwater	*Puffinus tenuirostris*
	Fork-tailed Storm-petrel	*Oceanodroma furcata*
	Leach's Storm-petrel	*Oceanodroma leucorhoa*
	Brown Pelican	*Pelecanus occidentalis*
	Brandt's Cormorant	*Phalacrocorax penicillatus*
	Double-crested Cormorant	*Phalacrocorax auritus*
	Pelagic Cormorant	*Phalacrocorax pelagicus*
	American Bittern	*Botaurus lentiginosus*
	Great Blue Heron	*Ardea herodias*
	Great Egret	*Ardea alba*
	Cattle Egret	*Bubulcus ibis*
	Green Heron	*Butorides virescens*
	Black-crowned Night-heron	*Nycticorax nycticorax*
	Turkey Vulture	*Cathartes aura*
	Greater White-fronted Goose	*Anser albifrons*
	Snow Goose	*Chen Ccaerulescens*
	Ross's Goose	*Chen rossii*
	Canada Goose	*Branta canadensis*
	Dusky Canada Goose	*Branta canadensis occidentalis, Baird*
	Brant	*Branta bernicla*
	Trumpeter Swan	*Cygnus buccinator*
	Tundra Swan	*Cygnus columbianus*
	Wood Duck	*Aix sponsa*

Species Group	Common Name	Scientific Name
	Gadwall	*Anas strepera*
BIRDS CONT.	Eurasian Wigeon	*Anas penelope*
	American Wigeon	*Anas americana*
	Mallard	*Anas platyrhynchos*
	Blue-winged Teal	*Anas discors*
	Cinnamon Teal	*Anas cyanoptera*
	Northern Shoveler	*Anas clypeata*
	Northern Pintail	*Anas acuta*
	Green-winged Teal	*Anas crecca*
	Canvasback	*Aythya valisineria*
	Redhead	*Aythya americana*
	Ring-necked Duck	*Aythya collaris*
	Greater Scaup	*Aythya marila*
	Lesser Scaup	*Aythya affinis*
	Harlequin Duck	*Histrionicus histrionicus*
	Surf Scoter	*Melanitta perspicillata*
	White-winged Scoter	*Melanitta fusca*
	Black Scoter	*Melanitta nigra*
	Long-tailed Duck	*Clangula hyemalis*
	Bufflehead	*Bucephala albeola*
	Common Goldeneye	*Bucephala clangula*
	Barrow's Goldeneye	*Bucephala islandica*
	Hooded Merganser	*Lophodytes cucullatus*
	Common Merganser	*Mergus merganser*
	Red-breasted Merganser	*Mergus serrator*
	Ruddy Duck	*Oxyura jamaicensis*
	Osprey	*Pandion haliaetus*
	White-tailed Kite	*Elanus leucurus*
	Bald Eagle	*Haliaeetus leucocephalus*
	Northern Harrier	*Circus cyaneus*
	Sharp-shinned Hawk	*Accipiter striatus*
	Cooper's Hawk	*Accipiter cooperii*
	Northern Goshawk	*Accipiter gentilis*
	Red-tailed Hawk	*Buteo jamaicensis*
	Rough-legged Hawk	*Buteo lagopus*
	Golden Eagle	*Aquila chrysaetos*
	American Kestrel	*Falco sparverius*
	Merlin	*Falco columbarius*
	Gyrfalcon	*Falco rusticolus*
	Peregrine Falcon	*Falco peregrinus*
	Prairie Falcon	*Falco mexicanus*
	Gray Partridge	*Perdix perdix*
	Ring-necked Pheasant	*Phasianus colchicus*
	Ruffed Grouse	*Bonasa umbellus*
	White-tailed Ptarmigan	*Lagopus leucurus*
	Blue Grouse	*Dendragapus obscurus*
	Wild Turkey	*Meleagris gallopavo*
	Mountain Quail	*Oreortyx pictus*

Species Group	Common Name	Scientific Name
	California Quail	*Callipepla californica*
BIRDS CONT.	Northern Bobwhite	*Colinus virginianus*
	Virginia Rail	*Rallus limicola*
	Sora	*Porzana carolina*
	American Coot	*Fulica americana*
	Sandhill Crane	*Grus canadensis*
	Black-bellied Plover	*Pluvialis squatarola*
	American Golden-Plover	*Pluvialis dominica*
	Pacific Golden-Plover	*Pluvialis fulva*
	Snowy Plover	*Charadrius alexandrinus*
	Semipalmated Plover	*Charadrius semipalmatus*
	Killdeer	*Charadrius vociferus*
	Black Oystercatcher	*Haematopus bachmani*
	Greater Yellowlegs	*Tringa melanoleuca*
	Lesser Yellowlegs	*Tringa flavipes*
	Solitary Sandpiper	*Tringa solitaria*
	Spotted Sandpiper	*Actitis macularia*
	Willet	*Catoptrophorus semipalmatus*
	Wandering Tattler	*Heteroscelus incanus*
	Whimbrel	*Numenius phaeopus*
	Long-billed Curlew	*Numenius americanus*
	Marbled Godwit	*Limosa fedoa*
	Ruddy Turnstone	*Arenaria interpres*
	Black Turnstone	*Arenaria melanocephala*
	Surfbird	*Aphriza virgata*
	Red Knot	*Calidris canutus*
	Sanderling	*Calidris alba*
	Semipalmated Sandpiper	*Calidris pusilla*
	Western Sandpiper	*Calidris mauri*
	Least Sandpiper	*Calidris minutilla*
	Baird's Sandpiper	*Calidris bairdii*
	Pectoral Sandpiper	*Calidris melanotos*
	Sharp-tailed Sandpiper	*Calidris acuminata*
	Rock Sandpiper	*Calidris ptilocnemis*
	Dunlin	*Calidris alpina*
	Stilt Sandpiper	*Calidris himantopus*
	Buff-breasted Sandpiper	*Tryngites subruficollis*
	Ruff	*Philomachus pugnax*
	Short-billed Dowitcher	*Limnodromus griseus*
	Long-billed Dowitcher	*Limnodromus scolopaceus*
	Common Snipe	*Gallinago gallinago*
	Wilson's Phalarope	*Phalaropus tricolor*
	Red-necked Phalarope	*Phalaropus lobatus*
	Red Phalarope	*Phalaropus fulicaria*
	South Polar Skua	*Catharacta maccormicki*
	Pomarine Jaeger	*Stercorarius pomarinus*
	Parasitic Jaeger	*Stercorarius parasiticus*
	Bonaparte's Gull	*Larus philadelphia*

Species Group	Common Name	Scientific Name
	Heermann's Gull	*Larus heermanni*
BIRDS CONT.	Mew Gull	*Larus canus*
	Ring-billed Gull	*Larus delawarensis*
	California Gull	*Larus californicus*
	Herring Gull	*Larus argentatus*
	Thayer's Gull	*Larus thayeri*
	Western Gull	*Larus occidentalis*
	Glaucous-winged Gull	*Larus glaucescens*
	Glaucous Gull	*Larus hyperboreus*
	Sabine's Gull	*Xema Sabini*
	Black-legged Kittiwake	*Rissa tridactyla*
	Caspian Tern	*Sterna caspia*
	Elegant Tern	*Sterna elegans*
	Common Tern	*Sterna hirundo*
	Arctic Tern	*Sterna paradisaea*
	Forster's Tern	*Sterna forsteri*
	Black Tern	*Chlidonias niger*
	Common Murre	*Uria aalge*
	Pigeon Guillemot	*Cepphus columba*
	Marbled Murrelet	*Brachyramphus marmoratus*
	Ancient Murrelet	*Synthliboramphus antiquus*
	Cassin's Auklet	*Ptychoramphus aleuticus*
	Rhinoceros Auklet	*Cerorhinca monocerata*
	Tufted Puffin	*Fratercula cirrhata*
	Rock Dove	*Columba livia*
	Band-tailed Pigeon	*Columba fasciata*
	Mourning Dove	*Zenaida macroura*
	Barn Owl	*Tyto alba*
	Flammulated Owl	*Otus flammeolus*
	Western Screech-owl	*Otus kennicottii*
	Great Horned Owl	*Bubo virginianus*
	Snowy Owl	*Nyctea scandiaca*
	Northern Pygmy-owl	*Glaucidium gnoma*
	Burrowing Owl	*Athene cunicularia*
	Spotted Owl	*Strix occidentalis*
	Barred Owl	*Strix varia*
	Long-eared Owl	*Asio otus*
	Short-eared Owl	*Asio flammeus*
	Northern Saw-whet Owl	*Aegolius acadicus*
	Common Nighthawk	*Chordeiles minor*
	Black Swift	*Cypseloides niger*
	Vaux's Swift	*Chaetura vauxi*
	White-throated Swift	*Aeronautes saxatalis*
	Black-chinned Hummingbird	*Archilochus alexandri*
	Anna's Hummingbird	*Calypte anna*
	Calliope Hummingbird	*Stellula calliope*
	Rufous Hummingbird	*Selasphorus rufus*
	Belted Kingfisher	*Ceryle alcyon*

Species Group	Common Name	Scientific Name
	Lewis's Woodpecker	*Melanerpes lewis*
BIRDS CONT.	Acorn Woodpecker	*Melanerpes formicivorus*
	Williamson's Sapsucker	*Sphyrapicus thyroideus*
	Red-naped Sapsucker	*Sphyrapicus nuchalis*
	Red-breasted Sapsucker	*Sphyrapicus ruber*
	Downy Woodpecker	*Picoides pubescens*
	Hairy Woodpecker	*Picoides villosus*
	Three-toed Woodpecker	*Picoides tridactylus*
	Black-backed Woodpecker	*Picoides arcticus*
	Northern Flicker	*Colaptes auratus*
	Pileated Woodpecker	*Dryocopus pileatus*
	Olive-sided Flycatcher	*Contopus cooperi*
	Western Wood-pewee	*Contopus sordidulus*
	Willow Flycatcher	*Empidonax traillii*
	Hammond's Flycatcher	*Empidonax hammondii*
	Dusky Flycatcher	*Empidonax oberholseri*
	Pacific-slope Flycatcher	*Empidonax difficilis*
	Say's Phoebe	*Sayornis saya*
	Ash-throated Flycatcher	*Myiarchus cinerascens*
	Western Kingbird	*Tyrannus verticalis*
	Eastern Kingbird	*Tyrannus tyrannus*
	Loggerhead Shrike	*Lanius ludovicianus*
	Northern Shrike	*Lanius excubitor*
	Cassin's Vireo	*Vireo cassinii*
	Hutton's Vireo	*Vireo huttoni*
	Warbling Vireo	*Vireo gilvus*
	Red-eyed Vireo	*Vireo olivaceus*
	Gray Jay	*Perisoreus canadensis*
	Steller's Jay	*Cyanocitta stelleri*
	Western Scrub-Jay	*Aphelocoma californica*
	Pinyon Jay	*Gymnorhinus cyanocephalus*
	Clark's Nutcracker	*Nucifraga columbiana*
	Black-billed Magpie	*Pica pica*
	American Crow	*Corvus brachyrhynchos*
	Northwestern Crow	*Corvus caurinus*
	Common Raven	*Corvus corax*
	Horned Lark	*Eremophila alpestris*
	Purple Martin	*Progne subis*
	Tree Swallow	*Tachycineta bicolor*
	Violet-green Swallow	*Tachycineta thalassina*
	Northern Rough-winged Swallow	*Stelgidopteryx serripennis*
	Cliff Swallow	*Petrochelidon pyrrhonota*
	Barn Swallow	*Hirundo rustica*
	Black-capped Chickadee	*Poecile atricapillus*
	Mountain Chickadee	*Poecile gambeli*
	Chestnut-backed Chickadee	*Poecile rufescens*
	Bushtit	*Psaltriparus minimus*
	Red-breasted Nuthatch	*Sitta canadensis*

Species Group	Common Name	Scientific Name
	White-breasted Nuthatch	*Sitta carolinensis*
BIRDS CONT.	Brown Creeper	*Certhia americana*
	Rock Wren	*Salpinctes obsoletus*
	Canyon Wren	*Catherpes mexicanus*
	Bewick's Wren	*Thryomanes bewickii*
	House Wren	*Troglodytes aedon*
	Winter Wren	*Troglodytes troglodytes*
	Marsh Wren	*Cistothorus palustris*
	American Dipper	*Cinclus mexicanus*
	Golden-crowned Kinglet	*Regulus satrapa*
	Ruby-crowned Kinglet	*Regulus calendula*
	Western Bluebird	*Sialia mexicana*
	Mountain Bluebird	*Sialia currucoides*
	Townsend's Solitaire	*Myadestes townsendi*
	Veery	*Catharus fuscescens*
	Swainson's Thrush	*Catharus ustulatus*
	Hermit Thrush	*Catharus guttatus*
	American Robin	*Turdus migratorius*
	Varied Thrush	*Ixoreus naevius*
	Wrentit	*Chamaea fasciata*
	Northern Mockingbird	*Mimus polyglottos*
	European Starling	*Sturnus vulgaris*
	American Pipit	*Anthus rubescens*
	Cedar Waxwing	*Bombycilla cedrorum*
	Orange-crowned Warbler	*Vermivora celata*
	Nashville Warbler	*Vermivora ruficapilla*
	Yellow Warbler	*Dendroica petechia*
	Yellow-rumped Warbler	*Dendroica coronata*
	Black-throated Gray Warbler	*Dendroica nigrescens*
	Townsend's Warbler	*Dendroica townsendi*
	Hermit Warbler	*Dendroica occidentalis*
	Palm Warbler	*Dendroica palmarum*
	Macgillivray's Warbler	*Oporornis tolmiei*
	Common Yellowthroat	*Geothlypis trichas*
	Wilson's Warbler	*Wilsonia pusilla*
	Yellow-breasted Chat	*Icteria virens*
	Western Tanager	*Piranga ludoviciana*
	Green-tailed Towhee	*Pipilo chlorurus*
	Spotted Towhee	*Pipilo maculatus*
	California Towhee	*Pipilo crissalis*
	Chipping Sparrow	*Spizella passerina*
	Brewer's Sparrow	*Spizella breweri*
	Clay-colored Sparrow	*Spizella pallida*
	Vesper Sparrow	*Pooecetes gramineus*
	Savannah Sparrow	*Passerculus sandwichensis*
	Grasshopper Sparrow	*Ammodramus savannarum*
	Fox Sparrow	*Passerella iliaca*
	Song Sparrow	*Melospiza melodia*

Species Group	Common Name	Scientific Name
	Lincoln's Sparrow	*Melospiza lincolnii*
BIRDS CONT.	Swamp Sparrow	*Melospiza georgiana*
	White-throated Sparrow	*Zonotrichia albicollis*
	Harris's Sparrow	*Zonotrichia querula*
	White-crowned Sparrow	*Zonotrichia leucophrys*
	Golden-crowned Sparrow	*Zonotrichia atricapilla*
	Dark-eyed Junco	*Junco hyemalis*
	Lapland Longspur	*Calcarius lapponicus*
	Black-headed Grosbeak	*Pheucticus melanocephalus*
	Snow Bunting	*Plectrophenax nivalis*
	Lazuli Bunting	*Passerina amoena*
	Red-winged Blackbird	*Agelaius phoeniceus*
	Tricolored Blackbird	*Agelaius tricolor*
	Western Meadowlark	*Sturnella neglecta*
	Yellow-headed Blackbird	*Xanthocephalus xanthocephalus*
	Brewer's Blackbird	*Euphagus cyanocephalus*
	Brown-headed Cowbird	*Molothrus ater*
	Bullock's Oriole	*Icterus bullockii*
	Gray-crowned Rosy-Finch	*Leucosticte tephrocotis*
	Pine Grosbeak	*Pinicola enucleator*
	Purple Finch	*Carpodacus purpureus*
	Cassin's Finch	*Carpodacus cassinii*
	House Finch	*Carpodacus mexicanus*
	Red Crossbill	*Loxia curvirostra*
	Common Redpoll	*Carduelis flammea*
	Pine Siskin	*Carduelis pinus*
	Lesser Goldfinch	*Carduelis psaltria*
	American Goldfinch	*Carduelis tristis*
	Evening Grosbeak	*Coccothraustes vespertinus*
	House Sparrow	*Passer domesticus*
MAMMALS	Virginia Opossum	*Didelphis virginiana*
	Masked Shrew	*Sorex cinereus*
	Vagrant Shrew	*Sorex vagrans*
	Montane Shrew	*Sorex monticolus*
	Baird's Shrew	*Sorex bairdi*
	Water Shrew	*Sorex palustris*
	Pacific Water Shrew	*Sorex bendirii*
	Trowbridge's Shrew	*Sorex trowbridgii*
	Shrew-mole	*Neurotrichus gibbsii*
	Townsend's Mole	*Scapanus townsendii*
	Coast Mole	*Scapanus orarius*
	California Myotis	*Myotis californicus*
	Western Small-footed Myotis	*Myotis ciliolabrum*
	Yuma Myotis	*Myotis yumanensis*
	Little Brown Myotis	*Myotis lucifugus*
	Long-legged Myotis	*Myotis volans*
	Fringed Myotis	*Myotis thysanodes*

Species Group	Common Name	Scientific Name
	Long-eared Myotis	*Myotis evotis*
MAMMALS CONT.	Silver-haired Bat	*Lasionycteris noctivagans*
	Big Brown Bat	*Eptesicus fuscus*
	Hoary Bat	*Lasiurus cinereus*
	Townsend's Big-eared Bat	*Corynorhinus townsendii*
	American Pika	*Ochotona princeps*
	Brush Rabbit	*Sylvilagus bachmani*
	Eastern Cottontail	*Sylvilagus floridanus*
	Nuttall's (Mountain) Cottontail	*Sylvilagus nuttallii*
	Snowshoe Hare	*Lepus americanus*
	Black-tailed Jackrabbit	*Lepus californicus*
	Mountain Beaver	*Aplodontia rufa*
	Yellow-pine Chipmunk	*Tamias amoenus*
	Townsend's Chipmunk	*Tamias townsendii*
	Yellow-bellied Marmot	*Marmota flaviventris*
	California Ground Squirrel	*Spermophilus beecheyi*
	Golden-mantled Ground Squirrel	*Spermophilus lateralis*
	Cascade Golden-mantled Ground Squirrel	*Spermophilus saturatus*
	Eastern Gray Squirrel	*Sciurus carolinensis*
	Eastern Fox Squirrel	*Sciurus niger*
	Western Gray Squirrel	*Sciurus griseus*
	Douglas' Squirrel	*Tamiasciurus douglasii*
	Northern Flying Squirrel	*Glaucomys sabrinus*
	Northern Pocket Gopher	*Thomomys talpoides*
	Western Pocket Gopher	*Thomomys mazama*
	Camas Pocket Gopher	*Thomomys bulbivorus*
	American Beaver	*Castor canadensis*
	Western Harvest Mouse	*Reithrodontomys megalotis*
	Deer Mouse	*Peromyscus maniculatus*
	Columbian Mouse	*Peromyscus keeni*
	Pinon Mouse	*Peromyscus truei*
	Dusky-footed Woodrat	*Neotoma fuscipes*
	Bushy-tailed Woodrat	*Neotoma cinerea*
	Southern Red-backed Vole	*Clethrionomys gapperi*
	Western Red-backed Vole	*Clethrionomys californicus*
	Heather Vole	*Phenacomys intermedius*
	White-footed Vole	*Phenacomys albipes*
	Red Tree Vole	*Phenacomys longicaudus*
	Montane Vole	*Microtus montanus*
	Gray-tailed Vole	*Microtus canicaudus*
	Townsend's Vole	*Microtus townsendii*
	Long-tailed Vole	*Microtus longicaudus*
	Creeping Vole	*Microtus oregoni*
	Water Vole	*Microtus richardsoni*
	Muskrat	*Ondatra zibethicus*
	Black Rat	*Rattus rattus*
	Norway Rat	*Rattus norvegicus*

Species Group	Common Name	Scientific Name
	House Mouse	*Mus musculus*
MAMMALS CONT.	Western Jumping Mouse	*Zapus princeps*
	Pacific Jumping Mouse	*Zapus trinotatus*
	Common Porcupine	*Erethizon dorsatum*
	Nutria	*Myocastor coypus*
	Coyote	*Canis latrans*
	Red Fox	*Vulpes vulpes*
	Gray Fox	*Urocyon cinereoargenteus*
	Black Bear	*Ursus americanus*
	Raccoon	*Procyon lotor*
	American Marten	*Martes americana*
	Fisher	*Martes pennanti*
	Ermine	*Mustela erminea*
	Long-tailed Weasel	*Mustela frenata*
	Mink	*Mustela vison*
	Wolverine	*Gulo gulo*
	American Badger	*Taxidea taxus*
	Western Spotted Skunk	*Spilogale gracilis*
	Striped Skunk	*Mephitis mephitis*
	Northern River Otter	*Lutra canadensis*
	Mountain Lion	*Puma concolor*
	Bobcat	*Lynx rufus*
	Elk	*Cervus elaphus*
	Mule Deer	*Odocoileus hemionus*
	White-tailed Deer	*Odocoileus virginianus*
	Columbian White-tailed Deer	*Odocoileus virginianus leucurus*
	Mountain Goat	*Oreamnos americanus*
MARINE MAMMALS	Northern (Steller) Sea Lion	*Eumetopias jubatus*
	California Sea Lion	*Zalophus californianus*
	Harbor Seal	*Phoca vitulina*
REPTILES	Snapping Turtle	*Chelydra serpentina*
	Painted Turtle	*Chrysemys picta*
	Western Pond Turtle	*Clemmys marmorata*
	Red-eared Slider Turtle	*Trachemys scripta*
	Northern Alligator Lizard	*Elgaria coerulea*
	Southern Alligator Lizard	*Elgaria multicarinata*
	Western Fence Lizard	*Sceloporus occidentalis*
	Western Skink	*Eumeces skiltonianus*
	Rubber Boa	*Charina bottae*
	Racer	*Coluber constrictor*
	Ringneck Snake	*Diadophis punctatus*
	California Mountain Kingsnake	*Lampropeltis zonata*
	Gopher Snake	*Pituophis catenifer*
	Western Terrestrial Garter Snake	*Thamnophis elegans*
	Northwestern Garter Snake	*Thamnophis ordinoides*
	Common Garter Snake	*Thamnophis sirtalis*

Species Group	Common Name	Scientific Name
	Western Rattlesnake	*Crotalus viridis*

2.1.5 Habitat Classification

The estuary includes a complex mosaic of interconnected and interacting habitat types. One of the difficulties in describing these habitat types is choosing a habitat classification system that adequately describes the habitats used by focal species and is acceptable to all stakeholders in the subbasin. For example, habitat type descriptions differ as a result of the resolution of the methods utilized to map and classify the habitat. Further, habitat mapping methods are designed to describe aquatic or terrestrial habitat types, but generally are not capable of adequately mapping both. Choosing the appropriate habitat classification system is further complicated by the diversity of habitats found throughout the lower Columbia River mainstem and estuary or by different area coverage of each habitat mapping effort. For the purposes of this subbasin assessment, a habitat classification was needed that could: describe aquatic habitats, describe terrestrial habitats, and provide a historical context for evaluating the change in estuary and mainstem (to Bonneville Dam) habitat types over time. There is not one habitat classification system that provides for all these needs; thus, we chose to utilize multiple habitat classification systems to describe estuary and mainstem habitat types as described below. The use of multiple habitat classification systems creates additional challenges because the habitat types among different classification systems are rarely directly comparable. However, we evaluated each habitat classification system to determine potential groupings of specific habitat types from each classification system, limiting the comparison to habitat types known to occur in the lower Columbia River and estuary.

2.1.5.1 Bathymetric Mapping

Bathymetry is a low resolution method that provides coarse delineations of habitat types. Habitat classification using bathymetry provides a means to segregate aquatic habitat based on depth criteria; additionally, published bathymetric mapping efforts provide a historical context for evaluating Columbia River estuary habitat change. Using bathymetric survey maps of the U.S. Coast Survey (now U.S. Geodetic Survey), five major types of estuary (i.e. rm 0-46.5) habitat were defined by the Columbia River Estuary Data Development Program (Thomas 1983) according to elevation and the dominant vegetation: tidal swamps, tidal marshes, shallow water/flats, medium depth water, and deep water. A cross-sectional view of these habitat types is depicted in Figure 2-4. Tidal swamps are those areas where the dominant vegetation is mostly shrub and woody species with elevations varying between mean high high water (MHHW) and the line of non-aquatic vegetation. Tidal marshes vary considerably depending on dominant low shrubs or emergent herbaceous vegetation and have been recorded slightly above mean low low water (MLLW) to slightly above MHHW. Shallow water/flats are defined as being between an elevation slightly above the MLLW mark to -6 ft MLLW. Medium depth water is between 6 ft and 18 ft below MLLW, while deep water is defined as 18 ft and deeper. Further, at a given elevation, there is an overriding influence of time and salinity in development of specific types of habitat. For example, tidal marsh habitat may be classified as a saltwater or freshwater marsh and each is characterized by distinctive vegetation as driven by salinity levels. Additionally, shallows/flats habitat may be present in an area formerly classified as medium depth water as a result of accretion; given time and further accretion, shallows/flats habitat may transition to tidal marsh.

Thomas (1983) also investigated five categories of non-estuarine habitat (i.e. developed floodplain, natural and filled uplands, non-tidal swamps, non-tidal marshes, and non-tidal water) to identify the fate of floodplain areas that were removed from the estuarine system. Developed floodplain habitat was defined as all diked floodplain converted to agriculture, residential, or other land use. Natural and filled uplands included those areas where measurable acreages have been filled, primarily through disposal of dredge material. Non-tidal swamps were areas of the diked floodplain that were never cleared or were cleared and converted back to swamp. Non-tidal marshes included areas of the diked floodplain that support emergent wetland vegetation; these were typically abandoned pastures dominated by rush and sedge. Non-tidal water was those areas of former tidal sloughs that were separated from the river by dikes and tidegates.

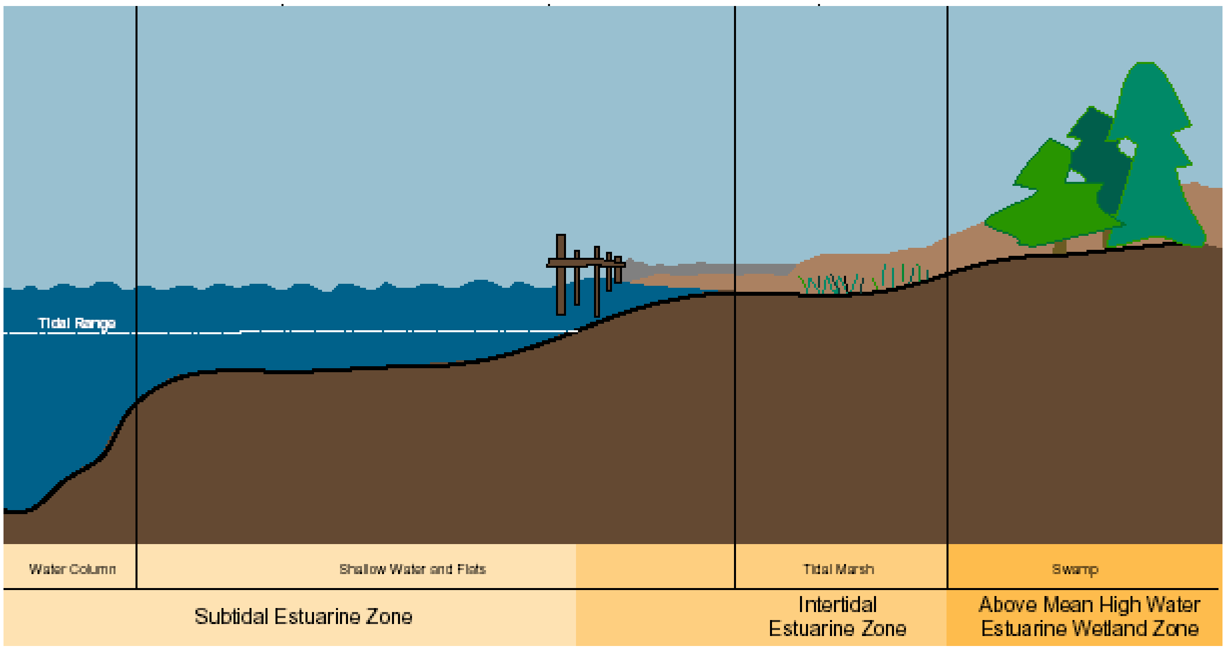


Figure 2-4. Cross-sectional depiction of general estuary habitat types (USACE 2001).

2.1.5.2 Satellite Imagery Habitat Mapping

Satellite imagery provides a high resolution habitat mapping method that principally uses vegetative communities to describe habitat types. Because of the use of vegetation, satellite imagery is generally not capable of distinguishing different types of aquatic habitats. Different satellite imagery technology are available that provide different levels of resolution; two of these technologies are compared in Garono et al. (2003b).

A widely accepted habitat classification system developed from satellite imagery is that of Johnson and O'Neil (2001); this habitat classification system describes wildlife habitats present in Washington and Oregon and provides a historical context for evaluating habitat change in lower Columbia River mainstem and estuary habitats. A total of 32 wildlife habitat types are delineated in this classification system (Table 2-2); each habitat type is further described based on geographic distribution, physical setting, landscape setting, structure, and composition. Johnson and O'Neil (2001) also provide information on other classification systems and key references, natural disturbance regimes, succession and stand dynamics, management and anthropogenic impacts, and status and trends to provide further insight for each habitat type. This habitat classification system has been utilized by the Northwest Habitat Institute for producing maps comparing historical and current wildlife habitat types in the lower Columbia River mainstem and estuary as part the NPCC subbasin planning process (IBIS 2003).

Table 2-2. Wildlife habitat types in Washington and Oregon determined by Johnson and O'Neil (2001).

Wildlife Habitat Types	Vegetative/Land Use/Marine Groupings
Westside Lowland Conifer-Hardwood Forest	*Alnus rubra-Acer macrophyllum* Upland Forests
	Picea sitchensis-Tsuga heterphylla Forests
	Pseudotsuga menziesii-Alnus rubra-Acer macrophyllus Forests
	Maritime *Tsuga heterphylla-Thuja plicata* Forests
	Forested Dunes
Westside Oak and Dry Douglas-fir Forest and Woodlands	Westside *Quercus garryana* Forests and Woodlands
	Westside *Quercus garryana-Pseudotsuga menziesii* Forests
	Westside Dry *Pseudotsuga menziesii* Forests
	Pseudotsuga menziesii-Arbutus menziesii Forests
Southwest Oregon Mixed Conifer-Hardwood Forests	*Abies concolor* Mixed conifer Forests
	Pinus jefferii Woodlands
	Pseudotsuga menziesii-Lithocarpus densiflorus Forests
	Southwest Oregon Low Elevation Mixed Conifer Forests
Montane Mixed Conifer Forests	*Abies amabilis-Tsuga heterophylla* Forests
	Abies lasiocarpa-Picea englemannii Forests
	Abies magnifica var. shastensis Forests and Woodlands
	Tsuga mertensiana Forests
	Tsuga mertensiana-Abies amabilis Forests
Eastside (Interior) Mixed Conifer Forest	Eastside *Abies grandis- Pseudotsuga menziesii* Forest
	Eastside *Pseudotsuga menziesii- Pinus ponderosa* Forest
	Eastside *Tsuga heterphylla-Thuja plicata* Forests
Lodgepole Pine Forests and Woodlands	*Pinus contorta* Grass understory
	Pinus contorta Shrub understory
	Pinus contorta Subalpine Forests
	Pinus contorta Woodlands and Forests on Pumice
Ponderosa Pine Forests and Woodlands	*Pinus ponderosa* Woodlands
	Eastside *Pinus ponderosa -Quercus garryana* Forest and Woodlands
Upland Aspen Forests	*Populus tremuloides* Upland Forests
Subalpine Parklands	Subalpine and Alpine Wetlands
	Pinus albicaulis-Abies lasiocarpa Woodlands and Parklands
	Tsuga mertensiana Parklands
Alpine Grasslands and Shrublands	Subalpine and Alpine Grasslands
	Alpine Dwarf Shrublands-Fellfields and Sedge Turf
Westside Grasslands	Westside *Festuca idahoensis var. romeri-Danthonia californica*
Ceanothus-Manzanita Shrublands	Chaparral
Western Juniper and Mountain Mahogany Woodlands	*Juniperus occidentalis* Scablands
	Juniperus occidentalis-Artemisia tridentata Tall Shrublands
	Juniperus occidentalis/ Bunchgrass
	Cercocarpus ledifolius
Eastside (Interior) Canyon Shrublands	Eastside Moist Deciduous Shrublands
Eastside (Interior) Grasslands	*Pseudoroegneria spicata* Grasslands
	Eastside Low-to-Mid-elevation *Festuca idahoensis* Grasslands
	Eastside Modified Grasslands
	Sporobolus cryptandrus-Aristida puppurea var.longiseta Grasslands
Shrub-steppe	*Artemisia tripartita* Shrub-steppe
	Artemisia cana Shrub-steppe
	Artemisia tridentata ssp. tridentata and ssp. wyomingensis Shrub-steppe
	Artemisia tridentata ssp. vaseyana Shrublands
	Purshia tridentata Shrub-steppe
	Sandy steppe and Shrub-steppe
Dwarf Shrub-steppe	*Artemisia rigida/Eriogonum spp./Poa secunda* Dwarf-Shrub Scabland
	Artemisia arbuscula Dwarf-Shrub-steppe

Eastside *Quercus garryana* Woodlands

Wildlife Habitat Types
Vegetative/Land Use/Marine Groupings

Desert Playa and Salt Scrub
- Alkali Grasslands and Wetlands
- *Atriplex confertifolia* Shrublands
- Mixed Saltdesert Shrub-Non-Playa
- Mixed Saltdesert Shrub-Playa
- *Sarcobatus vermiculatus* Shrublands

Agriculture, Pasture, and Mixed Environs*
- Cultivated Croplands
- Improved Pasture
- Modified Grasslands
- Orchard/Vineyard/Nursery
- Unimproved Pasture

Urban and Mixed Environs*
- High Density
- Moderate Density
- Low Density

Open Water-Lakes, Rivers, Streams
- Riverine
- Lacustrine-Open Water

Herbaceous Wetlands
- Graminoid Wet Meadow
- Freshwater Aquatic Beds
- Herbaceous and Sedge Wetlands

Westside Riparian - Wetlands
- *Alnus viridis ssp. sinuata-Acer circinatum* Shrublands
- Westside Riparian and Wetland Deciduous Forests
- *Picea sitchensis* Wetland Forests and Woodlands
- *Tsuga heterophylla-Thuja plicata* coniferous wetlands
- Westside Riparian/Wetland Shrublands
- Shrub/herbaceous Sphagnum Bogs
- Wooded Bogs

* Wildlife habitats were determined by an expert panel process.

Artemisia nova Dwarf-Shrublands

Wildlife Habitat Types
Vegetative/Land Use/Marine Groupings

Montane Coniferous Wetlands
- Westside Montane Coniferous Wetlands
- *Picea engelmannii* Forested Wetlands

Eastside (Interior) Riparian - Wetlands
- Eastside Midmontane *Alnus incana-Salix ssp.* Riparian Shrublands
- Eastside Lowland Riparian Shrublands
- Eastside *Populus balsamifera ssp. trichocarpa*
- *Alnus rhombifolia* Riparian
- *Pinus ponderosa* Riparian Woodlands
- *Populus tremuloides* Riparian/Wetland Forests and Woodlands

Coastal Dunes and Beaches
- Coastal Dune Grasslands
- Coastal Dune Shrublands

Coastal Headlands and Islets
- Coastal Headland Shrublands and Grasslands

Bays and Estuaries*
- Bays and Estuaries (includes Intertidal Marshes)

Inland Marine Deeper Waters*
- Puget Sound to Strait of Juan de Fuca

Marine Nearshore*
- Marine environment from shore line to 20m depth

Marine Shelf*
- Marine environment from 20m to 200m depth

Oceanic*
- Marine environment greater than 200m depth

The Lower Columbia River Estuary Partnership (LCREP) was interested in producing spatial data sets describing the location and distribution of estuarine and tidal freshwater habitat cover types along the Columbia River from the mouth to Bonneville Dam using a consistent method and data source (Garono et al 2003c). The habitat mapping focused on estuarine and tidal freshwater habitats; areas not located along the river and >175 ft elevation (for the eastern dataset) or >100 ft elevation (for the western dataset) were deleted from the habitat classification(Garono et al 2003c). The habitat types designated in this research differed from that of Thomas (1983) and Johnson and O'Neil (2001). In general, the vegetated habitat types are more specific than that of Thomas (1983) but less specific than that of Johnson and O'Neil (2001); the aquatic habitat types were less specific than Thomas (1983) and similar to that of Johnson and O'Neil (2001). However, in order to compare the habitats mapped in 2000 with a National Oceanic and Atmospheric Administration (NOAA) mapping dataset from1992, a more generalized list of habitat types were derived to achieve consistency between the two datasets (Garono et al. 2003a). This habitat change analysis provided a recent context for evaluating lower Columbia River mainstem and estuary habitat change.

The resulting habitat types from the merge of the 1992 and 2000 datasets include: herbaceous wetland, scrub-shrub wetland, forested wetland, herbaceous upland, scrub-shrub upland, deciduous forest upland, coniferous forest upland, mixed forest upland, unconsolidated shoreline, water, urban, and other (Garono et al 2003a). The following general guidelines defined the major habitat classes: herbaceous habitat types had >70% herbaceous cover, scrub-shrub habitat types had >70% woody vegetation <8 ft high, forest habitat types had >60% conifers of broad-leaved vegetation, mixed forest habitat types were defined based on the proportion of conifers/deciduous ranging from 40/60 to 50/50, and unconsolidated shoreline habitat had at least 70% of the area as exposed substrate (Garono et al. 2003c). It is not clear what criteria Garono et al. (2003a,c) utilized to distinguish between wetland and upland habitat.

2.1.5.3 WDFW Priority Habitats

WDFWs Priority Habitats and Species Program was initiated in 1989 and remains in use today. WDFW priority habitats are generally defined as habitat types with unique or significant value to many species. An area identified and mapped as priority habitat has one or more of the following attributes: comparatively high fish and wildlife density, comparatively high fish and wildlife species diversity, important fish and wildlife breeding habitat, important fish and wildlife seasonal ranges, important fish and wildlife movement corridors, limited availability, high vulnerability to habitat alteration, or unique or dependent species. A priority habitat may be described by a unique vegetation type or by a dominant plant species that is of primary importance to fish and wildlife (e.g., oak woodlands, eelgrass meadows). A priority habitat may also be described by a successional stage (e.g., old growth and mature forests). Alternatively, a priority habitat may consist of a specific habitat element (e.g., consolidated marine/estuarine shorelines, talus slopes, caves, snags) of key value to fish and wildlife.

Specific descriptions of the four WDFW Priority Habitats considered in this subbasin assessment follows. Old growth forest west of the Cascade crest are generally defined as stands of at least 2 tree species, forming a multi-layered canopy with occasional small openings, with at least 8 trees/acre that are >81 cm diameter at breast height (dbh) or >200 years old. Mature forests are defined as stands with average tree diameter >53 cm dbh; decay, number of snags, and quantity of large downed material is generally less than old growth forests. Riparian habitats are a general grouping that includes all areas adjacent to aquatic systems with flowing water that contain elements of both aquatic and terrestrial ecosystems. Freshwater wetlands are defined as transitional lands between terrestrial and aquatic systems where the water table is usually at or near the surface or the land is covered by shallow water; no vegetation is specified other than the presence of hydrophytic plants. Numerous conditions may satisfy the designation as rural natural open space: an area where a priority species resides or uses for breeding or regular feeding, a corridor connecting other priority habitats, or an isolated remnant of natural habitat larger than 10 acres and surrounded by agricultural development. Rural natural open space is a general habitat type that may or may not possess wetland, riparian, aquatic, or forested habitat attributes; thus, specific descriptions of habitat attributes and relationship to focal species habitat requirements is fairly subjective.

Little data are available regarding the relationship between historical and current habitat conditions of WDFW priority habitats; thus, we have no context in which to evaluate habitat change of WDFW priority habitats in the lower Columbia River mainstem and estuary.

Because of the general nature of these habitat designations, there may be considerable overlap among the characteristics of each habitat; thus, analysis of the specific relationships between these habitats and the focal species is problematic. For example, riparian habitats are a general grouping that include elements of aquatic and terrestrial environments; freshwater wetland habitats associated with flowing water may be a subset of the riparian category. Within the freshwater wetland category, there is uncertainty as to whether the wetland is dominated by herbaceous vegetation, shrubs, or trees; each of these wetlands provides very different habitat opportunities for the focal species. Additionally, the rural natural open space is also a general habitat type; unless some knowledge of a specific rural natural open space habitat is available, it is difficult to distinguish whether the habitat includes forest, riparian, wetland, or any combination of these habitat characteristics.

2.1.5.4 Relationship Among Habitat Classification Systems

Each habitat classification system described above was developed with a specific purpose; each system only partially satisfies the needs for this subbasin assessment (i.e. describe aquatic habitats, describe terrestrial habitats, and provide a historical context for evaluating the change in estuary and mainstem habitat types over time). For example, each system differs in the specificity of habitat types and the area covered by those habitat types. In order to completely describe the aquatic and terrestrial habitats throughout the lower Columbia River mainstem and estuary, the habitat classification systems were compared to establish similarities among them. However, because each habitat classification system was developed with different methods, there is no direct relationship among the habitat types used in each system and we relied heavily on professional judgment to determine the relationship among each classification system. We evaluated each habitat classification system to determine possible groupings of specific habitat types from each classification system (Table 2-3); we limited the comparison to habitats known to occur in the lower Columbia River and estuary. For example, wildlife habitats from Johnson and O'Neil (2001) that only occur in eastern regions of Washington or Oregon were not included in the comparison.

Table 2-3. Potential relationship of specific habitat types among the different habitat classification

systems.

Estuarine Habitat Types (Thomas 1983)	Wildlife Habitat Types (Johnson and O'Neil 2001)	LCREP Estuary and Tidal Freshwater Habitats (Garono et al. 2003a)	WDFW Priority Habitats
Deep Water	Open Water – Lakes, River, and Streams	Water	NA
Medium Depth Water	Open Water – Lakes, River, and Streams	Water	NA
Shallow Water/Flats	Open Water – Lakes, River, and Streams Bays and Estuaries	Water	NA
Tidal Marsh	Herbaceous Wetlands	Herbaceous Wetlands Scrub-Shrub Wetlands	Riparian
Tidal Swamp	Westside Riparian-Wetlands	Forested Wetland	Riparian
Non-estuarine Water	Open Water – Lakes, River, and Streams	Water	NA
Non-estuarine Marsh	Herbaceous Wetlands	Herbaceous Wetlands Scrub-Shrub Wetlands	Freshwater Wetland Riparian
Non- estuarine Swamp	Westside Riparian-Wetlands	Forested Wetland	Riparian Freshwater Wetland
Developed Floodplain	Agriculture, Pastures, and Mixed Environs Urban and Mixed Environs	Urban	Rural Natural Open Space
Natural and Filled Uplands	Coastal Dunes and Beaches	Unconsolidated Shore	NA
NA	Westside Lowland Conifer-Hardwood Forest	Coniferous Forest Upland	Old Growth/Mature Forest
NA	Westside Oak and Dry Douglas-fir Forest	Deciduous Forest Upland	Old Growth/Mature Forest
NA	Montane Mixed Conifer Forest	Mixed Forest Upland Coniferous Forest Upland	Old Growth/Mature Forest
NA	Westside Grasslands	Herbaceous Upland	Rural Natural Open Space

2.1.6 Estuary and Lower Mainstem Zones

The Columbia River estuary and lower mainstem consists of two major physiographic subsystems: the estuarine subsystem and the tidal freshwater subsystem (Johnson et al. 2003b). The estuary and lower mainstem are dynamic subsystems, resulting partially from interactions between seasonal flow and salinity-tidal regimes. Subsystem designation was based on efforts of the Columbia River Estuary Data Development Program (Simenstad et al. 1984). The estuarine subsystem extends from the Columbia River mouth to Puget Island (rm 0-46) and includes 7 distinct areas based on habitat structure, salinity concentration, and sediment composition: Entrance, Mixing Zone, Youngs Bay, Baker Bay, Grays Bay, Cathlamet Bay, and the Upper Estuary. A map of the estuarine subsystem boundaries is provided in Figure 2-5, however, a similar map was not available for the tidal freshwater subsystem. Boundary delineation of these areas is consistent with the estuary areas discussed by Thomas (1983). The freshwater subsystem extends from Puget Island to Bonneville Dam and is separated into 2 areas (i.e. rm 46-105 and rm 105-146). The distinct areas within the estuary and tidal freshwater subsystems are briefly described below based on Johnson et al. (2003b):

- **Entrance** – The area is dominated by subtidal habitat and has the highest salinity in the estuary. Historically, the Entrance was a high-energy area of natural fluvial land forms (e.g. Clatsop Spit, Trestle Bay), and a complex of channels, shallow water, and sand bars. The Entrance area supports the Columbia Plume, which creates a unique low-salinity, high productivity environment extending well into the ocean. The dynamic nature of the areas has changed as a result of dredging and jetty construction, which have limited wave action and the ocean-fed supply of sediment.
- **Mixing Zone** – The area is characterized by a network of mid-channel shoals and flats, such as Desdemona and Taylor Sands. The Mixing Zone has the highest variation in salinity within the estuary based on interactions between tide cycles and river flow. The estuary turbidity maximum (ETM), which is created through these interactions, is often located within the Mixing Zone. Urban development, primarily around Astoria, has moderately impacted intertidal and subtidal habitats in the area.
- **Youngs Bay** – The area is characterized by a broad flood plain and was historically abundant in tidal marsh and swamp habitat. Diking and flood control structures were used to convert floodplain habitat in the area to pasture. The remaining fragmented tidal marsh and tidal swamp habitats in Youngs Bay are thought to be different in structure and vegetative community than the historical condition of these habitats.
- **Baker Bay** – The area was historically a high energy area from ocean currents and wave action, which have been altered as a result of dredging and jetty construction. Additionally, migration of mid-channel islands toward the interior of Baker Bay has sheltered the area from wave action. As a result, tidal marsh habitat has recently started to develop in some areas while much of the historical tidal marsh and tidal swamp habitat has been lost because of dike construction in the floodplain. Because of proximity to the river mouth, Baker Bay consists primarily of brackish water.
- **Grays Bay** – Historically, water circulation in the area was a result of interactions between river flow and tidal intrusion. Pile dikes constructed adjacent to the main Columbia River navigation channel have decreased circulation in Grays Bay; this circulation change has caused flooding problems in the Grays and Deep River valley bottoms and has promoted tidal marsh habitat development in the accreting bay. Dike construction, primarily for pasture conversion, has isolated the main channel from its historical floodplain and eliminated much of the historical tidal swamp habitat.

- **Cathlamet Bay** – The area is characterized by some of the most intact and productive tidal marsh and swamp habitat remaining in the estuary; a large portion of Cathlamet Bay is protected by the Lewis and Clark National Wildlife Refuge. The western edge of Cathlamet Bay contains part of the brackish oligohaline zone, which is thought the be important during juvenile anadromous fish transition from fresh to salt water. Portions of Cathlamet Bay have lost substantial acreage of tidal swamp habitat as a result of dike construction; conversely, tidal marsh habitat has formed along the fringe of dredge disposal locations.
- **Upper Estuary** – The area is characterized by deep channels and steep shorelines on both sides of the river. The narrow channel structure produces an area dominated more by tidal swamp habitat and less edge habitat (tidal marsh). The Upper Estuary is typically dominated by freshwater, except during low river flow or large flood tides. Dike construction and clearing of vegetation has resulted in a substantial loss of tidal marsh habitat on Puget Island and within the Skamokawa and Elochoman floodplain.
- **Tidal Freshwater** – The tidal freshwater subsystem is distinct from the estuarine subsystem based on geology, vegetation, and climate. This region is influenced by major tributaries such as the Willamette, Cowlitz, Lewis, and Kalama Rivers. This area of the Columbia River mainstem is characterized by elongate islands that divide the river and form oxbow lakes, sloughs, and side channels (e.g. Sauvie Island and Scappoose Bay). The tidal freshwater subsystem was historically dominated by a combination of tidal plant communities, ash riparian forests, and marshy lowlands.

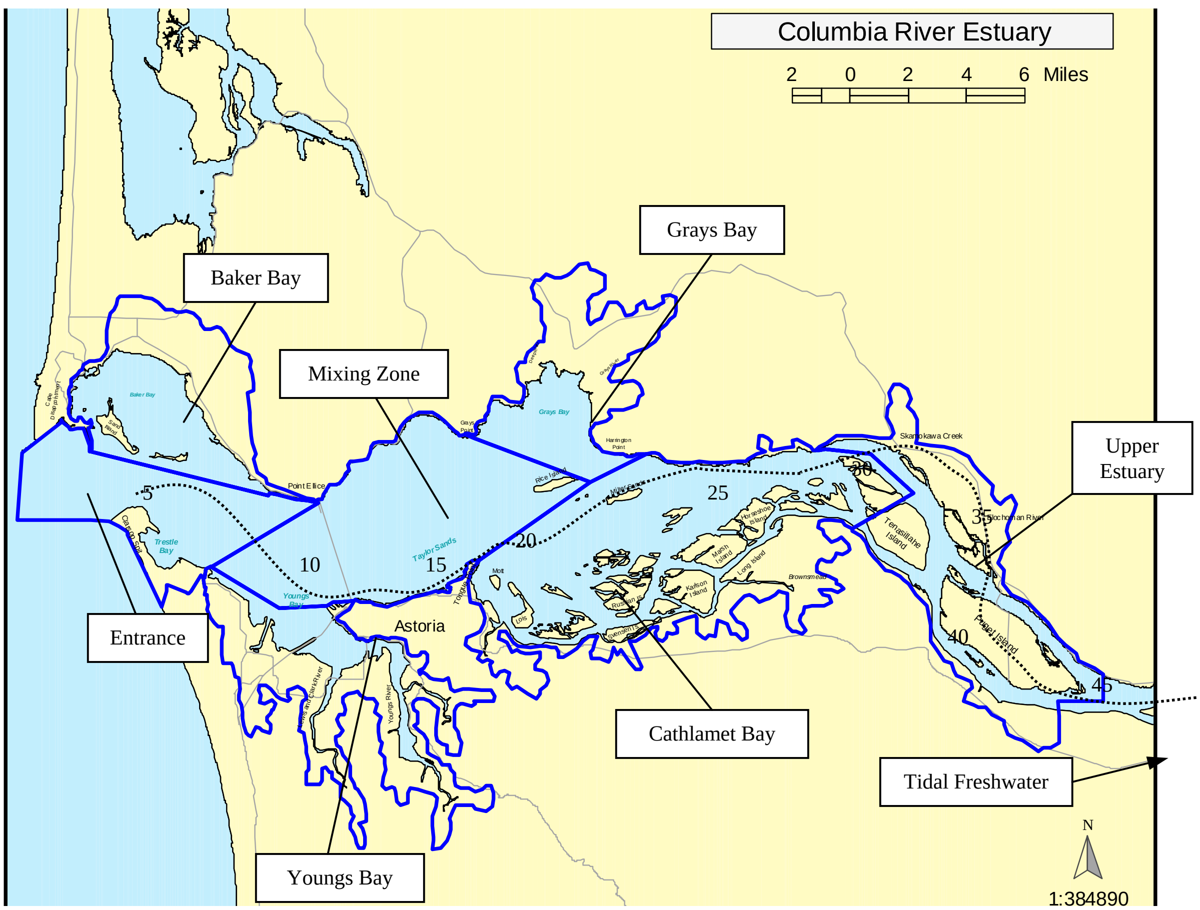


Figure 2-5. Approximate area boundaries of distinct physiographic areas within the Columbia River estuary based on Thomas (1983) and Johnson et al. (2003b). Dashed line represents an approximation of the main channel; numbers along this channel are approximate river mile measurements.

2.1.7 Major Land Uses

The size of the subbasin lends itself to an abundance of possible land uses. The area contains multiple population centers and political jurisdictions, including the largest Oregon population center (Portland) and the fourth largest in Washington (Vancouver; Figure 2-1). Nine counties are located wholly or partially within the subbasin as well as 14 port districts. Jurisdictional boundaries of many of these entities overlap. The following list is a brief description of the major land uses within the lower Columbia River mainstem and estuary subbasin (Marriott et al. 2001):

- Approximately 2.5 million people live in the basin; many others visit for recreation or business.
- Hundreds of fish and wildlife species reside in or migrate through the estuary; more than a dozen rare and endangered species utilize the lower river and estuary.
- Bonneville Dam generates power for the region and beyond as part of the Federal Columbia River Power System.
- Five deep-water ports support a shipping industry that transports 30 million tons of goods annually.
- Timber harvest occurs throughout the basin; six major pulp and paper mills contribute to the regions economy.
- Aluminum plants along the river produce 43% of the U.S.'s aluminum.
- Agriculture is widespread throughout the floodplain, including many fruit and vegetable crops as well as beef and dairy cattle.
- Although commercial fishing activity has declined in recent years, the industry continues to play a significant role in the region.
- Primary recreational activities include fishing, boating, hiking, and windsurfing.

2.1.8 Areas of Biological Significance

Numerous areas of special biological significance provide critical natural habitats and help maintain the delicate balance of the ecosystem. Since 1870, more than half of the tidal swamp and marsh areas in the lower river have been lost as a result of diking, draining, filling, dredging, and flow regulation. Since 1948, tidal wetland habitats in the lower 46 miles of the river have decreased by as much as 70%. Much of the remaining wetlands are protected by inclusion in the Lewis and Clark and the Julia Butler Hansen National Wildlife Refuges. In addition to the feeding, spawning, nursery, and migratory habitat they provide, these wetlands are critical to flood control and water quality. Specific areas of special biological significance in the Lower Columbia River Estuary Program include:

- Clatsop Spit in Fort Stevens State Park is a significant migratory shorebird feeding and nesting area for sanderlings
- Baker Bay, Youngs Bay, Trestle Bay, Grays Bay and Cathlamet Bay are especially productive areas for benthic organisms, anadromous fish and waterfowl
- Bald eagle nesting sites in the lower estuary
- High-quality wetlands in Pacific County

- Lewis and Clark National Wildlife Refuge, which includes most of the islands and the open water between RM 18 and 25; managed primarily for waterfowl
- Julia Butler Hansen National Wildlife Refuge, which includes the lower Elochoman River area in Washington
- Tenasillahee Island Research Natural Area; the upstream tip of the island consists of a spruce swamp that is a remnant of a once widespread habitat type in the program study area
- Puget Island Natural Area Preserve
- White Island Natural Area Preserve, black cottonwood-willow community, and high-quality surge-plain wetlands in Wahkiakum County
- Ridgefield National Wildlife Refuge
- Vancouver Lake Lowlands, including Shillapoo Wildlife Recreation Area
- Sauvie Island Wildlife Management Area
- Steigerwald Lake Wildlife Refuge
- Franz Lake Wildlife Refuge
- Pierce Island Natural Area Preserve and a high-quality, black cottonwood-Oregon ash community, both in Skamania County
- Pierce Ranch Wildlife Refuge

Other areas of special biological significance include: Bradwood Cliffs; Kerry Island; Big and Little Creek Estuary; Tansy Point; Tongue Point; Cooperage Slough; Russian Point Marsh; East Sand Island; Gnat Creek Marsh; Blind Slough Spruce Swamp; Burnside Marsh; Deer Island; Wallace Island; Prescott and Carr Slough; Wapato Bay; Scappoose Flats; Sandy Island; Burlington Bottom; Smith and Bybee Lakes; Virginia Lake; McGuire Island; Sandy River Delta; Gary, Flat, and Chatham Islands; Horsetail Creek Wetlands; and Rooster Rock State Park wetlands.

2.2 Focal Species

Focal species are those species that have special legal, ecological, cultural, or local status and are used to evaluate the health of the ecosystem and the effectiveness of management actions. In this section, we describe the process by which the focal species list was created (Section 2.2.1) and provide a brief description of each focal species life history and abundance trends (Sections 2.2.2 through 2.2.15).

2.2.1 Selection Process

Focal species selection followed the NPCC's *Technical Guide for Subbasin Planners* (NPCC 2001). The *Technical Guide* indicates that the assessment of focal species serves two functions:

- It provides insight on the status of species that warrant legal consideration because of Endangered Species Act (ESA) or treaty right considerations; and
- It serves a diagnostic function, with certain species used as an indicator of broad ecological health.

Further, focal species are used to evaluate the effectiveness of management actions and the health of the ecosystem. The *Technical Guide* offers four criteria for selecting focal species (in order of importance):

- Designation as Federal endangered or threatened species;
- Ecological significance;
- Cultural significance; and
- Local significance.

Within the Lower Columbia and Estuary subbasins, identification and selection of species has been a thoughtful and deliberative facet of the subbasin planning process. Early in 2001, the Lower Columbia Fish Recovery Board (LCFRB), together with the Washington Department of Fish and Wildlife, considered an initial set of 21 species for the 11 subbasins on the Washington State side of the Lower Columbia Region, including the mainstem and estuary. In 2003, the Lower Columbia River Estuary Program (LCREP; now called the Lower Columbia River Estuary Partnership) entered into an agreement with Oregon to participate with the LCFRB in the co-development of a subbasin plan for the Columbia Estuary and Columbia Lower Subbasins. A Planning Group[1] was formed to guide this effort. The Planning Group added three additional species not contemplated by the LCFRB (i.e. river otter, osprey, and bald eagle). Table 2-4 depicts the selection of species for the estuary/mainstem subbasin assessment and their relationship to selection criteria.

[1] NOAA Fisheries, US Fish & Wildlife Service, WA Dept of Fish & Wildlife, OR Department of Fish & Wildlife, LCREP, LCFRB, City of Portland, Clatsop County Economic Development, CREST, USACE, Washington & Oregon State (fill in others).

Table 2-4. Species Selection and Planning Context.

Species	ESA	Ecological[1]	Cultural	Economic[2]	Recreation[3]
Species of Primary Interest (Focal Species)					
Fall Chinook	X	X	X	X	X
Chum	X	X	X	X	X
Spring Chinook	X	X	X	X	X
Winter Steelhead	X	X	X	X	X
Summer Steelhead	X	X	X	X	X
Coho	X	X	X	X	X
Pacific Lamprey	X	X	X		
Bald Eagle	X	X	X		
CWT Deer	X	X[4]	X		
Green Sturgeon		X	X		
White Sturgeon		X	X	X	X
Species of Ecological Significance					
N. Pikeminnow		X		X[8]	X
Shad		X[7]		X	X
River Otter		X[9]			
Eulachon		X	X	X	X
Caspian Tern		X[6]		X	
Osprey		X			
Yellow Warbler		X[10]			
Red-eyed Vireo		X[10]			
Species of Management Interest					
Dusky Canada Goose				X[5]	
Sandhill Crane	X			X[5]	
Species of Recreational Significance					
Walleye		X[7]			X
Smallmouth Bass		X[7]			X
Channel Catfish		X[7]			X

1 May be positive or negative ecological impact; this column only indicates relative significance.

2 May be positive or negative economic impact; this column only indicates relative significance.

3Active recreation potential (e.g., harvest).

4 Likely ecologically important historically.

5 Seasonal crop damage.

6 Historically not present in estuary.

7 Non-native species.

8 Some economic importance for control program.

9 Indicator of ecosystem health.

10 Indicator of habitat type.

In the species selection process, it became evident that individual species were important to the subbasin planning process for different purposes and significance at the subbasin- and Columbia River Basin-scale. Some species, like summer steelhead, have basin-wide significance in terms of their legal, ecological, cultural, economic, and recreational significance. Other species, like the river otter, are of interest because of their value as an indicator of ecosystem health. Still others, like yellow warblers, are indicators of a specific habitat type.

The Planning Group decided to organize the list of species into broad categories that help convey the purpose and significance that individual species play in the planning process. All species will be addressed in the management plan and will have biological objectives and strategies developed for them, although the structure of the biological objectives and strategies may take different form due to inherent differences in their significance, ecological interactions, information available, and management structures in place.

Species of Primary Interest (Focal Species). This category of species will receive the highest level of attention and are considered the focal species for purposes of developing a subbasin plan that adheres to the standards of the Council. The ocean-type and stream-type salmonids play a major role in structure and content of the subbasin assessment because of their importance to all of the selection criteria, the absence of management plans in the estuary/mainstem, and the far-reaching implications of their life cycle requirements to various landscape-level processes and habitat conditions within and outside of the subbasins. Well developed recovery or management plans exist for bald eagle, CWT deer, pacific lamprey, and the green/white sturgeon. The plans augment this assessment and provide the basis for developing biological objectives and strategies for these species. The subbasin management plan will address the integration of the various species-specific management plans into a balanced approach for all focal species.

Species of Ecological Interest: This category of species is intended to inform subbasin planners on the general health of the estuary/mainstem in terms of quality of the environment, habitat diversity, or management issues. Each of these species will be addressed in the management plan. Native species include: Northern Pikeminnow, River Otter, eulachon, Caspian terns, Osprey, yellow warbler, and red-eyed vireo; non-native species include shad.

Species of Management Interest: This category of species is important from a management perspective and are indicative of a habitat type that is not represented elsewhere in the planning process (e.g., agricultural lands). Species include the Dusky Canada Goose and the Sandhill Crane (federally listed).

Species of Recreational Interest: This category of non-native species has recreational interest in the estuary/mainstem, as well as poorly understood ecological interactions with salmonids. They include walleye, smallmouth bass, and channel catfish.

Detailed descriptions of the biology and life history of each species are found elsewhere in the Technical Foundation (i.e. Volume I [for salmonids] or Volume III [for other species; except for river otter, bald eagle, and osprey, which were not part of the Technical Foundation]). The following sections are intended to briefly describe the life history of each focal species as it relates to potential use of lower Columbia River mainstem and estuary habitats.

2.2.2 Ocean-type Salmonids

Ocean-type salmonids represent the life history strategy that migrates downstream from the spawning area within days to months of emergence from the gravel. Early migrants may only be 30-40 mm fork length, while later migrants are usually larger, ranging from 50-80 mm fork length; subyearling migrants from the mid-Columbia and further up the basin tend to be considerably larger, ranging from 70-100 mm fork length (NMFS 2002). Ocean-type salmonid populations in the lower Columbia River include fall chinook and chum salmon. Ocean-type juvenile salmon commonly spend weeks to months rearing in the lower mainstem and estuary prior to reaching the requisite size for ocean entry and survival. Ocean-type salmon are oriented to low velocity, near-shore habitats; riparian/wetland areas in the mainstem and tidal marsh habitats in the estuary that are connected to the lower river (i.e. access not blocked via dikes) provide essential cover and feeding requirements of ocean-type juvenile salmon (Simenstad and Cordell 2000 as cited in USACE 2001, Bottom et al. 2001). They are often associated with substrates consisting of fines and sands, although this may be an artifact of the low velocity preference rather than a partiality for fine-grained substrates. As fish grow, ocean-type juvenile salmon utilize other habitat types (e.g. water column habitat) and are not as strongly associated with near-shore habitats.

2.2.2.1 Fall Chinook

Chinook salmon (*Oncorhynchus tshawytscha*) are the largest and most diverse of the Pacific salmon. Two runs of fall chinook return to Washington lower Columbia River tributaries: “tule” fall chinook, and “bright” late fall chinook. Tule fall chinook return from August through November to spawn almost immediately, typically in large tributary mainstems. Fall chinook have ocean-type life histories where juveniles gradually migrate downstream as subyearlings during their first spring and summer. Most tule fall chinook adults return after 2 to 3 years in the ocean where they range along the coasts of Washington, Oregon, and British Columbia. Bright late fall chinook return from August through October and spawn November through January. Life history is otherwise similar to tule fall chinook except the lower river bright fall run migrates farther north, and may spend up to 4 years in the ocean before returning.

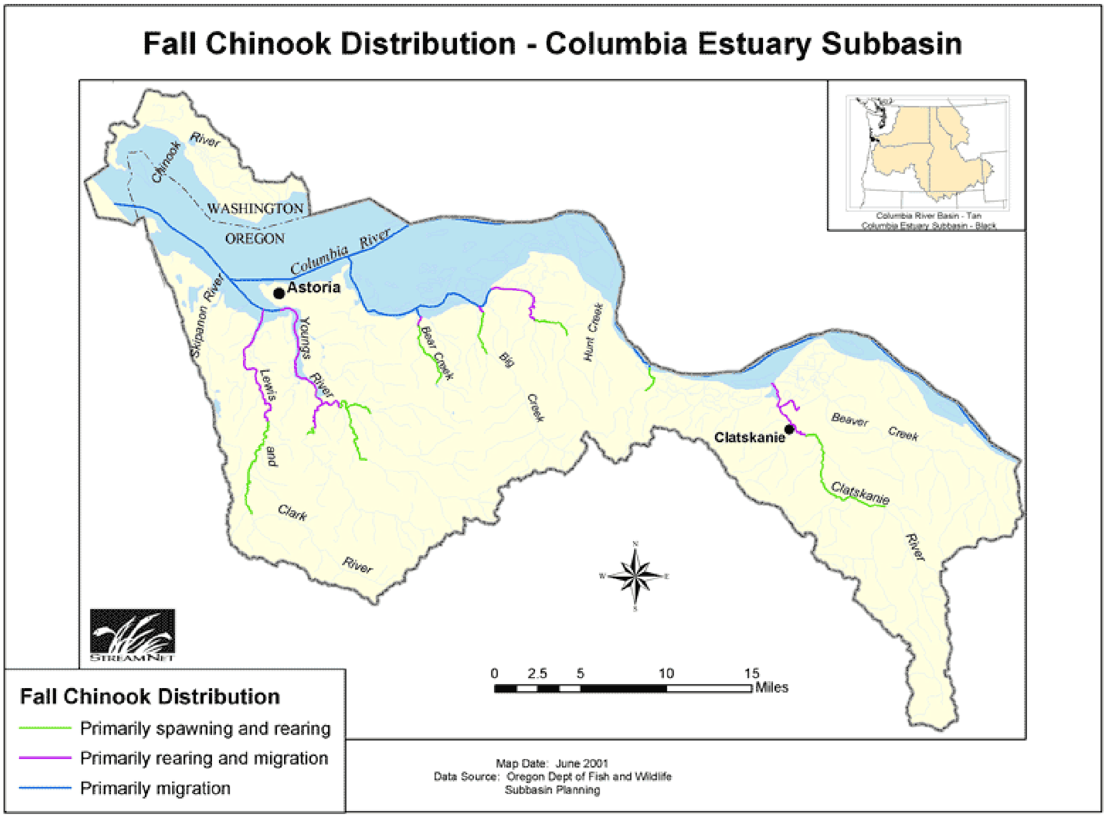
Lower Columbia River chinook populations were listed as threatened in 1999. Chinook salmon were historically present in all Washington lower Columbia tributaries. Tule fall chinook were widely distributed while bright fall chinook were limited to the Lewis River, and perhaps the mainstem Columbia near the present Bonneville Dam site. The Willamette/Lower Columbia Technical Recovery Team has identified 31 historical populations of chinook salmon in the Columbia River ESU. Washington accounts for 13 of 20 tule fall and 1 of 2 late fall chinook populations in this ESU; the other chinook populations originate in Oregon waters. All Washington lower Columbia chinook populations are below proposed recovery targets with the possible exceptions of Lewis late fall, Coweeman fall, and East Fork Lewis fall population. Current runs of tule fall chinook are dominated by hatchery-produced fish.

Fall chinook exhibit some variability in their timing of migration to the estuary. Some fall chinook fry migrate to the ocean soon after yolk resorption at 30-45 mm in length (Lister et al. 1971, Healey 1991). In most river systems, however, fry migrate at 60–150 days post-hatching or as fingerling in the late summer or autumn of their first year. When environmental conditions are not conducive to subyearling emigration, ocean-type chinook salmon may remain in fresh water for their entire first year.

In the Columbia River estuary, subyearling chinook salmon were captured in every month of the year and were distributed throughout freshwater, estuarine, and marine regions (Bottom et al. 1984). Reimers (1973), working in the Sixes River, Oregon, suggested that estuarine rearing is critical to fall chinook survival. Subyearling chinook were one of the most abundant species collected in the Columbia River estuary; Bottom et al. (1984) suggested that subyearling chinook abundance was partially related to their slow migration through the estuary (i.e. subyearling chinook were available for long periods of time in a variety of estuarine habitats). For example, subyearling chinook tagged and released in April and May were captured in the estuary through October (Bottom et al. 1984). Subyearling chinook moved through the estuary slower than other salmonids; in fact, migration rate appeared to decrease for about half the hatchery groups when they entered the estuary (Bottom et al. 1984). Generally, juvenile hatchery subyearling chinook released further upstream in the basin migrated at a faster rate than juveniles released lower in the system (Bottom et al. 1984). Subyearling chinook abundance was highest in the spring and summer months; during spring and summer, subyearling chinook were most frequently associated with water column and nearshore habitats while in the winter, they were more frequently associated with nearshore, shoals, and bay habitats (Bottom et al. 1984). Subyearling chinook represented 68% of the total catch of juvenile salmonids in the estuary (Bottom et al. 1984).

Diet of juvenile fall chinook varies considerably based on fish size and location in the river, estuary, and nearshore habitats (see Craddock et al. 1976, McConnell et al. 1978, Levy and Northcote 1982, McCabe et al. 1983, Bottom et al. 1984, Dawley et al. 1986, McCabe et al. 1986, Bottom and Jones 1990, Sherwood et al. 1990, Brodeur 1992, Miller and Simenstad 1997, Simenstad and Cordell 2000). For young chinook in the lower mainstem, Craddock et al. (1976) determined that diptera were the primary prey species during the winter and spring while zooplankton (primarily *Daphnia*) were the major prey item from July to October; similarly, Bottom et al. (1984) and Bottom and Jones (1990) reported that young chinook in the estuary primarily ate amphipods (*Corophium*), cladocerans (*Daphnia*), and diptera, with *Corophium* dominant in winter and spring and *Daphnia* dominant in summer.

Adult fall chinook primarily use the Columbia River estuary and lower mainstem as a migratory route to spawning areas (Figure 2-6). There is evidence of fall chinook spawning and subsequent rearing in Oregon tributaries in the estuary region and in Washington tributaries in the tidal freshwater region near Bonneville Dam (Figure 2-6). Recent spawning surveys indicate fall chinook spawning in the Columbia River mainstem below Bonneville Dam; however, these fish are expected to be hatchery strays and the National Marine Fisheries Service (NMFS) does not consider them to be part of the lower Columbia River fall chinook ESU. (For more information regarding the fall chinook life cycle, refer to the Technical Foundation, Volume I, section 3.2)



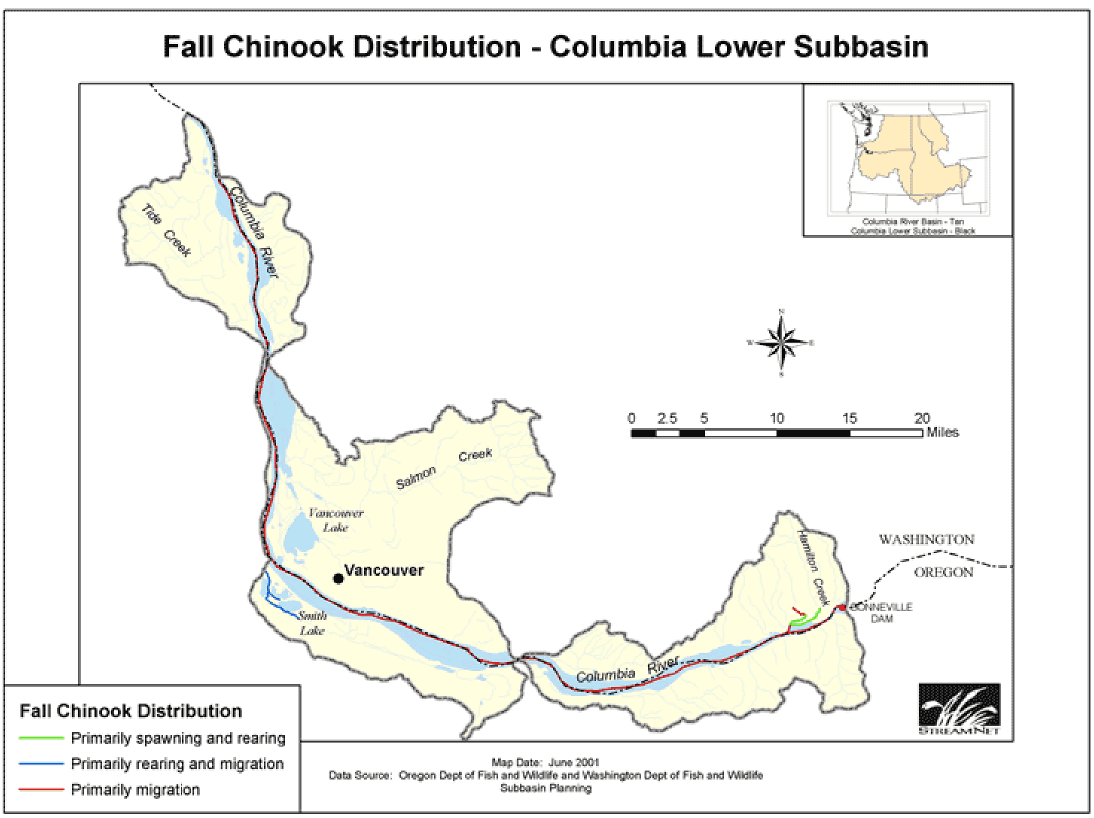


Figure 2-6. Adult fall chinook distribution in the Columbia Estuary and Columbia Lower

Subbasins.

2.2.2.2 Chum Salmon

Chum salmon (*Oncorhynchus keta*) return during fall (generally October/November) to spawn in the lowermost reaches of the Columbia River tributaries often just above tidewater. Chum fry migrate downstream almost immediately after emergence and spend most of their life in the estuary or ocean. Runs of over 1 million chum are believed to have once returned to the Columbia River. Annual runs now average 4,000 fish, about 3% of the historical run size. All naturally produced chum populations in the Columbia River and its tributaries in Oregon and Washington were listed as threatened in August 1999.

Chum salmon once migrated as far upstream as the Walla Walla River. Today, production is generally limited to areas downstream of Bonneville Dam, including Grays River, Hardy Creek, and Hamilton Creek, and in the mainstem Columbia River near Ives Island. The latter three populations are located immediately downstream of Bonneville Dam. The Willamette/Lower Columbia Technical Recovery Team has identified 16 historical populations of chum salmon in the Columbia River ESU. Of these, eight occur only in Washington, six occur only in Oregon, and two are shared between states. Chum populations have been largely extirpated for 14 of 16 historical populations. Significant populations exist only in the Grays River and the lower Columbia River Gorge tributaries and mainstem. All chum populations are below the lower bound of proposed recovery planning targets with the possible exception of the lower Gorge population.

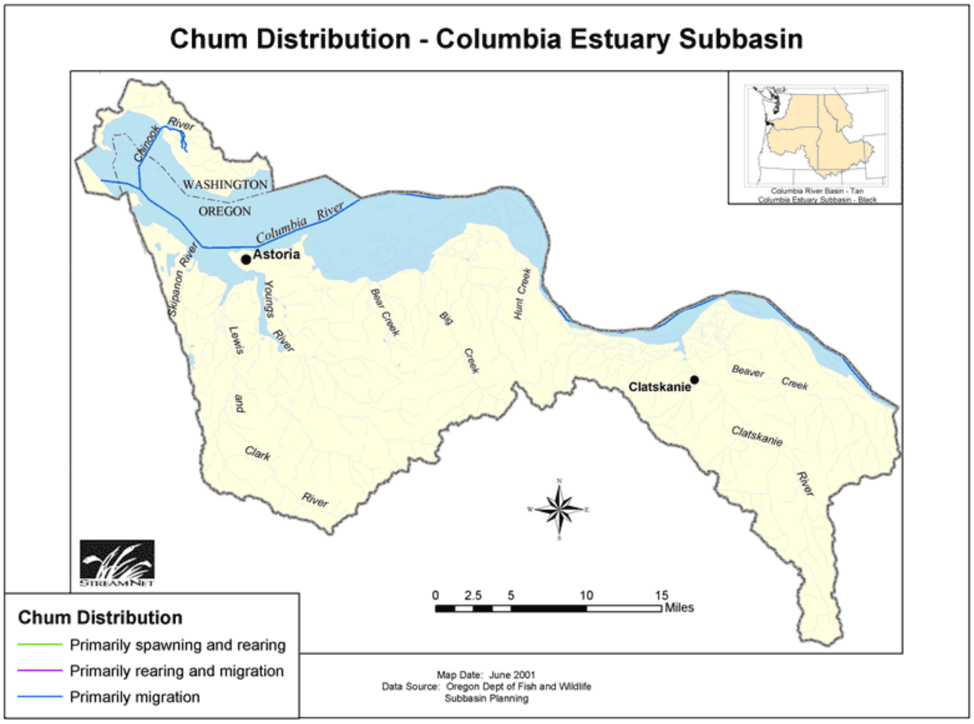
The period of estuarine residence appears to be the most critical phase in the life history of chum salmon and may play a major role in determining the size of the subsequent adult run back to fresh water (Mazer and Shepard 1962, Bakkala 1970, Mathews and Senn 1975, Fraser et al. 1978, Peterman 1978, Sakuramoto and Yamada 1980, Martin et al. 1986, Healey 1982, Bax 1983, Salo 1991).

Chum fry generally emigrate shortly after emergence; several factors influence the timing of downstream migration, including time of adult spawning, stream temperatures during egg incubation and after hatching, fry size and nutritional condition, population density, food availability, stream discharge volume and turbidity, physiological changes in the fry, tidal cycles, and day length (Simenstad et al. 1982, Salo 1991). In Washington, chum may reside in fresh water for as long as a month (Salo and Noble 1953, Bostick 1955, Beall 1972).

In the Columbia River estuary, juvenile chum salmon were a minor portion of the catch during sampling efforts of Bottom et al. (1984); chum, sockeye, and cutthroat collectively represented 1% of the total juvenile salmonid catch. Chum salmon juveniles were captured in the estuary during April and May during both years of the study; chum salmon were present in the estuary from February through June (Bottom et al. 1984). Juvenile chum salmon were primarily distributed within the freshwater or estuarine regions of the estuary, although there was one occurrence in the marine region (Bottom et al. 1984).

Diet varies considerably based on fish size and location in the river, estuary, and nearshore habitats (see Craddock et al. 1976, McConnell et al. 1978, Levy and Northcote 1982, McCabe et al. 1983, Bottom et al. 1984, Dawley et al. 1986, McCabe et al. 1986, Bottom and Jones 1990, Sherwood et al. 1990, Brodeur 1992, Miller and Simenstad 1997, Simenstad and Cordell 2000).

Chum salmon adults utilize the Columbia River estuary and lower mainstem for migration to spawning areas. Chum salmon are known to spawn in Washington tributaries associated with the Columbia Estuary and Columbia Lower Subbasins, such as the Chinook River or Hamilton Creek (Figure 2-7). Further, spawning and outmigration surveys have documented successful chum spawning in the lower mainstem Columbia River below Bonneville Dam along the north bank near the I-205 bridge. (For more information regarding the chum salmon life cycle, refer to the Technical Foundation, Volume I, section 3.1.)



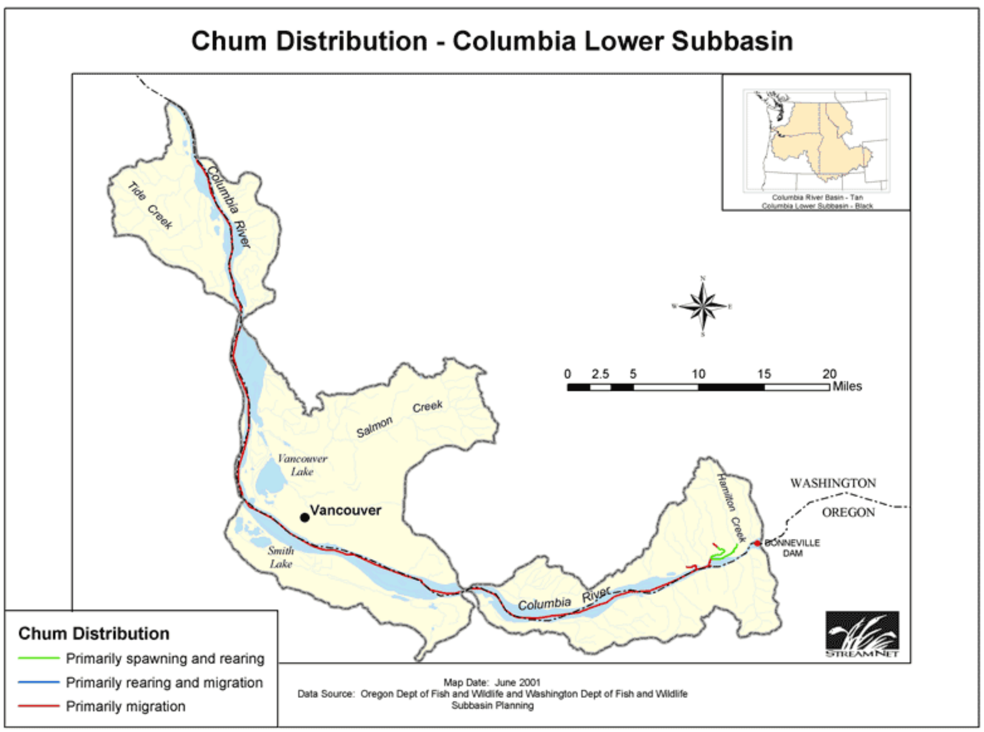


Figure 2-7. Adult chum salmon distribution in the Columbia Estuary and Columbia Lower Subbasins.

2.2.3 Stream-Type Salmonids

Stream-type salmonids represent the life history strategy that rear within the natal stream for months or years after emergence from the gravel and outmigrate during their second year of life. In general, stream-type juvenile salmon reach the lower mainstem and estuary at a relatively large size (> 80mm) and commonly spend less time than ocean-type salmonids rearing in the lower mainstem and estuary. Stream-type juvenile salmonids actively migrate through the lower Columbia River mainstem and estuary. Stream-type salmon are oriented to water column habitats and are typically found throughout the near surface water column (i.e. top 6 m); they tend to avoid low-velocity areas and are not associated with any specific substrate type. Stream-type salmonid populations in the lower Columbia River include spring chinook, winter steelhead, summer steelhead, and coho salmon.

Yearling salmonids have been documented eating the same types of organisms as subyearlings, although the composition and specific diet items likely differs. For example, Bottom et al. (1984) noted that adult *Diptera* and *Corophium spp*. were major prey items of both yearling and subyearling chinook; however, *Diptera* accounted for about 55% of yearling chinook diet while it accounted for about 8% of the diet of subyearling chinook. In the lower Columbia River and estuary, Dawley et al. (1986) and Bottom and Jones (1990) observed yearlings salmonids consuming diptera, cladocerans, and amphipods.

2.2.3.1 Spring Chinook

Chinook salmon (*Oncorhynchus tshawytscha*) are the largest and most diverse of the Pacific salmon. Spring chinook typically return to freshwater in March and April and migrate into small headwater streams to spawn in late summer. Spring chinook exhibit a stream-type life history where juveniles rear in tributary streams for one year before rapidly migrating downstream on the spring freshet. Most adults return after 2 to 4 years in the ocean where they migrate far to the north off Canada and Alaska.

Lower Columbia River chinook populations were listed as threatened in 1999. Chinook salmon were historically present in all Washington lower Columbia tributaries; spring chinook were present in the larger Cascade subbasins. The Willamette/Lower Columbia Technical Recovery Team has identified 31 historical populations of chinook salmon in the Columbia River ESU. Washington accounts for 7 of 9 spring chinook populations in this ESU; the other chinook populations originate in Oregon waters. All Washington lower Columbia spring chinook populations are below proposed recovery targets. Current runs of spring chinook are dominated by hatchery-produced fish.

Yearling chinook salmon were present in the estuary most months of the year and were distributed throughout the freshwater, estuarine, and marine regions (Bottom et al. 1984). Yearling chinook abundance was highest in April and May and was relatively low for most other months; they represented 8% of the catch of juvenile salmonids (Bottom et al. 1984). Yearling chinook were most frequently associated with water column and nearshore habitats; they were most susceptible to purse seine harvest in main channel sampling stations, indicating an affinity to water column habitat (Bottom et al. 1984). Yearling chinook migrated through the estuary faster than subyearlings but slower than steelhead (Bottom et al. 1984). More than half of the hatchery groups of yearling chinook appeared to decrease their migration rate through the estuary, however, only about a third increased in mean fork length (Bottom et al. 1984). As with other salmonids, juvenile hatchery yearling chinook released further upstream in the basin migrated at a faster rate than juveniles released lower in the system (Bottom et al. 1984).

Adult spring chinook utilize the estuary and lower mainstem primarily as a migration route to spawning locations. There is no evidence of spring chinook spawning in the lower mainstem or in tributaries of the Columbia Estuary and Columbia Lower subbasins (Figure 2-8). (For more information regarding the spring chinook life cycle, refer to the Technical Foundation, Volume I, section 3.2.)

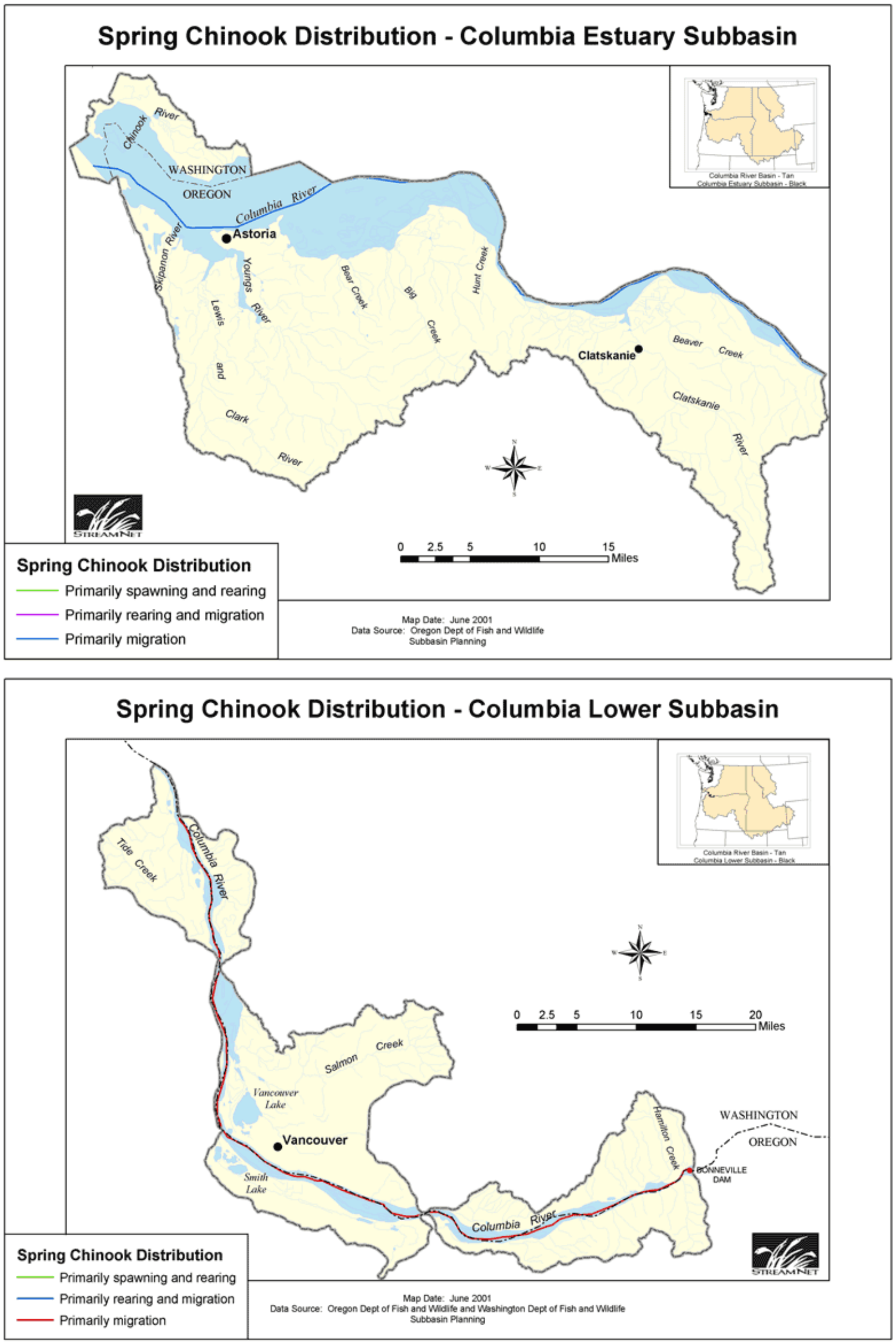


Figure 2-8. Adult spring chinook distribution in the Columbia Estuary and Columbia Lower Subbasins.

2.2.3.2 Steelhead

Steelhead (*Oncorhynchus mykiss*) are rainbow trout that migrate to and from the ocean. Resident and anadromous life histories are often found in the same population. Steelhead exhibit tremendous variability in life history with juveniles rearing for 1 to 4 years in freshwater before migrating seaward and as adults spending 1 to 3 years in the ocean. Steelhead generally migrate northward along the coast of Canada and Alaska before dispersing far out into the North Pacific.

Lower Columbia River steelhead are listed as threatened under the ESA. The Willamette/Lower Columbia Technical Recovery Team has identified 23 historical populations of steelhead in the Columbia River ESU. Washington accounts for 14 of 17 winter run steelhead and 5 of 6 summer run steelhead populations in this ESU. Three additional winter run populations of the unlisted Washington Coast ESU occur in lower Columbia subbasins included in this planning process. Small but significant steelhead populations remain in most Washington subbasins where they were historically present. All Washington lower Columbia winter steelhead populations are below proposed recovery planning targets with the possible exception of the Kalama winter steelhead population. All Washington lower Columbia summer steelhead populations are below proposed recovery planning targets with the possible exception of the Wind summer population.

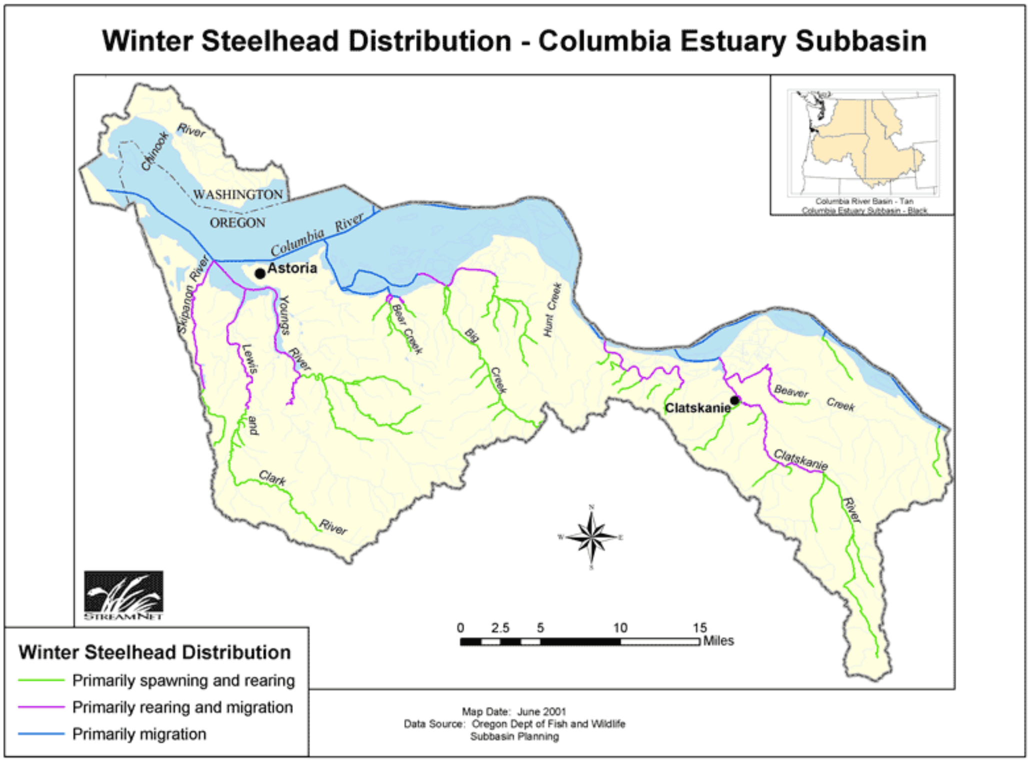
Steelhead in the Columbia River estuary consumed a relatively even proportion of *Corophium salmonis* (amphipod), *Corbicula manilensis* (bivalve), and adult *Diptera* (Bottom et al. 1984).

Juvenile steelhead were present in the Columbia River estuary from February to July of each year of sampling by Bottom et al. (1984); steelhead abundance was greatest in May and relatively low for other months (Bottom et al. 1984). Juvenile steelhead constituted 5% of the total juvenile salmonid catch (Bottom et al. 1984). Steelhead juveniles were distributed throughout the freshwater, estuarine, and marine regions of the estuary; they were most frequently associated with water column habitats (Bottom et al. 1984). Juvenile steelhead moved through the estuary more rapidly than other salmonids; based on catch data, they were present in the estuary for the shortest duration of any of the salmonid group (Bottom et al. 1984). Winter steelhead have been found to migrate at an average rate of 3.3 km/hr, traveling 134-143 km in 32 to 90 hours (Durkin 1982, Dawley et al. 1986 as cited in USACE 2001). Migration rate of many hatchery groups of juvenile steelhead increased through the estuary (Bottom et al. 1984). As with other salmonids, juvenile hatchery steelhead released further upstream in the basin migrated at a faster rate than juveniles released lower in the system (Bottom et al. 1984).

2.2.3.2.1 Winter Steelhead

Winter steelhead return to fresh water between December and May and generally spawn in late April and early May. Winter steelhead returned to the Cowlitz, Kalama, NF and EF Lewis, Washougal, and Wind. Where winter and summer runs occur in the same stream, winter steelhead tend to spawn lower in the watershed than summer steelhead.

Adult winter steelhead use the Columbia River estuary and lower mainstem for migration to spawning areas. Further, winter steelhead are known to spawn and rear in numerous small tributaries associated with the Columbia Estuary and Columbia Lower Subbasins (Figure 2-9). (For more information regarding the winter steelhead life cycle, refer to the Technical Foundation, Volume I, section 3.4.)



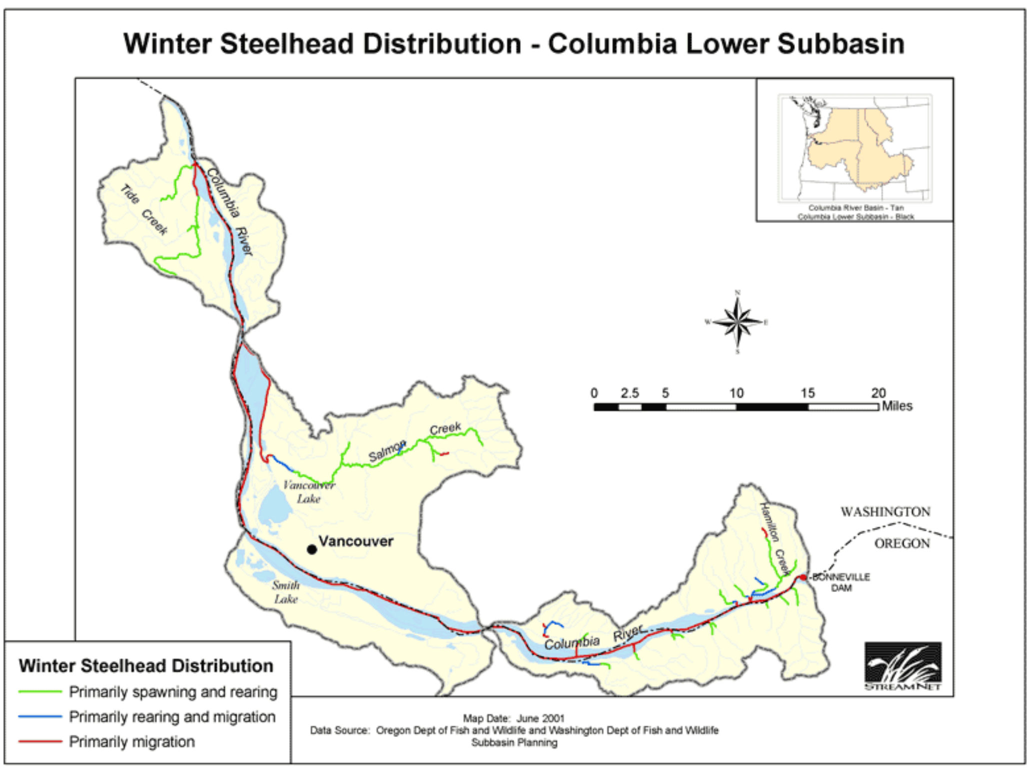
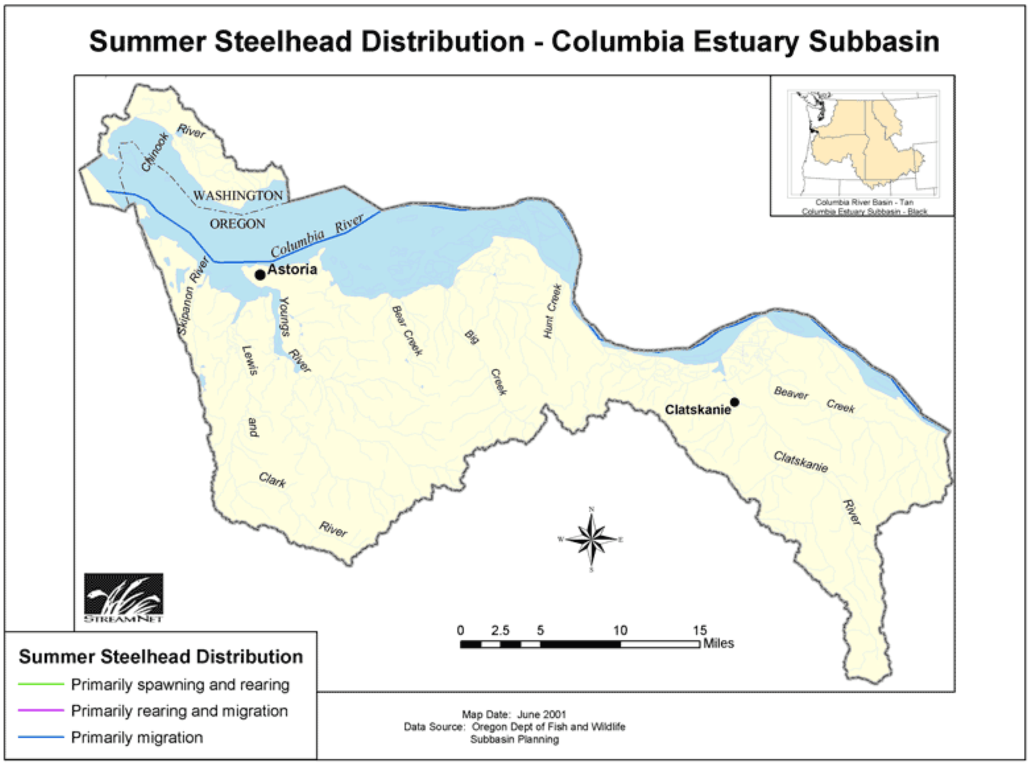


Figure 2-9. Adult winter steelhead distribution in the Columbia Estuary and Columbia Lower Subbasins.

2.2.3.2.2 Summer Steelhead

Summer steelhead return from the ocean between May and October and generally spawn between late February and early April. Watersheds that historically supported summer steelhead included the Kalama, North Fork Lewis, East Fork Lewis, Washougal, and Wind. Where summer and winter runs occur in the same stream, summer steelhead tend to spawn higher in the watershed than winter steelhead.

Adult summer steelhead use the Columbia River estuary and lower mainstem for migration to spawning areas. Further, there is evidence of summer steelhead spawning and rearing in small Oregon tributaries associated with the Columbia Lower Subbasin (Figure 2-10). (For more information regarding the summer steelhead life cycle, refer to the Technical Foundation, Volume I, section 3.4.)



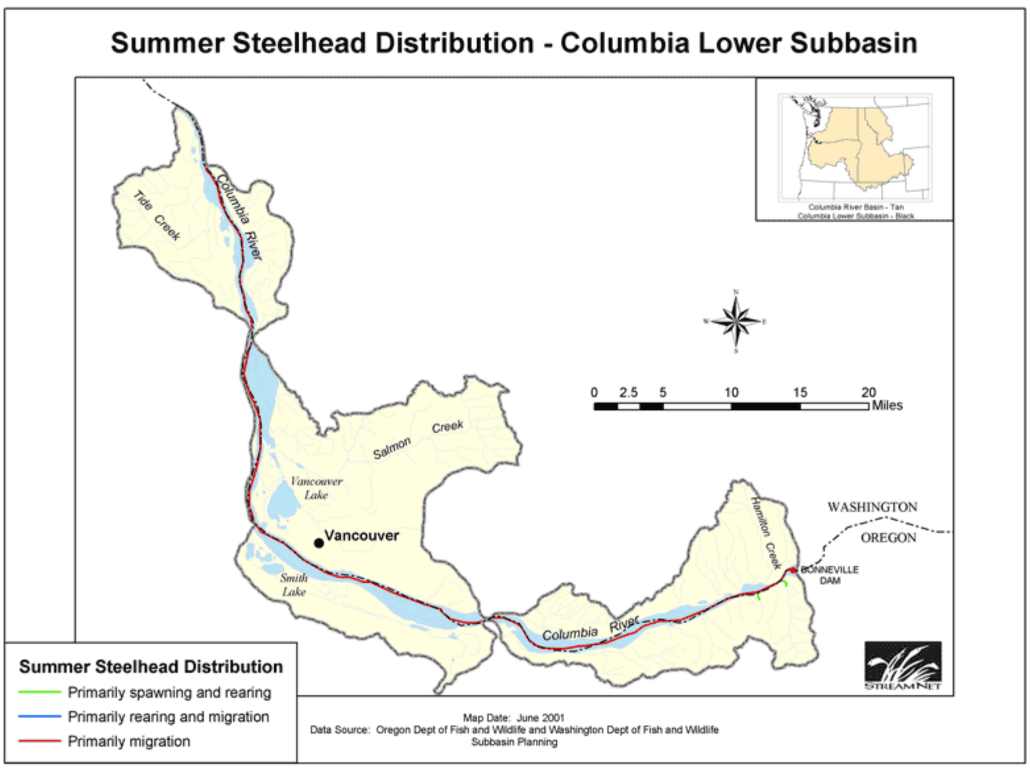


Figure 2-10. Adult summer steelhead distribution in the Columbia Estuary and Columbia Lower Subbasins.

2.2.3.3 Coho Salmon

Coho (Oncorhynchus kisutch) salmon spawn during fall in small streams with the onset of spawning typically tied to fall freshets in September and October. Coho adults are almost entirely 3-year olds although a few jacks return at age 2. Juvenile coho rear in freshwater for one year prior to migration during spring. Lower Columbia River coho runs include early and late returning stocks. Most early-run fish migrate south to mature in coastal Oregon waters. Most late-run coho migrate north into Washington coastal waters.

Coho are currently a candidate for listing under the ESA. Coho salmon historically returned to spawn in all accessible tributary reaches in the lower Columbia River basin. Today, coho populations in Washington tributaries of the lower Columbia River have been heavily influenced by extensive hatchery releases. Past fishery impacts were excessive for coho, however, current fishing impacts are relatively low as a result of implementation of selective fisheries. Tributary hydropower development has blocked significant coho habitat in the Cowlitz and Lewis basins. Current stream habitat conditions severely limit coho production.

Recent numbers of natural coho spawners are generally unknown although most wild populations are thought to have been extirpated or consist of no more than a few hundred fish. Approximately 13 Washington lower Columbia River subbasins were historically used by coho salmon according to the NOAA Fisheries status review and Washington's salmon stock inventory. Recovery targets have not yet been proposed for coho because of incomplete habitat and status information on which they could be based.

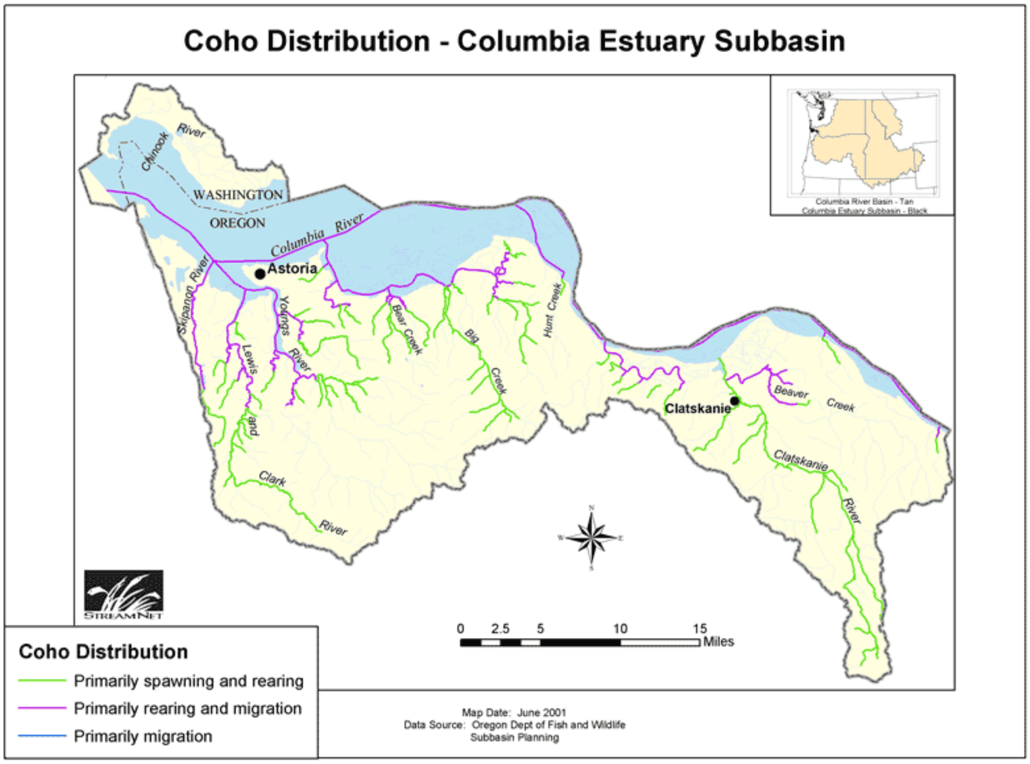
Most juvenile coho, in the region south of central British Columbia, migrate seaward as smolts in late spring, typically during their second year. Factors that tend to affect the time of migration include: the size of the fish, flow conditions, water temperature, dissolved oxygen levels, day length, and the availability of food (Shapovalov and Taft 1954). The size of coho smolts is fairly consistent over the species' geographic range; a FL of 100 mm seems to be the threshold for smoltification (Gribanov 1948).

Juvenile coho salmon were present in the Columbia River estuary from March to August of each year of sampling by Bottom et al. (1984); coho abundance was greatest in May and June and relatively low for other months (Bottom et al. 1984). Juvenile coho salmon comprised 18% of the total juvenile salmonid catch (Bottom et al. 1984). Coho juveniles were distributed throughout the freshwater, estuarine, and marine regions of the estuary; they were most frequently associated with water column habitats, however, tagged hatchery coho released in the lower Columbia (i.e. Grays River (rm 34) and Big Creek (rm 29)) were more likely to be found in shallow bays and intertidal areas than upriver coho (Bottom et al. 1984). Juvenile coho salmon moved through the estuary relatively quickly and appeared to increase their migration rate through the estuary (Bottom et al. 1984). As with other salmonids, juvenile hatchery coho released further upstream in the basin migrated at a faster rate than juveniles released lower in the system (Bottom et al. 1984).

The most common prey items of coho salmon in the Columbia River estuary were *Corophium salmonis* and *Corophium spinicorne* (amphipods) and adult *Diptera*; *Corophium salmonis* constituted over half of the coho diet (Bottom et al. 1984).

Adult coho salmon use the Columbia River estuary and lower mainstem for migration to spawning areas. Further, coho salmon are known to spawn and rear in numerous small tributaries associated with the Columbia Estuary and Columbia Lower Subbasins (Figure 2-11).

(For more information regarding the coho salmon life cycle, refer to the Technical Foundation, Volume I, section 3.3.)



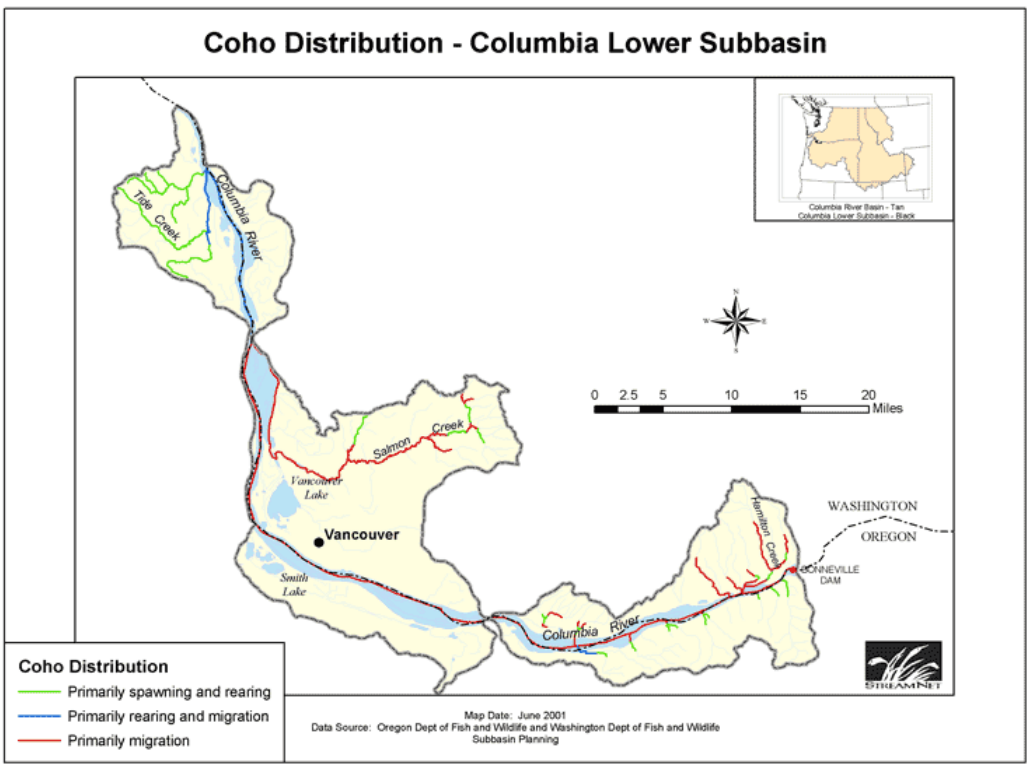


Figure 2-11. Adult coho salmon distribution in the Columbia Estuary and Columbia Lower Subbasins.

2.2.4 Pacific Lamprey

Pacific lamprey (*Lampetra tridentata*) are a native anadromous inhabitant of Pacific Northwest rivers including the Columbia. Lamprey spawn in small tributaries, historically as far upstream as Idaho and British Columbia, and die after spawning. Young lamprey, called ammocoetes, are algae filter feeders that burrow in sandy stream margins and side channels for up to 6 years before downstream migration. Adults are predators that feed only in the ocean and attach themselves to their prey with suction mouths.

Lamprey were historically an important food source for native peoples and a significant component of the Columbia River ecosystem. Spawning adults are a source of marine-derived nutrients in the freshwater and an important prey item for sturgeon and marine mammals. In fresh water, at least 7 aquatic and five avian species prey on juvenile lamprey. Relatively little is known about status and biology of Pacific lamprey. Most data suggests that populations in the Columbia basin have been declining concurrent with hydroelectric development and other habitat changes. Although adult lamprey can negotiate waterfalls, they apparently have difficulty in dam passage and juveniles migrating downstream do not appear to benefit from juvenile passage systems.

Adult Pacific lamprey entry into freshwater can vary from February (Kan 1975) to September (Beamish 1980, Scott and Crossman 1973). Habitat utilization of the lower Columbia River mainstem and estuary by adult lampreys is not known; likely, the lower Columbia River serves primarily as a migration corridor. Further, similar to most adult salmonids, lamprey feeding ceases during upstream migrations (Scott and Crossman 1973). The first juvenile life stage of lampreys, ammocoetes, burrow into sand and silt substrates after hatching where they filter feed on algae (Scott and Crossman 1973, Kostow 2002). Ammocoetes spend approximately 6 years rearing in freshwater; rearing begins downstream of the nest and, as ammocoetes grow, they gradually move downstream, generally at night, continuing to burrow and filter feed in fine substrates (Scott and Crossman 1973, Kostow 2002, Claire 2003). Because of this burrowing activity, ammocoetes may be an indicator of water quality or contaminants (Gustavo Bisbal, USFWS, personal communication). Older ammocoetes generally occupy the lower portions of river basins, and thus, may be found throughout the tidal freshwater portion of the lower Columbia. Pacific lamprey ammocoetes metamorphose into macrothalmia (physiological equivalent of a smolt) and begin the seaward migration; during this transformation, Pacific lamprey survive on lipid reserves and do not feed (Kostow 2002).

In the Columbia River estuary, juvenile Pacific lamprey were present from December to June; Pacific lamprey abundance was highest in December and was extremely low for the remainder of the year (Bottom et al. 1984). Juvenile Pacific lamprey abundance in the Columbia River estuary is relatively low compared to most other species captured (Bottom et al. 1984). Pacific lamprey juveniles were distributed throughout the freshwater, estuarine, and marine regions of the estuary, however, presence in the marine region was limited. In an analysis of estuary feeding groups, juvenile Pacific lamprey were grouped with white sturgeon, however, no data were collected regarding lamprey diet composition. This is consistent with the life history data presented above that indicates Pacific lamprey do not feed during their downstream migration to saltwater. Pacific lamprey life history data suggests use of Columbia River estuarine habitats is limited. (For more information regarding the Pacific lamprey life cycle, refer to the Technical Foundation, Volume III, section 2.0.)

2.2.5 *Sturgeon*

2.2.5.1 White Sturgeon

White sturgeon (*Acipenser transmontanus*) live in large rivers along the Pacific coast of North America and move freely between freshwater and the ocean where they may remain for variable but prolonged periods. Large sizes (over 12 feet and 1000 pounds) and long life spans (100 years or more) allow them to negotiate heavy current and outlast good and bad periods. These fish are bottom-oriented feeders that eat primarily shrimp and clams as young but graduate to a live fish diet as they get larger.

Sturgeon are an ancient order of fishes that have existed for hundreds of millions of years. Sturgeon species are found in most major river systems of the Northern Hemisphere but have been widely decimated by over fishing and dam construction. Their long lifespan and late age of maturity make sturgeon particularly susceptible to over fishing. Columbia River white sturgeon were severely over fished during the late 1800's prior to the adoption of significant fishery restrictions and recovery required decades. Mainstem dams block movements, fragment the habitat, and reduce anadromous prey. Sturgeon rarely use fish ladders which were engineered to pass the more surface-oriented salmon.

White sturgeon historically ranged all the way to the Canadian headwaters of the Columbia River and to Shoshone Falls in the upper Snake River. The lower Columbia population is among the largest and most productive sturgeon populations in the world and sustains excellent sport and commercial fisheries. However, many upriver populations have declined or disappeared. Bonneville reservoir continues to support a significant white sturgeon population although numbers and sizes are substantially less than in the lower river. Only the Kootenai River subpopulation of white sturgeon has been listed under the Endangered Species Act (endangered).

White sturgeon move freely between fresh and saltwater environments (DeVore et al. 1999); as a result, individual white sturgeon in the Columbia River below Bonneville Dam may exhibit any number of life history strategies (Bemis and Kynard 1997, Kynard 1997). Movements of adult white sturgeon in freshwater vary considerably and appear to be a function of access and seasonal food availability (Beamesderfer et al. 1995). In the lower Columbia River, DeVore and Grimes (1993) reported that adults often 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). DeVore et al. (1999) reported of 471 white sturgeon were originally tagged in the unimpounded lower Columbia River downstream from Bonneville Dam, sturgeon were recaptured in 23 separate locations outside the Columbia River Basin from the Fraser River, B.C., to the Sacramento River, CA, from 1976–97. Thus, adult white sturgeon may be found anytime throughout the lower Columbia River mainstem and estuary; extensive seasonal use of the estuary during summer is likely. White sturgeon often concentrate in deep water habitats, but are known to freely feed in a wide range of habitats throughout its range.

White sturgeon are communal, broadcast spawners (Wang et al. 1985; Conte et al. 1988; Paragamian et al. 2001, and references therein) that 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). In the lower Columbia River mainstem, white sturgeon are known to spawn in the free-flowing reach of the Columbia River Gorge below Bonneville Dam. Adhesive embryos settle to the substrate; white sturgeon larvae remain in the substrate until the yolk is absorbed (Brannon et al. 1985). White sturgeon that burrow into fine sediments commonly die as a result of suffocation. The larval swim-up dispersal stage of white sturgeon enter the water column and are subject to the influences of current (Brannon et al. 1985). Larvae seek substrates that provide cover and remain associated with these substrate until the yolk is absorbed and feeding is initiated (Brannon et al. 1985). Larvae begin exogenous feeding and metamorphose into juveniles at about 3-4 months after fertilization (Parsley et al. 2002). Juveniles feed on a variety of prey items, including chironomid larvae, amphipods, and mysis shrimp (Scott and Crossman 1973, Wydowski and Whitney 1979, Sprague et al. 1993). Thus, juvenile white sturgeon may also be found anytime throughout the lower Columbia River mainstem and estuary in a variety of different habitats.

In the Columbia River estuary, white sturgeon were part of a large group of benthic and epibenthic feeders present during the summer (Bottom et al. 1984). *Corophium salmonis* (amphipod) comprised the majority of the white sturgeon diet; other important diet items included *Neomysis mercedis* and *Macoma balthica* (Bottom et al. 1984).

In the Columbia River estuary, white sturgeon were captured all months of the year during sampling efforts by Bottom et al. (1984); catch was twice as high in the summer compared to the rest of the year. Although, white sturgeon catch was relatively low compared to other species present in the estuary (Bottom et al. 1984). White sturgeon distribution was limited to the freshwater and estuarine regions of the estuary; white sturgeon were not captured in the marine region of the estuary (Bottom et al. 1984). In the spring, white sturgeon were most frequently associated with channel bottom habitats in the freshwater region of the estuary; in the summer, white sturgeon were most frequently associated with water column and channel bottom habitats in the freshwater and estuarine regions of the estuary. (For more information regarding the white sturgeon life cycle, refer to the Technical Foundation, Volume III, section 1.0.)

2.2.5.2 Green Sturgeon

Green sturgeon (*Acipenser medirostris*) occur in the lower Columbia River but do not typically range far upstream from the estuary. NOAA Fisheries completed a status review for green sturgeon in 2003 and determined that listing under the Endangered Species Act was not warranted.

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). 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; no spawning occurs in the 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.

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).

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. 2002). 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. 2002). Larvae hatched in the laboratory are photonegative, exhibiting hiding behavior (Deng et al. 2002), 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. 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).

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.

2.2.6 Northern Pikeminnow

The northern pikeminnow *(Ptychocheilus oregonensis)* is native to freshwater lakes and rivers of the Pacific slope of western North America from Oregon to northern British Columbia. This opportunistic species has flourished with habitat changes in the mainstem Columbia River and its tributaries. Pikeminnow are of particular interest for their predation on juvenile salmon. Salmonids are an important food for large pikeminnow and millions of juvenile salmonids are estimated to fall prey each year. Predation can be especially intense in dam forebays and tailraces where normal smolt migration behavior is disrupted by dam passage. A pikeminnow management program has been implemented in the Columbia and Snake rivers in an attempt to reduce predation mortality by reducing numbers of the large, old pikeminnow that account for most of the losses. A bounty fishery program for recreational anglers is aimed at balancing pikeminnow numbers rather than eliminating the species and has also stimulated development of a popular fishery.

Northern pikeminnow are large (10-20 inches), long-lived (10-15 years), slow-growing predaceous minnows (Cyprinidae). Northern pikeminnow have successfully evolved in a range of dynamic lentic and lotic ecosystems and successfully adapted to their varied habitat conditions; they are considered opportunistic generalists that inhabit slow to moderately flowing streams and lakes. Based on known distribution and habitat usage, all life stages of northern pikeminnow may be found in many habitat types throughout the lower Columbia River mainstem; however, usage of estuarine habitats is minimal because of low salinity tolerance. This is consistent with data collected by Bottom et al. (1984). Northern pikeminnow distribution was limited to the freshwater region of the estuary; pikeminnow were not captured in the marine or estuarine regions of the estuary. Northern pikeminnow were present in the freshwater region of the estuary from June to October; pikeminnow abundance in the estuary was very low relative to other species captured (Bottom et al. 1984).

Beamesderfer (1992) attributed the widespread distribution and resiliency of northern pikeminnow to their relatively broad spawning and rearing habitat requirements. In the Columbia River downstream from its confluence with the Snake River, northern pikeminnow abundance 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; Beamesderfer et al. 1996). 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 that eggs hatch in 7 days at 65°F water, and that the young become free swimming within 14 days. Newly-emerged larval northern pikeminnow in the Columbia River drift downriver during July, generally at night. Although pikeminnow adapted to a variety of habitats, age-0 northern pikeminnow rearing in littoral habitats of the upper John Day Reservoir had significantly greater growth and lower mortality in 1994, a year with low flows, abundant instream vegetation, and high near-shore water temperatures. Parker et al. (1995) observed a similar relationship in pikeminnow age 2 and older; sex-specific growth coefficients were higher and sex-specific annual mortality rates were lower for pikeminnow in Columbia River reservoirs compared the free-flowing reach below Bonneville Dam. However, this may be a function of greater density of northern pikeminnow in the lower mainstem compared to the mainstem reservoirs.

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). (For more information regarding the northern pikeminnow life cycle, refer to the Technical Foundation, Volume III, section 4.0.)

2.2.7 Eulachon

Eulachon is the official common name for smelt (*Thaleichthys pacificus*) which swarm into the lower Columbia River and tributaries to spawn during winter and early spring. Eulachon are a small, anadromous forage fish inhabiting the northeastern Pacific Ocean from Monterey Bay, California, to the Bering Sea and the Pribilof Islands. Adults are typically 5 to 8 inches long and 3-5 years old. Most eulachon die after spawning. Huge schools of smelt spawn in the Columbia and Cowlitz mainstems during most years. Pulses of spawners are also seen sporadically in other tributaries including the Grays, Lewis, and Sandy Rivers.

Smelt support a popular sport and commercial dip net fishery in the tributaries, as well as a commercial gillnet fishery in the Columbia. They are used for food and are also favored as sturgeon bait. Smelt are also eaten in large numbers by other fishes including sturgeon, birds, and marine mammals. Smelt numbers and run patterns can be quite variable and low runs during the 1990's were a source of considerable concern by fishery agencies. Current patterns show a substantial increase in run size compared to the 1990's. The low returns in the 1990's are suspected to be primarily a result of low ocean productivity.

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.

Eulachon spawn primarily at night. Each female deposits approximately 17,000 to 60,000 eggs, depending on size of female (Morrow 1980). Fertilized eggs are adhesive and attach 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).

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 and Saalfeld 1955). After the yolksac is depleted eulachon will feed on pelagic plankton. Stomach samples of juvenile eulachon contained euphausiids (Barraclough 1964). Eulachon rear in near-shore marine areas from shallow to moderate depths. 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.

Eulachon spend the majority of life in salt water and little is known about this saltwater phase. Eulachon feed on plankton in salt water, but stop feeding when returning to fresh water. 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).

2.2.8 River Otter

The river otter (*Lutra canadensis*) is a top predator of most aquatic food chains that has adapted to a wide variety of aquatic habitats, from marine environments to high mountain lakes of North America (Toweill and Tabor 1982, Melquist and Hornocker 1983, Melquist and Dronkert 1987). The river otter is a year-round resident of the lower Columbia River mainstem and estuary (Howerton et al. 1984, Henny et al. 1996), although field observations and trapper data indicate that population numbers are relatively low (Howerton et al. 1984). Otters on the lower Columbia River concentrate their time in shallow, tidal influenced back waters, sloughs, and streams throughout the estuary. River otters exhibit differing degrees of social and spatial structure based on available habitat, shelter, and food (Reid et al. 1994b). Otter home ranges (approximately 11 river miles) are largely defined by local topography and overlap extensively within and among sexes, exhibiting varying degrees of mutual avoidance and tolerance depending on seasonal dispersion and availability of food and shelter (Reid et al. 1994b). However, otters do maintain territories within home ranges that are delineated by scent marking and latrine sites. Areas within territories are used almost exclusively by the defending otter, who excludes other otters of the same sex (i.e., females otter excludes other females and family groups while males exclude other males). Female river otters mate immediately after parturition during the months of March and April, with estrous lasting up to 46 days (Wright 1963, Melquist and Hornocker 1983). Fertilized eggs develop to the blastocyst stage and are arrested in development (delayed implantation) for up to 10 months (Hamilton and Eadie 1964, Tabor and Wight 1977). The duration of pregnancy after implantation occurs is approximately 2 months. Otter diets vary seasonally and generally consist of a wide variety of fish species and aquatic invertebrates such as crabs, crayfish, and mussels (Toweill 1974, Toweill and Tabor 1982, Melquist and Dronkert 1987, Reid et al. 1994a).

2.2.9 Columbian White-tailed Deer

The Columbian white-tailed deer (*Odocoileus virginianus leucurus*), a subspecies of the white-tailed deer, is on the federal Endangered Species List and is classified as endangered under Washington and Oregon state laws. This deer once ranged from Puget Sound to southern Oregon, where it lived in floodplain and riverside habitat. The conversion of much of its habitat to agriculture and unrestricted hunting reduced its numbers to a just a few hundred in the early 20th century. A few scattered populations remain and numbers have climbed to approximately 300-500 in the lower Columbia and 5,000 in the Roseburg area. Habitat conversion and losses coupled with the low productivity of the population are the currently the most important threats to population viability. Recovery goals identify the need to secure additional habitat for population re-introduction.

Columbian white-tailed deer are present in low-lying mainland areas and islands in the Columbia River upper estuary and along the river corridor. They are most closely associated with Westside oak/dry Douglas fir forest within 200m of a stream or river; however, Columbian white tails can be found breeding or feeding in any number of habitats (Westside lowland conifer-hardwood forest, Westside grasslands, Westside riparian wetlands, herbaceous wetlands, agriculture/pastures/mixed environments, urban/mixed environments; Johnson and O'Neil 2001). Columbian white-tailed deer are non-migratory; in the Columbian White-Tailed Deer National Wildlife Refuge, mean home range for females was about 390 acres and for males was 475 acres, with daily movements considerably smaller than these ranges (Gavin et al. 1984). The peak of breeding activity is generally around mid-November and peak of fawning is about mid-June (USFWS 1976). Columbian white-tailed deer diet consists of browse, forbs, and grasses; generally, browse is chosen in summer, fall, and winter, forbs are most heavily utilized in spring, summer, and early fall, while 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). (For more information regarding the Columbia white-tailed deer life cycle, refer to the Technical Foundation, Volume III, section 11.0.)

2.2.10 Caspian Tern

Caspian terns (*Sterna caspia*) are highly migratory species that are distributed throughout the world and are currently present in large numbers in the Columbia River estuary. The species is not listed, but is of conservation concern because there are relatively few breeding sites and because of significant predation of listed Columbia River salmonids.

Caspian terns have become increasingly abundant in the Columbia River estuary in recent years, becoming the largest breeding colony in North America (Carter et al. 1995). Breeding colony preference is for newly formed, flat, sandy, mid-channel islands, such as those formed via dredge spoils or accretion. There is considerable concern regarding Caspian tern consumption of juvenile salmonids, however, we have no mechanism to measure whether current tern predation differs significantly from historical predation. Further, management actions to discourage breeding on Rice Island and encourage breeding on East Sand Island appears to be decreasing the amount of tern predation on juvenile salmonids.

Caspian terns are highly migratory and exhibit cosmopolitan distribution (Harrison 1984). 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 area used by Caspian terns was created from dredge spoils from the navigational channel (Roby et al. 1998). 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). Recent management actions have successfully discouraged breeding on Rice Island while encouraging breeding on other estuary islands. 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). The timing of southward migration varies with region (Cuthbert and Wires 1999); 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).

Caspian terns breed in colonies and 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). Nest substrates vary from sand, sand-gravel, spongy marshy soil, or dead or decaying vegetation to hard soil, shell banks, limestone, or bedrock, but terns seem to preferred sand (Quinn and Sirdevan 1998). Caspian terns have been reported to fly up to 38 miles from the breeding colony while foraging (Gill 1976, Ryan et al. 2001, 2002); the Columbia River estuary colony appear to feed within the estuary (Collis et al. 1999, Collis et al. 2001). Caspian terns are piscivorous (Harrison 1984); fish may constitute up to 98% of the diet, particularly during periods of high fish abundance such as the peak of smolt outmigration (Roby et al. 1998). Breeding Caspian terns require one-third of their body weight of fish per day during the nesting season, which also coincides with the peak of smolt migration. Diet of the Rice Island colony is dominated by juvenile salmonids (Roby et al. 1998, Roby et al. 2002) while diet of the East Sand Island colony was primarily non-salmonids (Roby et al. 2002). Studies in 1990 and 1991 revealed that eggs of Caspian terns nesting at Rice and East Sand Islands were contaminanted with organochlorine compounds, including PCBs, DDE, dioxins, and furans, suggesting that their food source (primarily juvenile salmonids) may be contaminated with these coumpounds as well (USFWS 2002). (For more information regarding the Caspian tern life cycle, refer to the Technical Foundation, Volume III, section 10.0.)

2.2.11 Bald Eagle

Bald eagles (*Haliaeetus leucocephalus*) are distributed throughout North America, breeding in most of its range; abundance is highest along coastal areas of the northern conterminous states, Canada, Alaska, as well as Florida and South Carolina. Eagles have been observed to reach a maximum age of about 28 years in the wild (Schempf 1997 as cited in Stinson et al. 2001); captive birds have lived to age 47 (Stalmaster 1987 as cited in Stinson et al. 2001). In general, southern areas within this range are more important as wintering areas than breeding areas. In Washington, bald eagles are substantially more abundant in the cool, maritime region west of the Cascade Mountain range (Stinson et al. 2001).

Depending on the level of competition for food and nest sites, bald eagles may attempt to breed at age 3 or as late as age 8 (Gerrard et al. 1992, Bowman et al. 1995, Buehler 2000 as cited in Stinson et al. 2001). Bald eagles develop pair bonds that generally last until one eagle dies (Jenkins and Jackman 1993 as cited in Stinson et al. 2001). Eagles usually return annually to a nesting territory near a reliable food source; breeding adults will defend their territories from intruding eagles. As with breeding site fidelity, bald eagles seem to exhibit a relatively high annual fidelity to wintering areas (Harmata and Stahlecker 1993, Buehler 2000 as cited in Stinson et al. 2001). Communal night roosts are an important component of bald eagle wintering habitat. Eagles may also roost singly, in pairs or gather in large congregations of as many as 500 individuals at locations that are used year-after-year. Roosts may vary widely but studies have shown that communal night roosts provide a microclimate more favorable than available elsewhere in the vicinity (Keister et al. 1985, Stalmaster 1981, Knight et al. 1983, Stellini 1987 as cited in Stinson et al. 2001).

Bald eagle populations throughout its range exhibited a slow decline because of habitat loss, decreased abundance of winter foods, and harassment/hunting since the time of European settlement. Despite protection with the Bald Eagle Protection Act of 1940, harassment by humans continued because of misidentification with golden eagles, poisoning of bald eagles in conjunction with livestock predator control programs, and collection of bald eagle parts for black market collectors or native American ceremonial uses. The population decline accelerated dramatically after the early 1940s with the widespread use of organochlorine pesticides, particularly DDT (Elliot and Harris 2001-2002). By the 1960s, less than 700 breeding pairs were estimated to exist in the lower 48 states and bald eagles had been extripated from at least seven states within its historical range (Stinson et al. 2001).

The ban of DDT, habitat protection, reduced persecution, and reintroduction projects have aided in recovery of the North American population (Stinson et al. 2001). During the preceeding 25 years, the bald eagle population has doubled every 7-8 years. Most known populations have reached regional recovery goals where applicable, but populations remain below pre-European settlement abundance (Buehler 2000 as cited in Stinson et al. 2001). In Washington the most recent (1998) statewide survey recorded 664 occupied nest sites; this accounts for 12% of the known bald eagle territories across the lower 48 states (Stinson et al. 2001). A recent decline in nest occupancy rate and the occurrence of nest sites in developed areas suggests that nesting habitat in areas of western Washington is approaching saturation (Stinson et al. 2001).

Historically, bald eagles were common and locally abundant throughout Washington; accounts from 1890 indicate that bald eagles were especially abundant near the mouth of the Columbia River (Stinson et al. 2001). No historical population abundance or density estimates are available for bald eagles in Washington. The Washington and Oregon bald eagle populations were included for federal listing as endangered under the Endangered Species Act in 1978. Threats to the population identified at the time of listing included reproductive failure caused by organochlorine pesticides, widespread loss of suitable nesting habitat resulting from logging, housing development, and recreation, and persecution (primarily illegal shooting (USFWS 1978 as cited in Stinson et al. 2001). In 1994, the USFWS proposed to reclassify the bald eagle from endangered to threatened throughout its range; this reclassification was finalized in 1995. In 1999, the USFWS proposed to delist the bald eagle throughout its range, however, this delisting has not been finalized.

Breeding bald eagles require large trees near open water that is not subject to intense human activity and will generally select one of the largest trees in a stand for nesting (Anthony et al. 1982 as cited in Stinson et al. 2001). In Washington, 99% of all bald eagle nests are within 1 mile of a lake, river, or marine shoreline. The distance to open water varies somewhat with shore type; nests tend to be closer to marine shores and rivers than to lake shores. Eagles also require perches distributed throughout their nest territories; perches are prominent points which provide a view of the common foraging area. Because eagles exhibit consistent daily foraging patterns, they often use the same perches (Stalmaster 1987, Gerrard and Bortolotti 1988 as cited in Stinson et al. 2001).

Bald eagles breeding in the lower Columbia River region are year-yound residents and do not migrate during the winter (Garrett et al. 1988). All bald eagle nest sites in this area have been monitored for productivity since the late 1970s, and in recent years there were 96 occupied breeding territories (Isaacs and Anthony 2003). In addition, the area supports an additional wintering population of over 100 eagles. Studies in the early 1980's in the Columbia River estuary indicated eagle diet consisted of 90% fish, 7% birds, and 3% mammals (Watson et al. 1991 as cited in Stinson et al. 2001). Waterfowl were the most common avian prey in nests, while suckers (*Catostomus spp*.), American shad (*Alosa sapidissima*), and carp (*Cyprinus carpio*) were the most common fish prey items. Bald eagles will often steal prey from osprey and gulls, and have even been observed stealing marine invertebrates from sea otters (Watt et al. 1995 as cited in Stinson et al. 2001), and fish from river otters (Taylor 1992 as cited in Stinson et al. 2001). Diet of bald eagles can vary considerably, depending on the geographic location or the methods used to determine diet composition (Knight et al. 1990 as cited in Stinson et al. 2001).

The lower Columbia River bald eagle population is one of only two regional populations in Washington that has exhibited low reproductive success representative of a decreasing population (the other regional population was in Hood Canal). Significant concentrations of DDE, PCB, and dioxins were found in bald eagle eggs on the lower Columbia River (Anthony et al. 1993, USFWS 1999b, Mahaffy et al. 2001 as cited in Stinson et al. 2001); concentrations of these contaminants were above no-effect levels estimated for the species. Despite low reproduction success, the lower Columbia River bald eagle population has increased, likely as a result of recruitment of new adults from other areas. Although, the reproductive health of the lower Columbia population appears to be improving based on recent linear trend analysis (Stinson et al. 2001), bald eagle productivity and breeding success of pairs nesting below river mile 60 remains low, especially for those pairs nesting between river mile 13 to 31 (USFWS 1999b, Isaacs and Anthony 2003).

The density of nesting eagles depends on many factors that determine habitat quality, such as prey populations, human disturbance, and perhaps the availability of nest and perch trees. Occupied nests of adjacent nesting pairs are generally spaced closer in areas of high quality habitat. The seasonal home range that contains the foraging and nesting habitat of a pair averages about 2.6 mi^2 in the Puget Sound region (Watson and Pierce 1998 as cited in Stinson et al. 2001) and about 8.5 mi^2 in the Columbia River Estuary (Garrett et al. 1993 as cited in Stinson et al. 2001). However, most eagle activity in the lower Columbia River occurs within 0.2 mi^2 of the nest site (Garrett et al. 1993).

2.2.12 Osprey

The osprey (*Pandion haliaetus*) is a large piscivorous bird of prey that nests and feeds along the lower Columbia River in spring and summer. Ospreys have nearly worldwide breeding distribution; birds that breed in the Pacific Northwest migrate to wintering grounds in southern

Mexico and northern Central America (Martell et al. 2001). Ospreys nest in forested riparian areas along lakes, rivers, or coastlines; nests are situated atop trees, rock pinnacles, or artificial structures such as channel markers or power/light poles (Poole et al. 2002, Henny et al. 2003a). Adult pairs are thought to mate for life and return to the same area annually for breeding (Poole et al. 2002). Generally, adults spend approximately one month on the breeding grounds before egg laying (Henny et al. 2003a); egg incubation takes about 5 weeks and nestlings are ready to fly approximately 7-8 weeks after hatching (Poole et al. 2002). Along the lower Columbia River during 1997 and 1998, osprey productivity was estimated at 1.64 young/active nest, which is higher than the generally recognized 0.80 young/active nest needed to maintain a stable population (Henny et al. 2003a). Ospreys feed almost exclusively on fish and are not particular about the species of fish they consume (Poole et al. 2002). In the lower Columbia and Willamette Rivers, largescale suckers are an important part of the osprey's diet; ospreys remain close to the nest for feeding (Henny et al. 2003a, 2003b).

The osprey has several advantages as a monitoring species for the health of the Columbia River. The osprey population was studied in detail in 1997 and 1998, and the population nests all along the river up to Umatilla. These earlier data (size of nesting population by river segment, reproductive performance, and residue concentrations in eggs) provide the baseline for comparison with similar data collected in the future to help address contaminant trends over time. Furthermore, residue concentrations in eggs can be compared among locations along the river, such as above and below dams, cities, or other point sources of contaminants. For example, higher PCB concentrations in osprey eggs were detected below Bonneville Dam compared to concentrations above the dam. Other advantages for having the fish-eating osprey as a contaminant monitoring species include:

- Osprey feed primarily on fish close to their nest sites and integrate contaminant exposure in the local area,
- Osprey are at the top of the food chain and are susceptible to biomagnification effects of contaminants (e.g. many contaminants biomagnify from 10 to 100 fold from fish to osprey eggs (Henny et al. 2003b)), and
- Productivity of conspicuous nesters can be monitored in an attempt to establish a response that is linked to population processes.

2.2.13 Sandhill Crane

Historically, sandhill cranes (*Grus canadensis*) occupied a larger North American range than they do today. In Washington, sandhill cranes were historically described as "not common summer resident both sides of the Cascades" (Dawson and Bowles 1909). Evidence of breeding sandhill cranes in Washington was absent from 1941 to 1972, when a paired appeared at Conboy Lake NWR. Sandhill crane breeding habitat in Washington is limited when compared to the large wetland complexes in southern Oregon, northern California, or elsewhere in its range; thus, the potential breeding production in Washington is relatively small compared to other breeding locations. Sandhill cranes have been a state listed endangered species in Washington since 1981. The Yakama Indian Nation has listed the sandhill crane as sensitive (BIA 1993); it is also considered a species of cultural importance. In Oregon, the greater sandhill crane is categorized as vulnerable on the sensitive species list and in California, the greater sandhill crane is listed as threatened.

Sandhill cranes are represented by three subspecies: greater, Canadian, and lesser. The greater sandhill crane is the only subspecies that nests in Washington. The only known breeding sites in Washington 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). 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). In 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). 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). (For more information regarding the sandhill crane life cycle, refer to the Technical Foundation, Volume III, section 12.0.)

2.2.14 Yellow Warbler

Within Washington, yellow warblers (*Dendroica petechia*) are apparently secure and are not of conservation concern. Yellow warblers are an excellent indicator of riparian zone structure and function.

The yellow warbler is a long-distance neotropical migrant; spring migrants begin to arrive in the Pacific Northwest region in April but the peak of spring migration in the region 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). The yellow warbler is a riparian obligate species most strongly associated with wetland habitats that contain Douglas spirea and deciduous tree cover (Rolph 1998). Biological objectives for this species in the lowlands of western Oregon and western Washington 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 (Altman 2001). Yellow warblers are a locally common breeder at lower elevations along rivers and creeks in the Columbia Basin, although only possible breeding evidence has been observed along the lower Columbia River mainstem and estuary (Smith et al. 1997). Yellow warblers capture and consume a variety of insect and arthropod species, as well as wild berries, by gleaning from subcanopy vegetation (Lowther et al. 1999). (For more information regarding the yellow warbler life cycle, refer to the Technical Foundation, Volume III, section 15.0.)

2.2.15 Red-eyed Vireo

The red-eyed vireo (*Vireo olivaceus*) is common in western Washington. This songbird has been one of the most abundant birds in North America, although its numbers seem to have declined recently, possibly as a result of the destruction of wintering habitat in the neotropics, fragmentation of northern breeding forests, or other causes. The red-eyed vireo is secure, particularly in the eastern United States. Within Washington, the red-eyed vireo is common, more widespread in northeastern and southeastern Washington, and not a conservation concern. The red-eyed vireo is an excellent indicator of riparian zone structure and function.

The red-eyed vireo is a long-distance neotropical migrant; it breeds throughout North America and winters in South America (Bent 1965). The red-eyed vireo is locally common in riparian growth and strongly associated with tall, somewhat extensive, closed canopy forests of cottonwood, maple, or alder in the Puget Lowlands (C. Chappell pers. comm.) and along the Columbia River in Clark, Skamania, and Klickitat Counties; presence in the Columbia River estuary is not well documented. Biological objectives for this species in the lowlands of western Oregon and western Washington 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, and riparian woodland >164 ft (50 m) wide (Altman 2001). Vireos are primarily insectivorous, with 85% of their diet composed of insects and only 15% of vegetable material, mostly fruits and berries eaten in August–October. (For more information regarding the red-eyed vireo life cycle, refer to the Technical Foundation, Volume III, section 14.0.)

2.3 Habitat

The habitat discussion is divided into three sections: 2.3.1 Habitat-Forming Processes, 2.3.2 Habitat Change, 2.3.3 Historical vs. Current Habitat Condition. Section 2.3.1 Habitat-Forming Processes describes the physical processes that determine habitat formation in the Columbia River estuary and lower mainstem. Section 2.3.2 Habitat Change identifies the natural and anthropogenic factors that have contributed to habitat changes in the subbasin. Section 2.3.3 Historical vs. Current Habitat Condition describes estimates of acreage change of specific estuary and lower mainstem habitats, presenting results from multiple habitat mapping efforts and discussing the similarities and differences among the mapping efforts.

2.3.1 Habitat-Forming Processes

An estuary is the portion of a river that is influenced by ocean tides. The estuary is a complex interaction of river and tidal forces, a high-energy and dynamic physical and biological system, with high temporal variability in circulation, sedimentation and biological processes (Sherwood and Creager 1990). Habitat formation in the lower Columbia River mainstem and estuary are controlled by opposing hydrologic forces of ocean processes (tides) and river processes (discharge) as depicted in the conceptual model (Figure 2-12). As each of these hydrologic processes interact, the habitats that form are a function of time. These processes may be disturbed by storms, extreme hydrologic events, or catastrophic events such as earthquakes or volcano eruptions. Tides introduce marine-derived sediments to the estuary while river discharge carries freshwater sediments via bedload and suspended sediment. This supply of sediments influences the bathymetry of the estuary through the processes of erosion and accretion. Suspended sediment, along with the production of organic matter, determine the degree of water turbidity. The opposing processes of estuary outflow (river discharge) and inflow (tides) determine the salinity gradient and the type and location of available nutrients. River discharge also directly affects the level of woody debris recruitment to the estuary. Finally, the main components of the habitat formation process (bathymetry, water turbidity, salinity, nutrients, and woody debris) determine the location and type of habitats that form and persist throughout the estuary and lower mainstem.

The habitat-forming processes of accretion, erosion, salinity, and turbidity affect the distribution of plants throughout the estuary. Vegetation within each habitat comprises the majority of primary production in the estuary, via the production of organic matter within plant tissue and the export of dissolved organic matter. Primary productivity is driven by light; as turbidity increases, light through the water column decreases, which can result in less phytoplankton growth and can limit the depth of submerged plants.

Elevation partially controls the types of habitat created and maintained through the various habitat-forming processes (USACE 2001). There is a continuous elevation gradient from tidal swamp to water column habitat, with some elevation overlap between each habitat type. Defined elevation ranges for each habitat type (tidal swamp, tidal marsh, tidal flats, water column) are presented in Thomas (1983). At a given elevation, there is an overriding influence of salinity in the development of each type of habitat which controls the vegetation assemblage.

Habitat Forming Processes Submodel

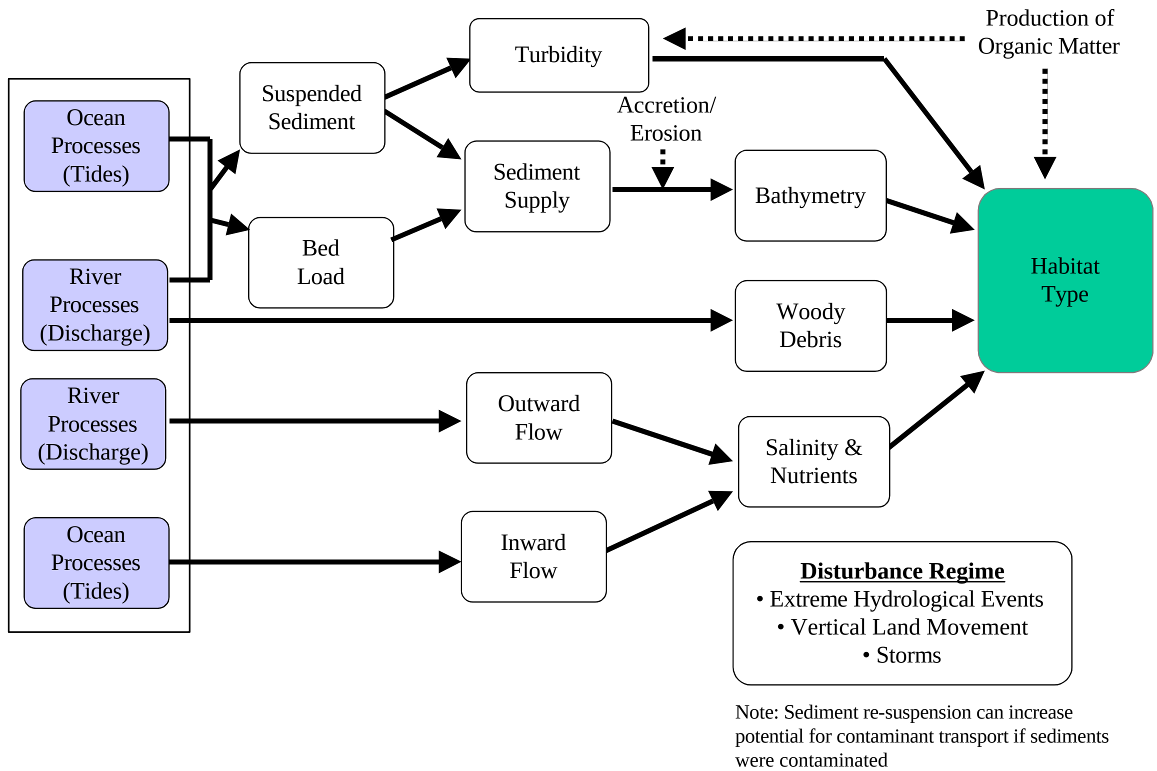


Figure 2-12. Conceptual model of habitat-forming processes in the Columbia River estuary (adapted from USACE 2001). Note that the function of time is not included in this particular model and time is an important controlling factor in the formation of habitat.

2.3.1.1 Hydrological Conditions

Flow affects from upstream dam construction and operation, irrigation withdrawals, shoreline anchoring, channel dredging, and channelization have significantly modified estuarine habitats and have resulted in changes to estuarine circulation, deposition of sediments, and biological processes (ISAB 2000, Bottom et al. 2001, USACE 2001, Johnson et al. 2003b). Flow regulation in the Columbia River basin has been a major contributor to the changes that have occurred in the estuary from historical conditions. The 21 dams built in the Columbia and Snake Rivers since 1933 have caused river flows to be altered substantially. Water losses from irrigation, reservoir evaporation, and climate change have resulted in annual flows at The Dalles, Oregon that are about 17% less than 19th century virgin flows (Bottom et al. 2001). Thus, the predevelopment flow cycle of the Columbia River has been modified by hydropower water regulation and irrigation withdrawal (Thomas 1983, Sherwood et al. 1990 as cited in Nez Perce et al. 1995, Weitkamp 1994, NMFS 2000c, Williams et al. 2000, Bottom et al. 2001, USACE 2001).

Spring freshet properties have been more highly altered than mean flow. Spring freshets are very important to the outmigration of juvenile salmonids; freshet flows stimulate salmon downstream migration and provide a mechanism for rapid migrations. Also, spring freshets (especially overbank flows) provide habitat, increase turbidity thereby limiting predation, and maintain favorable water temperatures during spring and early summer. Further, organic matter supplied by the river during the freshet season is a major factor maintaining the detritus-based food web. Additionally, reductions in freshet flows combined with flood-control diking and wetland development have disconnected the lower river from its floodplain. Consequently, substantial over-bank flows are rare compared to predevelopment flooding frequency, resulting in reduced large woody debris recruitment and riverine sediment transport to the estuary. Flow regulation in the Columbia has decreased spring freshet magnitude and increased flows over the rest of the year as a result of winter drawdown of reservoirs and filling of the reservoirs during the spring runoff season. The best historical record of Columbia River flow exists at the Dalles, Oregon, where a gauging station has recorded flow since 1878. About 97% of the flow of the total Columbia River flow passes the gauge at the Dalles. Average spring freshet flows at the Dalles since 1969 have been reduced by 50-55%, and winter flows (October–March) have increased by 35% (Bottom et al. 2001; Figure 2-13). This same pattern has been observed at Bonneville Dam (USACE 2001; Figure 2-14). Further, most of the spring freshet flow reduction is attributed to flow reduction, about 20% is a result of irrigation withdrawals, and only a small portion (5%) is connected to climatic change (Bottom et al. 2001).

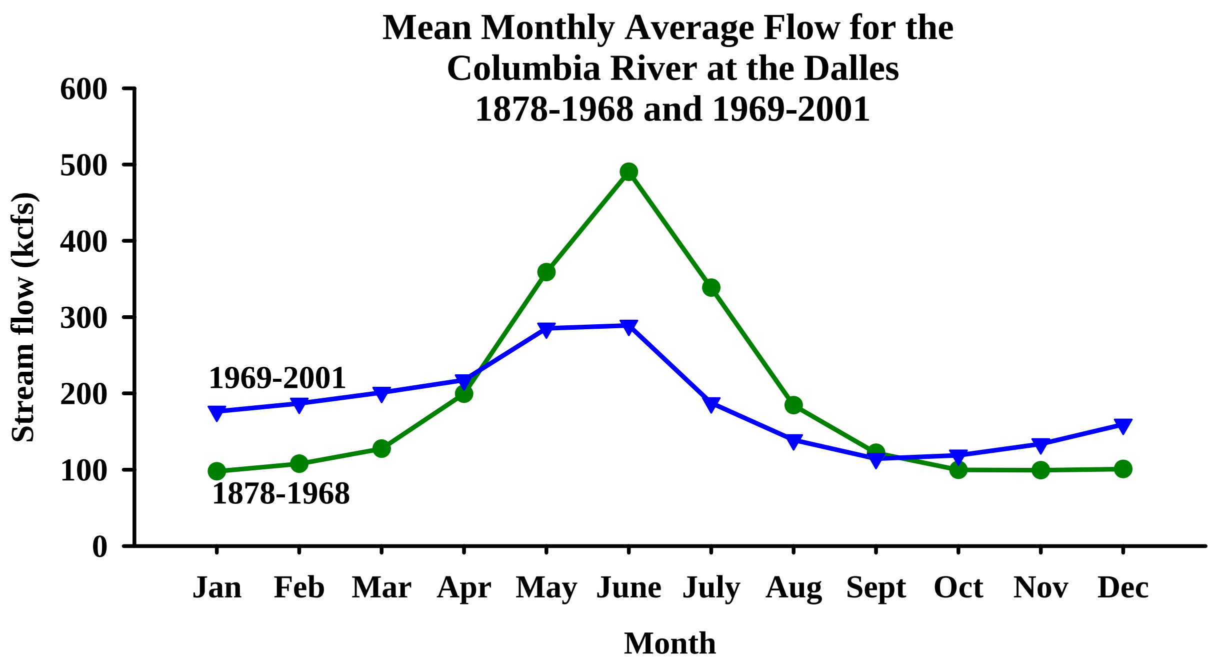


Figure 2-13. Mean monthly average flow at the Dalles. Construction of flow regulating dams has resulted in modification of the annual hydrograph of the Columbia River.

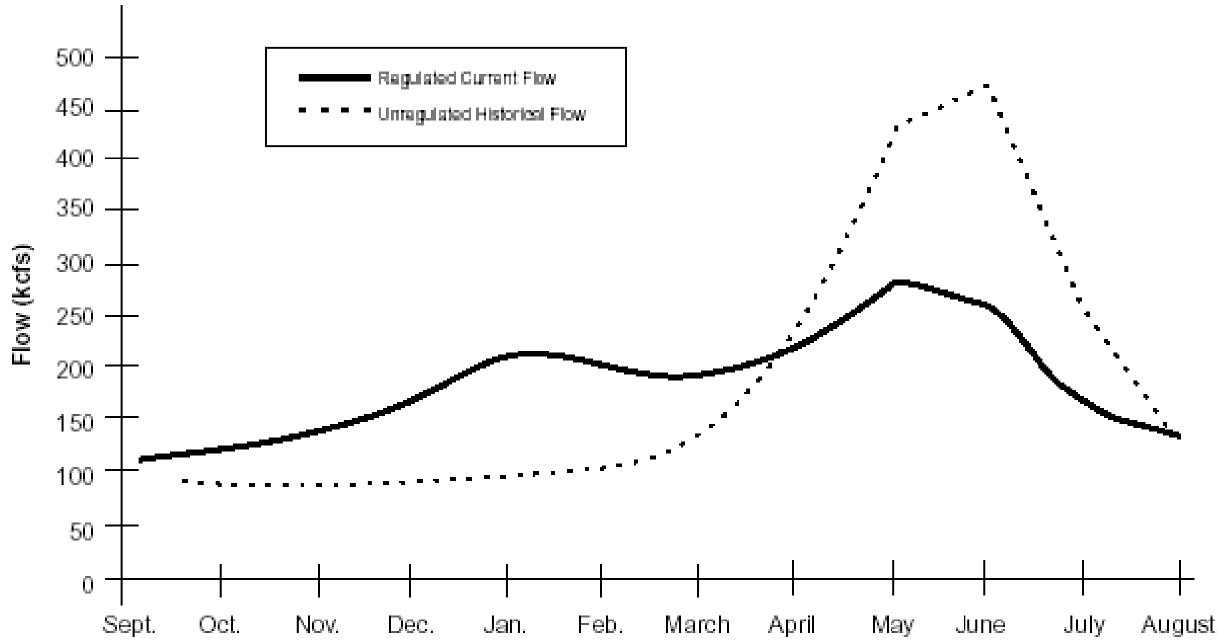


Figure 2-14. Current regulated mean monthly flow compared to historical unregulated mean monthly flow at Bonneville Dam (USACE 2001).

In addition to magnitude, the timing of maximum spring freshet flow has also changed as a result of hydropower operations and irrigation withdrawals. The mean predevelopment maximum spring freshet flow date was June 12 compared to the present mean date of May 29, an approximate 2 week shift in maximum spring freshet flow (Bottom et al. 2001, Jay and Naik 2002).

Finally, freshet styles have been affected by climate and human actions. There are three primary types of spring freshets based on the source of flow: large winter snowpack without considerable spring rain, normal winter snowpack with considerable spring rains, and large winter snowpack with considerable spring rains. The largest freshet flows on record have been associated with rain-on-snow events. Flow regulation is relatively effective in dampening freshets associated solely with snowpack; winter reservoir drawdown provides storage capacity for the steadily melting snowpack. However, heavy spring rains are more difficult to predict and flows are difficult to control because snowmelt rate is substantially higher. Although, the gradual warming of the region has made accumulation of low elevation spring snowpack less likely, decreasing the probability of spring freshets resulting from rain-on-snow events (Bottom et al. 2001, Jay and Naik 2002).

Total mainstem freshwater input at the head of the Columbia River estuary is best measured at Beaver, Oregon; flows at Beaver are the sum of flows for the interior and western Columbia River subbasins. The gauge there includes inputs from some substantial basins downstream of the Dalles (Willamette, White Salmon, Sandy, Lewis, etc.). Because dams from Bonneville upstream capture spring runoff in impoundments, flows from lower Columbia tributaries below Bonneville have become more important contributors to estuary flow during spring and winter runoff periods (Bottom et al. 2001). Average flow at Beaver is now substantially lower than pre-dam flows (Bottom et al. 2001; Figure 2-15).

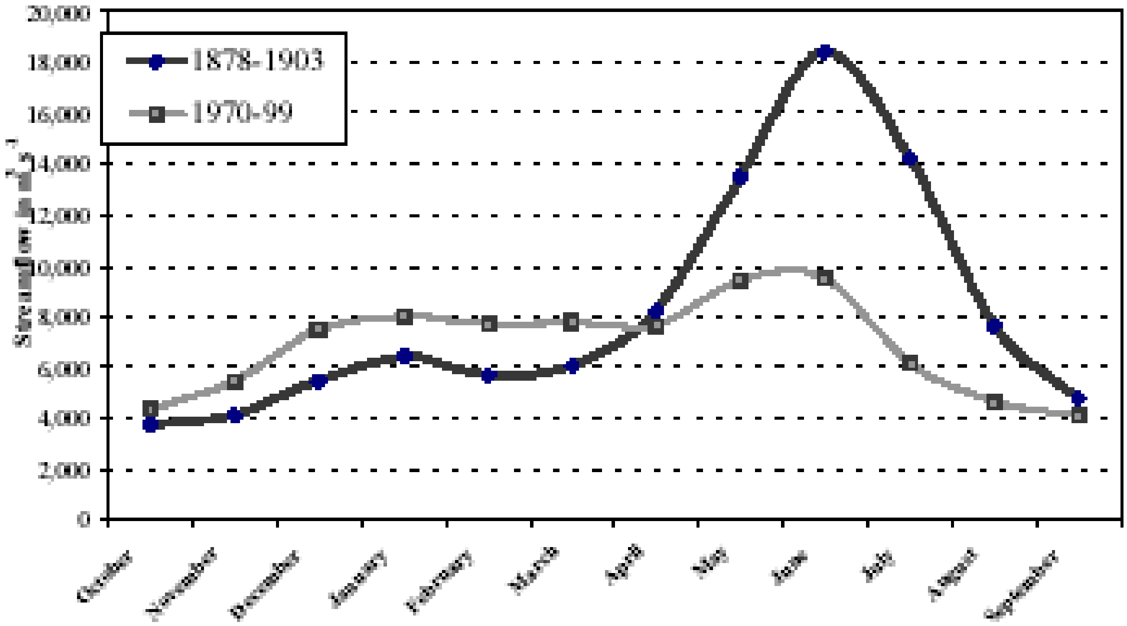


Figure 2-15. Comparison of historical (1878-1903 [data missing]) and recent (1970-1999) Columbia River annual flow cycle measured at Beaver, OR (Bottom et al. 2001).

Reduction of maximum flow levels, dredged material deposition, and diking measures have all but eliminated overbank flows in the Columbia River (Bottom et al. 2001). Overbank flows were historically a vital source of new habitats. Moreover, springtime overbank flows greatly increased habitat opportunity into areas that at other times are forested swamps or other seasonal wetlands. Historical bankfull flow level for the mainstem Columbia River below Vancouver was approximately 18,000 cubic meters per second (cms); current bankfull level is determined by the hydropower project flood level of about 24,000 cms. Historical bankfull flow levels were common prior to 1975 but are rare today; current bankfull flows have only been exceeded four times since 1948 (Figure 2-16). Further, the season when overbank flow is most likely to occur today has shifted from spring to winter, as western subbasin winter floods (not interior subbasin spring freshets) are now the major source of peak flows (Bottom et al. 2001, Jay and Naik 2002).

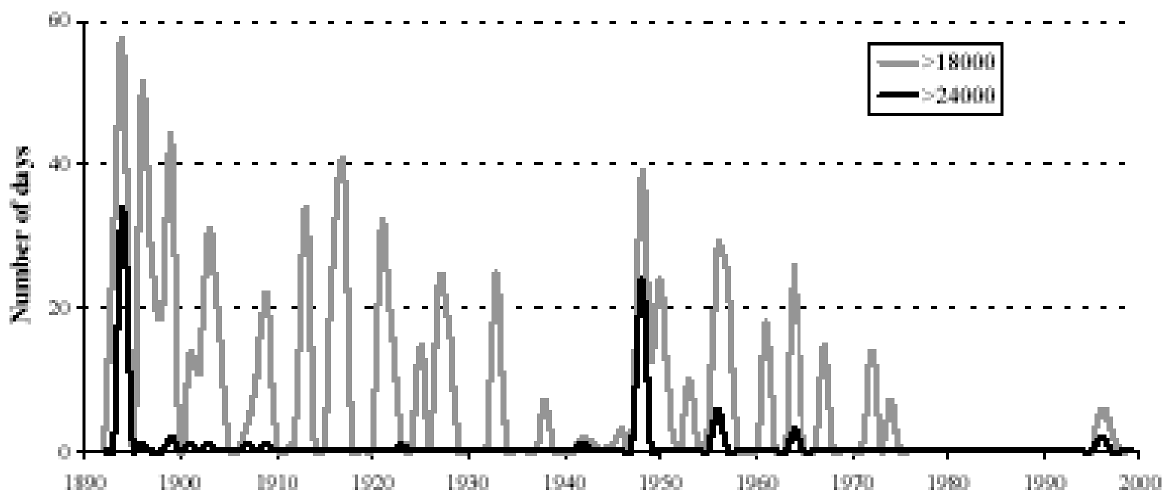


Figure 2-16. Frequency of mainstem Columbia River flow above historical bankfull (18,000 cms) and current bankfull (24,000 cms) flow levels.

2.3.1.2 Sediment Transport

Sediments in the estuary may be marine or freshwater derived; throughout the entire estuary, sediments comprise gravel (1%), sand (84%), silt (13%), and clay (2%) (Hubbell and Glenn 1973, Roy 1982 as cited in Moritz et al. 1999). Sediment transport in the lower mainstem and estuary is largely driven by the Columbia River's hydrologic cycle; most sediment transport coincides with the spring freshet, although high sediment concentrations can also be transported during infrequent winter floods (USACE 2002). Sediments are transported via sediment suspended in the water column or bed load movement. These mechanisms of sediment transport determine the sediment supply to the estuary, which determines the bathymetry of the estuary through the processes of erosion and accretion. Estuary bathymetry is one of the primary factors determing the types of habitat present in the estuary (USACE 2001; Figure 2-12). The following discussion is a brief synopsis of sediment transport mechanisms in the lower Columbia River and estuary; more detailed descriptions of sediment transport processes and estimates of lower Columbia River and estuary sediment budgets can be found in Whetten et al. (1969), Sherwood et al. (1984), Sherwood et al. (1990), Gelfenbaum et al. (1999), Moritz et al. (1999), USACE (1999), Buijsman (2000), Bottom et al. (2001), Kaminsky et al. (2002b), and USACE (2002), to name a few.

The entire Columbia River basin has two principal sediment sources: the upper watershed above the Snake River confluence that produces fine sediments from surficial deposits and the Cascades that supply coarse sediments (sand) resulting from erosion of volcanic material (Whetten et al. 1969 as cited in USACE 2002). Under average flow conditions, each sediment source was independently transported and deposited, with the upper basin sediments transported primarily as suspended sediment and the Cascade sediments transported primarily as bedload (Whetten et al. 1969 as cited in USACE 2002). Thus, sediment from either source may be present in the lower Columbia River and estuary.

Suspended sediment is supported by buoyancy and turbulence within the water column; because particles travel about the same speed as river velocity, they generally move substantial distances before depositing (USACE 2002). There are two main categories of suspended sediments: wash load and bed sediment load (USACE 2002). Wash load comprises silt and clay particles and is often generated from outside sources such as tributaries and local runoff (USACE 2002). Bed sediment load is composed of larger particles such as sand and is governed by the combination of the river's transport potential, the available particle (sand) supply, and the settling properties of the particles (USACE 2002). Sand constitutes about 95% of the total bed material found in the estuary and lower Columbia River mainstem (USACE 2002). However, sand typically constitutes less than 15% of the suspended sediment load, which is generally comprised of about 70-90% fine materials such as silt and clay (USACE 2002). The sand component of the suspended sediment may increase to over 30% when discharge exceeds 400,000 cfs (USACE 2002); however, flows of this magnitude are rare in the present era of water management.

Bed load movement describes the process of larger particles, such as sand or gravel, rolling or bouncing along the riverbed (USACE 2002). Because water velocity at the surface of the riverbed is slower than in the water column, bed load particles move slower than suspended sediments (USACE 2002). Further, bed load particles typically move intermittently and cover short distances during each movement (USACE 2002). Bed load movement typically occurs in a layer only a few sand grains thick (USACE 2002). Bed load movement shapes the riverbed into a series of sand waves; these waves continually move downstream as sand particles are eroded from the upstream face and deposited in the downstream trough (USACE 2002). Therefore, through this continually downstream movement, all the sand particles in a sand wave are eroded, transported, deposited, buried, and eventually eroded again (USACE 2002).

Currently, the most important sediment deposition conditions present in the estuary include shoaling in the navigation channel and deposition/accumulation of sand in low energy areas in the estuary and along the coast (USACE 2002). Shoaling in the navigation channel is a redistribution of bed sediments, rather than an accumulation of sediments, because it does not change the volume of bed material within a given reach (USACE 2002). Sediments generally accumulate in bays and shallow areas throughout the estuary (USACE 2002). Hubbell and Glenn (1973, as cited in USACE 2002) indicated that over 80% of the accumulated sediments was comprised of sand; although the percentage of accumulated silt increases in estuary bays relative to other shallow areas, sand was still the dominant material deposited.

Because sand sediments are vital to natural habitat formation and maintenance in the estuary, dredging and disposal of sand and gravel have been one of the major causes of estuarine habitat loss over the last century (Bottom et al. 2001); estimates of dredging volumes over time are depicted for different reaches in the lower Columbia River (Figure 2-17 and Figure 2-18). From 1958-1997, supply of sand to the estuary from upriver sources was estimated at 1.4 million cubic meters per year (Mm^3/yr; Gelfenbaum et al. 1999). Meanwhile, from 1956 to 1983, the US Army Corps of Engineers (USACE) removed an average of 0.9 Mm^3/yr from the Columbia River entrance and, from 1984 to 1998, the USACE removed an average of 2.5 Mm^3/yr (Kaminsky et al. 2000). Therefore, it is possible that much of the sand entering the estuary from upriver sources does not remain in the estuary and is disposed of in deep-water ocean sites or upland site outside the of the Columbia River littoral cell (Kaminsky et al. 2000, 2002b, Kaminsky 2002a). Further, because of flow regulation and river dredging operations, the sand removed from the lower river cannot be replenished in the absence of an unmitigated, catastrophic event, such as an extreme flood or volcanic eruption (Kaminsky 2002a). Present conditions of sand transport are one of net sand extraction from the river system, because the net supply of river sand has decreased by a factor of 3 over the historical period while the removal of sand has increased by a factor of 2.5 (Kaminsky 2002a). Future conditions of sand transport are not likely to improve in the next 20 years, based on the proposed dredging activities of the USACE; continued losses of Columbia River sand transport may exacerbate the present erosion trends in the coast and nearshore zone of the Columbia River littoral cell (Kaminsky 2002a).

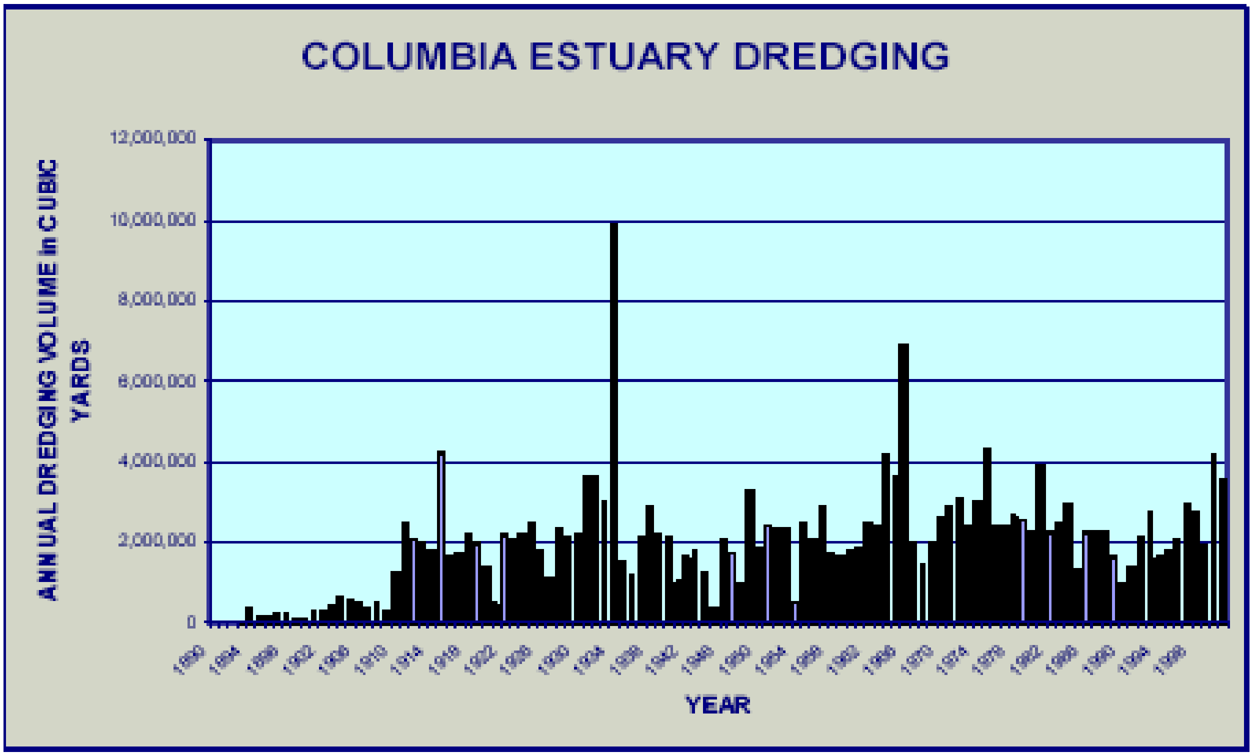


Figure 2-17. Volume of material dredged over time from the Columbia River between rm 3 and 40 (USACE 2002).

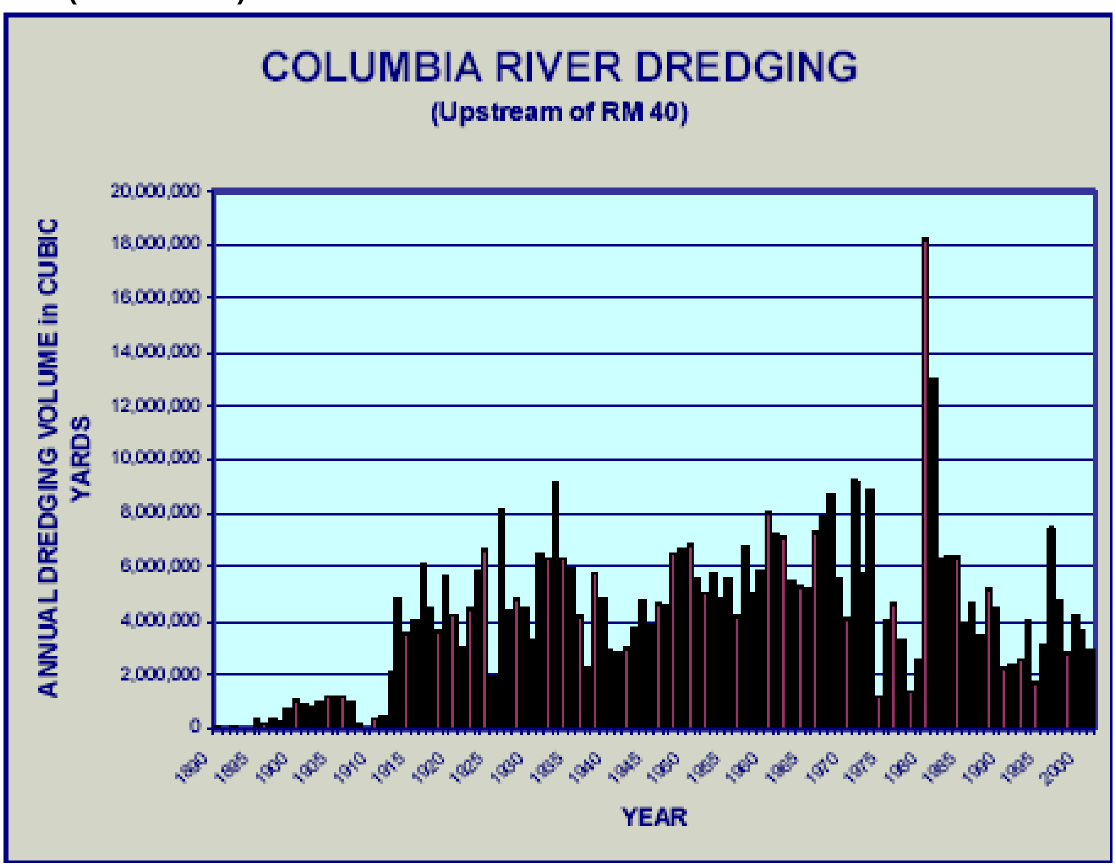


Figure 2-18. Volume of material dredged over time from the Columbia River between rm 40 and 106 (USACE 2002).

Dredging operations at the mouth of the Columbia River have become a topic of considerable debate because of the potential to affect shoreline erosion; consensus regarding the potential erosion effects of this dredging has not been reached (Kaminsky 2002a, 2002b, Kaminsky et al. 2002a, 2002b, Moritz 2002). One hypothesis is that current shoreline erosion cannot be attributed to dredging and disposal practices at the mouth of the Columbia River, as supported by records of dredging and disposal actions (Moritz 2002). For example, from 1905 to 1950, the mouth of the Columbia River navigation channel was maintained at a shallower depth than today and dredging at the mouth was sporadic (Figure 2-19; Moritz 2002). Further, all dredged sand from this time period was deposited either in high flow areas of the estuary or on the ebb-tidal shoal (Moritz 2002). Meanwhile, from 1950 to the present, about 4 million cubic yards are removed from the navigation channel at the mouth of the river; two thirds of all dredged sediment was placed within active sediment transport zones in the river mouth or in adjacent nearshore areas with a depth less than 18m (Moritz 2002). Moritz (2002) estimated that nearly 90% of all sediment dredged from the navigation channel at the mouth of the Columbia River has been deposited in a location where the sediment benefits littoral areas of the Columbia River littoral cell. In the most recent years (i.e. 1997 to present), 90% of the sand dredged from the navigation channel at the mouth of the Columbia River has been placed in two dispersive nearshore sites on the ebb-tidal shoal (Moritz 2002). To date, 80% of the dredge material deposited at these sites has been dispersed, of which less than 10% has been transported back to the navigation channel. Based on this history of dredge and disposal actions at the mouth of the Columbia River, Moritz (2002) suggested that dredging and disposal activities have helped maintain the ebb-tidal shoal and minimize shoreline erosion, rather than contribute to current erosion occurring in the Columbia River littoral cell.

The alternate hypothesis is that dredging and disposal practices at the mouth of the Columbia River have contributed to shoreline erosion within the Columbia River littoral cell (Kaminsky 2002a, 2002b, Kaminsky et al. 2002a, 2002b). Estimates of projected dredging operations indicate that about 6.7 million cubic yards of sand will be removed annually from the lower river, while the sand supply from upland sources is estimated at 1.95 million cubic yards annually, resulting in an annual net removal of about 4.75 million cubic yards of sand (Kaminsky 2002a). Sand transported via the Columbia River has previously served as a source for accreting sediments along Long Beach (Gelfenbaum et al. 1999); as the historical Columbia River sand supply decreases, the southern portion of the Long Beach peninsula is predicted to undergo net erosion (Kaminsky 2002a). Since 1997, the Southwest Washington Coastal Erosion Study's morphology beach monitoring program has documented net recession along the southern portion of the Long Beach peninsula (Kaminsky 2002a). Preliminary shoreline change modeling results indicate that current shoreline configuration is changing in response to reduced sediment supply, primarily from the ebb-tidal deltas at the mouths of the Columbia River and Grays Harbor (Kaminsky et al. 2002a, 2002b). Additionally, future shoreline position will likely be a function of sediment supply from the Columbia River, ebb-tidal deltas, and the nearshore ocean lower shoreface (Kaminsky et al. 2002b). Based on proposed future dredge operations and disposal sites, use of upland or deepwater ocean sites for dredge disposal may become more prevalent, which will contribute to the decrease in sediment supply from the Columbia River (Kaminsky et al. 2002b). Strategic utilization of dredged sand from navigation projects in the Columbia River, Willapa Bay, and Grays Harbor may be one of the only viable options for maintaining sediment budgets and natural sediment dispersal pathways to reduce erosion in the Columbia River littoral cell (Kaminsky 2002b).

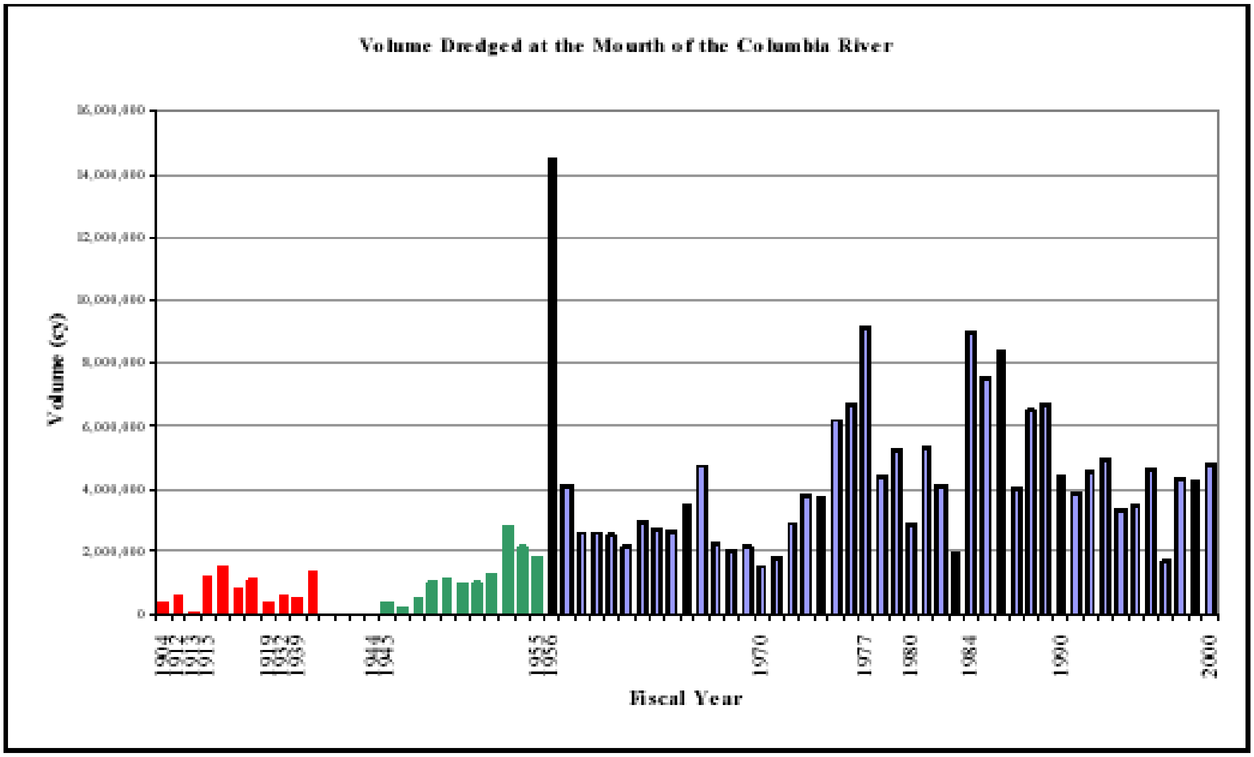


Figure 2-19. Volume of material dredged from the mouth of the Columbia River over time (USACE 2002).

The volume and type of sediment transported by the mainstem Columbia River has profound impacts on the estuary food web and species interactions within the estuary. For example, organic matter associated with the fine sediment supply maintains the majority of estuarine secondary productivity in the food web (Simenstad et al. 1990, 1995 as cited in Bottom et al. 2001). Also, turbidity (as determined by suspended sediments) affects estuary habitat formation, regulates primary production via affects on light penetration, and decreases predation on juvenile salmonids via decreased predator efficiency.

Sediment transport is non-linearly related to flow; thus, it is difficult to accurately apportion causes of sediment transport reductions into climate change, water withdrawal, or flow regulation (Jay and Naik 2002). However, the largest single factor in reduced sediment transport appears to be the reduction of spring freshet flow as a result of water regulation and irrigation withdrawal. Jay and Naik (2002) compared sediment transport data from the Columbia River at Beaver, Oregon, for the pre-1970 and post-1990 periods; they concluded that sand supply in the Columbia River remains available and has not reduced Columbia River sand transport. Findings of the USACE (1999, 2001, 2002) are consistent with this conclusion; they determined that there has been no substantial change in the river's sand supply. Further, the USACE (2002) suggested that sand supply in the Columbia River will unlikely become limiting to sediment transport because the riverbed is underlain by alluvial sand deposits that range in thickness from 100 ft. near Vancouver to 400 ft. in the estuary (Gates 1994 as cited in USACE 2002). Figure 2-20 depicts the estimated volume of sand transported in the Columbia River at Vancouver, Washington; years of high sand transport volume correspond with high flow years and recent era sand transport volumes are generally lower than historical sand transport volumes as a result of water regulation.

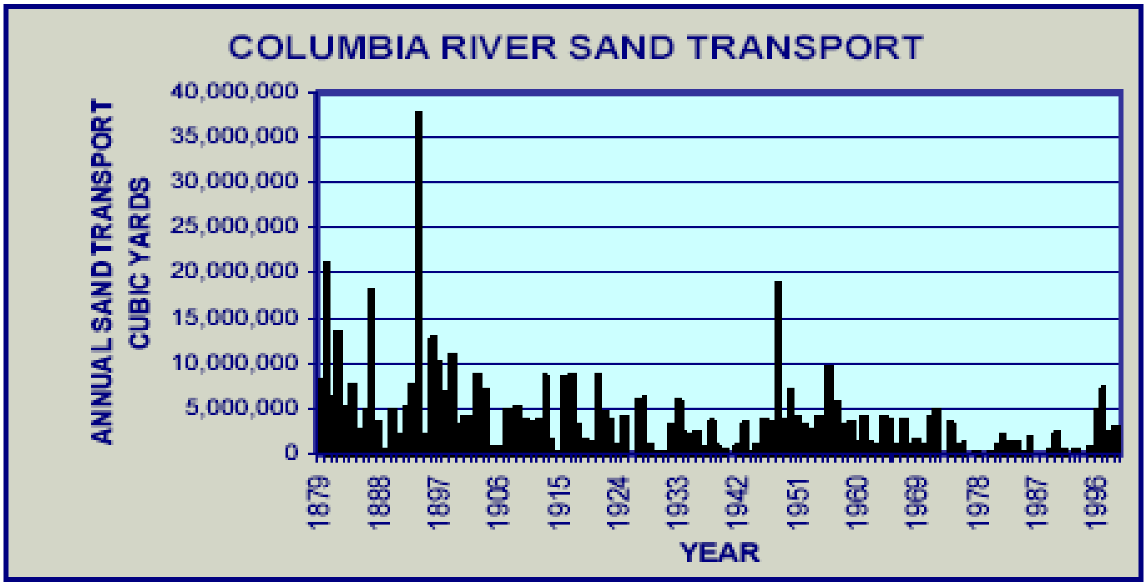


Figure 2-20. Total volume of sand transported annually in the Columbia River at Vancouver, Washington (Cited in USACE 2002; derived from Sherwood et al. 1990 and Bottom et al. 2001).

Recent analyses indicate a two-thirds reduction in sediment-transport capacity of the Columbia River relative to the pre-dam period (Sherwood et al. 1990, Gelfenbaum et al. 1999). Flow reductions affect estuary habitat formation and maintenance by reducing sediment transport (Bottom et al. 2001, USACE 2001). Moreover, the nature of sediments reaching the estuary has been altered. Research indicates that fine materials may be supply limited (which is rare in light of urban development, timber harvest, and agriculture), while sand transport is limited by discharge (Sherwood and Creager 1990). Regulated flows are usually sufficient to transport fine materials (silt, clay, fine sand), but not enough to transport sand and gravels. Thus, under regulated flow conditions, the reduction in sand transported to the estuary is disproportionately greater than reductions in flow and total sediment load (Bottom et al. 2001, Jay and Naik 2002); for example, the reduction in sand and gravel transport has been higher (>70% reduction compared to predevelopment flow) than for silt and clay transport (Bottom et al. 2001). Sand and gravel substrates are important components of preferred salmonid habitat in the estuary while organic matter associated with fine sediments is an important component of the food web.

Because of water velocity reductions, sediments and nutrients that would otherwise have been transported downstream accumulate in reservoirs (Robeck et al. 1954 and Puig et al. 1987 as cited in Weitkamp 1994). Thus, Columbia River reservoir construction has trapped much of the yearly upstream sediment load behind dams. Reservoirs also restrict bedload transport (i.e. movement of sediment along the riverbed when flow is sufficient). Historically, the amount of sediment supplied to the estuary was a function of the type of sediments available and river discharge. Changes in the sources of sedimentation and the regulation of upriver flows, coupled with entrapment of sediment behind dams, have changed sediment supply to the estuary. The idea of mainstem Columbia River reservoirs acting as sediment sinks is contrary to the findings of Whetten et al. (1969, as cited in USACE 2002); they found that sediment generally was not accumulating in mainstem reservoirs as a result of scour by high discharge.

Construction of the north and south jetties significantly increased sediment accretion in marine littoral areas near the mouth of the Columbia River and have decreased the inflow of marine sediments into the estuary. Ocean currents that formerly transported marine sediments into the estuary and Columbia River sediments along the marine littoral areas were disrupted as a result of jetty construction. Accretion, particularly in areas adjacent to the river mouth (i.e. Long Beach, Clatsop Spit), increased significantly in the late 1800s and early 1900s. Sediment accumulation rates have slowed since 1950, potentially as a result of reduced sediment supply from adjacent deltas or the Columbia River (Kaminsky et al. 1999). Because of the decreased sediment supply from the Columbia River and ebb-tidal deltas, recent modeling results indicate that the shorelines immediately north of the historical sediment source areas at the entrance to the Columbia River are susceptible to erosion in the future; accurate estimates of the Columbia River sediment supply are vital to realistic model predictions (Kaminsky et al. 2000). Conversely, Moritz (2002) suggested that the apparent widespread erosion within the Columbia River littoral cell is actually a localized re-distribution of sands resulting from the Columbia River ebb-tidal shoal that was initially pushed offshore after jetty construction and is now being forced toward an equilibrium through present day ocean currents/waves.

2.3.1.3 Salinity and Nutrients

River discharge (estuary outward flow), tidal processes (estuary inward flow), and channel depth determine the salinity gradient and the type and location of available nutrients (Figure 2-12). Columbia River flow may seasonally vary by an order of magnitude, which can significantly influence salinity intrusion and salinity stratification; salinity intrusion decreases while salinity stratification increases with higher river flows. Tides have complex effects on salinity; tide-induced turbulent vertical mixing inhibits salinity intrusion, while horizontal transport by tides is the primary salt transport mechanism during strong tides or low river discharge. The dependence of salinity intrusion on channel depth is strong; the controlling channel depth has doubled over the last 120 years. Bathymetric changes have likely caused the greatest changes in salinity intrusion and stratification, but reduced spring freshet flows have also substantially altered salinity intrusion length (Figure 2-21; Bottom et al. 2001).

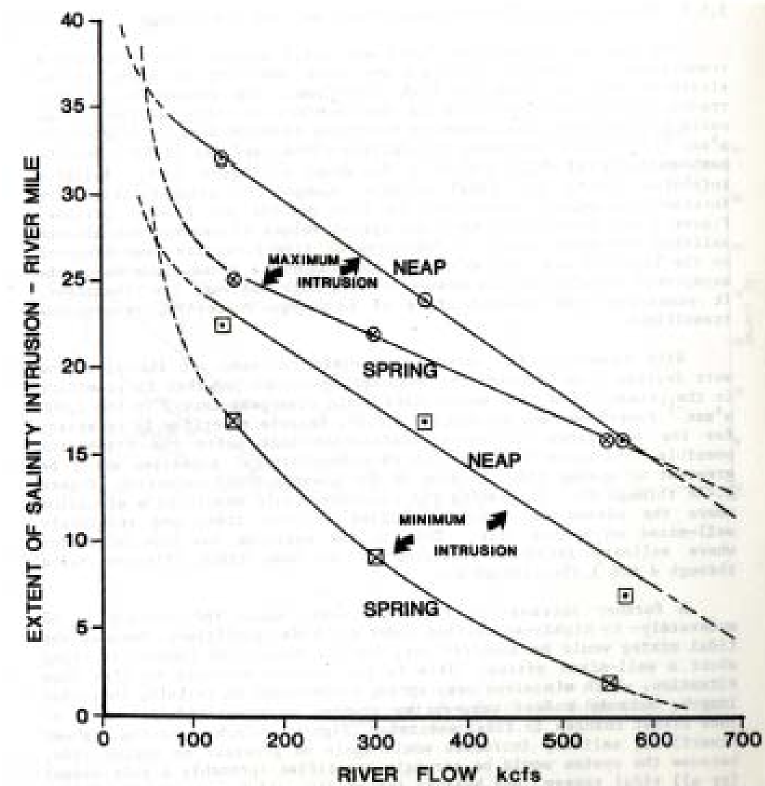


Figure 2-21. Maximum and minimum salinity intrusion distance in the Columbia River estuary, based on 1980 bathymetric conditions (Jay 1984 as cited in Bottom et al. 2001).

Operation of the Columbia River hydrosystem controls river flows and Columbia River flow affects salinity gradients. Increased river flow decreases the extent and duration of intrusion of salt water into the estuary, while decreased river flow does the opposite. Altered estuary bathymetry and flow have affected the extent and pattern of salinity intrusions into the Columbia River; stratification has increased and mixing has decreased (Sherwood et al. 1990 as cited in Williams et al. 2000).

The estuary turbidity maximum (ETM) is an area of elevated levels of suspended particulate material, in particular, river bed sediments, other particulate material, and associated bacteria. Suspension of material in the ETM is a result of turbulence caused by tidal forces pushing saline water upriver below the outflowing river water (Figure 2-22). The ETM is an critical zone of organic matter accumulation and cycling (Figure 2-23), especially in the current imported microdetritus-based food web as discussed in subsequent sections. In the Columbia River, the ETM appears to move upstream with the leading edge of the salt wedge during flood tides, then retreats with the salt wedge during ebb tides. The combination of tidal energy and river discharge determine the location, size, shape, and salinity gradients of the Columbia River ETM (Figure 2-24). As depicted in this figure, low river flow allows the ETM to migrate further upstream; this is particularly true during neap (flood) tides (Figure 2-24; Scenario 1 and 2). During high flows, river discharge maintains the ETM location closer to the river mouth (Figure 2-24; Scenario 3). The length of the ETM ranges from 0.5 to 3 miles and the location fluctuates up to 9 miles daily, based on river discharge and tide cycle. On the south bank, the ETM generally migrates between Youngs Bay and Tongue Point, while on the north bank, the ETM is usually on either side of Point Ellice (USACE 2001).

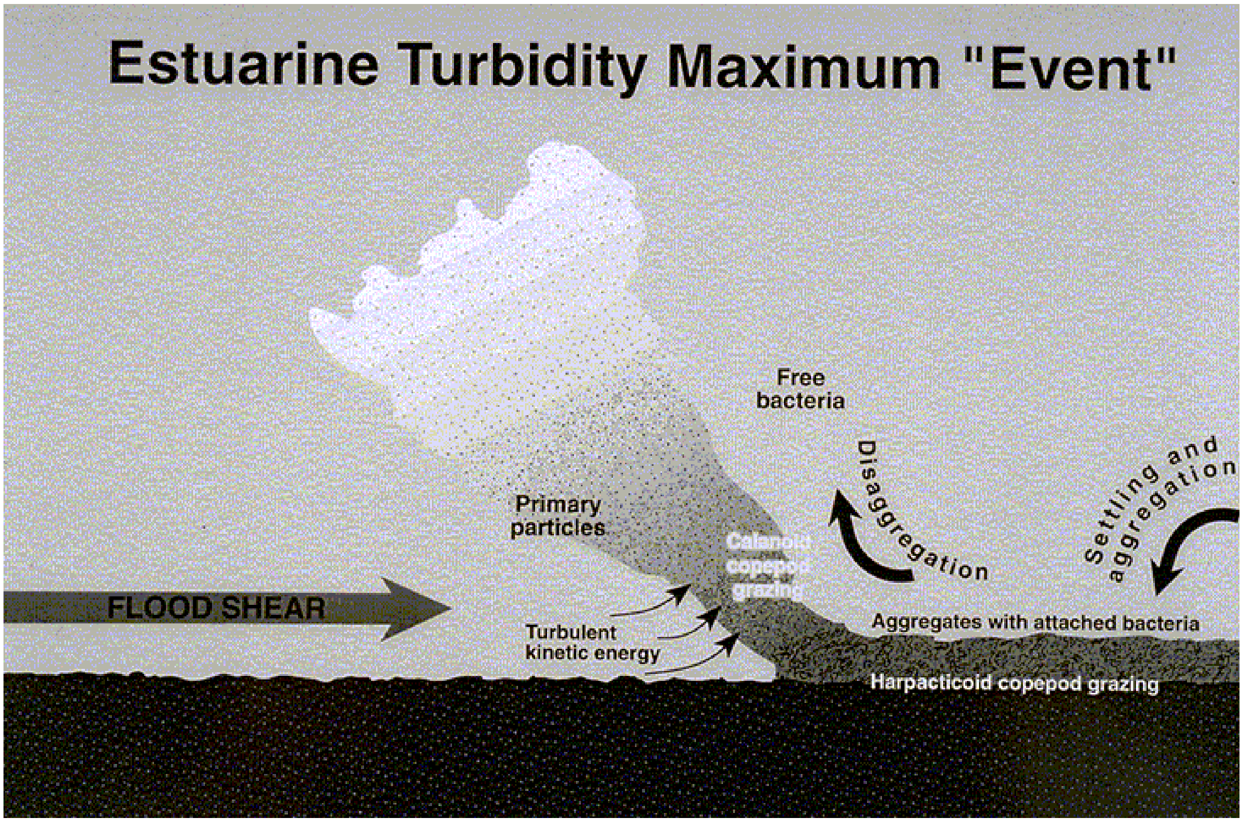


Figure 2-22. Diagram of an ETM "event" as tidal forces push salinity upriver beneath the outflowing river water (NSF 2003).

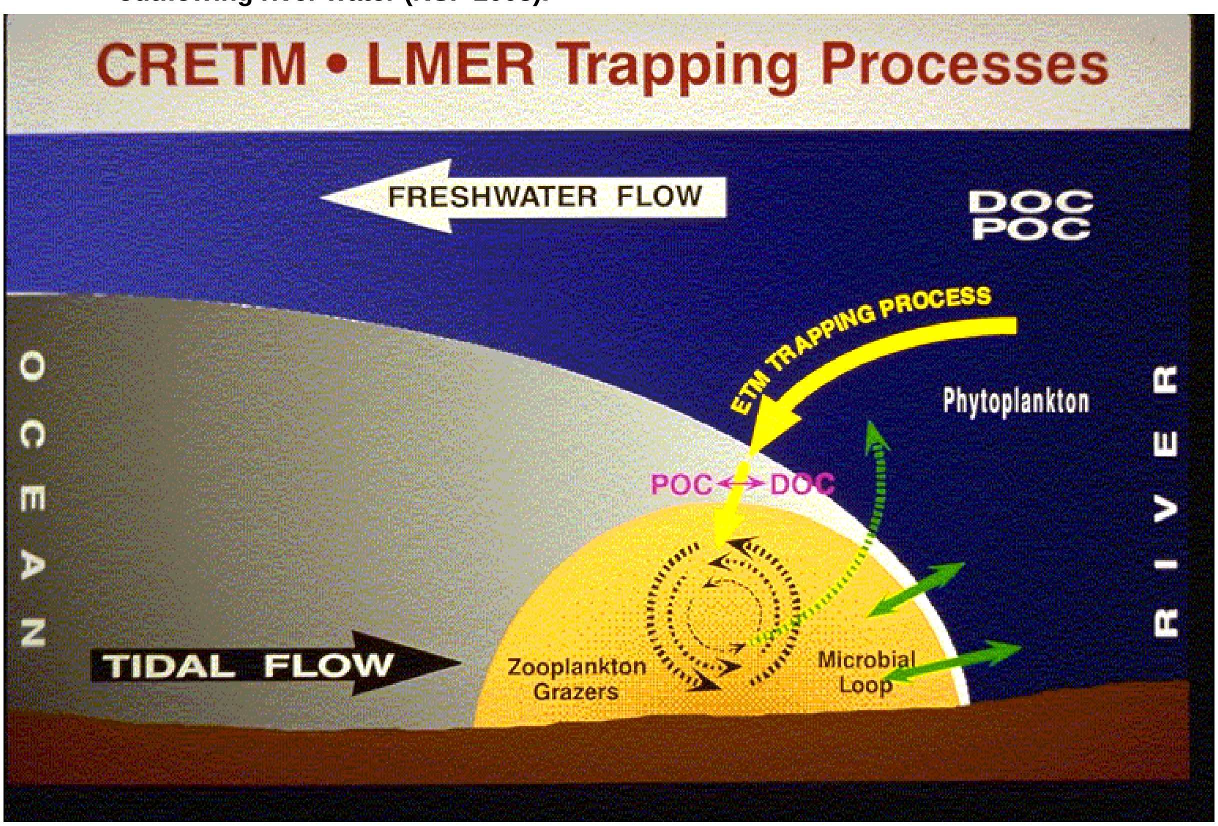
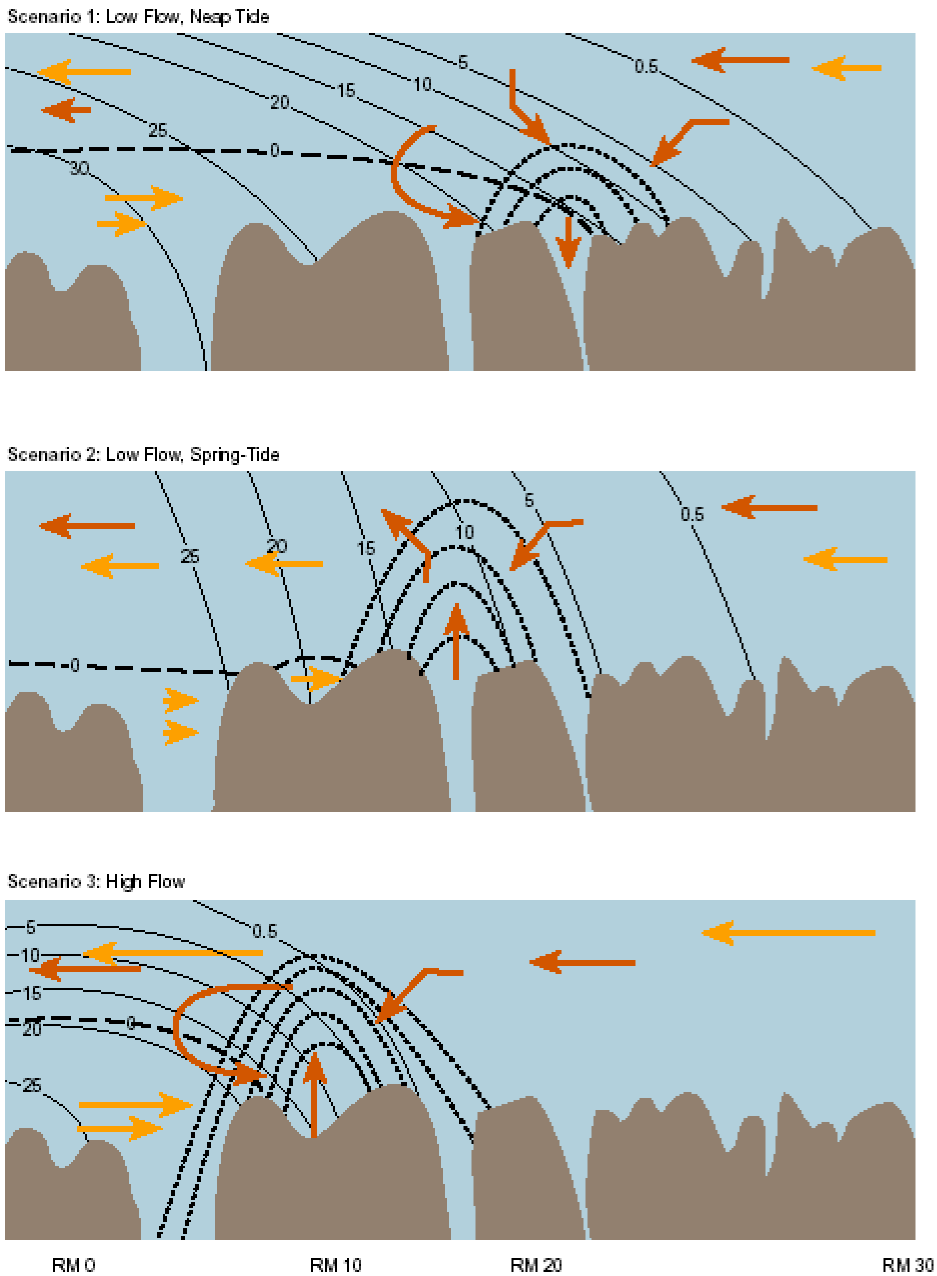


Figure 2-23. Diagram of biological activity within the ETM, illustrating the productivity of this area (NSF 2003).



Source: Simenstad, et al, 1994.

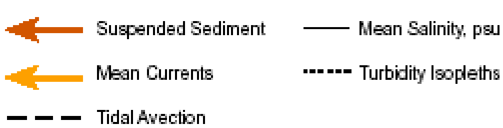


Figure 2-24. Variations in the estuary turbidity maximum (ETM) under different river flow and tide cycle conditions (USACE 2001).

Hydropower generation in the Columbia River has altered the amount and timing of water delivered to the Columbia River plume; the biological effects to juvenile salmonids of this altered flow pattern on the plume environment are largely under-studied (Bisbal and McConnaha 1998; Williams et al. 2000). Prior to hydrosystem operations, the coastal plume had well-defined seasonal directions: in winter, the plume extended toward the north while in the summer, the plume reversed and net transport was in a southwesterly direction, up to 400 km (Ebbesmeyer and Tangborn 1992 as cited in Bisbal and McConnaha 1998). Further, evidence suggests that the shift of freshet flows from the spring to the winter has altered sea surface salinities along a substantial part of the west coast of North America (Ebbesmeyer and Tangborn 1992 as cited in Williams et al. 2000). The nearshore environment, particularly that associated with the Columbia River plume, is important habitat to outmigrating juvenile salmonids (NMFS et al. 1998, Pearcy 1992 as cited in NMFS 2000c). Hydrologic conditions associated with the Columbia River plume creates a highly productive, low salinity zone compared to the surrounding ocean environment. Recent data suggests that juvenile salmonids are concentrated along this productive zone of the Columbia River plume during their early ocean existence (NMFS et al. 1998). Inter-annual variation in ocean recruitment of salmon is high and believed to be associated with annual variation in nearshore ocean physical and biological conditions (NMFS et al. 1998). Anthropogenic factors that may alter this productive plume environment, as well as management actions such as large releases of hatchery salmonids that may create competition for resources, can decrease survival during plume residence (Bisbal and McConnaha 1998).

Decreased spring flows and sediment discharges have reduced the extent, speed of movement, thickness, and turbidity of the Columbia River plume that previously extended far out and south of the river mouth during spring and summer (Ebbesmeyer and Tangborn as cited in Bisbal and McConnaha 1998; Barnes et al. 1972, Cudaback and Jay 1996, Hickey et al. 1998 as cited in NMFS 2000c). Although additional nutrients are available from upwelling during low river flows, low river discharge is unfavorable for juvenile salmonid survival because of reduced turbidity in the Columbia River plume (Pearcy 1992 as cited in NMFS 2000c). Decreased plume turbidity results in increased foraging efficiency of birds and fish predators, increased residence time of fish in the estuary and nearshore ocean environment where predation is high, decreased incidence of fronts with concentrated food resources for juvenile salmonids, and reduced overall total secondary productivity based on upwelled and fluvial nutrients (Pearcy 1992 as cited in NMFS 2000c). Further, decreased estuarine turbidity has allowed for increased predation on juvenile salmonids throughout the estuary (Junge and Oakley 1966, Bottom and Jones 1990 as cited in Nez Perce et al. 1995).

2.3.2 Habitat Change

Historically, environmental conditions in the Columbia River mainstem and estuary were controlled by ocean processes, Columbia River Basin landscape conditions, and riverine processes, which were influenced by climate and a host of natural processes and disturbance. The historical mainstem and estuary conditions were highly variable and the magnitude of environmental changes suggest major shifts in estuarine and riverine habitat conditions. Alterations to ocean and riverine processes have changed the amount and types of habitat in the lower Columbia River mainstem and estuary.

2.3.2.1 Climate

Variations in Columbia River discharge as a result of climate effects occur in time scales from years to centuries (Chatters and Hoover 1986, 1992 as cited in Bottom et al. 2001);

research on climate cycles as they affect ocean productivity and salmonid survival has focused on the Pacific Decadal Oscillation (PDO; typically 40-50 year cycle) and the El Niño-Southern Oscillation (ENSO; typically 3-7 year cycle; Mantua et al. 1997 as cited in Bottom et al. 2001). The Columbia Basin's climate response to these cycles is governed by the basin's latitudinal position; climate in the region displays a strong response to both the PDO and ENSO cycles (Mantua et al. 1997 as cited in Bottom et al. 2001, Mote et al. 2003). Warm phases of ENSO (i.e. El Niño) and PDO cycles correspond with winter and spring weather that is warmer and drier than average; cool phases of ENSO (i.e. La Niña) and PDO typify cooler, wetter weather (Mote et al. 2003). Climate effects of short-term El Niño cycles are strengthened during warm phases of the PDO, while La Niña effects are intensified during a cold PDO phase (Gershunov et al. 1999, Mote et al. 2003). Strong El Niño winters result in Columbia and Willamette River flows that are 91 and 92% of the long-term annual average, respectively; conversely, strong La Niña winters result in Columbia and Willamette River flows that are 110 and 111% of the long-term annual average, respectively (Bottom et al. 2001). When the ENSO and PDO cycle phases are out of sync (i.e. cool ENSO/warm PDO or warm ENSO/cool PDO), streamflow tends to be near the long-term average (Mote et al. 2003).

In addition to the substantial direct affects climate has on river flow, climate indirect effects on other factors are often striking as a result of the relationship between river flow and other factors. For example, sediment discharge increases more than linearly with flow; thus, as climate affects flow, the effects on sediment discharge are amplified (Bottom et al. 2001). Another possible magnification of climate effects is the organic matter supplied during high river discharge; the extent to which this organic matter supports estuarine secondary production depends largely on whether the material is trapped in circulation processes associated with the estuary turbidity maximum (Bottom et al. 2001). Despite our ability to measure changes in climate, Bottom et al. (2001) discussed the difficulty in separating climate versus anthropogenic effects on river discharge and sediment/nutrient discharge.

Current climate projections predict gradual warming of the region, potentially with higher precipitation, particularly in winter (Hamlet and Lettenmaier 1999, Mote et al. 2003); the predicted precipitation changes are well within the 20th century annual variability range. Mote et al. (2003) indicated that, of the predicted precipitation and temperature changes, temperature changes are likely more important because they shift river flow from summer to winter. The Columbia River, being a large, snowmelt-dominated watershed (Neal 1972), is not expected to be susceptible to increased risk of spring flooding, rather, will be strongly influenced by changes in low flow because of limited reservoir storage and anthropogenic demands on water (Callahan et al. 1999 and Miles et al. 2000 as cited in Mote et al. 2003). The predicted future climate conditions will possibly reduce the likelihood of spring freshets caused by heavy spring rain on late snowpack because warmer temperatures will not allow the accumulation of snow late into the spring. This freshet style (rain on snow) has historically produced the most substantial increases in river discharge (Bottom et al. 2001). A potential consequence of this climate change is heightened conflicts over water supply during the critical spring season as a result of increased water demand and decreased natural flows (Hamlet and Lettenmaier 1999, Mote et al. 2003).

Climate has substantial effects on nearshore and ocean productivity; variability in productivity as a result of climate has important implications for many focal species in the subbasin, particularly those that make extensive use of the lower estuary, nearshore ocean, or open ocean environments. For example, the timing of spring upwelling and spring phytoplankton blooms are largely determined by the character of upwelling winds (i.e. variable winds produce more upwelling) and the circulation and stratification of the upper ocean, which is significantly influenced by winter climate (Logerwell et al. 2003 as cited in Mote et al. 2003). Additionally, the PDO cycle has profound effects on ocean nutrient levels. The warm PDO phase results in warmer and more thermally stratified coastal waters off Washington and Oregon, causing poor nutrient conditions; the opposite is true for the cold PDO phase (Mote et al. 2003). It has been suggested that potential oceanic warming that results from the predicted climate change may push the range of some salmon north out of the Pacific Ocean entirely (Welch et al. 1998 as cited in Mote et al. 2003), however, the notion that ocean thermal limits alone determine salmon distribution is likely too simplistic (Walker et al. 2000 as cited in Mote et al. 2003). Thus, minor oceanic warming may not lead to drastic changes in salmonid distribution unless accompanied by substantial changes in the oceanic food web, such as prey distribution (Mote et al. 2003).

2.3.2.2 River Flow

The Columbia River has the largest annual flow of any river on the Pacific coast of North America. Historically, unregulated flows at the mouth ranged from 79,000 cfs to over 1 million cfs, with average flows about 273,000 cfs (Marriot et al. 2002). Currently, discharge at the mouth of the river ranges from 100,000 to 500,000 cfs, with an average of about 260,000 cfs (Marriot et al. 2002). Highest flows are experienced during and just after winter storms, generally from December through March. Flows and sediment load have been altered by construction of 31 irrigation and hydropower dams in the basin since 1890. Prior to human influence, the Columbia River estuary had extensive sand beds and variable river flows. However, the construction of upriver hydroelectric dams has dramatically changed the nature of the estuary, as these dams have translated into different flow rates and sediment discharges (Figure 2-13, Figure 2-14, Figure 2-15, and Figure 2-16). Moreover, channel deepening, use of jetties and dredging to stabilize channels, development of perennial wetland areas, and isolation of remaining wetlands from the mainstem river have altered the physical character of the Columbia River estuary and these changes have affected the biological systems that the estuary supports. Introduction of non-native species and degradation of water quality have also impacted the estuarine biota. All of these influences interact in complex ways. The quantitative estimates of habitat loss, however, do not reflect the qualitative losses that have also occurred, and which may have important effects on the salmon rearing capacity of the estuary.

Because of changes to flow and sediment transport and the various habitat alterations that have occurred in the estuary, the availability of shallow (10cm-2m depth), low velocity (<30 cm/s) habitats appears to decrease at a steeper rate with increasing flow compared to historical conditions (see also the physical process description of sediment transport in section 2.3.1.2). Further, the resilience of the estuary to increasing water depth with increasing flow appears to have decreased, likely as a result of disconnectedness with the historical floodplain. These conditions have decreased the shallow water refugia for juvenile salmonids and likely contribute to decreased survival during high flow conditions (NMFS 2000c).

2.3.2.3 Water Temperature

Many factors can cause high stream temperatures, but they are generally related to land-use practices rather than point source discharges. For example, some actions that result in high stream temperatures are the removal of riparian vegetation that directly shade streams, excessive water withdrawals for irrigation or other purposes, and warm irrigation return flows. Loss of wetlands and increases in groundwater withdrawals have decreased stream base flows, which contribute to increases in temperature. Other land uses that create shallower streams can also cause temperature increases. These land uses have occurred in some combination throughout the lower mainstem and estuary; however, the degree of water temperature increase within the lower mainstem and estuary as a result of these land uses is not completely understood. Water temperature alterations affect salmonid metabolism, growth rate, disease resistance, and the timing of adult migrations, fry emergence, and smoltification (NMFS 2000c).

2.3.2.4 Channel Confinement

The most significant habitat changes from historical to current conditions have been the loss of tidal marsh and tidal swamp habitat that are critical to juvenile salmonids, particularly small or ocean-type salmonids (Thomas 1983, USACE 2001, Johnson et al. 2003b). Thomas (1983) noted that diking has caused more of the estuary habitat changes documented in the historical/current habitat comparison than any other factor, anthropogenic or natural. This conclusion is consisted with the findings of Kukulka and Jay (2003) who indicated that dike removal alone would restore considerable amounts of shallow water estuary habitats. Further, diking entirely removes habitat from the estuarine system, while other anthropogenic factors change estuary habitats from one type to another (Thomas 1983). The degree to which estuary habitat types have been effected by diking is directly proportional to elevation; thus, the highest elevation habitat type (i.e. tidal swamp) has been impacted by diking the most (Thomas 1983).

Historically, floodwaters of the Columbia River inundated the margins and floodplains along the estuary, allowing juvenile salmon access to a wide expanse of low-velocity marshland and tidal channel habitats (Bottom et al. 2001). Flooding occurred frequently and was important to habitat diversity and complexity. Historical flooding also allowed more flow to off channel habitats (i.e. side channels and bays) and deposited more large woody debris into the ecosystem. Historically, seasonal flooding increased the potential for salmonid feeding and resting areas in the estuary during the spring/summer freshet season by creating significant tidal marsh vegetation and wetland areas throughout the floodplain (Bottom et al. 2001). In general, the river banks were gently sloping, with riparian and wetland vegetation at the higher elevations of the river floodplain becoming salmonid habitat during flooding river flows or flood tides. It is estimated that the historical estuary had 75 percent more tidal swamps than the current estuary because tidal and flood waters could reach floodplain areas that are now diked or otherwise disconnected from the main channel (USACE 2001, Johnson et al. 2003b).

Mainstem habitat in the Columbia and Willamette Rivers have for the most part been reduced to a single channel where floodplains have been reduced in size, off-channel habitat has been lost or disconnected from the main channel, and the amount of large woody debris has been reduced (NMFS 2000c). Most of the remaining mainstem habitats are affected by flow fluctuations associated with reservoir management (NMFS 2000c). Dikes prevent over-bank flow and affect the connectivity of the river and floodplain (Tetra Tech 1996); thus, the diked floodplain is higher than the historical floodplain and inundation of floodplain habitats only occurs during times of extremely high river discharge (Kukulka and Jay 2003). There is a critical level (i.e. the elevation of the diked floodplain) where water level must reach before substantial floodplain habitat are inundated; this threshold level varies between reaches (Kukulka and Jay 2003). Above this critical water level, large amounts of shallow water floodplain habitats become available with small increases in water level up to an optimum threshold (Kukulka and Jay 2003). With continued floodplain inundation above this threshold, availability of shallow water habitats decrease (Kukulka and Jay 2003), presumably because the shallow water habitats initially created at the critical water level no longer satisfy the depth criteria of shallow water habitat (0.1 to 2.0 m in this case). Under a modern bathymetry and flow regime scenario, the critical river discharge level in which significant shallow water habitats become available through floodplain inundation is relatively high and the frequency of occurrence of this river discharge is rare; thus, floodplain inudation is uncommon and availability of shallow water habitats is limited (Kukulka and Jay 2003). As is the case in the estuary (Bottom et al. 2001), loss of these vital mainstem floodplain habitats has likely reduced the productive capacity of the lower Columbia River for juvenile salmonids (particularly fry and subyearling smolts).

2.3.2.5 Channel Modifications

Development of a shipping channel has greatly affected the morphology of the estuary. The extensive use of jetties and diking to maintain the shipping channel has impacted natural flow patterns and large volumes of sediments are dredged annually. Dredged materials are disposed of in-water (in the ocean or in the flow adjacent to the shipping channel), along shorelines, or on upland sites. Annual maintenance dredging since 1976 has averaged 3.5 million cubic yards per year in the estuary. By concentrating flow in one deeper main channel, the development of the navigation channel has reduced flow to side channels and peripheral bays. Saltwater intrusion patterns have been reduced, and habitat types have been altered. Disposal of dredge materials has created barren land or islands that have indirectly increased avian predation on salmonids.

2.3.2.6 Contaminants

Industrial and urban development and agricultural practices in the lower Columbia River has resulted in pollutants accumulating in lower mainstem and estuary habitats, but the extent of detrimental effects of contaminants on juvenile salmonids is not clear. In general, contaminants affect survival by increasing stress, predisposing fish to disease, and interrupting physiological processes. Tributary water quality problems contribute to poor water quality where sediment and contaminants from the tributaries settle in mainstem and estuary habitats (NMFS 2000c). Further, the dampening of peak and sustained flood flows by hydrosystem operations has increased the accretion of sediments facilitating the accumulation of pollutants from the entire Columbia River basin in estuarine sediments (Sherwood et al. 1990 as cited in Nez Perce et al. 1996). Less water volume translates to less dilution and higher concentrations of pollutants; any stresses imposed on juvenile fish will be exacerbated by the presence of contaminants (Nez Perce et al. 1995).

The most recent data regarding contaminant effects on juvenile salmonids have been generated through assessment work for the USACE proposed channel deepening project (USFWS 1999a, NWFSC 2001, USACE 2001, NMFS 2002, USFWS 2002). Recent sampling of hatchery and wild juvenile salmon near Sand Island at the mouth of the Columbia River indicated the presence of contaminants in the food chain of juvenile salmonids (NMFS 2002). Elevated concentrations of DDT and PCBs were detected in both whole body and stomach content samples (NMFS 2002). The whole body concentrations of DDT and PCBs were among the highest concentrations measured at estuarine sites in Washington and Oregon; the whole body DDT levels were greater than and the whole body PCB levels were similar to concentrations detected in juvenile chinook salmon in the Duwamish estuary, which is a heavily contaminated industrial estuary near Seattle (NMFS 2002). Further, the presence of elevated concentrations of DDT and PCBs in stomach content samples is clear evidence that exposure to these contaminants is occurring in the estuary (NMFS 2002).

Studies of sub-lethal exposure of juvenile salmon to contaminants in urban estuaries suggest that these contaminants could affect the survival, growth, and fitness of salmon (Casillas et al. 1996). A series of experiments with natural and laboratory exposure of fall chinook salmon to hydrocarbon pollutants in Puget Sound estuaries demonstrated impaired growth, reduced immune defenses, and increased susceptibility to disease (Stein et al. 1995, Arkoosh et al. 1998a, Arkoosh et al. 1998b, Stehr et al. 2000). Water quality issues could reduce productivity for species that make extensive use of estuarine habitats for rearing, such as subyearling chinook, and chum salmon.

In the case of bald eagles, concentrations of PCBs, pesticides, and dioxins were found in bald eagle eggs collected along the lower Columbia River at concentrations associated with reduced breeding success based on eagles studied elsewhere (Anthony et al. 1993, USFWS 1999b). Reproductive problems for lower Columbia River bald eagles include eggshell thinning and a low number of young produced per occupied nest, which is considered a result of embryo dessication and mortality caused by bioaccumulative organochlorine contaminants such as DDE and PCBs (Anthony et al. 1993, USFWS 1999b). Eggshell thinning, generally attributed to DDE (a DDT derivative), was prevalent in eagle eggs and shell fragments collected along the river (Anthony et al. 1993, USFWS 1999b). Anthony et al. (1993) reported a significant relationship between eggshell thickness and breeding success among lower Columbia breeding pairs, but follow up studies in 1994 and 1995 did not show a significant relationship (USFWS 1999b). The latter studies also showed that the contaminants DDE and total PCBs declined in eggs sampled in 1994 and 1995 compared to eggs sample 10 years earlier. Even though egg concentrations have declined, values still exceeded no-effect levels estimated for the species. Recent increases in productivity and breeding success have been observed in lower Columbia River bald eagles and is likely a result of recruitment of eagles from outside regions and possibly improving contaminant conditions (USFWS 1999b, Isaacs and Anthony 2003). However, lower Columbia River eagles nesting below rm 60 continue to experience poor reproduction compared to bald eagles nesting elsewhere in Oregon and Washington. Productivity is lowest for bald eagles nesting between rm 13 and 31 (USFWS 1999b, Isaacs and Anthony 2003).

Osprey eggs collected in 1997 and 1998 along the lower 410 km of the Columbia River exhibited the highest DDE values reported for osprey in North America during the late 1980s and 1990s; additionally, DDE concentrations in eggs collected along the Columbia River were twice the concentration of eggs collected along the Willamette River in 1993 (Henny et al. 2003a, 2003b). Osprey eggshell thickness followed the classic semi-logarithmic response to DDE, as eggshell thickness decreased with increasing DDE concentration. Reproductive success was higher for nests that contained eggs with DDE concentration below 4,200 μg/kg; at this concentration, DDE results in 15% eggshell thinning (Wiemeyer et al. 1988 as cited in Henny et al. 2003a). Additionally, Henny et al. (2003a) noted that DDE concentrations in largescale suckers (a primary food item of osprey) in the Columbia River was double the levels detected in the Willamette River. Despite contaminant levels in osprey known to cause eggshell thinning, the lower Columbia River osprey population was increasing (Henny et al 2003a), but not as fast as the population nesting along the Willamette River (Henny et al. 2003b). The other contaminants found in osprey eggs (e.g., PCBs, dioxins, furans, mercury and other organochlorine pesticides), except for one egg with a high total dioxin-like activity calculated from PCBs and dioxins, appeared to be below any known effect levels for ospreys. During 1997 and 1998, osprey productivity was estimated at 1.64 young/active nest, which is higher than the generally recognized 0.80 young/active nest needed to maintain a stable population (Henny et al. 2003a).

Contaminant concentrations above available reference levels have been observed in river otter tissue samples; however, detrimental physiological effects have not been clearly established. For example, concentrations of organochlorines (i.e. PCBs, pesticides, dioxins, and furans) were higher in lower Columbia River otter samples compared to reference sites outside the lower Columbia River basin (Tetra Tech 1996). In general, observed contaminant concentrations in river otters increased with age; also, for age 2+ river otters, tissue contaminant levels decreased from rm 119.5 (near Vancouver/Portland) to rm 11.0 (Tetra Tech 1996). A number of physiological concerns were documented in river otters compared to otters from the reference sites: abnormal liver function, lower baculum weight and length, and lower mean testes weight (Tetra Tech 1996). However, when compared to previous tissue contaminant concentration data (Henny et al. 1981 as cited in Tetra Tech 1996), contaminant levels in river otter tissue in the 1990s indicate a major decline in PCB concentrations (Tetra Tech 1996). Further, data suggests that certain physiological problems may be temporary because organs of older males did not show significant size differences compared to reference animals (Tetra Tech 1996).

In the lower 150 miles of the mainstem Columbia River, the states of Oregon and Washington have found the following contaminants above guidance levels for fish tissue and sediment: organochlorines (including DDT, DDD, DDE, PCB, aldrin, dieldrin, trichlorobenzene, pyrene, and PAHs), and toxic metals (including mercury, cyanide, arsenic, chromium, iron, nickel, silver, zinc, cadmium, and copper; Tetra Tech 1993 as cited in Nez Perce et al. 1995). The U.S. Environmental Protection Agency has identified numerous water quality concerns for the Columbia River mainstem, including temperature, PCBs, dioxins, furans, pesticides, metals, and bacteria in the Columbia River estuary and temperature, PCBs, dioxins, furans, pesticides, metals, bacteria, dissolved oxygen, and total suspended solids in the Columbia River mainstem below Bonneville Dam (Nez Perce et al. 1995). However, two of the more widely known contaminants, DDT and PCBs, were much more prevalent in the lower Columbia River in the 1960s and early 1970s than they are today; their concentrations have continued to decline since 1972, when the use of DDT was banned (USACE 2001).

Data collected in the early to mid 1990s suggest that contaminant concentrations in water, sediment, or biota result in localized impairment throughout the lower Columbia River (i.e. Bonneville Dam to the mouth; Tetra Tech 1996). Metals concentration exceedance of sediment reference levels indicate possible localized effects to benthic organisms; further, some organic compounds (i.e. PCBs, DDT and derivatives, dioxins, and furans) detected in sediment and fish tissue are high enough to biomagnify through the food chain and cause adverse effects to piscivorous organisms (Tetra Tech 1996). In general, contaminant concentrations are higher in resident benthic-dwelling fish (such as largescale sucker) compared to migratory salmonids; thus, potential adverse physiological effects to biota, biomagnification to upper trophic level organisms, and human health risks associated with fish consumption are higher in benthic fish than salmonids (Tetra Tech 1996, USFWS 2003).

For years, the unmitigated flow of deicing agents from the Portland International Airport (PDX) directly into Columbia Slough has been a concern. Although PDX uses deicing agents in limited quantities, untreated flow of deicing agents can cause significant water quality problems. Deicing agents (typically a glycol mixture) are highly biodegradable and exert substantial biological oxygen demand when released to surface water. Biological oxygen demand decreases the dissolved oxygen level in the receiving surface water; decreased dissolved oxygen can stress organisms, making them less competitive and decreasing survival through a host of confounding factors. In 2003, PDX activated a glycol recovery system; the system combines underground monitoring, metering, storage, and aeration, as well as treatment by the City of Portland’s wastewater treatment plant. The glycol recovery system is intended to decrease glycol discharge levels to comply with the Oregon Department of Environmental Quality's total maximum daily load requirements for the Columbia Slough.

2.3.2.7 Restoration

Habitat actions proposed in the NMFS Biological Opinion on the Operation of the Federal Columbia River Power System (BiOp; NMFS 2000c) are intended to accelerate efforts to improve survival in priority areas while laying the foundation for long-term habitat strategies. The overarching objectives of the habitat strategy are: protect existing high quality habitat, restore degraded habitats and connect them to functioning habitats, and prevent further degradation of habitat and water quality. Specifically, Reasonable and Prudent Alternative (RPA) Actions 158 through 163 of the BiOp detail specific actions related to estuarine habitat while RPA Actions 156 and 157 address habitat issues within the lower mainstem (NMFS 2000c). An "Action Plan" has recently been published that outlines a plan for implementing the above RPA actions related to estuary and mainstem habitat restoration, as well as RPA actions that address planning, modeling, and research, monitoring, and evaluation needs described in the BiOp (BPA and USACE 2003).

Restoration of tidal swamp and marsh habitat in the estuary and tidal freshwater portion of the lower Columbia River has been identified as an important component of current and future salmon restoration efforts. RPA Action 160 in the BiOp called for an estuary restoration program with the goal of protecting and enhancing 10,000 acres of tidal wetlands and other key habitats over 10 years, beginning in 2001, with the intention of rebuilding productivity for ESA-listed salmon population in the lower 46 miles of the Columbia River. There is considerable uncertainty whether the 10,000 acres is the precise amount needed to produce desired increases in salmonid productivity or if the 10-year schedule is an appropriate time scale for recovery efforts. NMFS (2000c) identified the importance of continued monitoring and evaluation of the estuary restoration program and the 10,000-acre goal to ensure that habitats being restored are important for salmon survival and recovery. NMFS (2000c) also suggested examples of acceptable habitat improvement efforts, including but not limited to: acquiring diked lands, breaching levees, improving plant communities, reestablishing flow patterns, or enhancing connections between lakes, sloughs, side channels, and the main channel.

Dike removal could provide a sizable increase in shallow water habitat, even without restoration of historical flow regimes (Kukulka and Jay 2003). Dike removal alone provided more of an increase in shallow water habitat than flow restoration without dike removal. Restoration of natural flows increases the duration of shallow water habitat inundation in high-flow years, but individually does not restore the large size of the area historically inundated.

Management actions that seek to alter anthropogenic factors and restore natural habitat-forming processes need to be evaluated based on their impact on biological diversity and not simply on production of juvenile salmonids (Bisbal and McConnaha 1998). For example, changes in hydrosystem water management should attempt to provide benefits for the full range of salmonid life history patterns and not just the current majority. Restoration efforts need to move from the practice of management for average biological conditions to management for the full spectrum of possible biological variation (Williams et al. 1996 as cited in Bisbal and McConnaha 1998).

2.3.3 Historical vs. Current Habitat Condition

Current ecological conditions in the Columbia River estuary reflect years of anthropogenic impacts that have altered natural ecosystem inputs and processes and affected habitat conditions for all species that utilize the estuary. The extent of change of estuary habitat is highly dependent on location in the estuary and the type of habitat.

Significant effort has focused on quantifying the loss or change of habitats within the estuary and lower Columbia mainstem over time. The two methods employed to quantify habitat change include bathymetry and satellite imagery. Although there is some difficulty in comparing results of the two different methods, the underlying conclusion from both methods is that estuary and mainstem habitats have changed significantly as a result of human influence. Bathymetry is a low resolution method that provides coarse delineations of habitat types; further, bathymetry provides a means to segregate aquatic habitat based on depth criteria. Satellite imagery provides a high resolution habitat mapping method that principally uses vegetative communities to describe habitat types. Because of the use of vegetation, satellite imagery is generally not capable of distinguishing different types of aquatic habitats. Different satellite imagery technology are available that provide different levels of resolution; two of these technologies are compared in Garono et al. (2003b).

Using bathymetric survey maps of the U.S. Coast Survey (now U.S. Geodetic Survey), five major types of estuary (i.e. rm 0-46.5) habitat were defined by the Columbia River Estuary Data Development Program (Thomas 1983) according to elevation and the dominant vegetation: tidal swamps, tidal marshes, shallows/flats, medium depth water, and deep water. Change in acreage from 1870 to 1983 was estimated (Table 2-5). Additionally, Thomas (1983) investigated five categories of non-estuarine habitat (i.e. developed floodplain, natural and filled uplands, non-tidal swamps, non-tidal marshes, and non-tidal water) to identify the fate of floodplain areas that were removed from the estuarine system. Some estuary habitat has been lost and converted to non-estuarine habitat, while other habitats have been lost as result of succession to another estuarine habitat type (Thomas 1983). As a result, the relative proportions of the five estuary habitat types has changed considerably from 1870 to 1983. Also, the significance of loss of certain habitat types has been partially masked by the formation of these habitats elsewhere. Further, the geographic movement of estuary habitats is not clear from the quantification of total acreage change. For example, the total acreage of a certain habitat type within a particular estuary area may not have changed considerably from historical to current conditions, however, the location of this habitat type within the estuary area may be completely different. The habitat change within each estuary region and from one type to the next is discussed in the following subsections.

The Lower Columbia River Estuary Partnership (LCREP) was interested in describing the location and distribution of estuarine and tidal freshwater habitat cover types along the Columbia River from the mouth to Bonneville Dam using a consistent method and data source (Garono et al 2003c) as well as understanding recent habitat change in the estuary and lower Columbia River mainstem (Garono et al 2003a). The habitat mapping focused on estuarine and tidal freshwater habitats; areas not located along the river and >175 ft elevation (for the eastern dataset) or >100 ft elevation (for the western dataset) were deleted from the habitat classification (Garono et al 2003c). Although habitat change expressed as the percent of the 1992 area indicates considerable change from 1992 to 2000, the percent of total habitat comprised by each land cover class is similar in both 1992 and 2000. Further, it is important to note the losses and gains of each habitat type, as well as the transition among habitat types. For example, most of the loss of shrub-scrub wetland habitat was to either herbaceous or forested wetlands; the absolute loss of the shrub-scrub wetland habitat was offset by substantial transition of herbaceous wetlands to shrub-scrub wetlands. Much of the increase in deciduous forest upland habitat coverage was a result of transition of shrub-scrub upland, coniferous forest upland, or mixed forest upland habitats; this may be indicative of normal successional transitions. A considerable amount of the change in area of habitat cover was potentially explained by either natural habitat succession or error associated with differences in accuracy of the two data sets. In general, if a specific habitat type changed from 1992 to 2000, it remained within the larger designation of wetland or upland, that is, wetlands transitioned to other wetlands while uplands transitioned to other uplands.

Johnson and O'Neil (2001) developed a habitat classification system to describe wildlife habitats present in Washington and Oregon. The habitats described by Johnson and O'Neil (2001) have been used in the NPCC subbasin planning process throughout the Columbia Basin. Maps of many NPCC subbasins depicting the habitat coverage in 1850 and 1999 are currently available through the Interactive Biodiversity Information System (IBIS) website (http://ibis.nwhi.org).

Comparison of estuary and lower mainstem habitats describe by the three primary classification systems and mapping efforts (Thomas 1983, Johnson and O'Neil (2001)/IBIS (2003), Garono et al. 2003c) is difficult because of the different purposes of each effort. Further, each effort covered a different geographic area, encompassed different time periods, and utilized a different method or resolution. These differences may contribute to different results obtained during each effort. Nevertheless, we attempt to describe the changes in habitat in the Columbia River estuary and lower mainstem based on the findings of these habitat mapping projects. Regardless of the differences, each mapping project reached the conclusion that estuary and mainstem habitats have changed significantly as a result of human influence.

Other habitat inventory efforts include that of Christy and Putera (1992) and Graves et al. (1995) who extended the work of Thomas (1983); these mapping efforts used Geographic Information Systems methods (GIS) to delineated Thomas' (1983) estuary habitat types from rm 46.5 to rm 105. Finally, Johnson et al. (2003b) summarized many of the habitat inventory efforts to date (Thomas 1983, Graves et al. 1995, USACE 1996, Garono et al. 2002) to describe habitat changes in the Columbia River estuary and lower mainstem up to Bonneville Dam. A qualitative change in habitat characteristics by estuary area is included in Table 2-6. These studies are identified here primarily to inform the reader that other habitat mapping projects exists for the Columbia River estuary and lower mainstem.

Table 2-5. Estimated change in estuary habitats by region within the Columbia River estuary from rm 0 to rm 46 (Thomas 1983).

HABITAT TYPE Estuary Region	1870 Acreage	1983 Acreage	Change
DEEP WATER			
Entrance	8,900	10,580	+1,680 (19%)
Mixing Zone	8,450	8,360	-90 (1%)
Youngs Bay	810	850	+40 (5%)
Baker Bay	1,800	450	-1,350 (75%)
Grays Bay	2,270	1,690	-580 (26%)
Cathlamet Bay	6,390	5,590	-800 (13%)
Upper Estuary	6,520	5,060	-1,460 (22%)
TOTAL	**35,140**	**32,580**	**-2,560 (7%)**
MEDIUM DEPTH WATER			
Entrance	4,480	2,640	-1,840 (41%)
Mixing Zone	10,780	10,330	-450 (4%)
Youngs Bay	1,120	870	-250 (22%)
Baker Bay	4,700	1,350	-3,350 (71%)
Grays Bay	2,230	2,040	-190 (9%)
Cathlamet Bay	8,190	5,700	-2,490 (30%)
Upper Estuary	2,710	2,790	+80 (3%)
TOTAL	**34,210**	**25,720**	**-8,490 (25%)**
SHALLOW/TIDAL FLATS			
Entrance	2,980	1,680	-1,300 (44%)
Mixing Zone	9,540	9,490	-50 (1%)
Youngs Bay	4,400	3,860	-540 (12%)
Baker Bay	4,830	8,450	+3,620 (75%)
Grays Bay	3,790	4,330	+540 (14%)
Cathlamet Bay	13,330	14,250	+920 (7%)
Upper Estuary	1,770	2,710	+940 (53%)
TOTAL	**40,640**	**44,770**	**+4,130 (10%)**
TIDAL MARSH			
Entrance	0	250	+250
Mixing Zone	10	10	0
Youngs Bay	7,210	980	-6,230 (86%)
Baker Bay	1,640	730	-910 (56%)
Grays Bay	310	760	+450 (145%)
Cathlamet Bay	5,580	5,960	+380 (7%)
Upper Estuary	1,430	510	-920 (64%)
TOTAL	**16,180**	**9,200**	**-6,980 (43%)**
TIDAL SWAMP			
Entrance	0	0	0
Mixing Zone	0	0	0
Youngs Bay	3,000	130	-2,870 (96%)
Baker Bay	3,480	0	-3,480 (100%)
Grays Bay	4,410	510	-3,900 (88%)
Cathlamet Bay	7,950	4,060	-3,890 (49%)
Upper Estuary	11,180	2,250	-8,930 (80%)
TOTAL	**30,020**	**6,950**	**-23,070 (77%)**
TOTAL ESTUARY	**156,190**	**119,220**	**-36,970 (24%)**

Table 2-6. Qualitative description of the change in habitat characteristics from historical to current conditions by area, including a judgment of relative importance (adapted from Johnson et al. 2003b; L, M, and H refer to Low, Medium, and High).

Area	Tidal Exchange	Bathymetry	Salinity
Entrance	***L***-only a small area of historical marshes and swamps	***H***-very large increases in deep water area, and loss of medium and shallow depth areas	***L***-probably somewhat less dynamic, but still ocean-dominated
Mixing Zone	***L***-only a small area of historical marshes and swamps	***L***-little change in area, although high degree of shifting of locations	***M***-very dynamic salinity zone, probably altered by flow regulation
Youngs Bay	***H***-substantial loss of tidal marsh and swamp	***M***-loss of medium and shallow depth areas	***M***-very dynamic salinity zone, probably altered by flow regulation
Baker Bay	***H***-substantial loss of tidal marsh and swamp	***H***-substantial loss of deep and medium deep areas, and increase in shallow areas	***M***-very dynamic salinity zone, probably altered by flow regulation
Grays Bay	***H***-substantial loss of tidal swamp	***M***-shift from deepwater area to shallow flats	***L***-a small change in dilute salinity dynamics
Cathlamet Bay	***M***-loss of tidal swamps, but gain in tidal marsh	***M***-loss of deep and medium deep areas	***L***-a small change in dilute salinity dynamics
Upper Estuary	***H***-substantial loss of tidal swamp and marsh	***H***-loss of deep and gain in medium deep area, and substantial increase in shallow areas	***L***-a small change in dilute salinity dynamics
Tidal Freshwater Middle Reach (RM46-102)	***H***-substantial loss of tidal swamp and marsh, and non-tidal wetland	***H***-loss of shallow area, and gain in deep area	***L***-salinity not a factor
Tidal Freshwater Upper Reach (RM 102-146)	***H***-substantial loss of tidal swamp and marsh suspected, and gain in non-tidal wetland	***H***-loss of shallow area, and gain in deep area	***L***-salinity not a factor

2.3.3.1 Deep Water Habitat

Thomas (1983) documented a total loss of 2,560 acres of deep water habitat from 1870 to 1983; this represents a 7% loss of the 1870 acreage (Table 2-5). The most substantial losses of deep water habitat include 1,350 acres in the Baker Bay and 1,450 acres in the Upper Estuary. Loss of deep water habitat in Baker Bay represents the migration of Sand Island from the Entrance area to Baker Bay, which had occurred naturally by 1885. Jetty construction moderated the variability in water movement within the Entrance area, causing the retention of Sand Island in its present location (Thomas 1983). Further, maintenance dredging activities of the river bar and navigation channel in the Entrance area have contributed to increases of deep water habitat in this area (Thomas 1983). Although little change of deep water habitat acreage was observed in the Mixing Zone and Youngs Bay areas, location of deep water habitats in these areas has shifted as a result of migration of the channel (Thomas 1983). Loss of deepwater habitats in the subareas furthest upstream (Grays Bay, Cathlamet Bay, and Upper Estuary) was primarily a result of accretion that converted these habitats to medium depth or shallow/flats habitat. Deep water habitat losses were complemented by a 1,680 acre gain in the Entrance area; this increase in deep water habitat was also related to the migration of Sand Island.

In the Columbia Estuary Subbasin, there has been close to a complete loss of open water habitat from 1850 to 1999; as of 1999, only 878 acres of this habitat type remained in the subbasin (Table 2-7 and Figure 2-25). The open water habitat type does not have a water depth designation, thus, it is not clear which open water habitats comprise deep, medium, or shallow depths. Much of the historical open water habitat type was converted to the bays and estuaries habitat type (Table 2-7 and Figure 2-25). In the Columbia Lower Subbasin, a similar loss of open water habitat and conversion to bays and estuaries habitat occurred from the historical to current conditions (Table 2-8 and Figure 2-26). The apparent conversion of open water habitat to bays/estuaries habitat in these subbasins is a function of the different mapping data and methods used for the current and historical maps rather than an actual habitat conversion (Thomas O'Neil, Northwest Habitat Institute, personal communication). In the historical era mapping effort, the focus was on terrestrial habitats and the bays/estuaries habitat type was not even included in the habitat classification. Thus, although bays/estuaries habitat may have been present historically, location this habitat type was not mapped and much of the Columbia River corridor was classified as open water habitat. During the current era mapping effort, bays/estuaries habitat was classified and included in the terrestrial layer of the map while open water was included in the aquatic map layer. Because the terrestrial layer was overlayed on the aquatic layer, any bays/estuaries habitat in the same location as open water habitat would override the open water habitat type. Thus, on the current era map, bays/estuaries habitat may be overestimated and open water habitat may be underestimated.

In an analysis of recent habitat change, Garono et al (2003a) observed very little change in water habitat from 1992 to 2000 (Table 2-9). Again, there is no water depth designation to this water habitat type, so it is not clear which water habitats comprise deep, medium, or shallow depths.

2.3.3.2 Medium Depth Habitat

Except for the Upper Estuary area, Thomas (1983) documented a loss of medium depth water habitat in all areas of the estuary from 1870 to 1983 (Table 2-5). The collective loss of medium depth water habitat in the estuary was 8,490 acres, which represents about 25% of the 1870 acreage (Table 2-5). Substantial acreages of medium depth water were converted to deep water in the Entrance Subarea and to tidal flats in the Baker Bay Subarea; this is consistent with the migration of Sand Island and the maintenance of the navigation channel as described above. In Cathlamet Bay, considerable acreage of medium depth habitats was converted to shallows/flats through the process of accretion.

2.3.3.3 Shallow Water/Flats Habitat

The shallow water/flats habitat type is the only habitat where an estuary-wide increase in acreage occurred from 1870 to 1983 (Table 2-5). There are two basic processes by which shallow water/flats habitat can be created: accretion in deep/medium depth water habitats or erosion of tidal marsh or tidal swamp habitat (Thomas 1983). Formation of shallow water/flats habitat in the estuary from 1870 to 1983 have primarily been a result of the former process (Thomas 1983). The Entrance area showed the only substantial loss of shallow water/flats habitat while a large increase of this habitat type was observed in Baker Bay; these changes are consistent with the natural migration of Sand Island (Thomas 1983). Further, construction of the South Jetty resulted in considerable accretion of sand in the Entrance area; as a result, sand dunes have formed in areas that were formerly shallow water/flats habitat (Thomas 1983). In the more upstream areas (Grays Bay, Cathlamet Bay, and the Upper Estuary), losses of former medium and deep water habitats resulting from accretion have contributed to the increases in shallow water/flats habitat (Thomas 1983).

The shallow water/flats habitat defined by Thomas (1983) may have been mapped as open water, or bays/estuaries by Johnson and O'Neil (2001) and IBIS (2003) or as water by Garono et al. (2003c). As previously discussed, there is no depth designation to the general water habitat types of Johnson and O'Neil (2001) and Garono et al. (2003c); thus, comparison to the specific depth water habitats of Thomas (1983) is not appropriate. The bays and estuaries habitat type (Johnson and O'Neil 2001) was previously discussed in section 2.3.3.1; this habitat type appeared to replace much of the open water habitat in the estuary and lower mainstem (Figure 2-25 and Figure 2-26; IBIS 2003).

2.3.3.4 Tidal Marsh Habitat

Approximately 10,500 acres of 1870 tidal marsh acreage have been lost, however, the formation of about 3,500 acres of new tidal marsh resulted in the net loss of about 7,000 acres (Table 2-5). The 1870 estimate of tidal marsh acreage was difficult to determine because tidal marsh often occurred in a mosaic with tidal swamp habitat (Thomas 1983). In general, tidal marsh habitat loss is a result of extensive diking; high elevation tidal marshes have been diked more than lower elevation marshes (Thomas 1983). New tidal marsh formation has resulted primarily from vegetative colonization of disposed dredge material, but colonization has also occurred along natural shorelines and in shallow water/flats habitat (Thomas 1983). The location of tidal marsh habitat within each estuary area has changed as a result of modified flow regime, modified tidal action, and/or shipping channel development and maintenance.

In the Entrance area, the small gain of tidal marsh habitat has resulted from changes to wave action as a result of jetty construction (Thomas 1983). Formerly, wave action in the Entrance area prevented vegetative colonization (Thomas 1983). The jetties have resulted in decreased wave action, allowing the formation of tidal marsh habitat in the now sheltered area of Trestle Bay (Thomas 1983). In Baker Bay, the historical tidal marsh habitats have all been diked and therefore considered as lost (Thomas 1983). The 730 acres of tidal marsh habitat in Baker Bay in 1983 was all recently formed along shorelines in areas that were formerly exposed to wave action where vegetation could not colonize (Thomas 1983). A similar situation has occurred in Youngs Bay, where much of the historical tidal marsh habitat has been lost to diking and close to half of the 1983 tidal marsh habitat was recently formed (Thomas 1983). In Grays Bay, diking has not affected tidal marsh acreage because most diked areas were formerly tidal swamp; the gain of tidal marsh habitat in the Grays Bay area resulted from accretion in tide flats followed by bulrush colonization (Thomas 1983). A similar situation has occurred in Cathlamet Bay, however, the formation of tidal marsh habitats has occurred primarily in areas of dredge spoils deposition (Thomas 1983). In the Upper Estuary area, the net loss of tidal marsh habitat was the product of substantial losses of tidal marsh habitat on Tenasillahe Island as a result of diking that were offset by tidal marsh formation in areas of dredge spoils deposition (Thomas 1983).

The tidal marsh habitat defined by Thomas (1983) may have been mapped as herbaceous wetlands by Johnson and O'Neil (2001) and IBIS (2003; Table 2-3). In the Columbia Estuary Subbasin, almost 31,000 acres of herbaceous wetlands have been lost from 1850 to 1999; this represents a 67% loss of the 1850 acreage of herbaceous wetlands (Table 2-7 and Figure 2-25).

In the Columbia Lower Subbasin, approximately 140,000 acres of herbaceous wetlands have been lost from 1850 to 1999; this represents a 94% loss of the 1850 acreage of herbaceous wetlands (Table 2-8 and Figure 2-26). The percentage and absolute acreage loss of herbaceous wetlands determined by IBIS (2003) are considerably higher than the results of Thomas (1983); regardless, both mapping efforts document a substantial loss of the tidal marsh or herbaceous wetland habitat type.

The tidal marsh habitat defined by Thomas (1983) may have been mapped as herbaceous wetlands or scrub-shrub wetlands by Garono et al. (2003a;Table 2-3). The recent habitat change analysis by Garono et al. (2003a) documented an increase of 8,495 acres of herbaceous wetland habitat from 1992 to 2000; this increase represents 17% of the 1992 acreage (Table 2-9). Most of the herbaceous wetland habitat in 2000 was formerly scrub-shrub wetland (44%), forested wetland (31%), or urban areas (24%). Garono et al. (2003a) felt it was unlikely that urban habitats had converted to herbaceous wetlands from 1992 to 2000; rather, this result may be a function of the ability of the 2000 data set to better discriminate between actual urban areas and vegetated areas within and around urban areas. Conversely, Garono et al. (2003a) observed a loss of about 9,000 acres of scrub-shrub wetlands, which represent about 36% of the 1992 habitat acreage (Table 2-9). Most of the habitat loss of scrub-shrub wetlands was a result of conversion to herbaceous wetlands (44%) or forested wetlands (21%) (Garono et al. 2003a).

2.3.3.5 Tidal Swamp Habitat

Tidal swamp habitat was by far the most impacted estuarine habitat type; almost all of the 1870 tidal swamp habitat has been converted to one of the diked floodplain/non-tidal habitats described below (Thomas 1983). Loss of tidal swamp habitat alone was responsible for 62% of the total estuary habitat loss (Thomas 1983). Thomas (1983) reasoned that, because of their elevation and/or irregular tidal influence, tidal swamp habitat is the estuarine habitat most susceptible to diking. Historically, few tidal swamps were present in the Entrance and Mixing Zone areas, thus little change has been observed in these areas (Thomas 1983). There has been almost complete loss of all 1870 tidal swamp habitat from the Youngs Bay and Baker Bay areas; as a result, brackish water tidal swamps have been essentially eliminated from the estuary (Thomas 1983). In the areas furthest upstream, tidal swamp acreage losses have been extensive, however, a substantial amount of tidal swamp acreage is still present, particularly in the Cathlamet Bay area (Thomas 1983).

The tidal swamp habitat defined by Thomas (1983) may have been mapped as Westside riparian-wetlands by Johnson and O’Neil (2001) and IBIS (2003;Table 2-3). However, the Westside riparian-wetland habitat type typically occupies patches or linear strips within a forest matrix; other characteristics of this habitat type (Johnson and O’Neil 2001) indicate that it may differ substantially from the tidal swamp described by Thomas (1983). Nevertheless, Westside riparian-wetland appears to be the most closely related habitat type to tidal swamp. In the Columbia Estuary Subbasin, an increase of about 6,000 acres (i.e. 41% of 1850 acreage) of Westside riparian-wetlands occurred from 1850 to 1999 (Table 2-7 and Figure 2-25). Similarly in the Columbia Lower Subbasin, Westside riparian-wetland habitat acreage increased by about 3,000 acres (i.e. 24% of 1850 acreage) from 1850 to 1999 (Table 2-8 and Figure 2-26). This result was completely opposite that observed by Thomas (1983) for tidal swamp habitat. The increased acreage of Westside riparian-wetland from 1850 to 1999 is most likely a result of different resolutions between the mapping data rather than an actual increase in this wetland habitat type; the habitat change result for this habitat type would likely be much different if the resolution in the 1850 and 1999 data were similar (Thomas O’Neil, personal communication).

The substantial acreage loss of the tidal swamp and tidal marsh habitat types has important implications on juvenile salmonid survival in the estuary because evidence suggests salmonids, particularly ocean-type salmonids, depend on these habitats for food and cover requirements. Further, tidal marsh and swamp habitat acreage constituted 30% of the total 1870 acreage while these habitats comprise only 14% of the total 1983 estuarine habitat acreage.

The tidal swamp habitat defined by Thomas (1983) may have been mapped as forested wetlands by Garono et al. (2003a; Table 2-3). The recent habitat change analysis by Garono et al. (2003a) documented an increase of about 5,500 acres of forested wetland habitat from 1992 to 2000; this increase represents 49% of the 1992 acreage (Table 2-9). Most of the forested wetland habitat in 2000 was formerly scrub-shrub wetland (21%) or herbaceous wetland (9%); thus, the increase in forested wetland habitat appears to be partially explained by succession of other wetland habitats. The increase of forested wetland habitat is completely opposite that observed by Thomas (1983) for tidal swamp habitat; this difference is likely a result of the different time period, geographic area, and method used in each study.

2.3.3.6 Non-Estuarine Wetlands

Thomas (1983) estimated that about 7,000 acres of non-estuarine wetlands habitat (i.e. non-estuarine swamps, marsh, and water) were created in the estuary from 1870 to 1983; most of this area was formerly tidal swamps and, to a lesser extent, tidal marsh. Non-estuarine wetlands habitat was created in all estuary areas except the Mixing Zone (Table 2-10).

Similar habitat types defined by Johnson and O'Neil (2001) (i.e. Westside riparian-wetlands, herbaceous wetlands) or by Garono et al. (2003a) (i.e. forested wetlands, herbaceous wetlands) (Table 2-3) have already been discussed.

2.3.3.7 Forested Uplands

Forest upland habitats in the Columbia River estuary and lower mainstem characterized by Johnson and O'Neil (2001) include Westside (mesic) lowlands conifer-hardwood forest, Westside oak and dry Douglas fir forest, and montane mixed conifer forest. In the Columbia Estuary Subbasin and the Columbia Lower Subbasin, Westside lowlands conifer-hardwood forest increased by about 17,500 and 33,000 acres, respectively, from 1850 to 1999 (Table 2-7, Table 2-8, Figure 2-25, and Figure 2-26). In the analysis of more recent habitat change, Garono et al (2003a) documented an increase of about 4,500 acres of coniferous forest upland from 1992 to 2000 (Table 2-9). About half of the coniferous forest upland habitat in 1992 remained as such in 2000; much of the remaining coniferous forest upland habitat in 2000 was a result of conversion of mixed forest upland (26%), deciduous forest upland (18%), and scrub-shrub upland (18%).

In the Columbia Estuary Subbasin and the Columbia Lower Subbasin, montane mixed conifer forest decreased by about 4,500 and 2,500 acres, respectively, from 1850 to 1999 (Table 2-7, Table 2-8, Figure 2-25, and Figure 2-26). Most of the historical montane mixed conifer forest was recently classified as Westside lowlands conifer-hardwood forest in both subbasins; this may be an artifact of the different resolution of mapping data from 1850 to 1999. For the mixed forest upland habitat type, Garono et al (2003a) observed a loss of about 6,000 acres from 1992 to 2000 (Table 2-9); most of the lost mixed forest upland habitat was explained by the conversion to deciduous forest upland (26%) and coniferous forest upland (26%).

In the Columbia Lower Subbasin, Westside oak and dry Douglas fir forest habitat decreased by about 86,000 acres from 1850 to 1999 (Table 2-8 and Figure 2-26); this represents a 93% loss of this habitat type. Most of the Westside oak and dry Douglas fir forest habitat appears to have been converted to the agriculture, pastures, and mixed environs or the urban and mixed environs habitat types (Figure 2-26). Conversely, Garono et al. (2003a) documented a substantial increase in deciduous forest upland from 1992 to 2000; the increase of over 11,000 acres of this habitat represents a 429% change over the 1992 acreage (Table 2-9). The increase in deciduous forest upland habitat acreage in 2000 was a result of conversion of scrub-shrub upland, mixed forest upland, and coniferous forest upland.

2.3.3.8 Developed Floodplain

Thomas (1983) estimated that about 24,000 acres of developed floodplain habitat were created in the estuary from 1870 to 1983; most of this area was formerly tidal swamps and, to a lesser extent, tidal marsh. Developed floodplain habitat was not created in the Entrance or Mixing Zone areas; developed floodplain habitat was somewhat evenly distributed among the other five estuary areas (Table 2-10).

The developed floodplain habitat of Thomas (1983) is most closely related to the agriculture, pastures, and mixed environs and the urban and mixed environs habitat types of Johnson and O'Neil (2001). In the Columbia Estuary Subbasin, 16,887 acres of the agriculture, pastures, and mixed environs habitat type and 6,344 acres of the urban and mixed environs habitat type were created between 1850 and 1999 (Table 2-7 and Figure 2-25). Thus, the combined creation of these two habitat types from 1850 to 1999 (i.e. 23,231 acres) is extremely similar to the creation of developed floodplain habitat from 1870 to 1983 documented by Thomas (1983). In the Columbia Lower Subbasin, a considerable amount of the agriculture, pastures, and mixed environs habitat type (i.e. 110,041) and the urban and mixed environs habitat type (i.e. 89,900) were created between 1850 and 1999 (Table 2-8 and Figure 2-26).

In the analysis of more recent habitat change, Garono et al. (2003a) observed a decrease of about 2,000 acres of the urban habitat type from 1992 to 2000, which represents a decrease of about 14% of the 1992 acreage. As previously mentioned, Garono et al. (2003a) felt it was unlikely that urban habitat coverage had decreased from 1992 to 2000; rather, this result may be a function of the ability of the 2000 data set to better discriminate between actual urban areas and vegetated areas within and around urban areas.

The results presented by Thomas (1983) and IBIS (2003) are consistent with the habitat mapping data summarized by Johnson et al. (2003b). In the tidal freshwater portion of the lower mainstem from rm 46-102, there was a general increase in upland habitat complemented by a substantial loss of non-tidal water/wetland, tidal flats, and tidal marsh habitat types; similarly, from rm 105-146, there was an increase of non-tidal water/wetland and upland habitat balanced with a substantial loss of tidal flats and tidal marsh habitat types (Johnson et al. 2003b). In both reaches of the tidal freshwater portion of the lower mainstem, there was no available comparison category for tidal swamp habitat (Johnson et al 2003b).

2.3.3.9 Natural and Filled Uplands

The 1,900 acres of historical natural and filled upland habitat identified by Thomas (1983) was comprised mostly of sand dunes throughout the Entrance, Youngs Bay, and Baker Bay areas (Table 2-10). A considerable amount of natural and filled upland habitat was created from 1870 to 1983; some of this habitat was created as a result of accretion of sand in Baker Bay and along Clatsop Spit, however, most of this created habitat resulted from the disposal of dredge spoils (Thomas 1983).

The natural and filled upland habitat defined by Thomas (1983) may have been mapped as coastal dunes and beaches habitat by Johnson and O'Neil (2001) and IBIS (2003; Table 2-3). In the Columbia Estuary Subbasin, almost all of the historical coastal dunes and beaches habitat were lost by 1999 (Table 2-7 and Figure 2-25), which is contrary to the results of Thomas (1983). Numerous factors may explain this difference, including dissimilar habitat types and different time periods, geographic area, or methods used in each study.

The natural and filled upland habitat defined by Thomas (1983) may have been mapped as unconsolidated shore habitat by Garono et al. (2003a; Table 2-3). From 1992 to 2000, there was a loss of about 4,500 acres of unconsolidated shore habitat, which represents about 20% of the 1992 habitat acreage (Table 2-9). This result is also conflicts with the results of Thomas (1983) but is consistent with IBIS (2003).

Table 2-7. Historical (circa 1850) and current (1999) wildlife habitat types and acreage in the Columbia Estuary Subbasin (IBIS 2003).

	Acreage		
Habitat Name	**Historical (circa 1850)**	**Current (1999)**	**Change**
Westside (mesic) Lowlands Conifer-Hardwood Forest	303,217	320,712	+17,495 (6%)
Montane Mixed Conifer Forest	4,466	0	-4,466 (100%)
Agriculture, Pastures, and Mixed Environs	0	16,887	+16,887
Urban and Mixed Environs	0	6,344	+6,344
Open Water - Lakes, Rivers, and Streams	105,277	878	-104,399 (99%)
Herbaceous Wetlands	45,720	14,887	-30,833 (67%)
Westside Riparian-Wetlands	14,186	20,064	+5,878 (41%)
Coastal Dunes and Beaches	8,634	375	-8,259 (96%)
Coastal Headlands and Islets	741	510	-231 (31%)
Bays and Estuaries		101,022	+101,022
Marine Nearshore		562	+562
Total Acres:	**482,238**	**482,235**	

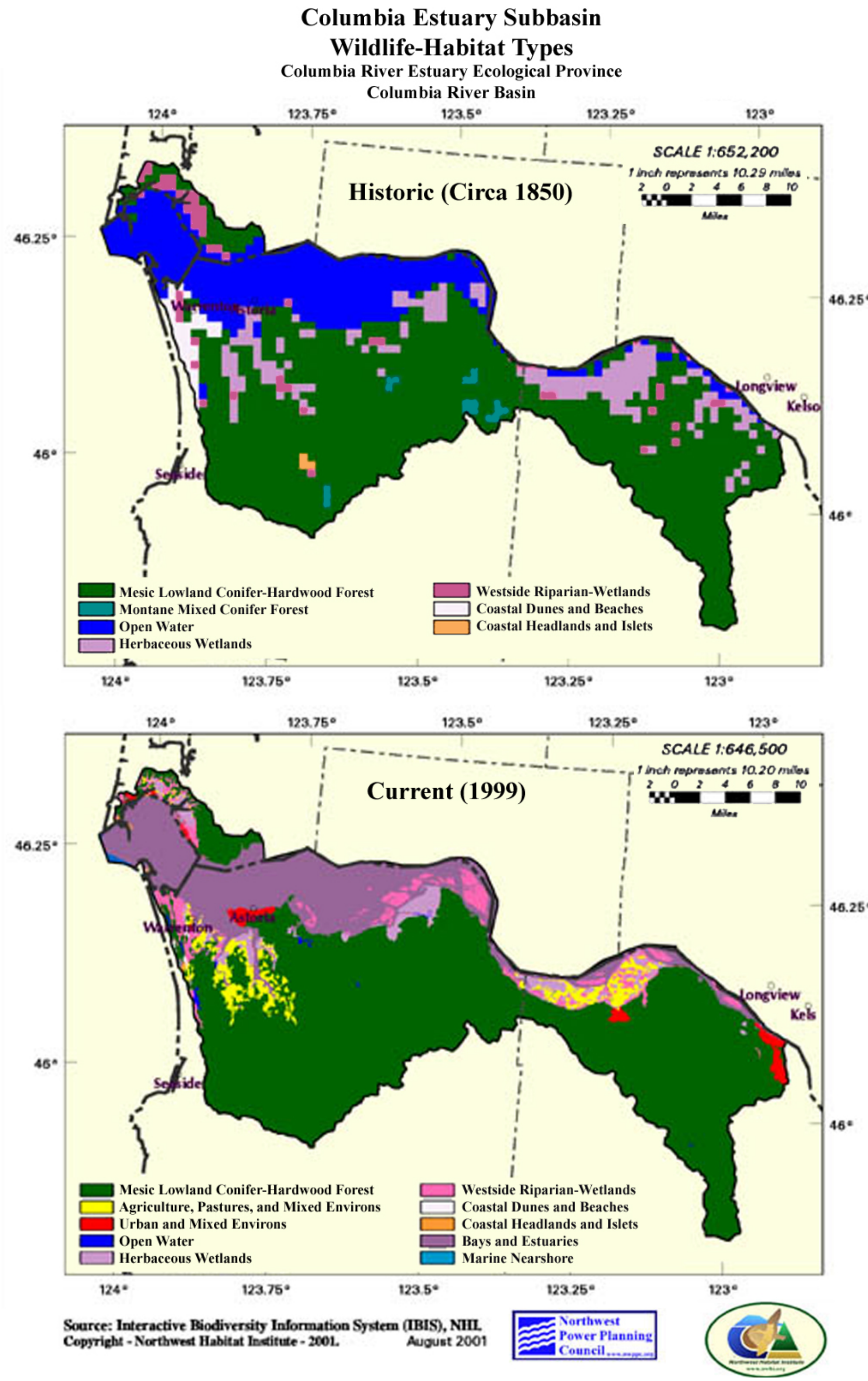


Figure 2-25. Historical (circa 1850) and current (1999) wildlife habitat types in the Columbia Estuary Subbasin (IBIS 2003).

Table 2-8. Historical (circa 1850) and current (1999) wildlife habitat types and acreage in the lower Columbia Lower Subbasin (IBIS 2003).

	Acreage		
Habitat Name	**Historical (circa 1850)**	**Current (1999)**	**Change**
Westside (mesic) Lowlands Conifer-Hardwood Forest	185,062	218,043	+32,981 (18%)
Westside Oak and Dry Douglas-fir Forest and Woodlands	92,444	6,206	-86,238 (93%)
Montane Mixed Conifer Forest	4,161	1,772	-2,389 (57%)
Alpine Grasslands and Shrublands	2,471	0	-2,471 (100%)
Westside Grasslands	2,965	0	-2,965 (100%)
Agriculture, Pastures, and Mixed Environs	0	110,041	+110,041
Urban and Mixed Environs	0	89,900	+89,900
Open Water - Lakes, Rivers, and Streams	44,350	841	-43,509 (98%)
Herbaceous Wetlands	149,521	9,413	-140,108 (94%)
Westside Riparian-Wetlands	12,982	16,086	+3,104 (24%)
Montane Coniferous Wetlands	0	1,912	+1,912
Bays and Estuaries	0	39,742	+39,742
Total Acres	**493,953**	**493,950**	

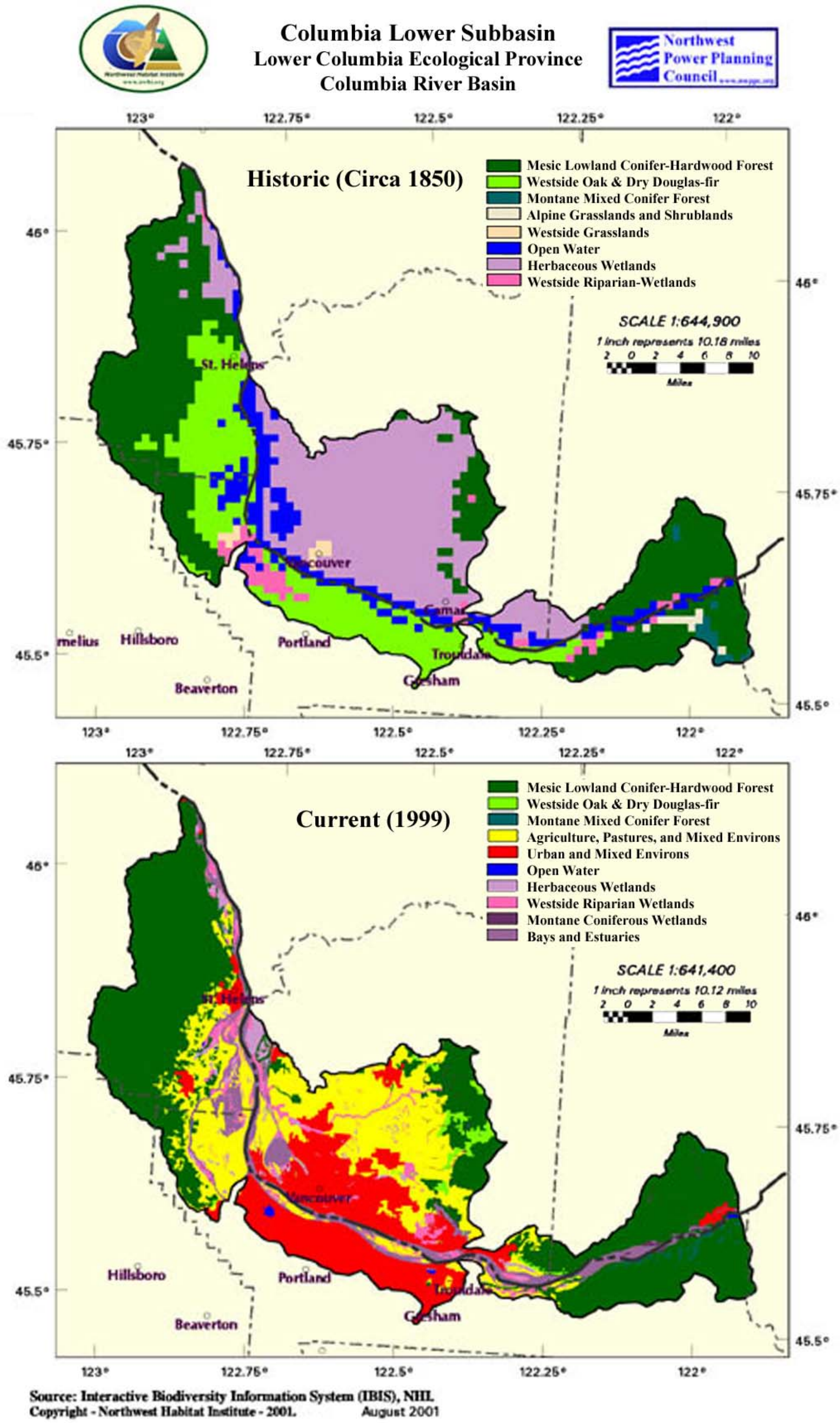


Figure 2-26. Historical (circa 1850) and current (1999) wildlife habitat types in the Columbia Lower Subbasin (IBIS 2003).

Table 2-9. Estimated change in Columbia River estuary habitat cover types from 1992 to 2000 (Garono et al. 2003a).

Land Cover Class	1992 Area (acres)	1992 % of Total	2000 Area (acres)	2000 % of Total	Change Area (acres)	Change % of 1992 Total
Herbaceous Wetland	50,106.0	18.1	58,601.0	21.1	8,495.0	17
Shrub-Scrub Wetland	24,781.7	8.9	15,810.5	5.7	-8,971.1	-36
Forested Wetland	11,101.9	4.0	16,580.7	6.0	5,478.8	49
Herbaceous Upland	6,568.5	2.4	11,415.3	4.1	4,846.7	74
Shrub-Scrub Upland	21,659.7	7.8	6,993.6	2.5	-14,666.2	-68
Deciduous Forest Upland	2,627.2	1.0	13,886.8	5.0	11,259.6	429
Coniferous Forest Upland	9,354.7	3.4	13,985.6	5.0	4,631.0	50
Mixed Forest Upland	11,403.4	4.1	5,274.2	1.9	-6,129.2	-54
Unconsolidated Shore	22,709.2	8.2	18,123.4	6.5	-4,585.8	-20
Urban	14,433.7	5.2	12,482.0	4.5	-1,951.6	-14
Water	102,758.9	37.0	102,871.0	37.0	112.2	0.1
Other			1,480.2	0.5	1,480.2	-

Table 2-10. Estimated change in non-estuarine habitats by region within the Columbia River estuary from rm 0 to rm 46 (Thomas 1983).

HABITAT TYPE Estuary Region	1870 Acreage	1983 Acreage	Change
DEVELOPED FLOODPLAIN			
Entrance	0	0	0
Mixing Zone	0	0	0
Youngs Bay	0	6,670	+6,670
Baker Bay	0	3,420	+3,420
Grays Bay	0	3,270	+3,270
Cathlamet Bay	0	4,150	+4,150
Upper Estuary	0	6,440	+6,440
TOTAL	**0**	**23,950**	**+23,950**
UPLANDS – NATURAL AND FILLED			
Entrance	530	1,300	+770 (145%)
Mixing Zone	0	590	+590
Youngs Bay	350	1,070	+720 (206%)
Baker Bay	1,050	1,600	+550 (52%)
Grays Bay	0	120	+120
Cathlamet Bay	0	920	+920
Upper Estuary	0	1,990	+1,990
TOTAL	**1,930**	**7,590**	**+5,660 (293%)**
NON-ESTUARINE SWAMP			
Entrance	0	130	+130
Mixing Zone	0	0	0
Youngs Bay		1,370	+1,370
Baker Bay	0	1,260	+1,260
Grays Bay	0	200	+200
Cathlamet Bay	0	110	+110
Upper Estuary	0	250	+250
TOTAL	**0**	**3,320**	**+3,320**
NON-ESTUARINE MARSH			
Entrance	0	360	+360
Mixing Zone	0	0	0
Youngs Bay	0	930	+930
Baker Bay	0	170	+170
Grays Bay	0	40	+40
Cathlamet Bay	0	430	+430
Upper Estuary	0	1,200	+1,200
TOTAL	**0**	**3,130**	**+3,130**
NON-ESTUARINE WATER			
Entrance	50	0	-50 (100%)
Mixing Zone	0	0	0
Youngs Bay	0	160	+160
Baker Bay	0	70	+70
Grays Bay	0	50	+50
Cathlamet Bay	0	270	+270
Upper Estuary	0	410	+410
TOTAL	**50**	**960**	**+910 (1,820%)**
TOTAL	**1,980**	**38,950**	**+36,970 (1,867%)**

2.4 Species/Habitat Interactions

Discussions of interactions between species and habitats are divided into multiple sections. Section 2.4.1 Focal Species Habitat Associations presents the general estuary and lower mainstem habitats associated with each focal species. Section 2.4.2 Salmonids provides a more detailed discussion of the known and suspected biological relationships between salmonids and the estuary and lower mainstem ecosystem. Sections 2.4.3 through 2.4.13 discuss the relationships between other focal species and the estuary and lower mainstem ecosystem.

2.4.1 Focal Species Habitat Associations

A species/habitat matrix was developed for the estuary focal species (Table 2-11); the matrix summarizes a qualitative assessment of potential species utilization within coarse estuary and mainstem habitats. Habitats were chosen for two reasons: 1) habitats were included in a current versus historical acreage comparison in the Columbia River estuary (Thomas 1983, Johnson et al. 2003b), or 2) habitats were considered important based on WDFW input. The utilization levels are based on professional interpretation of the reviewed literature and life history descriptions in the Technical Foundation; the utilization levels are an arbitrary qualitative scale that includes the following levels of habitat use: none, low, medium, high, and critical. The first four categories are self-explanatory; the critical designation indicates that the habitat type is critical to the survival of the particular life stage of the focal species. There are numerous habitat classification system available for fish and wildlife research (i.e. Rosgen stream channel typing, Cowardin wetland and deepwater habitat classification (Cowardin et al. 1979), wildlife habitat classification (Johnson and O'Neil 2001)); choice of the appropriate systems depends on the purpose of the project as described further in section 2.1.5. For this analysis, the coarse habitat types described in the current versus historical acreage comparison (i.e. Thomas 1983) provide a context to discuss potential effects of estuary habitat change over time on focal species. These estuary habitats have previously been defined in section 2.1.5.

Utilization levels of each habitat type by focal species is *not* intended to serve as the ultimate authority in determining importance of that habitat type. For example, the fish focal species do not utilize old growth/mature forests; however, the importance of this habitat type should not be ignored. More complete habitat associations specifically for wildlife species have been developed by Johnson and O'Neil (2001) and continue to be updated on the Northwest Habitat Institute's (NHI) webpage (www.nwhi.org; Table 2-12). This table presents the known wildlife focal species habitat associations within the Columbia Estuary and Columbia Lower Subbasins; thus, focal fish species are not included. The wildlife-habitat type column follows the classification of Johnson and O'Neil (2001) and is consistent with the habitats presented in Table 2-7 and Table 2-8. The association type column provides a qualitative description of the level of association between the species and the habitat. The activity type column describes the behavior that occurs within the habitat type. Finally, the confidence level indicates the level of certainty of the relationship between the species and the habitat type. NHI has also determined if a relationship exists among wildlife species and the various life stages of salmonids; those wildlife focal species that interact with salmonids are presented in Table 2-13. The relationship type column indicates the degree and repeatability of the relationship between the focal species and salmonids. The salmonid stage column describes the salmonid life stage affected by the relationship with the wildlife focal species.

Table 2-11. Likelihood of focal species utilization within various lower Columbia River mainstem and estuary habitat types.

				Riverine/Estuarine Habitat			Transition Habitat			Upland Habitat		
				Estuary Habitat Classification (Thomas 1983, Johnson et al. 2003b)					WDFW Priority Habitat Classification			
				Deep Water	Medium Depth Water	Tidal Flats	Tidal Marsh	Tidal Swamp	Riparian	Old Growth/ Mature Forest (see Note below)	Freshwater Wetland (i.e. isolated from river corridor)	Rural Natural Open Space
				Percent Habitat Change from 1870 to 1983 (Thomas 1983, Johnson et al. 2003b)								
Species	Primary Life Stage	Level of Use	Primary Season of Use	-13	-19	+10	-49	-74	-	-	-	-
Ocean-type salmonid[a]	Subyearling Juveniles	Migratory	Spring-Fall	◒	◕	◕	●	●	●	○	○	○
Stream-type salmonid[a]	Yearling Smolt	Migratory	Summer	◕	◕	◕	◕	◕	◕	○	○	○
Pacific Lamprey[b]	Ammocoetes or Macrothalmia	Migratory or Resident	Potentially Year-round	◕	◕	◔	◕	◕	◕	○	○	○
White Sturgeon[c]	Juveniles and Adults	Migratory or Resident	Year-round	●	◒	◒	◔	◔	◔	○	○	○
Northern Pikeminnow[d]	Juveniles and Adults	Migratory or Resident	Year-round	◒	◕	◒	◕	◕	◕	○	○	○
River Otter[e]	Juveniles and Adults	Resident	Year-round	◒	◒	◒	◕	◕	◕	◔	◕	◒
Caspian Tern[f]	Juveniles and Adults	Resident	Spring to Fall	◕	◕	◕	◔	◔	◔	○	○	◒
Bald Eagle/ Osprey[g]	Juveniles and Adults	Resident	Spring to Fall	◒	◒	◒	◔	◕	◕	◕	◒	◔
Yellow Warbler[h]	Juveniles and Adults	Resident	Spring to Fall	○	○	○	◔	●	●	◔	◕	◔
Red-eyed Vireo[i]	Juveniles and Adults	Resident	Spring to Fall	○	○	○	◔	●	●	◒	◕	◔
Sandhill Crane[j]	Juveniles and Adults	Resident	Winter	○	○	◒	◕	○	◕	○	◕	◕
Columbian White-tailed Deer[k]	Juveniles and Adults	Resident	Year-round	○	○	○	◒	◕	◕	◕	◒	◕

Note: Use of multiple habitat classification systems is problematic; considerable overlap occurs between habitat designations in different classifications. The habitat types used in the comparison of current and historical habitat conditions (Johnson et al. 2003b) are very general and are not intended to fully describe the vegetation components of the habitat. The WDFW Priority Habitats may be general or specific, depending on the category. For example, old growth/mature forests are described by specific tree diversity, density, and canopy layers but have no elevation specifications. Therefore, old growth forests could be a subset of tidal swamps or part of the upland region. In fact, the 74% loss of tidal swamp habitat may have consisted primarily of old growth tidal swamps and the importance of old growth habitats in the lower mainstem and estuary should not be underestimated. On the other hand, the WDFW riparian habitat category is very general and may encompass habitats categorized as tidal marsh or tidal swamp. Finally, use of the word "tidal" implies some influence of inflowing saltwater on the lower Columbia River mainstem and estuary habitats. In the Columbia River, the influence is generally realized as fluctuating water levels and not as substantial changes in salinity levels over the tidal cycle; many tidal areas in the lower Columbia River remain dominated by freshwater. In general, salinity can have an over-riding influence on estuary and mainstem habitats as it controls plant and animal species assemblages that occur in specific areas because most species have very specific salinity tolerance.

Qualitative Scale of Habitat Use:

- ● Critical
- ◕ High
- ◒ Medium
- ◔ Low
- ○ None

[a] Estuary habitats are utilized primarily by outmigrating juvenile salmonids, except for cutthroat trout that have been observed to occupy estuarine and tidewater habitats for the entire ocean residence period. The importance of the estuary and mainstem littoral habitats varies and is roughly equivalent to the amount of time each species utilizes the estuary and lower mainstem. Generally, salmonids that emigrate as fry or sub-yearlings (i.e. ocean-type chinook and chum salmon) use the estuary extensively for rearing, while salmonids that emigrate as yearlings spend less time in the estuary.
[b] Pacific lamprey do not feed during the transformation from ammocoetes to macrothalmia, which occurs around the time of migration from freshwater to saltwater. Although little is known about Pacific lamprey utilization of estuary or lower mainstem habitats, lampreys are not expected to spend much time in the lower mainstem or estuary.
[c] White sturgeon have been observed congregating in the Columbia River estuary during summer, presumably in relation to food availability. However, white sturgeon are likely present in the lower mainstem and estuary throughout much of the year. Estuary and lower mainstem habitat usage likely varies by age, with younger fish using nearshore or medium depth habitats and adults using deepwater habitats.
[d] Northern pikeminnow are freshwater species and are not known to use estuarine habitats. Northern pikeminnow are warm water species that inhabit the medium and deep water habitats of the Columbia River mainstem.
[e] River otter juveniles and adults are closely associated with aquatic habitats; pups are usually born in a subterranean burrow and begin to swim at about 2 months. River otters feed in water and on land; otters have been observed traveling long distances over land.
[f] Caspian terns can nest in a variety of substrates among an assortment of vegetation types; nests are commonly on sandy substrates in close proximity to abundant fish resources. Breeding Caspian terns almost exclusively eat fish; feeding occurs in near-shore and mid-channel habitats.
[g] Osprey may be found in various estuary and lower mainstem habitats. Presence is most likely in tidal swamps or riparian areas where adequate nest sites exist in proximity to aquatic habitats where fish/birds are abundant and available for consumption.
[h] Possible breeding evidence of yellow warblers has been documented in the Columbia River estuary and along the lower mainstem. If present, yellow warblers would most likely be found in tidal swamp, riparian, or freshwater wetland habitats because they are a riparian obligate species most strongly associated with wetlands that contain Douglas spirea and deciduous tree cover.
[i] Red-eyed vireos are relatively abundant in the Puget Sound and northeast Washington; there has been no confirmed breeding in the Columbia River estuary while possible breeding evidence has been documented along the mainstem near Bonneville. If present, red-eyed vireos would most likely be found in tidal swamp, riparian, or freshwater wetland habitats where woody species satisfy the canopy height and density requirements.
[j] The Columbia River estuary and lower mainstem is generally a migratory stop for sandhill cranes that breed in the Central Valley of California; up to 1,000 sandhill cranes have wintered on lower Columbia River bottomlands in recent years.
[k] Columbian white-tailed deer are generally associated with riparian and wetland habitats; their strongest habitat association is with oak and Douglas fir forest in close proximity to a stream or river.

Table 2-12. Wildlife focal species habitat associations in the Columbia Estuary and Columbia Lower Subbasins (IBIS 2003).

Focal Species	Wildlife-Habitat Type	Association Type	Activity Type	Confidence Level	Comments
Columbian White-tailed Deer	Mesic Lowlands Conifer-Hardwood Forest	Generally Associated	Feeds and Breeds	High	none
	Westside Oak and Dry Douglas-fir Forest and Woodlands	Closely Associated	Feeds and Breeds	High	Strong association with oak within 200 meters of a stream or river.
	Agriculture, Pastures, and Mixed Environs	Generally Associated	Feeds and Breeds	High	none
	Urban and Mixed Environs	Generally Associated	Feeds and Breeds	High	none
	Herbaceous Wetlands	Generally Associated	Feeds	High	none
	Westside Riparian-Wetlands	Generally Associated	Feeds and Breeds	High	none
Caspian Tern	Open Water - Lakes, Rivers, and Streams	Closely Associated	Feeds and Breeds	High	Nests on sandbars and dredge spoil islands within rivers.
	Herbaceous Wetlands	Closely Associated	Feeds	High	none
	Coastal Dunes and Beaches	Closely Associated	Other (see comments)	High	O = roosting/resting.
	Coastal Headlands and Islets	Generally Associated	Other (see comments)	High	O = roosting/resting.
	Bays and Estuaries	Closely Associated	Feeds	High	none
	Marine Nearshore	Closely Associated	Feeds	High	none
Bald Eagle	Mesic Lowlands Conifer-Hardwood Forest	Generally Associated	Reproduces	High	Could breed in this habitat where near open water habitats.
	Westside Oak and Dry Douglas-fir Forest and Woodlands	Generally Associated	Reproduces	High	Could breed in this habitat where near open water habitats.
	Montane Mixed Conifer Forest	Generally Associated	Reproduces	High	Could breed in this habitat where near open water habitats.
	Agriculture, Pastures, and Mixed Environs	Generally Associated	Feeds	High	none

	Urban and Mixed Environs	Generally Associated	Feeds and Breeds	High	Could breed in this habitat where near open water habitats, and if suitable nest structures are available.
	Open Water - Lakes, Rivers, and Streams	Closely Associated	Feeds	High	none
	Herbaceous Wetlands	Generally Associated	Feeds	High	none
	Westside Riparian-Wetlands	Generally Associated	Feeds and Breeds	High	none
	Coastal Dunes and Beaches	Present	Feeds	High	none
	Coastal Headlands and Islets	Generally Associated	Feeds and Breeds	High	none
	Bays and Estuaries	Generally Associated	Feeds and Breeds	High	Requires some sort of structure to place nest on, such as old pilings, if breeding is to occur in this habitat.
	Marine Nearshore	Generally Associated	Feeds	High	none
Osprey	Mesic Lowlands Conifer-Hardwood Forest	Generally Associated	Reproduces	High	Could breed in this habitat where near open water habitats.
	Westside Oak and Dry Douglas-fir Forest and Woodlands	Generally Associated	Reproduces	High	Could breed in this habitat where near open water habitats.
	Montane Mixed Conifer Forest	Generally Associated	Reproduces	High	Could breed in this habitat where near open water habitats.
	Agriculture, Pastures, and Mixed Environs	Present	Reproduces	High	Could breed in this habitat where near open water habitats, and if suitable nest structures are available.
	Urban and Mixed Environs	Generally Associated	Reproduces	High	Could breed in this habitat where near open water habitats, and if suitable nest structures are available.
	Open Water - Lakes, Rivers, and Streams	Closely Associated	Feeds	High	none
	Westside Riparian-Wetlands	Generally Associated	Feeds and Breeds	High	none
	Coastal Headlands and Islets	Generally Associated	Reproduces	Moderate	none
	Bays and Estuaries	Generally Associated	Feeds and Breeds	Moderate	Requires some sort of structure to place nest on, such as old pilings, if breeding is to occur in this habitat.
	Marine Nearshore	Generally Associated	Feeds	Moderate	none

River Otter	Urban and Mixed Environs	Present	Feeds	High	Might be found in marinas.
	Open Water - Lakes, Rivers, and Streams	Closely Associated	Feeds and Breeds	High	Dens placed in banks.
	Herbaceous Wetlands	Closely Associated	Feeds and Breeds	High	none
	Westside Riparian-Wetlands	Closely Associated	Feeds and Breeds	High	none
	Coastal Dunes and Beaches	Generally Associated	Feeds and Breeds	Moderate	Uses this habitat in the Puget Sound, Hood Canal, etc., but not likely to use outer coast beaches.
	Coastal Headlands and Islets	Present	Feeds and Breeds	High	Only where this habitat is near estuaries, coastal bogs, or along the Puget Sound and Strait of Juan de Fuca. Not likely on the outer coast.
	Bays and Estuaries	Closely Associated	Feeds and Breeds	High	none
	Marine Nearshore	Generally Associated	Feeds	High	Puget Sound, Hood Canal etc. only, not outer coast.
Sandhill Crane	Agriculture, Pastures, and Mixed Environs	Closely Associated	Feeds and Breeds	High	Also includes staging areas; must have roosting areas within the range.
	Herbaceous Wetlands	Closely Associated	Feeds and Breeds	High	none
Yellow Warbler	Westside Riparian-Wetlands	Closely Associated	Feeds and Breeds	High	none
Red-eyed Vireo	Mesic Lowlands Conifer-Hardwood Forest	Present	Feeds and Breeds	Moderate	Requires a hardwood component.
	Westside Riparian-Wetlands	Closely Associated	Feeds and Breeds	Moderate	Range of red-eyed vireo overlaps that of large black cottonwood groves.

Table 2-13. Focal species relationship to salmonids (IBIS 2003).

Common Name	Relationship Type	Salmonid Stage	Comments
Caspian Tern	Strong, consistent	Saltwater - smolts, immature adults, and adults	none
	Strong, consistent	Freshwater rearing - fry, fingerling, and parr	none
Bald Eagle	Indirect	Incubation - eggs and alevin	Feed on birds that feed on salmon.
	Indirect	Freshwater rearing - fry, fingerling, and parr	Feed on birds that feed on salmon.
	Strong, consistent	Carcasses	none
	Indirect	Saltwater - smolts, immature adults, and adults	Feed on birds that feed on salmon.
	Strong, consistent	Spawning - freshwater	none
	Indirect	Carcasses	Feed on birds that feed on salmon.
	Strong, consistent	Saltwater - smolts, immature adults, and adults	none
Osprey	Strong, consistent	Saltwater - smolts, immature adults, and adults	none
	Strong, consistent	Spawning - freshwater	none
	Strong, consistent	Freshwater rearing - fry, fingerling, and parr	none
River Otter	Strong, consistent	Freshwater rearing - fry, fingerling, and parr	none
	Strong, consistent	Spawning - freshwater	none
	Strong, consistent	Carcasses	none

2.4.2 Salmonids

Estuaries are important for many species, particularly anadromous salmonids. For example, anadromous salmonids that survive to reproduce migrate through the estuary at least twice during their life cycle; the estuary serves as a vital transition zone during the physiological acclimation from freshwater to saltwater (Simenstad et al. 1994b, Thorpe 1994 as cited in Bottom et al. 2001). Further, estuaries provide juvenile salmonids an opportunity to achieve the critical growth necessary to survive in the ocean (Neilson and Geen 1986, Wissmar and Simenstad 1988 as cited in Nez Perce et al. 1995); estuarine habitats serve as a productive feeding area, free of marine predators.

Many studies indicate that estuarine conditions are important in salmonid survival rates; however, to date researchers have not been able to specifically agree on what attributes of the estuary confer enhanced survival to salmon. Certain general physical and biological functions performed by estuaries, however, can be assumed to have direct impacts on salmon as they transition from their natal river basins to seawater.

2.4.2.1 Conceptual Models

The natural forces of ocean tides and river flows have been influenced by anthropogenic factors. The basic habitat-forming processes (i.e. physical forces of the ocean and river) create the conditions that define the estuarine and mainstem freshwater habitats. The created habitat types provide an opportunity for the primary plant production that serves as the base of complex food webs. All of these pathways combine to influence the growth, survival, and, eventually, the production of juvenile salmonids moving through the lower Columbia River (USACE 2001). These processes and pathways are generally described in the juvenile salmonid production conceptual model as illustrated below (Figure 2-27 and Table 2-14). The conceptual model was developed to describe juvenile salmonid production in the Columbia River estuary; it does not address the premise that population structure and life history diversity may be equally as important in determining salmonid survival. Further, although the conceptual model was developed with an ecosystem focus, it needs to be scrutinized to determine applicability to other tidal freshwater and estuary species. The foundational basis for a wildlife species conceptual model has been developed by Johnson and O'Neil (2001).

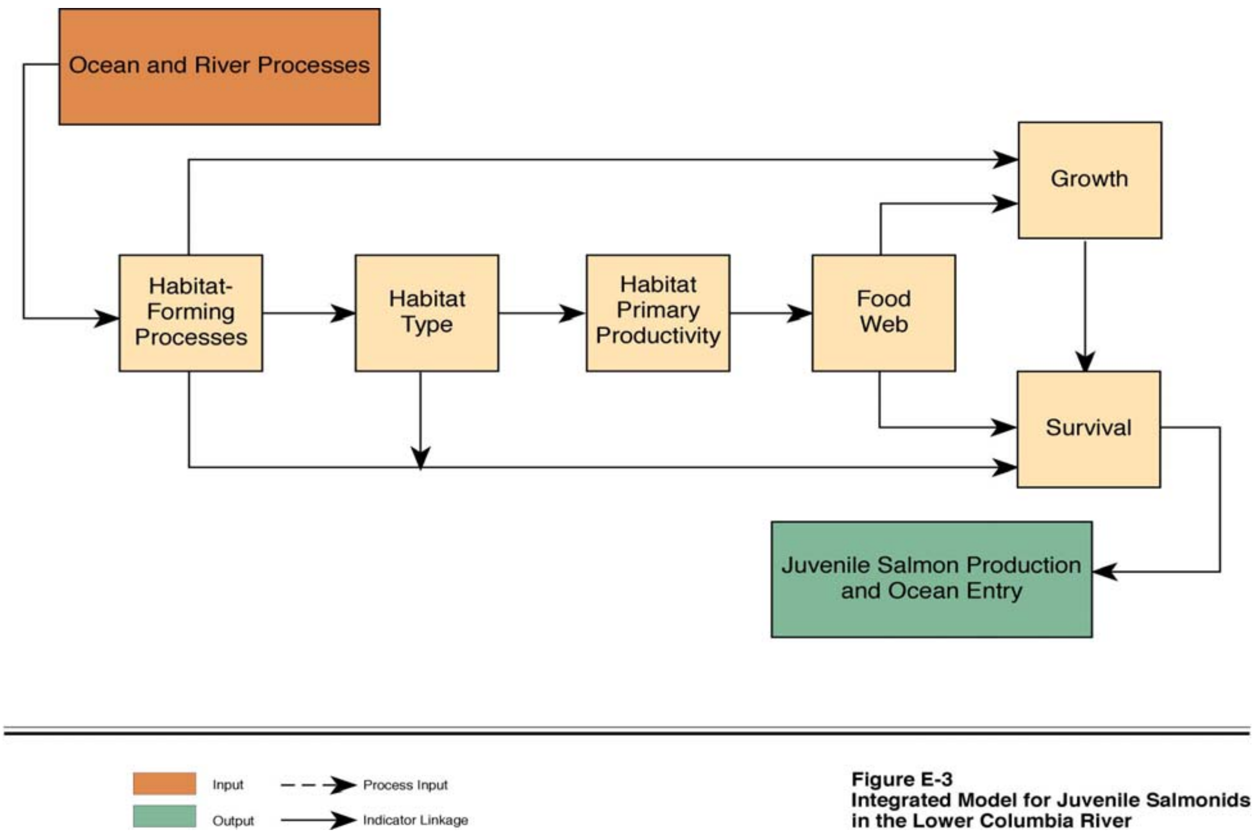


Figure 2-27. Conceptual model of the major components affecting juvenile salmonid production in the Columbia River estuary (USACE 2001).

Table 2-14. Conceptual model pathways and components for juvenile salmonid production in the Columbia River estuary (USACE 2001).

Model Pathways	Pathway Description	Model Components	Component Description
Habitat-Forming Processes	Physical processes that define the living conditions and provide the requirements fish naturally need within the river system are included in the Habitat-Forming Processes Pathway.	Suspended Sediment	Sand, silt, and clay transported in the water column
		Bedload	Sand grains rolling along the surface of the riverbed
		Woody Debris	Downed trees, logs, root wads, limbs
		Turbidity	Quality of opacity in water, influenced by suspended solids and phytoplankton
		Salinity	Saltwater introduced into freshwater areas through tidal ocean process
		Accretion/ Erosion	Deposited/carved sediments
		Bathymetry	Topographic configuration of the riverbed
Habitat Types	This pathway describes definable areas that provide the living requirements for fish in the Lower Columbia River.	Tidal Marsh and Swamp	Areas between mean lower low water (MLLW) and mean higher high water (MHHW) dominated by emergent vegetation (marsh) and low shrubs (swamp) in estuarine and riverine areas.
		Shallow Water and Flats	Areas between 6-foot bathymetric line (depth) and MLLW
		Water Column	Areas in the river where depth is greater than 6feet
Habitat Primary Productivity	This pathway describes the biological mass of plant materials that provides the fundamental nutritional base for animals in the river system.	Light	Sunlight necessary for plant growth
		Nutrients	Inorganic source materials necessary for plant growth
		Imported Phytoplankton Production	Material from single-celled plants produced upstream above the dams and carried into lower reaches of the river
		Resident Phytoplankton Production	Material from single-celled plants produced in the lower reaches of the river
		Benthic Algae Production	Material from simple plant species that inhabit the river bottom
		Tidal Marsh and Swamp Production	Material from complex wetland plants (hydrophytes) present in tidal marshes and swamps

Food Web	The Food Web pathway shows the aquatic organisms and related links in a food web that supports growth and survival of salmonids.	Deposit Feeders	Benthic organisms such as annelid worms that feed on sediments, specifically organic material and detritus
		Mobile Macroinvertebrates	Large epibenthic organisms such as sand shrimp, crayfish, and crabs that reside and feed on sediments at the bottom of the river
		Insects	Organisms such as aphids and flies that feed on vegetation in freshwater wetlands, tidal marshes, and swamps
		Suspension/Deposit Feeders	Benthic and epibenthic organisms such as bivalves and some amphipods that feed on or at the interface between sediment and the water column
		Suspension Feeders	Organisms that feed from the water column itself, including zooplankton
		Tidal Marsh Macrodetritus	Dead and decaying remains of tidal marsh and tidal swamp areas that are an important food source for benthic communities
		Resident Microdetritus	Dead and decaying remains of resident phytoplankton and benthic algae, an important food source for zooplankton
		Imported Microdetritus	Dead remains of phytoplankton from upstream that serve as a food source for suspension and deposit feeders
Growth	The Growth Pathway highlights the factors involved in producing both the amount of food and access by fish to productive feeding areas.	Habitat Complexity, Connectivity, and Conveyance	Configuration of habitat mosaics that allow for movement of salmonids between those habitats
		Velocity Field	Areas of similar flow velocity within the river
		Bathymetry and Turbidity	River bottom and water clarity conditions that influence the ability of salmonids to locate their prey
		Feeding Habitat Opportunity	Physical characteristics that affect access to locations that are important for fish feeding
		Refugia	Shallow water and other low energy habitat areas used for resting and cover
		Habitat-Specific Food Availability	Ability of complex habitats to provide feeding opportunities when fish are present

Survival	The Survival Pathway is a summary of key factors controlling or affecting growth and migration.	Contaminants	Compounds that are environmentally persistent and bioaccumulative in fish and invertebrates
		Disease	Pathogens (viruses, bacteria, and parasites) that pose survival risks for salmon
		Suspended Solids	Sand, silt, clay, and organics transported within the water column
		Stranding	Trapping of young salmonids in areas with no connectivity to water column habitat
		Temperature and Salinity Extremes	Temperature or salinity conditions that are problematic to salmonid survival
		Turbidity	Water clarity as it pertains to potential for juvenile salmonids to be seen by predators
		Predation	Potential for piscivorous mammals, birds, and fish to prey on salmonids
		Entrainment	Trapping of fish or invertebrates into hopper or pipeline dredges

The general conceptual model has been separated into component parts. The first figure in the series (Figure 2-28) is the juvenile salmon growth and survival conceptual model developed by Bottom et al. (2001); this conceptual model incorporates the premise that salmonid population structure and life history diversity plays an important role in salmonid survival. The next series of diagrams (Figure 2-29, Figure 2-30, and Figure 2-31) describe a conceptual model for juvenile salmonid production in the lower Columbia River and estuary detailed in USACE (2001); the conceptual model represents a 6 month collaborative effort by the USACE, Battelle Marine Science Laboratories, Parametrix, Inc., the Port of Portland, NMFS, USFWS, Limno-Tech, Inc., University of Washington, and the Sustainable Ecosystems Institute Science Panel. This conceptual model presents some of the same concepts as the Bottom et al. (2001) model. As previously mentioned, the conceptual model presented here was developed specifically for juvenile salmon production; the model needs to be scrutinized for applicability to other focal species. In regards to wildlife focal species, Johnson and O'Neil (2001) have explored possible components that would serve as a foundation for wildlife conceptual models, although such models have not been iteratively developed.

As described in the overall conceptual model for juvenile salmon production in the estuary (Figure 2-27), the type of available habitat determines the food web, which then drives salmon growth, survival, and ultimate production from the estuary. Within the food web, the available habitat determines the amount and type of primary productivity and hence, the base of the food web (Figure 2-29). In turn, the food web base determines the amount and type of prey species available to juvenile salmonids and therefore influences growth and survival (Figure 2-29). Salmonid growth is also influenced by habitat-forming processes and the types of habitat available as these provide refuge and affect each individuals energy costs (Figure 2-30). Growth is also affected by temperature and other compounding factors such as hatchery practices that may result in density-dependent competition as a result of large releases of hatchery fish (Figure 2-28). Finally, all of the base components of the conceptual model (i.e. habitat-forming processes, habitat type, food web, and growth) in conjunction with the physiological condition and adaptive behaviors of juvenile salmonids determines the ultimate production from the estuary (Figure 2-31).

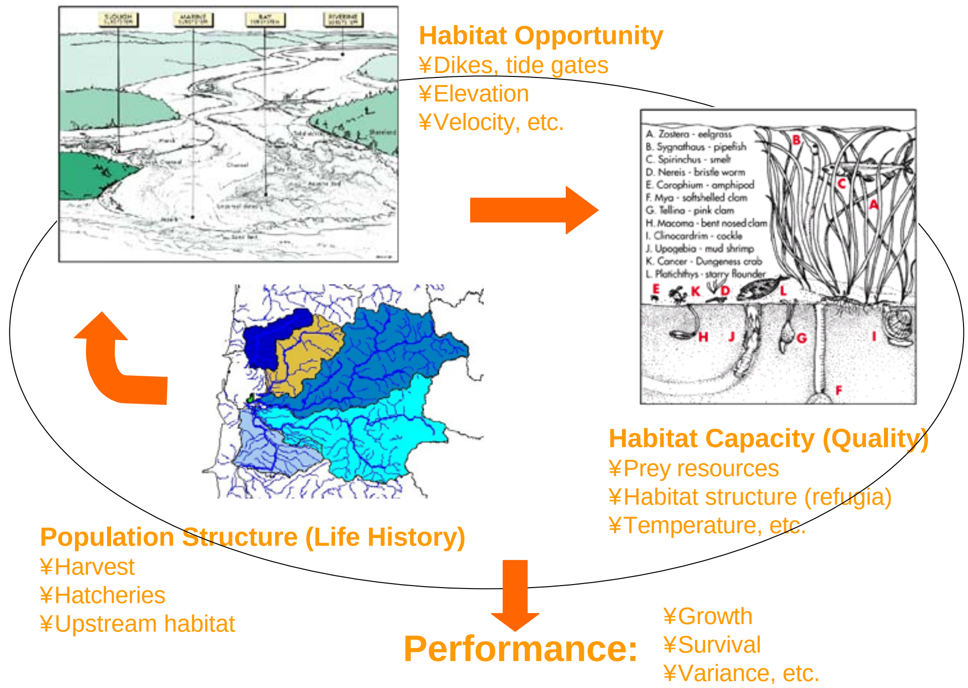


Figure 2-28. Conceptual model for juvenile salmon growth and survival in the Columbia River estuary (from Bottom et al. 2001).

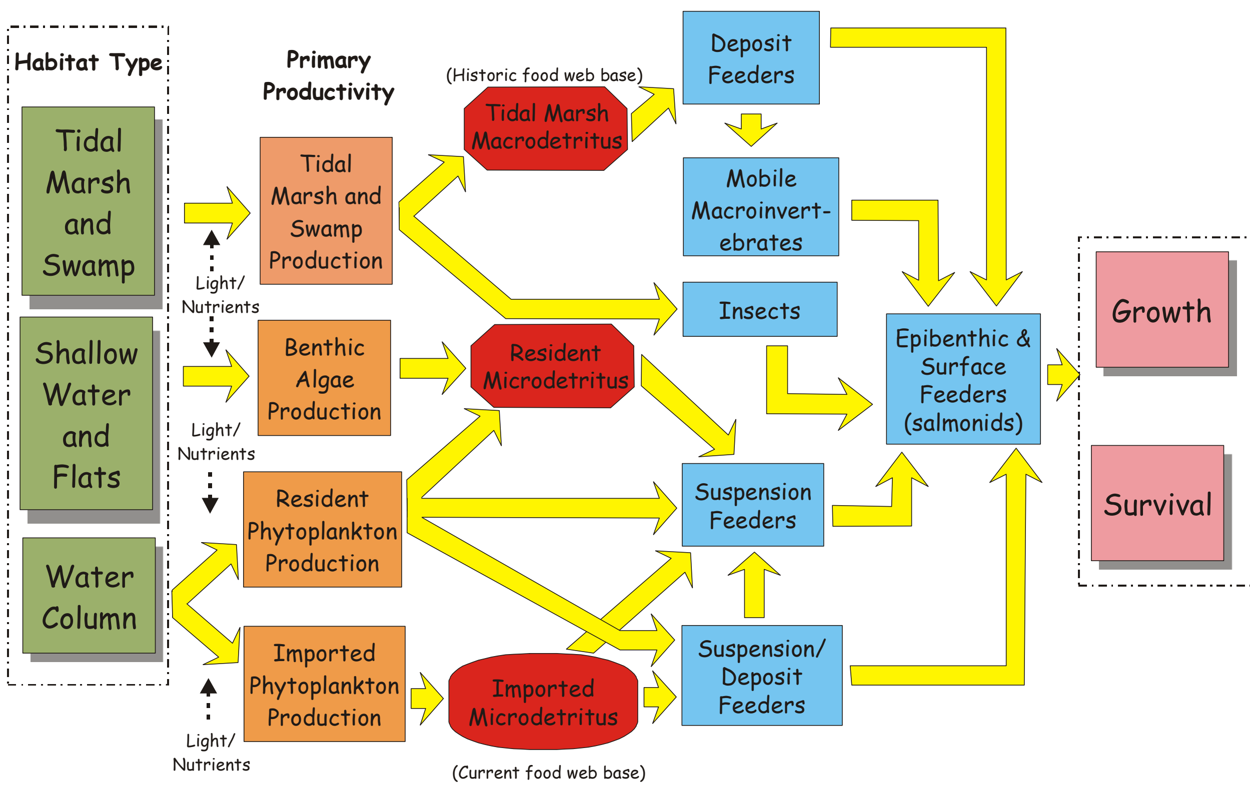


Figure 2-29. Conceptual model of the Columbia River estuary food web (adapted from USACE 2001).

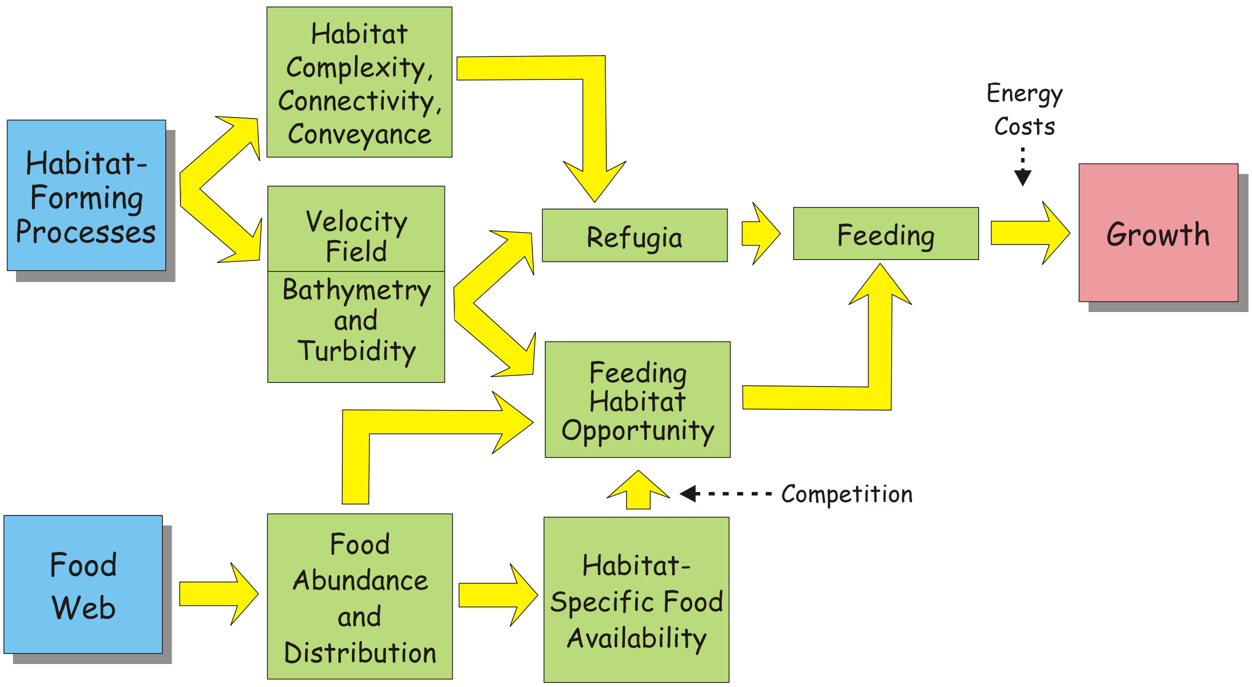


Figure 2-30. Conceptual model of juvenile salmonid growth in the Columbia River estuary (adapted from USACE 2001).

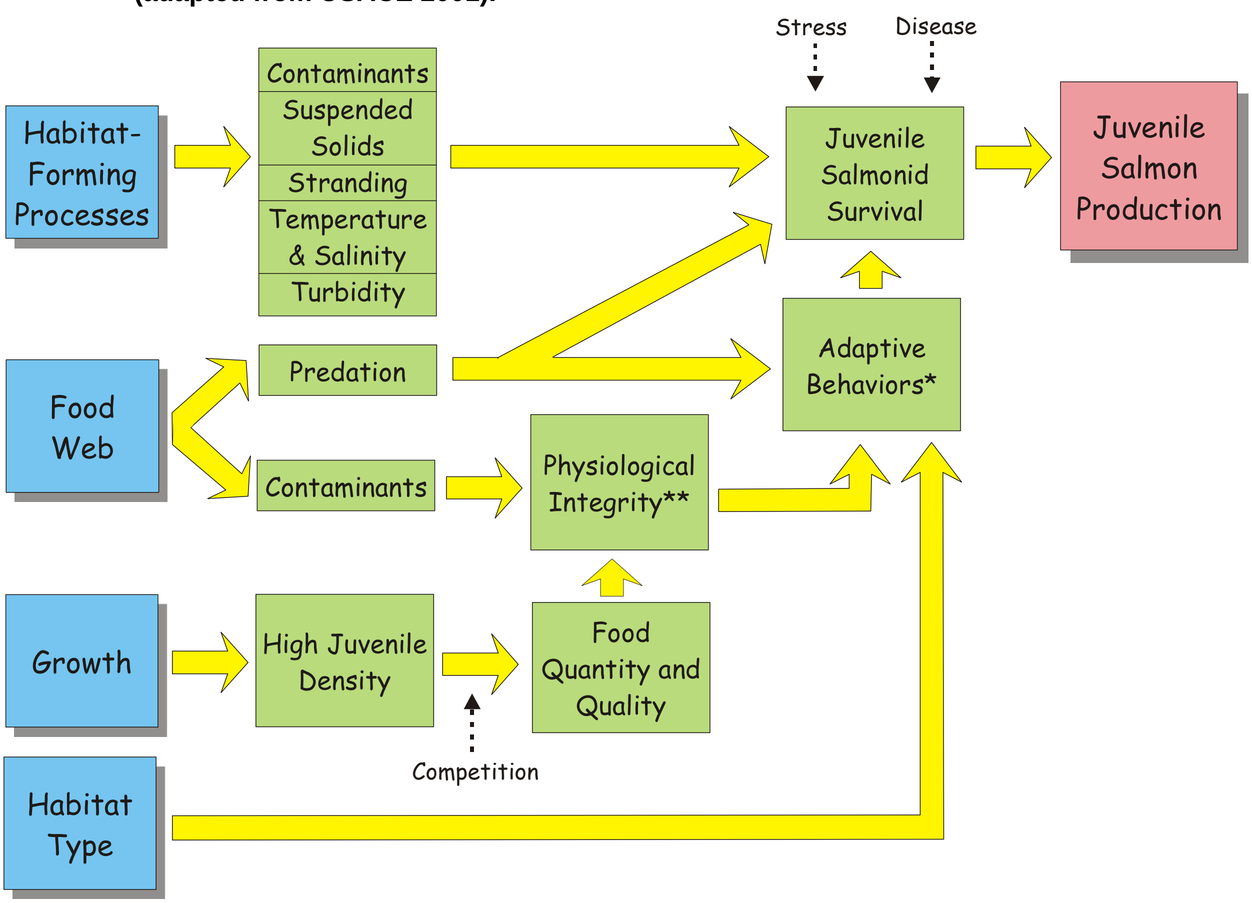


Figure 2-31. Conceptual model of juvenile salmonid production from the Columbia River estuary (adapted from USACE 2001).

2.4.2.2 Habitat Requirements

The freshwater habitat requirements for juvenile salmonids are well studied and understood; however, the estuarine habitat requirements of juvenile salmonids are just now coming into focus. In describing estuarine habitat requirements, juvenile salmonids are divided into two primary life-history types: ocean- and stream-type. Johnson et al. (2003b) recently presented the current understanding of the estuarine physical habitat requirements and threshold levels of juvenile salmon (Table 2-15). Note that the studies related to salinity, water temperature, and turbidity addressed threshold levels of exposure in which salmonids could survive. Although threshold levels may be similar among ocean- and stream-type salmonids, there is not complete agreement among researchers on the preferred ranges of these parameters for different salmonids.

Table 2-15. Physical habitat requirements and threshold levels of juvenile salmonids in relation to various habitat parameters (adapted from Weitkamp et al. 2001, as cited in Johnson et al. 2003b).

Parameter	Ocean-Type	Stream-Type
Water Depth	Surface waters	Surface to 6 m
Currents	Less than ~9 cm/second Less than 30 cm/second for current threshold modeling (1)	Found throughout a wide range of current velocities and tend to avoid low-velocity areas
Substrate	Varies (mostly sand/silt)	Varies, but tends to be associated with the water column to a greater extent than with substrate type
Salinity	Upon hatching 15-20 ppt (2) Juveniles 30 ppt seawater (3) Chinook fry of 1.5 gram could survive and grow in seawater (4)	Generally same as ocean-type
Water Temperature	Can tolerate brief periods of 15-20˚C; Lethal at approximately 22˚C (5) Sub-lethal effects can occur below lethal threshold, but vary.	Coho preferred range of 12-14˚C; Upper lethal temperature was 25˚C (6)
Turbidity	LC_{50} for coho (summer conditions) 1.2 g/l (7) LC_{50} for chum (summer conditions) 2.5 g/l (8)	Generally same as ocean-type

(1) Bottom et al. 2001, (2) Wagner et al. 1969, (3) Tiffan et al. 2000, (4) Clark and Shelbourn 1985, (5) Brett 1971, Lee and Rinne 1980, (6) Brett 1952, (7) Noggle 1978, (8) Smith 1978.

2.4.2.3 Habitat Utilization

Juvenile salmon may be found in the Columbia River estuary at all times of the year, as different species, life history strategies and size classes continually move into tidal waters; Wissmar and Simenstad (1998) estimated that there may be as many as 35 potential life history strategies for ocean-type chinook. Peak estuary entrance varies among and within species. Rich (1920) reported that juvenile migration span of any given chinook brood in the Columbia River basin is around 18 months – from fry that emigrate to the estuary soon after emerging in December to yearlings that do not leave until late their second spring. Myers and Horton (1982) have suggested that differentiation of life history forms may be a mechanism for partitioning limited estuarine habitats. Duration of estuarine residence varies from species to species. Coho, stream-type chinook salmon, steelhead, and anadromous cutthroat trout typically rear in fresh water for a year or more, and move rapidly through the estuary on their way seaward (<6 weeks). Chum and ocean-type chinook salmon make greater use of the estuary. Chum salmon typically live in the estuary for several weeks, and ocean type chinook that migrate to the estuary as fry can reside in estuaries for up to 2 months or more (Healey 1982).

Estuaries have important impacts on juvenile salmonid survival. Estuaries provide juvenile salmonids an opportunity to achieve the critical growth necessary to survive in the ocean (Neilson and Geen 1986, Wissmar and Simenstad 1988 as cited in Nez Perce et al. 1995). Estuarine habitats provide young salmonids with a productive feeding area, free of marine predators, where smolts can undergo physiological changes necessary to acclimate to the saltwater environment. Studies conducted by Emmett and Schiewe (1997) in the early 1980s have shown that favorable estuarine conditions translate into higher salmonid survival. During this research, juvenile coho and chinook smolts were collected and released in the river, in the estuary, in the transition zone outside the estuary, and in the ocean; efforts were replicated over multiple years. In both coho and chinook, smolts released in the estuaries consistently contributed in higher rates to commercial fisheries or returned at higher rates than smolts released outside the estuaries.

Studies show that habitat use of juvenile salmon within the estuary environment is size related. Fry less than 60 mm usually occupy shallow, protected habitats such as salt marshes, tidal creeks and intertidal flats (Levy and Northcote 1982, Myers and Horton 1982, Simenstad et al. 1982, Levings et al. 1986 as cited in Bottom et al. 2001). Fish 60-100 mm move to slightly deeper shoals and channels further from the shoreline (Healey 1982, 1991, Myers and Horton 1982, as cited in Bottom et al. 2001). Fish greater than 100 mm can be found in both deep and shallow water habitats. (Levy and Northcote 1982, Myers and Horton 1982, Simenstad et al. 1982, Bottom et al. 1984). These generalizations hold true more during the day than at night, when schooling fry or fingerlings may be seen venturing into deeper waters (Schreiner 1977, Kjelson 1982, Bax 1983, Healey 1991, Salo 1991, as cited in Bottom et al. 2001). Moreover, salmonids must continually adjust their habitat distribution in relation to twice-daily tidal fluctuations and seasonal and anthropogenic variations in river flow. Juveniles have been observed to move from low-tide refuge areas in deeper channels to salt marsh habitats at high tide and back again (Healey 1982). These patterns of movement suggest that access to suitable low-tide refuge near marsh habitat may be an important factor in production and survival of salmonid juveniles in the Columbia River estuary.

2.4.2.4 Habitat Availability

Using a model that incorporated predevelopment and current river flows and estuarine bathymetry, the habitat availability for juvenile salmonids in the historical and present estuary were simulated and compared. Based on the velocity criteria ($< 0.3 m\ s^{-1}$), habitat opportunity has declined in the upriver tidal freshwater mainstem and the upper estuary peripheral bays (Cathlamet and Grays) while it has not changed substantially in the lower regions of the estuary (lower estuary mainstem and lower estuary peripheral bays [Youngs and Baker]). Based on the depth criteria (0.1 to 2 m), habitat opportunity has increased in all regions compared to historical conditions, except in the upriver tidal freshwater mainstem. However, limitations in the representation of historical and modern bathymetry in the model limit confidence in the depth criteria results and prevent comparison of historical and current habitat opportunity based on the combined depth and velocity criteria. Despite model limitations, results indicate that the availability of suitable juvenile salmonid estuary habitat varies in response primarily to bathymetry, but also to river discharges and tides. Also, seasonal and inter-annual variability is important particularly in upper regions of the estuary where habitat opportunity is reduced during freshet flows (Bottom et al. 2001).

Small changes in salinity distribution may have significant effects on the ecology of fishes in the estuary, including salmonids. Salinity distribution, as affected by tidal flow and river discharge, determines the location of the ETM; salinity and the ETM are primary factors explaining seasonal species distributions and the structure of entire fish, epibenthic, and benthic invertebrate prey species assemblages throughout the Columbia River estuary (Haertel et al. 1969, Bottom and Jones 1990, Jones et al. 1990 as cited in NMFS 2000c and Simenstad et al. 1994b as cited in USACE 2001). By altering the distribution of preferred habitats within specific salinity ranges and the particular array of species that salmon encounter at different locations during their estuarine residence, small changes in salinity structure may have significant effects on the estuarine food web and fish production in the estuary. In particular, small changes in the distribution and gradient of oligohaline salinities may change the type of habitats available when juvenile salmon make the critical physiological transition from fresh to brackish water (NMFS 2000c).

2.4.2.5 Habitat Connectivity

Within the estuary, rapid changes in salinity gradients, water depths, and habitat accessibility impose important energetic and ecological constraints that salmonids do not encounter in freshwater (Bottom et al. 2001). Areas of adjacent habitat types distributed across the estuarine salinity gradient may be necessary to support annual migrations of juvenile salmonids (Simenstad et al. *in press* as cited in Bottom et al. 2001). As subyearlings grow, they move across a spectrum of salinities, depths, and water velocities. For species like chum and ocean type chinook salmon that rear in the estuary for extended time periods, a broad range of habitat types in the proper proximities to one another may be necessary to satisfy feeding and refuge requirements within each salinity zone. If juvenile salmonid life cycles require specific spatial and temporal sequences, then areas suitable for supporting salmonids may remain unused if their connectivity with other habitats is lost (Bottom et al. 1998). That is, the connectedness of these habitats likely determines whether juvenile salmonids are able to access the full spectrum of habitats they require.

Juxtaposition of high-energy areas with ample food availability and sufficient refuge habitat is a key habitat structure necessary for high salmonid production in the estuary. In particular, tidal marsh habitats, tidal creeks and associated complex dendritic channel networks may be especially important to subyearlings as areas of both high insect prey density, and as potential refuge from predators afforded by sinuous channels, overhanging vegetation and undercut banks (McIvor and Odum 1988). Salmonid production in estuaries is supported by detrital food chains (Healey 1979, 1982). Therefore habitats that produce and/or retain detritus, such as tidal wetlands emergent vegetation, eelgrass beds, macro algae beds and epibenthic algae beds, are particularly important (Sherwood et al. 1990). Historically, before the Columbia River was isolated from its floodplain, influx of organic matter occurred regularly during spring freshets.

The importance of proximate availability of feeding and refuge areas may hold true even for species that move more quickly through the estuary. For example, radio tagged coho in Grays Harbor estuary moved alternatively from low velocity holding habitats to strong current passive downstream movement areas (Moser et al. 1991). Consistent with these observations, Dittman et al. (1996) suggest that habitat sequences at the landscape level may be important even for species and life history types that move quickly through the estuary during the important smoltification process, as salmon gather the olfactory cues needed for successful homing and these cues may depend on the environmental gradients experienced during migrations.

2.4.2.6 Habitat Capacity

Diking and filling activities in the estuary have likely reduced the rearing capacity for fry and sub-yearling life stages by decreasing the tidal prism and eliminating emergent and forested wetlands and floodplain habitats adjacent to shore (Bottom et al. 2001, NMFS 2000c). Dikes throughout the lower Columbia River and estuary have disconnected the main channel from a significant portion of the wetland and floodplain habitats. Further, filling activities (i.e. for agriculture, development, or dredge material disposal) have eliminated many wetland and floodplain habitats. Because fry and subyearling smolts rely heavily on emergent and forested wetlands and floodplain habitats for food and refugia, reduction of these habitats have reduced the capacity for these salmonid life stages.

Both large woody debris and sand/gravel substrates are important factors defining the quantity and quality of estuarine habitat for salmonids; changes in flow cycles, flow magnitude, and habitat isolation has decreased the availability of these estuarine habitat components to juvenile salmonids (Sherwood et al. 1990 as cited in Nez Perce et al. 1995, NMFS 2000c, Bottom et al. 2001, USACE 2001). *Anecdotal* observations indicate that salmonids congregate near large woody debris; feeding may be enhanced because of the deposition of organic matter and the production of small benthic prey species. Much of the habitat that served as the large woody debris source (i.e. tidal swamps/wetlands, freshwater riparian forests and forested wetlands) has been disconnected from the lower river and estuary via diking and subsequent development. Decreases in flow decreases bedload transport of sand and large woody debris movement; recruitment of these important habitat features to estuary habitats has decreased. Restoration of lost estuary wetland habitat and historical flow patterns may benefit recovery of depressed salmonid stocks (ISAB 2000, NMFS et al. 2000).

2.4.2.7 Migration and Spawning

Hydrologic effects of the Columbia River hydrosystem include water level fluctuations, altered seasonal and daily flow regimes, reduced water velocities, and reduced discharge volume. Altered flow regimes can affect the migratory behavior of juvenile salmonids. For example, water level fluctuations associated with hydropower peak operations may reduce habitat availability, inhibit the establishment of aquatic macrophytes that provide cover for fish, and strand juveniles during the downstream migration. Reservoir drawdowns reduce available habitat which concentrates organisms, potentially increasing predation and disease transmission (Spence et al. 1996 as cited in NMFS 2000c).

Altered flow regimes can affect the spawning success of mainstem Columbia River spawners. For example, reservoir drawdowns in the fall for flood control produces high flow for fall spawners; fish may spawn in areas that are dewatered during the winter or spring, potentially resulting in complete egg mortality (NMFS 2000c).

2.4.2.8 Food Web Structure

There is a general inference today that the capacity of the Columbia River estuary to produce salmonids has decreased from historical levels. Losses of emergent marsh and forested wetland habitats have been substantial, and may be a major factor affecting the capacity of the estuary to support juvenile salmon. Studies show that small subyearling ocean-type chinook salmon occupy shallow habitats and feed extensively on emergent insects. The diet composition and distribution of small juveniles far into shallow tidal channels and sloughs at high tide suggest that these fish are rearing in direct association with vegetated edges of estuarine wetlands (Simenstad et al. 2000). However, habitat alterations such as artificial river confinement and water regulation through hydrosystem operations have restricted access to some productive Columbia River estuary floodplain habitats. The current estuary food web does not support the same diversity of salmon life-history types that occurred historically (Bottom et al. 2001).

Juvenile salmonids are part of a complex food web in the lower Columbia River mainstem and estuary (Figure 2-29; USACE 2001). Plant primary productivity is the base of the food web; plant material can be incorporated into the food web via direct consumption or through decaying material and consumption by detritivores (Jones et al. 1990 as cited in USACE 2001). Salmonids consume prey species supported by resident plant material and resident and imported plankton and detritus. The relative amount of available prey species depends on the abundance of estuary habitat types as well as the input of imported detrital material from upstream sources. The contribution of imported detritus is controlled primarily by reservoir production and flow rates from Bonneville Dam. Subyearling salmonids feed primarily on benthic prey items available in near-shore habitats while yearling salmonids feed primarily on zooplankton available in the water column; larvae and adult floating insects appear to be important prey items for most salmonids. Prey availability and consumption varies with tide stage; prey items inhabiting shallow habitats become more available during high tides while planktonic prey items appear to be equally available at different tide levels. Further, food availability may be negatively affected by the temporal and spatial overlap of juvenile salmonids from different locations; competition for prey may develop when large releases of hatchery salmonids enter the estuary (Bisbal and McConnaha 1998), although this issue remains unresolved (Lichatowich 1993 as cited in Williams et al. 2000).

Estuaries provide juvenile salmonids an opportunity to achieve the critical growth necessary to survive in the ocean (Neilson and Geen 1986, Wissmar and Simenstad 1988 as cited in Nez Perce et al. 1995). Juvenile chinook salmon growth in estuaries is often superior to river-based growth (Rich 1920a, Reimers 1971, Schluchter and Lichatowich 1977). Ability of the Columbia River estuary to support juvenile salmonid growth and maximize survival to the time of ocean entry depends on habitat productivity and access to productive habitats (Figure 2-30 and Figure 2-28; Brodeur et al. 2000, Bottom et al. 2001). The estuarine food web was historically macrodetritus-based because of the abundance of emergent, forested, and other wetland rearing areas throughout the estuary (Figure 2-29; Bottom et al. 2001, USACE 2001, Johnson et al. 2003b); these areas have been largely been lost or disconnected from the river via dike construction and subsequent development. Emergent plant production in the estuary has decreased by 82% and benthic macroalgae production by 15% (Sherwood et al. 1990 as cited in Nez Perce et al. 1995). The loss of wetland habitats combined with the development of reservoirs throughout the Columbia River has shifted the food web to a microdetritus base, primarily in the form of imported phytoplankton production from upriver reservoirs that dies upon exposure to salinity in the estuary (Bottom and Jones 1990 as cited in Nez Perce et al. 1995, Bottom et al. 2001, USACE 2001). Imported phytoplankton are found within the water column and support a pelagic food web that is less accessible to small juvenile salmonids that inhabit the edge habitat throughout the lower Columbia River and estuary (Sherwood et al. 1990 as cited in USACE 2001, USACE 2001).

While the macrodetritus-based food web was historically distributed throughout the lower river and estuary, the modern microdetritus-based food web is focused on the spatially confined turbidity maximum region of the estuary (Bottom et al. 2001). This modified food web benefits exotic species (American shad) and lower estuary forage fish (northern anchovy, Pacific herring, longfin smelt) and is a disadvantage to anadromous salmonids (Bottom and Jones 1990 as cited in Nez Perce et al. 1995, Bottom et al. 2001, USACE 2001). Although these forage fish are found in the diet of larger juvenile salmonids in the lower estuary and nearshore ocean, the presence of these forage fish unlikely satisfies smaller ocean-type salmonid feeding requirements in the estuary (Bottom and Jones 1990 as cited in USACE 2001). Survival tradeoffs between juvenile salmon estuary feeding requirements and estuary food web support of feeding requirements in the lower estuary and nearshore ocean are unknown.

Habitat alterations such as artificial river confinement have restricted access to some productive Columbia River estuary floodplain habitats. Further, water regulation through the hydrosystem operation has decreased seasonal freshet flows. Flow volumes that create over bank flows are rare, which further restricts access to any existing riparian wetland or forest habitat. Thus, because of the alteration of the estuary food web and the restricted habitat access, productive capacity of the estuary has decreased from historical levels (Bottom et al. 2001, NMFS 2000c, USACE 2001, Johnson et al. 2003b).

The role and importance of microbial communities in the modern day estuary food web has recently been the focus of a considerable amount of research. As discussed previously in section 2.3.1.3, the ETM traps particulate material of river and ocean origin. The residence time of these particles within in the ETM is believed to be 2-4 weeks, compared to the 1-2 day residence time of water (Neal 1972 as cited in Crump et al. 1999) or associated free-living bacteria outside the ETM (Crump et al. 1999). The circulation and trapping processes in the ETM facilitates attachment among particles, forming rapidly settling macroaggregates. In the Columbia River estuary, bacteria associated with these particles are believed to be a primary food source within the food web (Baross et al. 1994, Crump et al. 1998 as cited in Crump et al. 1999). For example, particle-attached bacteria accounted for about 90% of the heterotrophic bacterial activity in the Columbia River estuary water column; these bacteria were 10-100 times more active than free-living bacteria outside the ETM (Crump and Baross 1996, Crump et al. 1998 as cited in Crump et al. 1999). Crump et al. (1999) noted that a large part of the particle-attached bacteria in the ETM were unrelated to river or coastal ocean bacteria, suggesting these ETM bacteria developed in the estuary. Further, these bacteria attached to ETM particles are extremely important degraders of particulate organic matter in the system and serve as the food base for detritivorous copepods (the dominant grazer; Simenstad et al. 1994a) as well as rotifers and protozoa (Crump and Baross 1996). These organisms are often important in the diet of salmonids.

The estuarine food web can be highly variable because of differential pulses of organic matter and the varied distributions of food sources across estuarine habitats (Wissmar and Simenstad 1998 as cited in USACE 2001). Because of seasonal changes in habitats and prey resources caused by changes in habitat-forming processes, salmonids encounter a seasonally varying array of habitat conditions and prey resources. Consequently, the contribution of the estuary for juvenile salmonid survival, growth, passage, and smolting varies seasonally, especially when salmonids localize their rearing and movements in a specific estuarine region or habitat (USACE 2001).

The marine fish community off the mouth of the Columbia River has changed since the 1980s and is structured by physical oceanographic characteristics (such as salinity, temperature, and chlorophyll). The distribution and abundance of the nearshore marine predator and forage fish community affects the amount of predation on juvenile salmonids; that is, high prey densities increases the probability of predation on juvenile salmonids while high forage fish density decreases the probability of predation on juvenile salmonids (NMFS 1998, NMFS et al. 1998).

2.4.2.9 Changes in Salmonid Life History and Estuarine Residence Time

The physical habitat requirements of juvenile salmonids are related to their life history pattern (i.e. stream-type vs. ocean-type). The primary factors that describe physical habitats are water depth, water velocity, and substrate type, while secondary physical factors are water temperature, salinity, and turbidity (USACE 2001). Salmonids adapt to relatively wide ranges within the secondary physical factors. Anthropogenic factors may create artificial selection (Sheridan 1995 as cited in Bisbal and McConnaha 1998), add challenges for salmonids to maintain their biological diversity and ability to withstand environmental variation, and thus, can alter the biological structure of salmonid populations and reduce the variation in life history patterns (Bisbal and McConnaha 1998). Most mitigation efforts are optimized based on juvenile fish adundance, rather than life history diversity, such as the process of transporting emigrating juvenile salmonids past Columbia River dams (Bisbal and McConnaha 1998).

Rich (1920) investigated juvenile chinook life history and migration in the estuary from 1914-1916. He collected 1365 fish and discussed scale patterns of fish captured at different locations or known to be of a specific origin (such as a specific tributary or hatchery). From this scale pattern analysis, Bottom et al. (2001) classified the life history patterns described below (Table 2-16). Rich’s 1916 data show that chinook fry were present in the estuary from late March through September and chinook fingerlings were present from April to December (Bottom et al. 2001). Based on a comparison of average fork lengths of different sample groups, Rich (1920) indicated that growth in the estuary was particularly rapid in June, July, and August; however, Rich (1920) cautioned that average fork length comparisons may not represent actual growth rates because sampling in successive months likely measured entirely different groups of fish.

Table 2-16. Chinook life history types from scale analysis of fish captured in the Columbia River estuary (from Bottom et al. 2001 based on Rich 1920).

Life-history Type Collected	Rearing Behavior	% of Total
Fry	Fish that moved into the estuary as fry shortly after emergence	33%
Fingerling		
Smolts and recent arrivals	Fish too recently arrived at the estuary to have evidence of estuarine rearing on their scales. Includes both smolts and fingerlings bound for estuarine rearing habitats	28%
Fluvial-rearing	Fluvial rearing as fry and as fingerling. Includes fish that reared near their natal areas, and also fish that migrated into larger rivers downstream from their natal sites to rear, but which did not rear in the estuary	6%
Estuarine-rearing	Fish that reared for a short time in their natal stream then migrated to the estuary to rear	25%
Fluvial and estuarine-rearing	Fish with evidence of either adfluvial or estuarine rearing, but with scale patterns that did not lend themselves to identifying a fish in either category with certainty.	8%

Bottom et al. (2001) used historical and contemporary fish surveys to assess changes in use of estuarine environments by juvenile salmon. They conclude that population structure and life history diversity of subyearling ocean-type chinook salmon has simplified significantly since the early part of last century. Historically, chinook salmon in the Columbia River exhibited a wide diversity of life history types, using streams, rivers, the estuary, and perhaps the Columbia River plume as potential rearing areas. For ocean-type chinook salmon, there may be as many as 35 potential life history strategies (Wissmar and Simenstad 1998 as cited in USACE 2001, Bottom et al. 2001). Bottom et al. (2001) suggest that human affects on the environment have caused chinook life history patterns to be more constrained and homogenized than historical data show. Data collected by Rich (1920) show several forms of ocean-type chinook life histories, based on scale patterns, length, and time of capture. Groups of fish migrated to the estuary as either fry or fingerlings, in the spring or fall. Individual fingerlings arrived in the estuary throughout the year. Some fish remained in the estuary for extended periods of time while others migrated seaward rapidly. Fish from a brood represented a continuum of rearing and migrant behaviors spanning an 18-month period. However, even the work of Rich (1920) may have underestimated the historical diversity of estuarine rearing behaviors because many changes in the basin had already occurred. Migration timing and size of juvenile salmonids entering the estuary are important factors affecting stock life histories, maturation, and ultimate survival (Reimers 1973, Schluchter and Lichatowich 1977, Groot and Margolis 1991 as cited in Nez Perce et al. 1995).

By contrast, today ocean-type chinook with estuarine rearing life histories are not a primary life history form observed by managers and resource users. Most modern day ocean-type chinook fit into one of three groups: subyearling migrants that rear in natal streams, subyearling migrants that rear in larger rivers and/or the estuary, yearling migrants. Today, fish enter the estuary later (by at least two weeks) in pulses that coincide with hatchery releases. Subyearling chinook abundance in the estuary is limited; most chinook are yearlings with a homogeneous size distribution. Abundance patterns of juvenile chinook in the estuary now reflect hatchery management practices more than historical migration behavior. Although, current life history diversity may be underestimated because most research has focused on the migration and survival of hatchery yearlings and large natural subyearlings, not smaller subyearlings. Nevertheless, the uniform sizes and rapid estuary migrations of chinook salmon compared to historical observations suggests a loss of diversity and type of life history responses (Bottom et al. 2001). Hatchery practices have promoted a decrease in life history diversity; restoration efforts need to consider habitats/life histories that may be limited or non-existent today (Bottom et al. 2001).

Also, the size range of fish sampled today is smaller than the historical ranges. Size distributions indicate that historical juvenile entry continued from spring through fall, with some extended estuarine residence. The flux of chinook entering the estuary included fry that migrated to the estuary in the fall and may have overwintered in the lower reaches (Bottom et al. 2001). Smaller fish historically present in the fall are poorly represented in modern sampling studies. Today's chinook are composed of relatively few fry, and many larger subyearlings that likely do not reside in the estuary for extended periods. Bottom et al. (2001) suggest that the data indicate that ocean-type chinook with estuarine rearing life histories are now substantially reduced in proportion relative to their historical levels. The authors caution, however, that present day diversity may be underestimated by inclusion of data from modern monitoring programs that emphasize migration and survival of hatchery yearlings and subyearlings, but did not sample in many shallow-water habitats where smaller size ranges of juveniles would be more likely to be found.

2.4.3 Pacific Lamprey

Juvenile lamprey depend on sand and silt substrates, thus, habitat forming processes and conditions that create this habitat characteristic are likely beneficial to juvenile lamprey survival. Anthropogenic factors that introduce more sand and silt to a river's substrate (i.e. riparian zone development, logging, road building either within the subbasin or in upriver locations) may contribute to the development of habitat preferred by juvenile lamprey. Further, the altered Columbia River flow regime resulting from water regulation has decreased the flow volumes capable of transporting large volumes of sand/silt to the estuary/ocean; thus, sand and silt substrates are more likely to remain in the mainstem compared to historical conditions.

2.4.4 Sturgeon

Hard-bottom, high-velocity, structured habitats with adequate interstitial space are critical as spawning and incubation substrate as well as juvenile predation refuge and feeding areas for white sturgeon (Parsley et al. 1993; Perrin et al. 1999; Parsley et al. 2002; Secor et al. 2002). White sturgeon juveniles that burrow into fine sediments commonly die as a result of suffocation. Maintenance of these preferred white sturgeon habitats are important to the species continued productivity in the lower Columbia River and estuary. Anthropogenic factors that continue to introduce more sand and silt to the river's substrate (i.e. riparian zone development, logging, road building) likely decreases the availability of preferred white sturgeon habitat. Further, the decreased the flow volumes resulting from Columbia River water regulation has decreased sand/silt transport to the estuary/ocean; thus, sand and silt substrates are more likely to remain in the mainstem, adversely affecting white sturgeon. These habitat changes have likely occurred in mainstem and distributary channel habitats.

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 et al. 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. 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.

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.

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).

2.4.5 Northern Pikeminnow

Northern pikeminnow abundance in the mainstem Columbia River below the Snake River confluence is greatest in the lower mainstem from the estuary to the Dalles Dam (Beamesderfer et al. 1996). Although Northern pikeminnow have relatively broad spawning and rearing requirements, they seem to prefer low velocity water with clean rocky substrate. Anthropogenic factors that contribute to sedimentation and the altered flow regime of the Columbia River likely have decreased the availability of these preferred pikeminnow habitats. However, Northern pikeminnow have successfully adapted to the changing habitat conditions in the lower Columbia River mainstem as evidenced by their current abundance; it is anticipated that Northern pikeminnow will continue to thrive under the current trend of habitat alterations.

2.4.6 Eulachon

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.

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 of the study 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.

2.4.7 River Otter

In the estuary, river otters are concentrated in shallow water tidal sloughs and creeks associated with willow-dogwood and Sitka spruce habitats located primarily in the Cathlamet Bay area and along the Oregon riverbank (Howerton et al. 1984); otters likely inhabit similar areas throughout the tidal freshwater area of the lower Columbia. Dikes throughout the estuary have disconnected substantial amounts of side channel and floodplain habitats from the mainstem. However, the Cathlamet Bay area remains as one of the most intact and productive tidal marsh and swamp habitat throughout the entire estuary. Because river otters are capable of traveling over land, it is not understood how the loss of habitat connectivity of side channel and floodplain habitat has affected species' behaviors such as foraging, resting, mating, and rearing.

2.4.8 Columbian White-tailed Deer

Columbian white-tailed deer are most closely associated with Westside oak/dry Douglas fir forest within 200m of a stream or river; however, they can be found breeding or feeding in any number of habitats (lowland forest, grasslands, riparian wetlands, agriculture/pastures/mixed environments, urban/mixed environments; Johnson and O'Neil 2001). Agriculture and urban development throughout the lower Columbia River and estuary decreased the acreage of some of these habitats while increasing the acreage of others; thus, the net effect on Columbian white-tailed deer is difficult to quantify but appears to have negatively affected population abundance. Establishment of the Columbian White-Tailed Deer National Wildlife Refuge and other recovery efforts have focused on providing deer with appropriate contiguous habitat.

2.4.9 Caspian Tern

Caspian terns were not historically present in the Columbia River estuary. Management actions (i.e. periodic deposition of dredge spoils forming flat, sandy, unvegetated, mid-channel habitats) have created preferred habitat, encouraging colonization by Caspian terns. The altered Columbia River flow regime as a result of water regulation will likely produce variable effects on the presence of preferred Caspian tern habitat. For example, reduced peak flows are less likely to erode or inundate newly created dredge spoils islands; thus, sand substrates may remain stationary for long periods of time, but, without periodic inundation, vegetational succession begins and Caspian terns do not adapt well to the presence of vegetation in the breeding area. Further, decreased peak river flows and decreased wave action as a result of jetty construction have generally increased the amount of accretion throughout the estuary, which has increased the presence of the preferred newly formed, flat, sandy habitats of Caspian terns.

2.4.10 Bald Eagle

In western Washington, nest trees are most often old-growth Douglas-fir (Pseudotsuga menziesii) and Sitka spruce (Picea sitchensis) near the coast (Grubb 1976 as cited in Stinson et al. 2001), with a higher component of mature grand fir (Abies grandis) and black cottonwood (Populus balsamifera) around Puget Sound (Watson and Pierce 1998 as cited in Stinson et al. 2001). Assuming the presence of an adequate food supply, the single most critical habitat factor associated with bald eagle nest locations and success is the presence of large trees (Watson and Pierce 1998 as cited in Stinson et al. 2001). Thus, loss or alteration of nesting habitat as a result of natural events (e.g., fire, windstorms, etc.) or human-caused alterations (e.g., timber harvest, development) that results in permanent loss of nest trees or potential nesting habitat or prevents trees from attaining the size capable of supporting a nest, has the potential to reduce the number of nesting territories in Washington. Further, roost sites and perch sites also are often associated with large trees, so availability of this mature forest habitat determines potential bald eagle territories.

Declines in salmonid abundance has likely negatively affected bald eagles. Because the time of spawning for most Columbia River salmon runs is from August to January, declines in salmon runs have probably primarily affected the distribution and abundance of post-breeding and wintering bald eagles. Supplementation of salmon runs through hatchery fish generally does not replace the carcasses that historically provided food for bald eagles. Likewise, abundance of many seabirds and waterfowl have declined in recent years; loss of this prey base has also likely negatively affected eagles (Stinson et al. 2001).

Contaminant-free prey is necessary to maintain the reproductive health and survival of bald eagles. Organochlorine compounds and derivatives are still present in the Columbia River estuary and lower mainstem as result of industrialization within the subbasins. Often, contaminants are re-released in the ecosystem during river dredging. Bald eagles in the Columbia River estuary have exhibited chronic low nest productivity, apparently because of a variety of contaminants, including DDE, PCB’s, and dioxins (Anthony et al. 1993 as cited in Stinson et al. 2001). Residual DDT and PCBs continue to accumulate and concentrate as individuals consume contaminated prey. Some eagles may contain elevated levels of DDE in their tissues that prevents successful reproduction, or their territory may contain contaminated prey that continues to affect the resident eagles (Jenkins and Risebrough 1995 as cited in Stinson et al. 2001).

2.4.11 Osprey

Breeding osprey are concentrated in forested riparian areas, generally nesting atop trees or rock pinnacles. Osprey have adapted with human development and have been observed nesting on artificial structures such as channel markers or utility/light poles; recent data (late 1990s) suggests that the osprey population along the lower Columbia River mainstem may be increasing. Although habitat alterations do not appear to be having significant detrimental effects on osprey along the lower Columbia River, Columbia River osprey eggs contained the highest concentration of DDE (derivative of formerly banned pesticide DDT) reported in North America in the late 1980s and 1990s (Henny et al. 2003); DDE adversely affects eggshell thickness and decreases breeding success.

2.4.12 Sandhill Crane

The lower Columbia River mainstem and estuary is not a historical breeding or overwintering area for sandhill cranes. Agricultural development throughout the lower Columbia River floodplain has likely attracted overwintering sandhill cranes; for the last seven or eight years, an average of a few hundred, but up to 1,000 cranes have overwintered in the lower Columbia River floodplain. Reclamation of agricultural land for habitat restoration projects may discourage overwintering by sandhill cranes, although future development of herbaceous wetlands may provide adequate winter habitat for sandhill cranes currently using the region.

2.4.13 Yellow Warbler and Red-eyed Vireo

The yellow warbler and red-eyed vireo are both riparian obligate species; warblers prefer shrub-dominated habitats and vireos prefer dense, closed canopy forests. Habitat alterations along the lower Columbia River corridor have likely been more damaging to the possible presence of red-eyed vireos as opposed to yellow warblers. Dense riparian forests along the lower Columbia River are likely less abundant than shrub-dominated wetland habitat. Neither species is likely greatly affected by the disconnectedness of floodplain habitat from the mainstem.

2.5 Ecological Relationships

Ecological relationships describe species-species relationships and species-environment relationships; paramount to these relationships are the effects to the specific life stage of focal species, if known. Two general categories of interspecies relationships exist: native-native interactions (Section 2.5.1) and native-exotic interactions (Section 2.5.2). Each of these categories are addressed separately below; each section addresses predation and competition aspects of species interactions. Additionally, the discussion of exotic species addresses full scale ecosystem alterations.

2.5.1 Native Species Interactions

2.5.1.1 Predation

Significant numbers of salmon are lost to fish, bird, and marine mammal predators during migration through the mainstem Columbia River. Predation likely has always been a significant source of mortality but has been exacerbated by habitat changes. Piscivorous birds congregate near dams and in the estuary around man-made islands and consume large numbers of outmigrating juvenile salmon and steelhead (Roby et al. 1998). Caspian terns, cormorants, and gull species are the major avian predators (NMFS 2000a). While some predation occurs at dam tailraces and juvenile bypass outfalls, by far the greatest numbers of juveniles are consumed as they migrate through the Columbia River estuary. Native fishes, particularly northern pikeminnow, prey on juvenile salmonids. Marine mammals prey on adult salmon, but the significance is unclear.

Caspian terns are native to the region but were not historically present in the lower Columbia River mainstem and estuary; they have recently made extensive use of dredge spoil habitat and are a major predator of juvenile salmonids in the estuary. The terns are a migratory species whose nesting season coincides with salmonid outmigration timing. Since 1900, the tern population has shifted from small colonies nesting in interior California and southern Oregon to large colonies nesting on dredge spoil islands in the Columbia River and elsewhere (NMFS 2000c). Many of these Columbia River dredge spoils islands were created as a result of dredging the navigational channel after the eruption of Mt. St. Helens in 1980; however, Rice Island was initially constructed from dredge spoils around 1962 (Geoffrey Dorsey, USACE, personal communication). Caspian terns did not nest the estuary until 1984 when about 1,000 pairs apparently moved from Willapa Bay to nest on East Sand Island. Those birds (and others) moved to Rice Island in 1987 and the colony expanded to 10,000 pairs. Diet analysis has shown that juvenile salmonids make up 75% of food consumed by Caspian terns on Rice Island. Roby et al. (1998) estimated Rice Island terns consumed between 6.6 and 24.7 salmonid smolts in the estuary in 1997, and that avian predators consumed 10-30% of the total estuarine salmonid smolt population in that year. However, there are no data to compare historical and modern predation rates or predator populations; thus, effects of these unique predator populations in relation to historical losses of juvenile salmon to predation cannot be adequately quantified (Bottom et al. 2001). Also, recent management actions have been successful in discouraging Caspian tern breeding on Rice Island while encouraging breeding on East Sand Island, which may decrease predation on juvenile salmonids. Further, current predation studies are limited because of the unknown effects hatchery rearing and release programs have had on salmon migration behavior and predator consumption. Nevertheless, evidence suggests that current predator populations could be a substantial limiting factor on juvenile salmon survival (Bottom et al. 2001).

Pikeminnow have been estimated to consume millions of juveniles per year in the lower Columbia, as outlined in the following table.

Table 2-17. Projected abundance of northern pikeminnow, salmonid consumption rates, and estimated losses of juvenile salmonids to predation (NMFS 2000b).

Location	Length (km)	Predator Number	Consumption Rate (smolts/predator day)	Estimated Losses (millions/year)
Estuary to Bonneville Dam	224	734,000	0.09	9.7
Bonneville Reservoir	74	208,000	0.03	1.0

Pikeminnow numbers likely have increased as favorable slack-water habitats have been created by impoundment and flow regulation. In unaltered systems, pikeminnow predation is limited by smolt migratory behavior; the smolts are suspended in the water column away from the bottom and shoreline habitats preferred by pikeminnow. However, dam passage has disrupted juvenile migratory behavior and provided low velocity refuges below dams where pikeminnow gather and feed on smolts. The diet of the large numbers of pikeminnow observed in the forebay and tailrace of Bonneville Dam is composed almost entirely of smolts. Pikeminnow also concentrate at dam bypass outfalls and hatchery release sites to prey on injured or disoriented fish, and pikeminnow eat many healthy smolts as well. Predation rates on salmonids are often much lower in areas away from the dams, although large numbers of predators in those areas can still impose significant mortality.

In 1990, responding to observed predation problems, a pikeminnow management program was instituted that pays rewards to anglers for pikeminnows over a prescribed size. Through 2001, over 1.7 million pikeminnow had been harvested, primarily in a sport reward fishery. Modeling results project that potential predation on juvenile salmonids by northern pikeminnow has decreased 25% since fishery implementation (NMFS 2000a). By paying only for pikeminnow over a certain size, the program takes advantage of their population characteristics—they are relatively long-lived and only the large individuals are fish predators. Relatively low exploitation rates of only 10-20% per year compound over time to substantially reduce pikeminnow survival to large predaceous sizes.

Seals and sea lions (particularly harbor seals [*Phoca vitulina*], Steller sea lion [*Eumetopias jubatus*], and California sea lion [*Zalophus californianus*]) are common in and immediately upstream of the Columbia River estuary and are regularly observed up to Bonneville Dam. Seals and sea lions are regularly reported to prey on adult salmon and steelhead, although diet studies indicate that other fish comprise the majority of their food. Large numbers of pinnipeds might translate into significant salmon mortality despite this occasional use. However, it is difficult to interpret the significance of this mortality factor for salmon, considering that large pinniped populations have always been present in the Columbia River. However, current marine mammal predation may be proportionally more significant, since all sources of mortality on depressed stocks become more important. Their numbers were reduced by hunting (including bounty hunters) and harassment from the late 1800s until the Federal Marine Mammal Protection Act (FMMPA) was adopted in 1972. Their numbers have significantly increased since the adoption of FMMPA. Fishers historically viewed seals and sea lions as competitors and the old Columbia River Fisherman Protection Associations funded a control program. These mammals can be troublesome to sport and commercial fishers by taking hooked or net-caught fish before they can be landed.

2.5.1.2 Competition

The productivity of the Columbia River estuary likely has decreased over time as a result of habitat degradation, which initially would appear to increase the likelihood for competition among salmonids in the estuary especially during times of high juvenile abundance. However, historical natural abundance of juvenile salmonids in the lower mainstem and estuary was far greater than the current abundance, even considering large hatchery releases of juvenile salmonids. Thus, it is possible that decreases in estuary habitat productivity are of the same magnitude as decreases in salmon abundance, suggesting that salmonid density dependent mechanisms in the estuary are no more likely today than they were historically.

Because ocean-type chinook salmon spend more time in the estuary, they may be sensitive to changes in the productivity of the estuary environment than stream-type chinook salmon. Estuaries may be “overgrazed” when large numbers of ocean-type juveniles enter the estuary en masse (Reimers 1973, Healey 1991). Food availability may be negatively affected by the temporal and spatial overlap of juvenile salmonids from different locations; competition for prey may develop when large releases of hatchery salmonids enter the estuary (Bisbal and McConnaha 1998), although this issue remains unresolved (Lichatowich 1993 as cited in Williams et al. 2000). Reimer (1971) suggested a density dependant mechanism affects growth rate and hypothesized that fall chinook growth in the Sixes River was poor from June to August because of the large population in the estuary at this time and that the increased growth rate in September to November resulted from reduction in population size and a better utilization of the whole estuary.

The potential exists for large-scale hatchery releases of fry and fingerling ocean-type chinook salmon to overwhelm the production capacity of estuaries (Lichatowich and McIntyre 1987). However, Witty et al. (1995) could not find any papers or studies that evaluated specific competition factors between hatchery and wild fish in the Columbia River estuary. Simenstad and Wissmar (1983) cautioned that the estuary condition may limit rearing production of juvenile chinook, and many other studies have demonstrated the importance of the estuary to early marine survival and population fitness. However, rivers such as the Columbia, with well-developed estuaries, are able to sustain larger ocean-type populations than those without (Levy and Northcote 1982).

The intensity and magnitude of competition in estuaries depends in part on the duration of residence of hatchery and natural juvenile salmonids. One would expect summer/fall chinook from the mid-Columbia region to use the estuary for a period that probably depends upon their size when they arrive (Chapman et al. 1994). Chapman et al. (1994) conclude that the survival of juveniles transported to below Bonneville Dam at a size too small to ensure high survival at sea may depend upon growth in the estuary for successful ocean entry. Meanwhile, some workers (Reimers 1973; Neilson et al. 1985) have suggested that the amount of time spent in estuaries may relate to competition for food. Chapman et al. (1994) suggested that, if large numbers of hatchery fish are present in the Columbia River estuary, growth and survival of wild subyearling chinook could be reduced. However, Levings et al. (1986) reported that the presence of hatchery chinook salmon did not affect residency times and growth rates of wild juveniles in a British Columbia estuary and that hatchery fish used the estuary for about half the length of time that wild fry were present (40-50 days).

Natural populations of salmon and steelhead migrate from natal streams over an extended period (Neeley et al. 1993; Neeley et al. 1994); they also enter the estuary over an extended period (Raymond 1979). Hatchery fish are generally—but not always—released over a shorter period resulting in a mass emigration into natural environments. In recent years, managed releases of water, commonly called water budgets, have been used to aid mass and fast migration of hatchery and wild smolts through the migration corridor. Decisions regarding the mode of travel in the migration corridor (i.e., in river migration or collection/transportation) are made by managers to expedite movement of smolts to the estuary. Water budget management combined with large releases of hatchery fish result in large numbers of juvenile salmon and steelhead in the estuary during spring months when the estuary productivity is low. Fish that arrive in the estuary later in the season may benefit from increased food supplies. Chapman et al. (1994) note that subyearling chinook released later in the summer returned at significantly higher rates than subyearlings released early in the summer.

There is substantial overlap in estuarine habitat usage among chum and chinook salmon fry (Levy and Northcote 1982), suggesting significant potential for competition between these two salmonids. However, possible interactions between chum and chinook seems to be minimized by differences in migration timing and estuary residence periods; chum fry typically precede chinook in the estuary and spend a relatively short amount of time in the estuary compared to chinook (Levy and Northcote 1982).

2.5.2 Non-Indigenous Species Interactions

Introductions of aquatic non-indigenous species has become the focus of increasing concern and research; their increasing predominance in species assemblages indicate major changes in aquatic ecosystems (OTS 1993, Cohen and Carlton 1995, Smith 2001 as cited in Waldeck et al. 2003). Globally, there is an increasing rate of aquatic non-indigenous species introductions; this increase has been attributed to the increased speed and range of world trade, which facilitates the volume, variety, and survival of intentionally or unintentionally transported species. All aquatic non-indigenous species introductions in the lower Columbia River represent permanent alterations of the biological integrity of the ecosystem for numerous reasons: impacts of introduced species are unpredictable, introduced species alter food web dynamics, and introduced species are a conduit for diseases and parasites (Waldeck et al. 2003). Further, it has been hypothesized that changes in the Columbia River estuary and lower mainstem ecosystem as a result of hydrosystem development and water regulation have affected the successful establishment of aquatic non-indigenous species (Cordell et al. 1992 as cited in Draheim et al. 2002, Weitkamp 1994). The lower Columbia River ecosystem may still be adjusting to these major flow alterations; this adjustment period may benefit aquatic non-indigenous species (Weitkamp 1994).

Draheim et al. (2002) performed a literature review of aquatic non-indigenous species introductions in the Columbia River estuary and lower mainstem to Bonneville Dam; the authors also presented a 2001-2003 sampling plan for aquatic non-indigenous species. A final report on these sampling efforts was not available at the time of publication of this report, however, an interim report has been produced (Waldeck et al. 2003). A complete list of aquatic non-indigenous and cryptogenic (i.e. obscure or unknown origin) species to date was compiled in Draheim et al. 2002; the non-indigenous list includes plants (16), mammal (1), amphibians (1), fish (37), Annelida (2), Amphipoda (3), Cirripedia (1), Copepoda (3), Cumacea (1), Decapoda (4), Isopoda (1), Bivalvia (2), and Gastropoda (1), and the cryptogenic list includes Annelida (29), Amphipoda (3), Copepoda (1), Isopoda (1), Nemertea (1), and plants (2). However, limited information is available regarding the ecological interactions of many of these species; thus, only a select few are discussed in the sections below. In general, non-native fish species are dominated by species that have been intentionally introduced, whereas, most invertebrates are the result of unintentional introductions (Draheim et al. 2002). Further, fish introductions in the lower Columbia River increased in a linear fashion in the 1900s while non-indigenous invertebrate introductions seem to be increasing exponentially (Waldeck et al. 2003).

2.5.2.1 Predation

Walleye (*Stizostedium vitreum*) are voracious predators of fishes, including juvenile salmonids. On a fish-per-fish basis, walleye are every bit as damaging as pikeminnow, but walleye are considerably less abundant. Originally introduced into the upper Columbia basin, walleye, since the 1970s, have gradually spread downstream throughout the lower mainstem. Significant numbers of walleye have become established in Bonneville Reservoir and between Bonneville Dam and the estuary. Walleye population sizes are quite variable and driven by periodic large year classes that occur during warm low flow springs. Walleye are subject to a small directed sport fishery but were not included in the sport reward fishery because projected exploitation effects on salmonids were low. Unlike pikeminnow, most walleye predation occurs in smaller individuals not readily caught by anglers and unaffected by the compounding effects of annual exploitation.

Other introduced fishes—including smallmouth bass (*Micropterus dolomeiui*) and channel catfish (*Ictalurus punctatus*)—also have been found to consume significant numbers of juvenile salmonids. However, these species are more significant problems in upstream areas than in the lower river where their abundance is low.

2.5.2.2 Competition

American shad (*Alosa sapidissima*) have grown to substantial populations since introduction into the Columbia River system in 1885 (Welander 1940, Lampman 1946); in recent years, 2-4 million adults have been counted annually at Bonneville Dam. Although the construction of dams in shad-producing streams has been blamed in part for the decimation of East Coast stocks of American shad (Walburg and Nichols 1967 as cited in Weitkamp 1994), Weitkamp (1994) suggested that dams in the Columbia River system may partially be responsible for the shad's rapid population growth. American shad can successfully navigate some dams (Miller and Sims 1983 as cited in Weitkamp 1994); the completion of the Dalles Dam in 1956 (and subsequent inundation of Celilo Falls) extensively expanded the range of American shad into the upper Columbia and Snake Rivers (Stober et al. 1979 as cited in Weitkamp 1994). Further, the transition of the estuarine food web from a macrodetritus to microdetritus base (i.e. increased importation of plankton from upstream reservoirs) has benefited zooplanktivores, including American shad (Sherwood et al. 1990).

Because of 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 pikeminnow, smallmouth bass, and walleye (Petersen et al. In press). A pattern is slowly emerging that suggests the existence of American shad is changing trophic relationships with the Columbia River. Because of their abundance, consumption rates and patterns of American shad may have modified the estuarine food web. One study found that in the Columbia River estuary and lower mainstem (up to RKm 62) shad diet overlapped with subyearling salmonid diets, which may indicate competition for food. Juvenile shad and subyearling 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 pikeminnow, which in turn are significant predators of juvenile salmonids (Petersen et al. In press).

In the Columbia River estuary, American shad were described as year-round residents (Bottom et al. 1984). Subyearling shad were captured in all areas of the estuary, primarily from August to December (Bottom et al. 1984). Yearling shad were captured throughout the year in all areas of the estuary with all gear types (Bottom et al. 1984), indicating widespread temporal and spatial distribution. Two-year old American shad were also captured throughout the year in all areas of the estuary, but they were more common in the freshwater and estuarine regions (Bottom et al. 1984). In the January, yearling American shad were distributed throughout the freshwater and estuarine areas of the estuary in water column and channel bottom habitats while 2-year olds were also present in freshwater and estuarine areas, primarily in water column habitats (Bottom et al. 1984). In the spring (April to June), a large pelagic assemblage was identified that included subyearling and yearling American shad, subyearling and yearling salmonids, and Pacific herring (Bottom et al. 1984); thus, there is overlap in habitat usage by American shad and juvenile salmonids during the season of high juvenile salmonid abundance. In August, yearling and 2-year old American shad were associated with water column habitats in the marine, estuarine, and freshwater areas of the estuary (Bottom et al. 1984). Diet analysis indicated that subyearling American shad most frequently preyed upon calanoid, cyclopoid, and harpacticoid copepods and *Daphnia spp*. (Bottom et al. 1984). Meanwhile, yearling and 2-year old American shad most frequently consumed calanoid copepods, *Corophium salmonis,* and harpacticoid copepods; to a lesser extent, cyclopoid copepods and *Corbicula manilensis* were also consumed (Bottom et al. 1984). In the spring, yearling American shad consumed primarily calanoid copepods, although up to 25% of their diet consisted of *Corophium salmonis*; *Corophium salmonis* was the primary prey item (up to 75%) of subyearling and yearling salmonids present in the estuary during the same season (Bottom et al. 1984). In the summer, *Daphnia spp.* are a major prey item of subyearling and yearling American shad; *Daphnia spp*. are also the primary prey item of subyearling chinook salmon during the summer, comprising over 75% of the diet (Bottom et al. 1984).

Commercial harvest has been considered as a means to reduce the abundance of American shad in the Columbia River, however, harvest has been restricted because the shad spawning run coincides with the timing of depressed runs of summer and spring chinook, sockeye, and summer steelhead (WDFW and ODFW 2002).

The banded killifish (*Fundulus diaphanous*) was likely introduced illegally into the Columbia River basin (Farr and Ward 1993 as cited in Weitkamp 1994) sometime around 1970 (Weitkamp 1994). Although not abundant initially, densities of 375 fish per hectare have been observed at Miller Sands in summer and fall (Hinton et al. 1990 as cited in Weitkamp 1994). In its native range, the banded killifish is a midwater and surface feeder, preying primarily on cladocerans and ostracods, although, it consumes mollusks and flatworms to a lesser extent (Smith 1985 as cited in Weitkamp 1994). Although there may be some diet overlap among banded killifish, salmonids, and other fish in the estuary and lower mainstem, its impacts on native fish and the estuarine ecosystem are largely unknown (Weitkamp 1994). Changes to the estuary ecosystem resulting from development and operation of the hydropower system may have contributed to increased survival and range extension of banded killifish (Weitkamp 1994). Weitkamp (1994) suggested that the banded killifish’s limited distribution in shallow water habitats and its small size may limit the potential ecological impact in the estuary; however, continued growth of the population would warrant further investigation.

2.5.2.3 Ecosystem Alteration

Significant changes in estuary faunal communities have occurred through species introductions, but, for the most part, the effects of these species introductions have not been assessed. Several nonnative invertebrate species have expanded their populations dramatically since introduction, particularly the Asian bivalve, *Corbicula fluminea*. First discovered in the Columbia River estuary in 1938 (Ingram 1948), it was likely unintentionally introduced from ship ballast (Weitkamp 1994). This bivalve has expanded from the estuary far into the lower mainstem reservoirs and tributaries (Bottom et al. 2001). Densities exceeding 10,000 m^2 have been recorded in Cathlamet Bay, however, densities of 100-3,000 m^2 are more typical in the estuary (Emmett et al. 1986, Hinton et al. 1990 as cited in Weitkamp 1994); density elsewhere in the basin is not known. *C. fluminea* has been shown to outcompete native bivalves and are very tolerant of variable environmental conditions (i.e. can withstand considerable ranges and fluctuations in temperature, dissolved oxygen, flow velocity, water level, and contaminant concentrations) (Sinclair 1971, Gardner et al. 1976). Lauritsen (1986) suggests that large numbers of *C. fluminea* can have an affect on phytoplankton biomass and nutrient cycling. Because of their abundance, consumption rates and patterns of *C. fluminea* may have modified the estuarine food web. However, the influences of *C. fluminea* in the Columbia River estuary ecosystem and on native bivalves are poorly understood. Unpublished data from the California Department of Fish and Game showed that while these nonindigenous species were never prominent in the diets of juvenile salmonids, they seasonally made up the principle stomach contents of other pelagic fishes, such as American shad, herring, stickleback and smelt species (Bottom et al. 2001).

The calanoid copepod *Pseudodiaptomus inopinus* was recently introduced (around 1990) in the Columbia River estuary, likely from cargo ship ballast water originating from the Indo-Pacific region (Weitkamp 1994). The moderated peak flows and warmer water temperatures resulting from hydrosystem operation and other anthropogenic activities has facilitated success of this copepod in the estuary (Cordell et al. 1992 as cited in Weitkamp 1994). Cordell et al. (1992 as cited in Weitkamp 1994) identified *P. inopinus* as the third most abundant zooplanktor in the estuary; densities of 17,000 m^2 were recorded. *P. inopinus*, as well as other zooplanktors, is associated with the ETM, although ETM sampling has shown that *P. inopinus* is associated with different physical attributes of the ETM than the two most abundant zooplanktors in the estuary, *Eurytemora affinis* and *Scottolana canadensis* (Cordell et al. 1992 as cited in Weitkamp 1994). This spatial segregation suggests a reduced potential for competition between these native and exotic zooplankton (Cordell et al. 1992 as cited in Weitkamp 1994); however, the abundance of *P. inopinus* suggests that it may have substantial impact on the estuary ecosystem (Weitkamp 1994).

Ecosystem effects of non-indigenous aquatic plants are a concern for many resource managers. Of particular interest in the Columbia River estuary and lower mainstem are four plants considered noxious weeds: purple loosestrife (*Lythrum salicaria*), Eurasian water milfoil (*Myriophyllum spicatum*), parrot feather (*Myriophyllum aquaticum*), and Brazilian elodea (*Egeria densa*). Because much of the information regarding these aquatic nuisance plants was derived from the Washington Department of Ecology webpage, the following paragraphs identify known distribution within Washington. These, and other non-indigenous macrophytes, may also be a significant concern on the Oregon side of the lower mainstem and estuary, however, specific information regarding the status and distribution within Oregon was not found. Additionally, Wahkiakum County, Washington, recently published a management plan that discusses in detail the issue of aquatic vegetation management as well as known distribution of select non-indigenous aquatic plants in the Columbia River estuary (AquaTechnex 2003).

Purple loosestrife, native to Eurasia, was originally introduced to the eastern seaboard of North America in the early 1800s from European ship ballast and as a valued medicinal herb; expansion westward coincided with increased transportation systems and various commercial uses, such as horticulture and forage cultivation for beekeepers. In Washington, purple loosestrife was first collected in 1929 from Lake Washington; it has since spread to most areas of the state. Purple loosestrife generally occurs in shallow, fresh and brackish water, and may grow in wetlands, ponds/lakes, stream banks, and ditches. Purple loosestrife is a successful colonizer of any wet, disturbed site; it quickly adapts to environmental changes and can expand its range rapidly. The primary ecological effect of purple loosestrife is that it disrupts wetland ecosystems by displacing native plants and eventually displacing the animals that rely on the native flora for food, nesting, or cover. Purple loosestrife spreads aggressively and is very difficult to control; combinations of cutting and herbicide application have produced mixed results, depending on the season and duration of treatment. Biological control agents through the use of leaf-eating beetles or root-mining weevils show considerable promise for controlling purple loosestrife (WDOE 2003).

Eurasian water milfoil, native to Europe, Asia, and northern Africa, may have first been introduced to North America in the late 1800s at Chesapeake Bay; expansion of the plant throughout much of North America is thought to largely be a result of boating activity from one waterbody to the next. In Washington, the first known record of Eurasian water milfoil was a 1965 herbarium specimen from Lake Meridian in King County. Eurasian water milfoil is extremely adaptable and thrives in a variety of environmental conditions, such as still or flowing water, salinity up to 15 parts per thousand, water depth up to 10 meters, pH from 5.4-11, and survival under ice; it appears to grow best on fine-textured, inorganic substrates. Eurasian water milfoil negatively affects aquatic ecosystems in a number of ways. First, the dense canopies produced by Eurasian water milfoil shade out native vegetation, creating monospecific stands that provide poor habitat for fish and wildlife. Second, plant sloughing, leaf turnover, and decomposition at the end of the growing season increases phosphorus and nitrogen loading to the water column, affecting water quality. Third, dense canopies of Eurasian water milfoil affect water quality by increasing pH, increasing water temperature, and decreasing oxygen under the dense mats. Eurasian water milfoil also has many societal impacts; it often disrupts recreational activities such as fishing, boating, or water skiing. Further, Eurasian water milfoil can negatively impact power generation or irrigation withdrawals by clogging dam trash racks or water intake pipes. Numerous methods have been effectively used to control Eurasian water milfoil; success of each method depends on a number of factors, including duration of application and appropriateness of the method to the local environment. For example, covering sediments with an opaque fabric works well in localized areas but is not appropriate for large scale control programs. Water level drawdown has proven effective at dessicating plants in cold or dry climates, but this method is not effective in wet climates, such as western Oregon and Washington. Numerous herbicides have effectively controlled Eurasian water milfoil, provided the duration and concentration of application is sufficient. Finally, biological controls, particularly a native North American weevil, have been successfully used to control Eurasian water milfoil (WDOE 2003).

Parrot feather, native to the Amazon River in South America, has naturalized worldwide, particularly in warmer climates; its worldwide introduction has resulted primarily because of widespread use as an indoor/outdoor aquaria or aquatic garden plant. In the United States, parrot feather is present throughout the southern states and along both coastlines. In Washington, presence of parrot feather was first reported in 1944; parrot feather appears to be present in coastal lakes and streams, as well as the Southwest Washington portion of the Columbia River. Parrot feather is prevalent throughout the Longview/Kelso area drainage system, as well as many drainage ditches in Wahkiakum County. Able to colonize slow moving or still water, parrot feather is commonly found in freshwater lakes, ponds, streams, or canals. Parrot feather is well adapted to high nutrient environments and grows best when rooted in shallow water, although it is known to occur as a floating plant in nutrient-rich lakes. Although parrot feather provides cover for some aquatic organisms, generally it negatively alters the physical and chemical characteristics of its environment. Dense parrot feather stands alter aquatic ecosystems by shading the water column algae that previously served as the base of the aquatic food web. Further, parrot feather provides choice mosquito larvae habitat, which has created substantial problems in areas of high parrot feather occurence. Parrot feather is difficult to control; combinations of herbicides and mechanical controls (i.e. cutting or water drawdown) have produced mixed results. Further, because of its high tannin content, most grazers find parrot feather unpalatable. At present, biological control agents are not available, although research on multiple beetles and weevils show promise for parrot feather control. Additionally, fungal control options are currently under development (WDOE 2003).

Brazilian elodea, native to South America, is now distributed virtually worldwide, particularly because of its popularity as an aquarium plant. First reported in the United States in 1893 on Long Island, New York, Brazilian elodea has spread rapidly in fresh inland water throughout the U.S.; it was first reported in Washington in the early 1970s at Long Lake, Kitsap County. Brazilian elodea is distributed throughout many lakes, sloughs, and drainage ditches of western Washington, however, it has not been reported growing in eastern Washington lakes. Brazilian elodea may be rooted in water depths up to 20 feet or can be found drifting; it is adapted to both still and flowing water and thus can be found in lakes, ponds, ditches, and slow moving streams. Brazilian elodea forms dense monospecific stands that likely provide little benefit to native fish and wildlife; the dense stands restrict water movement and trap sediments, which affects water quality. Numerous methods have been effectively used to control Brazilian elodea; success of each method depends on a number of factors, much like that of Eurasian water milfoil. Thus, covering sediments with an opaque fabric works well in localized areas but not large scale control programs. Also, water level drawdown is not effective in wet climates, such as western Oregon and Washington. Numerous herbicides have effectively controlled Brazilian elodea. Additionally, a fungus that damaged Brazilian elodea in laboratory tests shows promise as a biological control. Finally, grass carp find Brazilian elodea particularly palatable and have been successfully employed as a management tool; however, use of grass carp has been limited because they are generally considered unsuitable for waterbodies where inlets and outlets cannot be screened (WDOE 2003).

Invasions of exotic cordgrasses (*Spartina alterniflora*, *S. anglica*, *S. densiflora*, and *S. patens*) have caused ecosystem changes in estuaries worldwide; each of these species are known to occur along the West coast of North America (Ayres et al. 2003). These species thrive in areas of accreting sediments, where they out-compete native vegetation (Daehler and Strong 1996). Although not known to be an immediate concern in the Columbia River estuary, *S. alterniflora* and *S. anglica* have caused significant changes in Willapa Bay, WA (Ayres et al. 2003). Cordgrasses disperse by floating seed and clonal fragments (Huiskes et al. 1995); such dispersal has been observed in Washington where *S. alterniflora* has spread from Willapa Bay to Grays

Harbor 30 km to the north (Ayres et al. 2003). Thus, significant potential exists for dispersal of these exotic cordgrasses to the Columbia River estuary.

Although not currently known to occur anywhere in the Columbia River basin, zebra mussels (*Dreissena polymorpha*) are a concern of Federal and State agencies throughout the Pacific Northwest (BPA 2002). Zebra mussels are an extremely prolific, freshwater mollusk native to the Caspian Sea (USGS 2002). In North America, they were first discovered in the Great Lakes in 1988 and have since spread to all the Great Lakes, as well as the major river systems in the Midwest (Hebert et al. 1991 as cited in USGS 2002). Introduction to the Great Lakes was likely a result of ballast water discharge; dispersal to river systems outside the Great Lakes may be a result of zebra mussels attaching to boats that are trailed from infested waters to other locations (USGS 2002). Under cool, humid conditions, zebra mussels can stay alive for several days out of water (USGS 2002), thus are capable of being transported long distances. During routine inspections at agricultural inspection stations, zebra mussels have been found attached to the hull or in the motor compartment of trailered boats crossing into California (USGS 2002). Many biological impacts of zebra mussels in North America are not yet known, primarily because many effects may still be developing (USGS 2002). However, zebra mussels have the potential to outcompete and eliminate native mussels (Nalepa 1994 and Schloesser and Nalepa 1994, as cited in USGS 2002), consume sizeable amounts of algae and increase water clarity, and alter macrophyte plant communities as a result of changes in water clarity (Skubinna et al. 1995 as cited in USGS 2002). In the Great Lakes, zebra mussels initially appear to be having little effect on fish populations, although it may be soon to determine because of their recent introduction (USGS 2002). Zebra mussels are well known for their ability and affinity to colonize and foul water supply pipes to many different types of industrial facilities; this colonization reduces effective pipe diameter and flow through these water pipes (USGS 2002). Although many methods have been used to control zebra mussels, each has varying levels of success under specific applications (USGS 2002).

2.6 Knowledge Gaps

There is an abundance of knowledge gaps in our current understanding of the physical processes of the estuary and lower mainstem and how these processes relate to the biological requirements of focal species. Faced with the challenge of recovering ESA-listed populations, recovery efforts need to progress in the face of uncertainty, recognizing our current limitations. Section 2.6.1 Uncertainty reminds us that there are many things we do not know regarding focal species relationships to the estuary and lower mainstem ecosystem. Section 2.6.2 Physical Process Models briefly describes the ongoing research efforts to increase our understanding of estuarine physical processes. Section 2.6.3 Current Research Needs identifies the future direction necessary to increase our understanding of the biological requirements of salmonids in relation to the estuary and lower mainstem ecosystem.

2.6.1 Uncertainty

Habitat requirements of non-salmonid fishes and wildlife focal species as they relate to Columbia River estuary habitat conditions and the processes that form and maintain those habitats are largely understudied. A considerable amount of information is available in the Pacific Northwest regarding habitat classification, habitat conservation, and wildlife-habitat relationships (Brown 1985, Ruggiero et al. 1991, WDNR 1996, WDNR 1998, Johnson and O'Neil 2001), however, none of these efforts have focused specifically on the interaction of wildlife focal species and the Columbia River estuary and lower mainstem. Gaumer et al. (1985) and Buchanon et al. (2001) generally discussed the dynamics of estuary habitats and relationship of different wildlife species to this habitat; again, the relationship of wildlife focal species and the Columbia River estuary were not specifically addressed.

Throughout this qualitative analysis, there are multiple inferences regarding the expected or likely relationship between salmonids and the habitat conditions or habitat-forming processes in the Columbia River estuary or lower mainstem. Much of what we know about the effects of changing habitat conditions on salmonid habitat requirements in the estuary is based on limited estuary-specific research or is speculative based on salmon and habitat relationships in non-tidal freshwater.

The issue of uncertainty is a significant challenge; as a result, US Army Corps of Engineers organized a workshop in March 2003 to review past and ongoing research in the Columbia River estuary, identify data gaps and key future research needs, and prioritize the identified research needs related to Columbia River salmonids (R2 2003). Although this workshop focused on salmonids, it is quite likely that many of these research needs apply to all focal species included in this assessment.

The key biological relationships in which we need a clearer understanding include:

- Specific relationships between salmonid life history strategies and habitat requirements, especially for ESA-listed species.
- Juvenile salmon usage and ecology in the tidal freshwater portion of the Columbia River estuary (i.e. Puget Island [rm 46] to Bonneville Dam [rm 146]).
- Specific linkages between biological and physical processes in various estuary habitat types.
- Inventory of current size, quality, and accessibility of habitat preferred by juvenile salmonids.

- Survival rates and growth indices for various salmonid life history types and the relationship to estuary habitats.
- Food web structure and linkages to estuary habitat types.
- Habitat forming processes required to maintain habitat types utilized by salmon.
- Relationship of structural and functional ecosystem components, including the natural variability associated with each.

2.6.2 Physical Process Models

Considerable effort has focused on developing predictive capabilities to describe the physical processes in the Columbia River estuary and lower mainstem. Considerable knowledge has been gained through this effort, however, connection of physical process models to biological requirements of salmonids and other focal species remain largely based on professional assumptions. The programs described below are not strictly physical process models, as identified in each program's description available on the internet. Note that the LCFRB was not involved in the development of the physical process models outlined here; these models are merely presented as an introduction to our current level of understanding of physical processes within the Columbia River estuary and lower mainstem and to highlight our current inability to connect physical process models with biological processes.

2.6.2.1 CORIE

The CORIE program is administered through the Oregon Graduate Institute, School of Science and Engineering, which is part of the Oregon Health and Science University. The following exerpts describing the CORIE program were taken directly from the CORIE website (http://www.ccalmr.ogi.edu/CORIE/):

> *CORIE is a pilot environmental observation and forecasting system (EOFS) for the Columbia River. It integrates a real-time sensor network, a data management system and advanced numerical models. Through this integration, we seek to characterize and predict complex circulation and mixing processes in a system encompassing the lower river, the estuary and the near-ocean. The acquired knowledge is transformed into data products designed to provide objective insights on the spatial and temporal variability of the Lower Columbia River.*
>
> *As a scientific tool, CORIE is designed to advance the emerging field of environmental information systems, and the understanding of river-dominated estuaries and plumes.*
>
> *The scientific objectivity and breadth of products of CORIE also gives the region's natural resource management and regulation community powerful new planning and analysis tools to improve policies and decisions.*
>
> *Early applications of CORIE have, in particular, addressed issues combining salmon habitat and passage, hydropower management, navigation improvements and habitat restoration. These applications show that there is a role for objective science to engender consensus across agencies with conflicting mandates. They also suggest that coordinating resources of multiple users of a waterway in the development of a shared scientific infrastructure, readily adaptable to evolving needs, might be a practical way to develop affordable management tools.*

Rapidly advancing performance and declining costs of electronic and computer technology will soon make EOFS economically feasible. The experience of systems like CORIE will encourage and provide paradigms for the development of national and international networks of EOFS, to the benefit of science and society.

The CORIE modeling system integrates models and field controls. Focus is on the simulation of 3D circulation, in a region centered in the estuary and plume, but extending from Bonneville Dam to the Eastern North Pacific.

CORIE simulations include (a) short term forecasts, (b) actual past conditions (referred to as hindcasts), (c) characteristic climatology conditions, and (d) scenario conditions. River, atmospheric, and ocean forcings are complied, in some cases in quasi-real time, from a variety of sources.

2.6.2.2 Columbia River Estuary Turbidity Maxima Research Project

The Columbia River Estuary Turbidity Maxima (CRETM) is a US National Science Foundation (NSF) Land-Margin Ecosystem Research (LMER) Project; the project is an ecosystem-scale, interdisciplinary investigation of the role of estuarine turbidity maxima (ETM) in shaping the food web of the Columbia River estuary. The following exerpt describing the CRETM program was taken directly from the CRETM website (http://depts.washington.edu/cretmweb/CRETM.html):

Our fundamental research goal is to understand how circulation phenomena in the estuary, called estuarine turbidity maxima (ETM), trap particles and promote biogeochemical, microbial and ecological processes that sustain a dominant pathway in the estuary's food web. To study this relationship between the physics of ETM and these various processes requires a resolutely interdisciplinary approach, and a complex, highly-orchestrated suite of field and laboratory measurements and experiments. The CRETM-LMER team involves scientists from six distinct disciplines-geophysics, sedimentology, geochemistry, microbiology, primary production biology, and zooplankton and food web ecology-to characterize ETM process. But, we depend upon hydrodynamic and ecosystem process modelers to help us synthesize our understanding about how the ETM and associated estuarine processes act as a "living" system that is fundamental to the way the estuary behaves.

2.6.3 Current Research Needs

A research, monitoring, and evaluation (RME) plan for the Columbia River estuary and plume was recently developed (Johnson et al. 2003a) for the purpose of fulfilling certain requirements of Reasonable and Prudent Alternatives 160, 161, and 163 of the 2000 Biological Opinion on the Operation of the Federal Columbia River Power System (NMFS 2000c). The three primary goals of this RME plan are: 1) Status Monitoring – quantify status/trends in listed salmonid usage/survival in the Columbia River estuary and plume, 2) Action Effectiveness – quantify effects of habitat restoration efforts on listed salmonids in the Columbia River estuary and plume, and 3) Uncertainties – resolve uncertainties regarding salmonid recovery efforts in the Columbia River estuary and plume (Johnson et al. 2003a). To the extent possible, future development of an RME plan for the lower Columbia River and estuary by the LCFRB should attempt to be consistent with and not duplicate the work of Johnson et al. (2003a).

During the recent lower Columbia River and estuary research needs workshop (R2 2003), research needs were categorized by priority and expected time needed for completion. Note that the LCFRB was not involved in the development of the research needs presented here, but we are simply presenting the findings of the collaborative workshop. Although this workshop focused on salmonids, it is quite likely that many of these research needs apply to all focal species included in this assessment. Four categories of research were identified: high priority/immediate, high priority/10-year window, high priority/long term, and medium priority. The following research topics were taken directly from the workshop proceedings report (R2 2003):

High priority research needs that could be addressed now include:

- *Move from a collection of available conceptual frameworks to an integrative implementation framework, where we combine what we have learned in the various conceptual frameworks to identify the most important areas for restoration actions, and what are the most likely avenues for success.*
- *Implement selected restoration projects as experiments, so that we can learn as we go.*
- *Implement pre- and post-restoration project monitoring programs, to increase the learning.*
- *"Mining" of existing, underutilized data to minimize the risk of collecting redundant or unnecessary data, and to compare with current and projected conditions.*
- *Make more use of ongoing PIT tagging and other tagging and marking studies and data to determine origin and estuarine habitat use patterns of different stocks.*
- *Collect additional shallow water bathymetry data for refining the hydrodynamic modeling, and identifying/evaluating potential opportunities for specific restoration projects.*
- *Determine operational and hydrologic constraints for the FCRPS, so that we have a better understanding of feasibility and effectiveness of modifying operations.*
- *Identify and implement off-site mitigation projects in CRE tributaries.*
- *Establish a data and information sharing network so that all researchers have ready and up-to-date access.*
- *Increased genetic research to identify genotypic variations in habitat use.*

High priority research needs that appear to be feasible within the present 10-year window of opportunity, but may not be implemented immediately or lead directly to projects in the near term include:

- *Understanding salmonid estuarine ecology, including food web dynamics.*

- *Understanding sediment transport and deposition processes in the estuary.*

- *Understanding juvenile and adult migration patterns.*

- *Identifying restoration approaches for wetlands and developing means for predicting their future state after project implementation.*

The following items were identified as high priority, but are considered long-term efforts (i.e. will likely take the longest to complete before a tangible product is developed):

- *Improve our understanding of the linkages between physical and biological processes to the point that we can predict changes in survival and production in response to selected restoration measures.*

- *Improve our understanding of the effect of toxic contaminants on salmonid fitness and survival in the CRE and ocean.*

- *Improve our understanding of the effect of invasive species on restoration projects and salmon and of the feasibility to eradicate or control them.*

- *Improve our understanding of the role between micro- and macro-detrital inputs, transport, and end-points.*

- *Improve our understanding of the biological meaning and significance of the Estuarine Turbidity Maximum relative to restoration actions.*

- *Identify end-points where FCRPS BO RPA action items are individually and collectively considered to be satisfied, so that the regulatory impetus is withdrawn.*

The following research needs were identified as medium priority (i.e. they may provide additional insights, but we currently have a reasonable idea of the most important features based on preceding work):

- *Increasing our understanding of how historical changes in the estuary morphology and hydrology have affected habitat availability and processes.*

2.7 Hypothesis Statements

The ultimate goal of this subbasin assessment is to assemble the technical information necessary to develop biological objectives for the Columbia Estuary and Lower Columbia Subbasins. The subbasin assessment concludes with the develop a working hypothesis that establishes the basis for the future management plan. The NPPC defines the working hypothesis as follows:

> *The working hypothesis is a collection of component hypotheses – a set of key assumptions that are based on assessment data and analysis. The overall working hypothesis describes a scientific understanding of the subbasin and contains the key assumptions relating to species-habitat relationships and/or the effectiveness of strategies to modify the elements of the environment. A working hypothesis summarizes a scientifically based understanding of the subbasin at the time the management plan is developed and begins to bridge the gap between the science and strategies. By developing a working hypothesis, you will have an explicit scientific rationale to considering alternative biological objectives and strategies. It will be used to evaluate and derive biological objectives and strategies in relation to the subbasin vision. Finally, the working hypothesis provides the elements necessary for scientific review of the subbasin plan by the Council and the Independent Scientific Advisory Board.*

The NPPC suggests that the working hypothesis is best developed around a scientific model such as Ecosystem Diagnosis and Treatment (EDT; NPPC 2001); however, EDT, or other similar models have not been parameterized for the estuary or lower mainstem. Therefore, in this assessment, hypotheses were developed based on scientific evidence and professional judgement. The hypothesis statements collectively represent our current understanding of the primary issues in the estuary and lower mainstem. Because the hypotheses are supposed to serve as the foundation of the management plan and directly link to biological objectives, in some cases the hypothesis statements needed to make a quantum leap to bridge the gap between our current level of understanding and the desired conditions in the subbasins.

As part of the implementation process of the subbasin plan, the working hypothesis will be tested and refined through research, monitoring, and evaluation. It is vital that subbasin planners reach an agreement on the working hypothesis, or set of alternative hypotheses, in order to develop the management plan. The following series of component hypothesis statements are intended to collectively serve as the NPPC 'working hypothesis' for the Columbia Estuary and Lower Columbia Subbasins based on the currently available scientific information. Note that the hypothesis statements do not take the classic form of a scientific hypothesis (i.e. if…then); they are formulated to address the NPPC hypothesis definition.

Hypothesis Statement 1 – Complex and dynamic interactions between physical river and oceanographic processes, as modulated by climate and human activities, affect the general features of fish and wildlife habitat in the Columbia River estuary and lower mainstem.

Habitat formation in the lower Columbia River mainstem and estuary is controlled by opposing hydrologic forces: ocean processes (tides) and river processes (discharge). These processes may be disturbed by storms, extreme hydrologic events, or catastrophic events such as earthquakes or volcano eruptions. Tides introduce marine-derived sediments to the estuary while river discharge carries freshwater sediments via bedload and suspended sediment. This supply of sediments influences the bathymetry of the estuary through the processes of erosion and accretion. Suspended sediment, along with the production of organic matter, determine the degree of water turbidity. The opposing processes of estuary outflow (river discharge) and inflow (tides) determine the salinity gradient and the type and location of available nutrients. River discharge also directly affects the level of woody debris recruitment to the estuary. Finally, the main components of the habitat formation process (bathymetry, water turbidity, salinity, nutrients, and woody debris) determine the location and type of habitats that form and persist throughout the estuary and lower mainstem.

As described in section 2.6.2, numerous on-going research projects are focused on describing the physical processes within the Columbia River estuary. For example, the CORIE program is an environmental observation and forecasting system for the Columbia River that seeks to characterize and predict complex circulation and mixing processes in the ecosystem encompassing the lower river, the estuary, and the near-ocean. Another project (CRETM) has focused its research efforts on understanding how circulation processes in the Columbia River estuary trap particles and promote biogeochemical, microbial, and ecological processes that comprise a dominant pathway in the estuarine food web.

Tide cycles (magnitude and periodicity) are natural processes that are partially influenced by storms and wind action but are largely beyond the dominion of human actions. However, the effects of tide cycles and tidal action have been altered by human intervention. For example, construction of the north and south jetties at the mouth of the Columbia River has decreased wave action in the lower river and has altered the hydrologic regime at the river/ocean interface; the result has been varying patterns of erosion and accretion compared to historical conditions.

River discharge is affected by precipitation, temperature, and water regulation/withdrawals. Sherwood et al. (1990) [as cited in Bottom et al. 2001] estimated that the 40% decrease in maximum spring freshet flow compared to historical conditions is because of water regulation (75%), irrigation withdrawal (20%), and climate change (5%). Changes in river discharge has decreased the freshwater-derived sediment supply and woody debris as well as altered the salinity gradient and nutrient distribution throughout the estuary (Sherwood et al. 1990 as cited in Williams et al. 2000, Bottom et al. 2001, USACE 2001). Artificial channel confinement has altered river discharge and hydrology, as well as disconnected the river from much of its floodplain, thereby eliminating much of the woody debris supply. Additionally, channel manipulations for transportation or development have also had substantial influence on river discharge and hydrologic processes in the river.

Evaluation of anthropogenic factors is complicated by climate effects. Variations in Columbia River discharge as a result of climate effects occur in time scales from years to centuries (Chatters and Hoover 1986, 1992 as cited in Bottom et al. 2001). The Columbia Basin’s climate response to climatic cycles is governed by the basin’s latitudinal position; climate in the region displays a strong response to both the PDO and ENSO cycles (Mantua et al. 1997 as cited in Bottom et al. 2001). The El Niño weather pattern produces warm ocean temperatures and warm, dry conditions throughout the Pacific Northwest. The La Niña weather pattern is typified by cool ocean temperatures and cool/wet weather patterns on land. Climate directly affects river flow and observed changes to flow are often substantial. Further, El Niño patterns result in poor ocean productivity in the Pacific Northwest and California, as was observed in the mid 1990s. The effects of poor estuary and mainstem habitats are exaggerated during periods of low ocean productivity.

Current climate projections predict gradual warming of the region, potentially with higher precipitation, particularly in winter (Hamlet and Lettenmaier 1999). The predicted future climate conditions will possibly reduce the likelihood of spring freshets caused by heavy spring rain on late snowpack because warmer temperatures will not allow the accumulation of snow late into the spring. This freshet style (rain on snow) has historically produced the most substantial increases in river discharge (Bottom et al. 2001). However, despite our ability to measure changes in climate, Bottom et al. (2001) discussed the difficulty in separating climate versus anthropogenic effects on river discharge and the habitat-forming processes it governs.

Hypothesis Statement 2 – Human activities have altered how the natural processes interact, changing habitat conditions for fish and wildlife in the Columbia River estuary and lower mainstem.

Anthropogenic factors have substantially influenced the current habitat conditions in the lower Columbia River mainstem and estuary. The primary anthropogenic factors that have determined estuary and lower mainstem habitat conditions include hydrosystem construction and operation (i.e. water regulation), channel confinement (primarily diking), channel manipulation (primarily dredging), and floodplain development and water withdrawal for urbanization and agriculture. Generally, these anthropogenic factors have influenced estuary and lower mainstem habitat conditions by altering hydrologic conditions, sediment transport mechanisms, and/or salinity and nutrient circulation processes. Often, there are no simple connections between a single factor and a single response, as many of the factors and responses are interrelated.

Flow effects from upstream dam construction and operation, irrigation withdrawals, shoreline anchoring, channel dredging, and channelization have significantly modified estuarine habitats and have resulted in changes to estuarine circulation, deposition of sediments, and biological processes (ISAB 2000, Bottom et al. 2001, USACE 2001, Johnson et al. 2003b). Flow regulation in the Columbia River basin has been a major contributor to the changes that have occurred in the estuary from historical conditions. The predevelopment flow cycle of the Columbia River has been modified by hydropower water regulation and irrigation withdrawal (Thomas 1983, Sherwood et al. 1990 as cited in Nez Perce et al. 1995, Weitkamp 1994, NMFS 2000c, Williams et al. 2000, Bottom et al. 2001, USACE 2001).

Flow regulation in the Columbia has decreased spring freshet magnitude and increased flows over the rest of the year as a result of winter drawdown of reservoirs and filling of the reservoirs during the spring runoff season. The historical flow records at the Dalles, Oregon, Bonneville Dam, and Beaver, Oregon, demonstrate that spring freshet flows have been reduced by about 50% and winter flows have increased about 30% (Figure 2-13, Figure 2-14, and Figure 2-15, respectively). Most of the spring freshet flow reduction is attributed to flow reduction, about 20% is a result of irrigation withdrawals, and only a small portion (5%) is connected to climatic change (Bottom et al. 2001).

Reduction of maximum flow levels, dredged material deposition, and diking measures have all but eliminated overbank flows in the Columbia River (Bottom et al. 2001), resulting in reduced large woody debris recruitment and riverine sediment transport to the estuary. Overbank flows were historically a vital source of new habitats. Moreover, historical springtime overbank flows greatly increased habitat opportunity into areas that at other times are forested swamps or other seasonal wetlands. Historical bankfull flow levels were common prior to 1975 but are rare today; current bankfull flows have only been exceeded four times since 1948 (Figure 2-16). Further, the season when overbank flow is most likely to occur today has shifted from spring to winter, as western subbasin winter floods (not interior subbasin spring freshets) are now the major source of peak flows (Bottom et al. 2001, Jay and Naik 2002).

Thomas (1983) suggested that channel confinement (i.e. diking) is particularly detrimental to estuary habitat capacity because it entirely removes habitat from the estuarine system, while other anthropogenic factors change estuary habitats from one type to another. The lower mainstem and estuary habitat in the Columbia River has, for the most part, been reduced to a single channel where floodplains have been reduced in size, off-channel habitat has been lost or disconnected from the main channel, and the amount of large woody debris has been reduced (NMFS 2000c). Dikes prevent over-bank flow and affect the connectivity of the river and floodplain (Tetra Tech 1996); thus, the diked floodplain is higher than the historical floodplain and inundation of floodplain habitats only occurs during times of extremely high river discharge (Kukulka and Jay 2003). It is estimated that the historical estuary had 75 percent more tidal swamps than the current estuary because tidal and flood waters could reach floodplain areas that are now diked or otherwise disconnected from the main channel (USACE 2001, Johnson et al. 2003b).

Development and maintenance of the shipping channel has greatly affected the morphology of the estuary. The extensive use of jetties and pile dikes to maintain the shipping channel has impacted natural flow patterns and large volumes of sediments are dredged annually. Dredged materials are disposed of in-water (in the ocean or in the flow adjacent to the shipping channel), along shorelines, or on upland sites. Dredge disposal in upland or deepwater sites reduces the amount of sediment available for habitat formation in the estuary as well as sediments that supply shoreline areas in the Columbia River littoral cell. Annual maintenance dredging since 1976 has averaged 3.5 million cubic yards per year in the estuary. By concentrating flow in one deeper main channel, the development of the navigation channel has reduced flow to side channels and peripheral bays.

Sediments in the estuary may be marine or freshwater derived; sediments are transported via sediment suspended in the water column or bed load movement. Riverine sediments available for transport has decreased as a result of dam construction: reservoirs restrict bedload movement and trap upstream supply of sediments. Sand sediments are vital to natural habitat formation and maintenance in the estuary; dredging and disposal of sand and gravel have been one of the major causes of estuarine habitat loss over the last century (Bottom et al. 2001). Conversely, the USACE (2002) suggests that sediment deposition conditions exist in the estuary, particularly shoaling in the navigation channel and deposition/accumulation of sand in low energy areas in the estuary and along the coast. Shoaling in the navigation channel is a redistribution of bed sediments, rather than an accumulation of sediments, because it does not change the volume of bed material within a given reach (USACE 2002).

Sediment transport is non-linearly related to flow; thus, it is difficult to accurately apportion causes of sediment transport reductions into climate change, water withdrawal, or flow regulation (Jay and Naik 2002). However, the largest single factor in reduced sediment transport appears to be the reduction of spring freshet flow as a result of water regulation and irrigation withdrawal. Recent analyses indicate a two-thirds reduction in sediment-transport capacity of the Columbia River relative to the pre-dam period (Sherwood et al. 1990, Gelfenbaum et al. 1999). Therefore, flow reductions affect estuary habitat formation and maintenance by reducing sediment transport (Bottom et al. 2001, USACE 2001). The reduction in sand and gravel transport has been higher (>70% reduction compared to predevelopment flow) than for silt and clay transport (Bottom et al. 2001), which has important implications for habitat formation and food web dynamics.

Construction of the north and south jetties significantly increased sediment accretion in marine littoral areas near the mouth of the Columbia River. Ocean currents that formerly transported Columbia River sediments along the marine littoral areas were disrupted as a result of jetty construction. Accretion, particularly in areas adjacent to the river mouth (i.e. Long Beach, Clatsop Spit), increased significantly in the late 1800s and early 1900s. Sediment accumulation rates have slowed since 1950, potentially as a result of reduced sediment supply from adjacent deltas or the Columbia River (Kaminsky et al. 1999). Because of the decreased sediment supply from the Columbia River and ebb-tidal deltas, recent modeling results indicate that the shorelines immediately north of the historical sediment source areas at the entrance to the Columbia River are susceptible to erosion in the future (Kaminsky et al. 2000).

River discharge, tidal processes, and channel depth determine the salinity gradient and the type and location of available nutrients. Altered estuary bathymetry and flow have affected the extent and pattern of salinity intrusions into the Columbia River; stratification has increased and mixing has decreased (Sherwood et al. 1990 as cited in Williams et al. 2000). The dependence of salinity intrusion on channel depth is strong; the controlling channel depth has doubled over the last 120 years. Bathymetric changes have likely caused the greatest changes in salinity intrusion and stratification, but reduced spring freshet flows have also substantially altered salinity intrusion length (Bottom et al. 2001). The combination of tidal energy and river discharge determine the location, size, shape, and salinity gradients of the Columbia River ETM; the organic matter accumulation and cycling associated with the ETM is especially important in the current imported microdetritus-based food web.

Industrial development in the lower Columbia River has resulted in pollutants accumulating in the estuary habitats; in general, contaminants affect survival by increasing stress, predisposing fish to disease, and interrupting physiological processes. Accumulation of contaminants in the lower mainstem and estuary have been exacerbated by tributary water quality problems (NMFS 2000c) and reduced peak and sustained flood flows in the lower river (Sherwood et al. 1990 as cited in Nez Perce et al. 1996). In the lower 150 miles of the mainstem Columbia River, many contaminants have been detected above guidance or regulatory levels for fish tissue, sediment, and water (Nez Perce et al. 1995, Tetra Tech 1996). However, two of the more widely known contaminants, DDT and PCBs, were much more prevalent in the lower Columbia River in the 1960s and early 1970s than they are today; their concentrations have continued to decline since 1972, when the use of DDT was banned (USACE 2001).

The degree to which habitat forming processes and anthropogenic factors determine the present day abundance of different habitat types depends on the habitat type and the processes by which they are formed. Further, total change in habitat acreage represents the sum of habitat loss and habitat formation throughout the estuary. Thus, the significance of loss of certain habitat types has been partially masked by the formation of these habitats elsewhere. Further, the geographic movement of estuary habitats is not clear from the quantification of total acreage change. For example, the total acreage of a certain habitat type within a particular estuary area may not have changed considerably from historical to current conditions, however, the location of this habitat type within the estuary area may be completely different.

Thomas (1983) documented substantial changes to estuary habitats from historical to current condtions as summarized below. Estuary-wide tidal marsh and tidal swamp acreage has decreased 43% and 77%, respectively, from 1870 to 1983 (Table 2-5), primarily as a result of dikes and levees that have disconnected the main channel from these floodplain habitats and also from water regulation that has decreased historical peak flows that previously provided water to these habitats. Losses of tidal marsh habitat has been most extensive in Youngs Bay, where a loss of over 6,000 acres was observed (Table 2-5). Extensive tidal swamp habitat losses have been observed in all estuary areas that this habitat was historically present (Table 2-5). Losses of medium and deep water habitat acreage have been less severe (25% and 7%, respectively; Table 2-5). Acreage of medium depth water habitat was lost in all areas of the estuary except the upper estuary, where a slight increase in acreage was observed; acreage loss was highest in the entrance, Cathlamet Bay, and Baker Bay areas of the estuary (Table 2-5). Similarly, deep water habitat acreage was lost in most areas of the estuary; losses were highest in the Baker Bay and upper estuary areas (Table 2-5). Meanwhile, approximately 1,700 acres of deep water habitat were added to the entrance area of the estuary (Table 2-5). The only estuary habitat type that realized a net increase in acreage from 1870 to 1983 was shallows/flats habitat (10%; Table 2-5). This increase in acreage was primarily a result of water regulation that has decreased historical peak flows that often eroded tidal flat habitats and also from decreased wave action and erosion after construction of the jetties at the mouth of the river. A substantial loss of shallows/flats habitat was observed in entrance area of the estuary; much of this habitat was converted to medium or deep water habitat. In total, 36,970 acres (23.7%) of the estuarine habitat acreage has been lost from 1870 to 1983. During this period, lost estuarine habitats were converted to the following non-estuarine habitats: developed floodplain (23,950 acres), natural and filled uplands (5,660 acres), non-estuarine swamp (3,320 acres), non-estuarine marsh (3,130 acres), and non-estuarine water (910 acres; Table 2-10).

Hypothesis Statement 3 – Although rates of obvious physical habitat change in the Columbia River estuary and lower mainstem have slowed in recent years, current physical and biological processes are likely still changing such that current habitat conditions represent a degraded state of equilibrium.

It is likely that the trends in wetland habitat loss have slowed in recent years; partially because much of the available habitat has already been removed and partially because current day development is highly scrutinized for potential effects on ESA-listed species and their habitats. Further, some restoration efforts are specifically focused on restoring or preserving tidal wetlands and other key salmon habitats, thus, the potential exists for reversing the habitat loss trend for this habitat type. Conversely, current water regulation practices continue to encourage the habitat-forming processes responsible for the 10% increase in tidal flat habitat.

Garono et al. (2003a) described the Columbia River estuary as “a shifting mosaic of land cover types”. Although Garono et al. (2003a) observed considerable movement from one habitat cover class to another from 1992 to 2000, specific wetland habitats were generally categorized as other wetland habitats while specific upland habitat classes remained within the general upland class (i.e. wetlands remained wetlands and uplands remained uplands, although dominant vegetation or other distinguishing characteristic may have changed). Further, Garono et al. (2003a) indicated that some of the observed habitat changes from 1992 to 2000 were likely a result of differences in mapping accuracy or were consistent with successional transition. Thus, most habitat changes in recent years can be characterized as an alteration of one wetland habitat type to another as opposed to the complete loss of wetland habitats that were observed historically.

The habitat alterations that have occurred since pre-development times have degraded the quality and quantity of habitat in the estuary and lower mainstem. Because this historical trend in habitat loss appears to have slowed recently, the estuary and lower mainstem habitat conditions are in a degraded state of equilibrium. This emphasizes the urgency of the current need to implement habitat restoration actions to reverse the trend of habitat loss.

Hypothesis Statement 4 – Our current understanding of the interrelationships among fish, wildlife, and limiting habitat conditions in the estuary and lower mainstem is not robust and does not offer sufficient resolution to allow managers to make informed decisions to benefit recovery and sustainability of natural resources.

Habitat requirements of non-salmonid fishes and wildlife focal species as they specifically relate to Columbia River estuary and lower mainstem habitat conditions and the processes that form and maintain those habitats are largely understudied. For example, Buchanon et al. (2001) generally discussed the dynamics of estuary habitats and relationship of different wildlife species to this habitat, however, this work was not specific to the Columbia River estuary.

Our current understanding of causal relationships between salmonids and the habitat conditions or habitat-forming processes in the Columbia River estuary or lower mainstem are only slightly clearer than that of wildlife or non-salmonid fishes. Much of what we know about the effects of changing habitat conditions on salmonid habitat requirements in the estuary is based on limited estuary-specific research or is speculative based on known salmon and habitat relationships in non-tidal freshwater. For example, researchers have developed considerable predictive capabilities to describe the physical processes in the Columbia River estuary and lower mainstem through projects such as CORIE or CRETM (section 2.6.2), however, connection of physical process models to biological requirements of salmonids and other focal species remain largely based on professional assumptions.

To address the issue of uncertainty, a scientific workshop was convened in March 2003 to review past and ongoing research in the Columbia River estuary, identify data gaps and key future research needs, and prioritize the identified research needs related to Columbia River salmonids (R2 2003). Although this workshop focused on salmonids, it is quite likely that many of these research needs apply to all focal species included in this assessment. Specific research needs that have repeatedly been identified include the need for: linkages of physical process models with biological processes, clearer understanding of sediment transport, hydrology, and bathymetry, connectivity of estuary habitats, connectivity of research efforts, and collaboration among researchers.

In summary, continued research is vital to the progress and success of restoration and recovery efforts in the Columbia River estuary and lower mainstem. Research and monitoring can provide a clearer understanding of the relationships between biological and physical processes in the estuary and lower mainstem; it also serves as a tool for evaluating and recalibrating implemented restoration and recovery actions. However, there is a limit to our ability to understand certain complex biological interactions as discussed below.

Hypothesis Statement 5 – Exotic species are capitalizing on the Columbia River estuary and lower mainstem habitats and they have impacted ecosystem processes and relationships.

The increasing predominance of exotic species in species assemblages indicates major changes in aquatic ecosystems (OTS 1993, Cohen and Carlton 1995, Smith 2001 as cited in Waldeck et al. 2003). Globally, there is an increasing rate of aquatic non-indigenous species introductions; this increase has been attributed to the increased speed and range of world trade, which facilitates the volume, variety, and survival of intentionally or unintentionally transported species. This observation appears to hold true in the Columbia River where fish introductions in the lower Columbia River increased in a linear fashion in the 1900s while non-indigenous invertebrate introductions seem to be increasing exponentially (Waldeck et al. 2003). The nature of exotic species introductions in the lower Columbia River are changing from the historical intentional introduction of game or food fish species to the unintentional introduction of species that have unknown or negative impacts on the ecosystem (Draheim et al. 2002). Future prevention of exotic species introductions is vital to maintaining the current balance of ecological relationships in the Columbia River estuary and lower mainstem.

The current biotic community in the Columbia River estuary and lower mainstem is fundamentally different today than it was historically because of the introduction of exotic species. All exotic species introductions in the lower Columbia River represent permanent alterations of the biological integrity of the ecosystem for numerous reasons: impacts of introduced species are unpredictable, introduced species alter food web dynamics, and introduced species are a conduit for diseases and parasites (Waldeck et al. 2003). Although the list of known exotic species in the lower Columbia River is currently greater than 70 (Draheim et al. 2002), limited information is available regarding the ecological interactions of many of these species.

Altered habitats in the Columbia River estuary and lower mainstem ecosystem as a result of hydrosystem development and water regulation have facilitated the successful establishment of aquatic non-indigenous species (Cordell et al. 1992 as cited in Draheim et al. 2002, Weitkamp 1994). The lower Columbia River ecosystem may still be adjusting to these major flow alterations and this adjustment period may benefit aquatic non-indigenous species (Weitkamp 1994).

There are many opposing philosophies regarding the control and/or eradication of exotic species based on differing political or social values. For example, some believe that introduced game fish should be removed from the Columbia River to restore the historical fish species assemblage, while others believe that introduced game fish should be protected and enhanced to ensure future social and economic benefits from recreational fisheries. Regardless of differing social values, there is often little that can be done to eradicate exotic species once a population has been established. The greatest success for removing exotic species occurs if the species is detected shortly after introduction and a population has not yet become established. Otherwise, the most we can generally expect from exotic species control efforts is to maintain the current community structure, attempt to limit the current abundance levels of exotic species, and diligently establish controls to prevent future exotic species introductions.

Hypothesis Statement 6 – Of all native fish and wildlife species utilizing the Columbia River estuary and lower mainstem habitat, salmonids appear the most distressed.

Despite substantial changes to the Columbia River estuary and lower mainstem ecosystem, many species have stable or increasing abundance trends. Some of these species may be considered a conservation concern as outlined in the body of this chapter. Regardless of their current abundance trend, implementation of an ecosystem-based approach to recovery of ESA-listed species indicates that an evaluation of effects of each recovery action on these species is warranted. The status and abundance trends of these species in the Columbia River estuary and lower mainstem is summarized below:

- The lower Columbia white sturgeon population is among the largest and most productive in the world. The deep water habitats in which sturgeon are commonly associated remain available throughout the lower mainstem and estuary. Hydrosystem development and operation 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). Further, sturgeon have demonstrated substantial variability in feeding locations; white sturgeon have potentially benefited from changes to the estuarine food web.

- NOAA Fisheries completed a status review for green sturgeon in 2003 and determined that listing under the Endangered Species Act was not warranted. Green sturgeon spend most of their life in near-shore marine and estuarine waters from Mexico to southeast Alaska (Houston 1988; Moyle et al. 1995). 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. These fish may be seeking warmer summer river waters in the northern part of their range.
- The northern pikeminnow population has flourished with habitat changes in the mainstem Columbia River and its tributaries. The highest density of northern pikeminnow in the mainstem Columbia River below the Snake River confluence is found in the lower mainstem from the Dalles to the estuary. A pikeminnow management program has been implemented in the Columbia and Snake rivers in an attempt to reduce predation mortality of juvenile salmonids by reducing numbers of the large, old pikeminnow that account for most of the losses. A bounty fishery program for recreational anglers is aimed at balancing pikeminnow numbers rather than eliminating the species and has also stimulated development of a popular fishery.
- Eulachon numbers and run patterns can be quite variable; low runs during the 1990's were a source of considerable concern by fishery agencies. Current patterns show a substantial increase in run size compared to the 1990's. The low returns in the 1990's are suspected to be primarily a result of low ocean productivity. Eulachon support a popular sport and commercial dip net fishery in the tributaries, as well as a commercial gillnet fishery in the Columbia. They are used for food and are also favored as sturgeon bait. Nevertheless, 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). Additionally, dredging has the potential to impact adult and juvenile eulachon (Larson and Moehl 1990); dredging operations in the lower Columbia River have made local substrate too unstable for the incubation of eulachon eggs. Thus, future dredging operations should be scheduled to avoid eulachon spawning areas during peak spawning times (Romano et al. 2002).
- Field observations and trapper data indicate the river otter population abundance in the lower Columbia River mainstem and estuary was relatively low in the early 1980s (Howerton et al. 1984); low abundance may be the normal equilibrium level for river otters in this region. River otters are concentrated in shallow water tidal sloughs and creeks associated with willow-dogwood and Sitka spruce habitats located primarily in the Cathlamet Bay area. Although dikes throughout the estuary have disconnected substantial amounts of side channel and floodplain habitats from the mainstem, the Cathlamet Bay area remains as one of the most intact and productive tidal marsh and swamp habitat

throughout the entire estuary. Further, because river otters are capable of traveling over land, it is not understood how the loss of habitat connectivity of side channel and floodplain habitat has affected species' behaviors such as foraging, resting, mating, and rearing. Contaminants in river otter tissue may have adverse physiological effects, however, data suggests that the effects may be temporary (Tetra Tech 1996).

- Habitat conversion, losses, and isolation coupled with the low productivity of the population are the currently the most important threats to Columbian white-tailed deer population viability. Nevertheless, the Columbian white-tailed deer population appears stable at low numbers and shows initial indicators of increasing abundance and productivity. In 1999, the USFWS proposed to delist the Columbian white-tailed deer throughout the entire range, however, public concern over delisting motivated USFWS to withdraw the delisting proposal. Columbian white-tailed deer are present in low-lying mainland areas and islands in the Columbia River upper estuary and along the river corridor. They are most closely associated with Westside oak/dry Douglas fir forest within 200m of a stream or river; acreage of this habitat type has decreased substantially from historical to current conditions. Restoration of contiguous preferred habitat is vital to population recovery.
- The Caspian tern breeding population in the estuary has increased significantly from historical to current conditions as a result of the formation of mid-channel islands, primarily from dredge spoil disposal. The largest breeding colony of Caspian terns in North America is currently located in the Columbia River estuary, a location where terns historically did not breed. Terns are a conservation concern because very few breeding colonies exist; thus, terns are susceptible to catastrophic events, disease, or other factors that may affect terns during the breeding season.
- The Washington and Oregon bald eagle populations were included for federal listing as endangered under the Endangered Species Act in 1978. In 1994, the USFWS proposed to reclassify the bald eagle from endangered to threatened throughout its range; this reclassification was finalized in 1995. In 1999, the USFWS proposed to delist the bald eagle throughout its range, however, this delisting has not been finalized. Bald eagle population in the Columbia River estuary and lower mainstem have suffered from low reproductive success because of contaminants in the ecosystem that have caused eggshell thinning. Despite this, the population has been slowly increasing, presumably as a result of adult recruitment from adjacent populations. Bald eagles are strongly associated with large trees during nesting, perching, and roosting; thus, the loss of mature forest habitats in the Columbia River estuary and lower mainstem has likely decreased the acreage of potential eagle territories.
- The osprey population along the lower Columbia River mainstem has increased slightly in recent years. Although forest habitats used for nesting have likely decreased, osprey have adapted to nesting on man-made structures. Contaminant levels in osprey tissue are high enough to result in decreased egg thickness, however, the increasing population in recent years suggests that young production is not a limiting factor.
- The lower Columbia River mainstem and estuary is not a historical breeding or overwintering area for sandhill cranes. Sandhill cranes currently do not breed in the area, but agricultural development throughout the lower Columbia River floodplain has attracted overwintering sandhill cranes. 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). Reclamation of agricultural land for habitat restoration projects may discourage overwintering by sandhill cranes, although future development of herbaceous wetlands may provide adequate winter habitat for sandhill cranes currently using the region.

- Within Washington, yellow warblers are apparently secure and are not of conservation concern; likewise, the red-eyed vireo is common, more widespread in northeastern and southeastern Washington, and not a conservation concern. The yellow warbler and red-eyed vireo are both riparian obligate species; warblers prefer shrub-dominated habitats and vireos prefer dense, closed canopy forests. Habitat alterations along the lower Columbia River corridor have likely been more damaging to the possible presence of red-eyed vireos as opposed to yellow warblers because dense riparian forests along the lower Columbia River are likely less abundant than shrub-dominated wetland habitat. However, there are no data to compared historical and current breeding populations in the Columbia River estuary and lower mainstem.
- The only non-salmonid focal species population currently experiencing a decreasing trend is that of Pacific lamprey. However, Pacific lamprey life history suggests that survival and production through the estuary has principally been unaffected by changing habitat conditions. For example, juvenile lamprey feeding during the outmigration is thought to be limited. The sand and silt substrates important to juvenile survival remain available. The estuary may provide juvenile lamprey with cues that facilitate successful adult return migrations, as has been observed in salmonids. Adults are expected to use the estuary and mainstem primarily as a migration corridor. The Columbia River estuary and lower mainstem altered habitat conditions is not expected to be the primary factor in declining Pacific lamprey populations.

Hypothesis Statement 7 – The Columbia River estuary and lower mainstem ecosystem is critical to expression of salmon life history diversity and spatial structure which support population resilience and production.

Estuaries have important impacts on juvenile salmonid survival. Estuaries provide juvenile salmonids an opportunity to achieve the critical growth necessary to survive in the ocean (Neilson and Geen 1986, Wissmar and Simenstad 1988 as cited in Nez Perce et al. 1995, Aitkin 1998 as cited in USACE 2001). Juvenile chinook salmon growth in estuaries is often superior to river-based growth (Rich 1920a, Reimers 1971, Schluchter and Lichatowich 1977). Estuarine habitats provide young salmonids with a productive feeding area, free of marine predators, where smolts can undergo physiological changes necessary to acclimate to the saltwater environment.

Juxtaposition of high-energy areas with ample food availability and sufficient refuge habitat is a key habitat structure necessary for high salmonid production in the estuary. In particular, tidal marsh habitats, tidal creeks and associated complex dendritic channel networks may be especially important to subyearlings as areas of both high insect prey density, and as potential refuge from predators afforded by sinuous channels, overhanging vegetation and undercut banks (McIvor and Odum 1988). Furthermore, areas of adjacent habitat types distributed across the estuarine salinity gradient may be necessary to support annual migrations of juvenile salmonids (Simenstad et al. *in press* as cited in Bottom et al. 2001). For example, as subyearlings grow, they move across a spectrum of salinities, depths, and water velocities. For species like chum and ocean type chinook salmon that rear in the estuary for extended time periods, a broad range of habitat types in the proper proximities to one another may be necessary to satisfy feeding and refuge requirements within each salinity zone. Additionally, the connectedness of these habitats likely determines whether juvenile salmonids are able to access the full spectrum of habitats they require (Bottom et al. 1998).

Juvenile salmonids must continually adjust their habitat distribution in relation to twice-daily tidal fluctuations as well as seasonal and anthropogenic variations in river flow. Juveniles have been observed to move from low-tide refuge areas in deeper channels to salt marsh habitats at high tide and back again (Healey 1982). These patterns of movement reinforce the belief that access to suitable low-tide refuge near marsh habitat is an important factor in production and survival of salmonid juveniles in the Columbia River estuary.

The importance of proximate availability of feeding and refuge areas may hold true even for species that move more quickly through the estuary. For example, radio tagged coho in Grays Harbor estuary moved alternatively from low velocity holding habitats to strong current passive downstream movement areas (Moser et al. 1991). Consistent with these observations, Dittman et al. (1996) suggest that habitat sequences at the landscape level may be important even for species and life history types that move quickly through the estuary during the important smoltification process, as salmon gather the olfactory cues needed for successful homing and these cues may depend on the environmental gradients experienced during migrations.

Hypothesis Statement 8 – Changes in Columbia River estuary and lower mainstem habitat have decreased the productivity of the ecosystem for salmonids and contributed to their imperiled status.

Natural and anthropogenic factors have negatively altered the habitat-forming processes, available habitat types, and the estuarine food web, resulting in decreased salmonid survival and production. Studies conducted by Emmett and Schiewe (1997) in the early 1980s have shown that favorable estuarine conditions translate into higher salmonid survival. The most significant habitat effects have resulted from modified river flow, channel manipulations, and contaminant effects. River flow, although influenced by many factors, will be discussed in detail in the next hypothesis statement addressing hydropower system effects; the other habitat effects will be addressed below.

Salmonid production in estuaries is supported by detrital food chains (Healey 1979, 1982). Therefore habitats that produce and/or retain detritus, such as tidal wetlands emergent vegetation, eelgrass beds, macro algae beds and epibenthic algae beds, are particularly important (Sherwood et al. 1990). Diking and filling activities in the estuary have likely reduced the rearing capacity for juvenile salmonids by decreasing the tidal prism and eliminating emergent and forested wetlands and floodplain habitats adjacent to shore (Bottom et al. 2001, NMFS 2000c). Dikes throughout the lower Columbia River and estuary have disconnected the main channel from a significant portion of the wetland and floodplain habitats. Further, filling activities (i.e. for agriculture, development, or dredge material disposal) have eliminated many wetland and floodplain habitats. Thus, diking and filling activities have eliminated the emergent and forested wetlands and floodplain habitats that many juvenile salmonids rely on for food and refugia, as well as eliminating the primary recruitment source of large woody debris that served as the base of the historical food chain. The current estuary food web is microdetritus based, primarily in the form of imported phytoplankton production from upriver reservoirs that dies upon exposure to salinity in the estuary (Bottom and Jones 1990 as cited in Nez Perce et al. 1995, Bottom et al. 2001, USACE 2001). The historical macrodetritus-based food web was distributed throughout the lower river and estuary, but the modern microdetritus-based food web is focused on the spatially confined ETM region of the estuary (Bottom et al. 2001). This current food web is primarily available to pelagic feeders and is a disadvantage to epibenthic feeders, such as salmonids (Bottom and Jones 1990 as cited in Nez Perce et al. 1995, Bottom et al. 2001, USACE 2001).

Habitat alterations in the lower Columbia River mainstem and estuary have increased the abundance of predators of juvenile salmonids (see Hypothesis Statement 11, section 0). Evidence suggests that predation related mortality of juvenile salmonids during outmigration is substantial, thereby limiting survival and abundance of salmonids.

Juvenile salmon collected by NOAA Fisheries at East Sand Island near the mouth of the Columbia River contained relatively high concentrations of DDTs and PCBs. Studies of sub-lethal exposure of juvenile salmon to contaminants in urban estuaries suggest that these contaminants could affect the survival, growth, and fitness of salmon (Casillas et al. 1996). Water quality issues could reduce productivity for species that make extensive use of estuarine habitats for rearing, such as ocean-type salmonids like fall chinook and chum salmon. Further, proposed future dredging operations in the lower Columbia River and estuary may locally force contaminants into the water column or expose contaminanted sediments, which may have detrimental effects if juvenile salmonids were present.

Additionally, the decreased habitat diversity and modified food web has decreased the ability of the lower Columbia River mainstem and estuary to support the historical diversity of salmonid life history types. Historically, chinook salmon in the Columbia River exhibited a wide diversity of life history types, using streams, rivers, the estuary, and perhaps the Columbia River plume as potential rearing areas. Bottom et al. (2001) identified several forms of ocean-type chinook life histories, based on the scale pattern, length, and time of capture data collected by Rich (1920). Wissmar and Simenstad (1998) and Bottom et al. (2001) suggest there may be as many as 35 potential ocean-type chinook salmon life history strategies. Bottom et al. (2001) suggested that human affects on the environment have caused chinook life history patterns to be more constrained and homogenized than historical data show. Most modern day ocean-type chinook fit into one of three groups: subyearling migrants that rear in natal streams, subyearling migrants that rear in larger rivers and/or the estuary, yearling migrants. Today, ocean-type chinook with estuarine rearing life histories are not a primary life history form observed by managers and resource users; most chinook are yearlings with a homogeneous size distribution. Abundance patterns of juvenile chinook in the estuary now reflect hatchery management practices more than historical migration behavior. Further, food availability may be negatively affected by the temporal and spatial overlap of juvenile salmonids from different locations; competition for prey may develop when large releases of hatchery salmonids enter the estuary (Bisbal and McConnaha 1998), although this issue remains unresolved (Lichatowich 1993 as cited in Williams et al. 2000).

Hypothesis Statement 9 – Construction and operation of the Columbia River hydropower system has contributed to changes in Columbia River estuary and lower mainstem habitat conditions that have reduced salmonid population resilience and inhibited recovery.

Contruction and operation of the hydropower system has had profound effects on Columbia River estuary and lower mainstem habitats. The primary effects of the hydropower system include decreased mean annual river flow, reversal of the historical hydrograph, reduction of the amount and type of sediments available for transport, and alteration of the type of nutrients and organic material available for transport.

Hydrologic effects of the Columbia River hydrosystem include water level fluctuations, altered seasonal and daily flow regimes, reduced water velocities, and reduced discharge volume. Altered flow regimes can affect the migratory behavior of juvenile and adult salmonids. For example, water level fluctuations associated with hydropower peak operations may reduce habitat availability, inhibit the establishment of aquatic macrophytes that provide cover for fish, and strand juveniles during the downstream migration. Reservoir drawdowns reduce available habitat which concentrates organisms, potentially increasing predation and disease transmission (Spence et al. 1996 as cited in NMFS 2000c).

Water regulation, as part of hydropower system operations, has drastically reduced historical spring freshet flows and altered juvenile salmon outmigration behavior. Often, historical lower Columbia River spring freshet flows were approximately four times the winter low flow levels. Today, spring freshet flows are only about twice the winter low flow level, which is now generally higher as a result of reservoir drawdown in winter. Spring freshets are very important to the outmigration of juvenile salmonids; freshet flows stimulate salmon downstream migration and provide a mechanism for rapid migrations. Also, spring freshets (especially overbank flows) provide habitat, increase turbidity thereby limiting predation, and maintain favorable water temperatures during spring and early summer. Further, organic matter supplied by the river during the freshet season is a major factor maintaining the detritus-based food web. Today, the contribution of imported detritus is controlled primarily by reservoir production and flow rates from Bonneville Dam.

Because of changes to flow and sediment transport and the various habitat alterations that have occurred in the estuary, the availability of shallow (10cm-2m depth), low velocity (<30 cm/s) habitats appears to decrease at a steeper rate with increasing flow compared to historical conditions. These conditions have decreased the shallow water refugia for juvenile salmonids and likely contribute to decreased survival during high flow conditions (NMFS 2000c).

Altered flow regimes can also affect the spawning success of mainstem Columbia River spawners. For example, reservoir drawdowns in the fall for flood control produces high flow for fall spawners; fish may spawn in areas that are dewatered during the winter or spring, potentially resulting in complete egg mortality (NMFS 2000c).

Historically, floodwaters of the Columbia River inundated the margins and floodplains along the estuary, allowing juvenile salmon access to a wide expanse of low-velocity marshland and tidal channel habitats (Bottom et al. 2001). Flooding occurred frequently and was important to habitat diversity and complexity. Historical flooding also allowed more flow to off channel habitats (i.e. side channels and bays) and deposited more large woody debris into the ecosystem. Historically, seasonal flooding increased the potential for salmonid feeding and resting areas in the estuary during the spring/summer freshet season by creating significant tidal marsh vegetation and wetland areas throughout the floodplain (Bottom et al. 2001). These conditions rarely exist today as a result of hydropower system water regulation.

Columbia River mainstem reservoirs trap sediments and nutrients, as well as reduce sediment bedload movement, thereby reducing sediment and nutrient supply to the lower Columbia River. The volume and type of sediment transported by the mainstem Columbia River has profound impacts on the estuary food web and species interactions within the estuary. For example, organic matter associated with the fine sediment supply maintains the majority of estuarine secondary productivity in the food web (Simenstad et al. 1990, 1995 as cited in Bottom et al. 2001). Also, turbidity (as determined by suspended sediments) affects estuary habitat formation, regulates primary production via affects on light penetration, and decreases predation on juvenile salmonids via decreased predator efficiency. Further, the type of sediment transported has profound effects on habitat formation. The reduction in sand and gravel transport has been higher (>70% reduction compared to predevelopment flow) than for silt and clay transport (Bottom et al. 2001). Sand and gravel substrates are important components of preferred salmonid habitat in the estuary.

Hypothesis Statement 10 – Predation has always been a significant source of juvenile salmonid mortality in the lower Columbia River mainstem and estuary but habitat changes resulting from human activities have substantially altered predator concentration and distribution, particularly Caspian terns and northern pikeminnow.

Significant numbers of salmon are lost to fish, bird, and marine mammal predators during migration through the mainstem Columbia River. Predation has always been a substantial source of mortality but is expected to have increased significantly in recent years because of increased abundance of predator populations that have responded to habitat changes resulting from human activities. For example, piscivorous birds congregate near dams and in the estuary around man-made islands and consume large numbers of outmigrating juvenile salmon and steelhead (Roby et al. 1998). Caspian terns, cormorants, and gull species are the major avian predators (NMFS 2000a). While some avian predation occurs at dam tailraces and juvenile bypass outfalls, by far the greatest numbers of juveniles are consumed as they migrate through the Columbia River estuary.

Caspian terns are native to the region but were not historically present in the lower Columbia River mainstem and estuary; they have recently made extensive use of dredge spoil habitat and are a major predator of juvenile salmonids in the estuary. The terns are a migratory species whose nesting season coincides with salmonid outmigration timing. Since 1900, the tern population has shifted from small colonies nesting in interior California and southern Oregon to large colonies nesting on dredge spoil islands in the Columbia River and elsewhere (NMFS 2000c). Caspian terns did not nest the estuary until 1984 when about 1,000 pairs apparently moved from Willapa Bay to nest on East Sand Island. Those birds (and others) moved to Rice Island (constructed from dredge spoils) in 1987 and the colony expanded to 10,000 pairs. Diet analysis has shown that juvenile salmonids make up 75% of food consumed by Caspian terns on Rice Island. However, there are no data to compare historical and modern predation rates or predator populations; thus, effects of this unique predator population in relation to historical losses of juvenile salmon to predation cannot be adequately quantified (Bottom et al. 2001). Further, recent management actions have been relatively successful in discouraging Caspian tern breeding on Rice Island while encouraging breeding on East Sand Island, which may decrease predation on juvenile salmonids. Also, current predation studies are limited because of the unknown effects hatchery rearing and release programs have had on salmon migration behavior and predator consumption. Nevertheless, evidence suggests that current predator populations could be a substantial limiting factor on juvenile salmon survival (Bottom et al. 2001).

Native fishes, particularly northern pikeminnow, prey on juvenile salmonids during outmigration. Pikeminnow numbers likely have increased as favorable slack-water habitats have been created by hydropower system water impoundment and flow regulation. In unaltered systems, pikeminnow predation is limited by smolt migratory behavior; the smolts are suspended in the water column away from the bottom and shoreline habitats preferred by pikeminnow. However, dam passage has disrupted juvenile migratory behavior and provided low velocity refuges below dams where pikeminnow gather and feed on smolts. The diet of the large numbers of pikeminnow observed in the forebay and tailrace of Bonneville Dam is composed almost entirely of smolts. Pikeminnow also concentrate at dam bypass outfalls and hatchery release sites to prey on injured or disoriented fish, and pikeminnow eat many healthy smolts as well. Northern pikeminnow have been estimated to consume millions of juvenile salmon per year in the lower Columbia River; an estimated 9.7 million juvenile salmonids are consumed annually from Bonneville Dam to the estuary (NMFS 2000b). Predation rates on salmonids are often much lower in areas away from the dams, although large numbers of predators in those areas can still impose significant mortality.

Hypothesis Statement 11 – Density dependent factors might affect salmonid productivity in the Columbia River estuary and lower mainstem under some conditions, but their current significance is unclear.

The productivity of the Columbia River estuary likely has decreased over time as a result of habitat degradation, which initially would appear to increase the likelihood for competition among salmonids in the estuary especially during times of high juvenile abundance. In situations of decreased habitat availability, reducing access to habitat at critical stages may be a limiting factor in production and recovery of depressed salmonid populations (Fresh et al. 2003). However, historical natural abundance of juvenile salmonids in the lower mainstem and estuary was far greater than the current abundance, even considering the large hatchery releases of juvenile salmonids that occur today. Thus, at our current level of understanding, the importance of density dependent mechanisms in the estuary, if they exist, are not clear.

Recent research in the Skagit River, WA, suggests that density dependent mechanisms are operating in the estuarine portion of that system (Greene et al. *in press* as cited in Fresh et al. 2003). For example, research has identified a density dependent limit to the number of juveniles in the estuary relative to the abundance of juvenile salmonids in the entire system. Greene et al. (*in press* as cited in Fresh et al. 2003) further demonstrated that variability in nearshore Puget Sound conditions (i.e. extension of the Skagit Bay estuary) accounted for significant variability in adult returns of Skagit Bay chinook salmon; moreover, incorporating density dependence helped to clarify the relationship between nearshore conditions and adult returns. Although research in the Skagit River estuary points toward density dependent mechanisms, applicability to the Columbia River estuary is unknown; Fresh et al. (2003) indicated that this information was forthcoming for the Columbia River estuary.

Estuaries may be "overgrazed" when large numbers of ocean-type juveniles enter the estuary en masse (Reimers 1973, Healey 1991). Food availability may be negatively affected by the temporal and spatial overlap of juvenile salmonids from different locations; competition for prey may develop when large releases of hatchery salmonids enter the estuary (Bisbal and McConnaha 1998), although this issue remains unresolved (Lichatowich 1993 as cited in Williams et al. 2000). Reimer (1971) suggested a density dependant mechanism affects growth rate and hypothesized that fall chinook growth in the Sixes River was poor from June to August because of the large population in the estuary at this time and that the increased growth rate in September to November resulted from reduction in population size and a better utilization of the whole estuary.

The potential exists for large-scale hatchery releases of fry and fingerling ocean-type chinook salmon to overwhelm the production capacity of estuaries (Lichatowich and McIntyre 1987). However, Witty et al. (1995) could not find any papers or studies that evaluated specific competition factors between hatchery and wild fish in the Columbia River estuary. Rivers such as the Columbia, with well-developed estuaries, are able to sustain larger ocean-type populations than those without (Levy and Northcote 1982).

Natural populations of salmon and steelhead migrate from natal streams over an extended period (Neeley et al. 1993; Neeley et al. 1994); they also enter the estuary over an extended period (Raymond 1979). Hatchery fish are generally—but not always—released over a shorter period resulting in a mass emigration into natural environments. Managed releases of water combined with large releases of hatchery fish result in large numbers of juvenile salmon and steelhead in the estuary during spring months when the estuary productivity is low. Some workers (Reimers 1973; Neilson et al. 1985) have suggested that the amount of time spent in estuaries may relate to competition for food. Fish that arrive in the estuary later in the season may benefit from increased food supplies. Chapman et al. (1994) note that subyearling chinook released later in the summer returned at significantly higher rates than subyearlings released early in the summer.

In summary, the existence of density dependent mechanisms among salmonids in the Columbia River estuary and lower mainstem are equivocal. Although capacity of the estuary to support juvenile salmonids has decreased from historical conditions (see section 2.4.2.8), abundance of salmonids has also decreased substantially. To date, we have limited ability to quantify the lower mainstem and estuary capacity and, therefore, have limited knowledge of how many salmonids can be present in the estuary/mainstem at any given time (i.e. different seasons, flow conditions, nutrient levels, macroinvertebrate abundance, etc.) before significant competition for resources results. It is clear that the capacity of the estuary/mainstem ecosystem has decreased relative to historical conditions, however, it is not clear whether this decreased habitat capacity has resulted in density dependent mechanisms that limit salmonid production at current salmonid abundance levels.

Hypothesis Statement 12 – Habitat restoration efforts are capable of significantly improving conditions for fish and wildlife species in the Columbia River estuary and lower mainstem.

Habitat actions proposed in the NMFS Biological Opinion on the Operation of the Federal Columbia River Power System (BiOp; NMFS 2000c) are intended to accelerate efforts to improve survival in priority areas while laying the foundation for long-term habitat strategies. The overarching objectives of the habitat strategy are: protect existing high quality habitat, restore degraded habitats and connect them to functioning habitats, and prevent further degradation of habitat and water quality. Specifically, Reasonable and Prudent Alternative (RPA) Actions 158 through 163 of the BiOp detail specific actions related to estuarine habitat while RPA Actions 156 and 157 address habitat issues within the lower mainstem (NMFS 2000c). An "Action Plan" has recently been published that outlines a plan for implementing the above RPA actions related to estuary and mainstem habitat restoration, as well as RPA actions that address planning, modeling, and research, monitoring, and evaluation needs described in the BiOp (BPA and USACE 2003).

Restoration of tidal swamp and marsh habitat in the estuary and tidal freshwater portion of the lower Columbia River has been identified as an important component of current and future salmon restoration efforts. Reasonable and Prudent Alternative Action 160 in the NMFS Biological Opinion on the Operation of the Federal Columbia River Power System (NMFS 2000c) called for an estuary restoration program with the goal of protecting and enhancing 10,000 acres of tidal wetlands and other key habitats over 10 years, beginning in 2001, with the intention of rebuilding productivity for ESA-listed salmonid populations in the lower 46 miles of the Columbia River. There is considerable uncertainty whether the 10,000 acres is the precise amount needed to produce desired increases in salmonid productivity or if the 10-year schedule is an appropriate time scale for recovery efforts. Thus, NMFS (2000c) identified the importance of continued monitoring and evaluation of the estuary restoration program and the 10,000-acre goal to ensure that habitats being restored are important for salmon survival and recovery. NMFS (2000c) also suggested examples of acceptable habitat improvement efforts, including but not limited to: acquiring diked lands, breaching levees, improving plant communities, reestablishing flow patterns, or enhancing connections between lakes, sloughs, side channels, and the main channel.

Dike removal could provide a sizable increase in shallow water habitat, even without restoration of historical flow regimes (Kukulka and Jay 2003). Dike removal alone provided more of an increase in shallow water habitat than flow restoration without dike removal. Restoration of natural flows increases the duration of shallow water habitat inundation in high-flow years, but individually does not restore the large size of the area historically inundated.

Management actions that seek to alter anthropogenic factors and restore natural habitat-forming processes need to be evaluated based on their impact on biological diversity and not simply on production of juvenile salmonids (Bisbal and McConnaha 1998). For example, changes in hydrosystem water management should attempt to provide benefits for the full range of historical salmonid life history patterns and not just the primary life history patterns currently observed. Restoration efforts need to move from the practice of management for average biological conditions to management for the full spectrum of possible biological variation (Williams et al. 1996 as cited in Bisbal and McConnaha 1998).

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