A REVIEW OF THE EFFECTS OF DAMS ON THE COLUMBIA RIVER ESTUARINE ENVIRONMENT, WITH SPECIAL REFERENCE TO SALMONIDS

by

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EXECUTIVE SUMMARY

Assessing the effects of dams on the Columbia River estuary is complicated. Most natural or anthropogenic processes in the estuary are highly interactive and dynamic, such that the specific role of a single process may change over time and location, and is consequently often difficult to assess. Numerous factors in addition to dams have affected the estuary, most importantly navigational improvements, diking, and increases in the human population, all of which may cause effects similar to those caused by dams. Aside from long-term catch data for commercially important species, there are essentially no biological records from the estuary until after nine dams were already constructed on the river. This means that almost all postulated biological changes in the estuary are based on speculation rather than on results of scientific investigations, making the ecological significance of these changes very difficult to determine. From existing literature, an attempt has been made to identify the potential impacts of dams on the estuary. However, given the above-mentioned complications, most of the impacts we describe should be considered as pathways through which dams may have impacted the estuary, rather than specific effects caused by dams.

Dams are thought to affect the physical environment of the estuary primarily via flow regulation. Flow regulation circumvents floods, which are important physical and biological structuring mechanisms in riverine systems by transporting large amounts of sediment through the estuary, providing physical energy for circulation, and promoting biological production. With the suppression of large floods by dams, downstream sediment transportation decreases, *in situ* estuarine production may decline, and the importance of floods as an evolutionary selective pressure diminishes. In the Columbia River estuary, the suppression of large floods

is thought to be partially responsible for the currently high accretion[1] rate in the estuary. Suppression of floods also decreases the fluvial energy available for water movement and alters circulatory patterns and salinity intrusion. Because the intrusion of salt water into the estuary depends on the amount of fresh water resisting the salt water, decreased maximum flows and increased minimum flows regulated by dams have decreased the seasonal variability of saltwater intrusion.

Decreased variability in saltwater intrusion affects the distribution of most estuarine organisms because their distributions are determined primarily by salinity tolerance. This may have allowed range extensions or altered the distributions of many species because areas that were formerly subjected to seasonally intolerable salinity levels for those species would now be habitable throughout the year. In contrast, species which formerly held a competitive advantage because of their tolerance to highly variable salinity levels may have lost this advantage, decreasing their distribution. Suppression of floods may also partially stabilize the population fluctuations of weak swimmers, such as zooplankters and larval fish, that would otherwise be periodically transported out of the estuary. In this respect, the selective pressure for maintaining a population in the estuary in the face of large floods has decreased, and populations of species formerly maintained at low levels because of periodic flushing from the estuary may expand.

Less variation in the range of salinity and the decreased ability needed to withstand large floods may have also benefitted exotic species, which may not be able to tolerate the environmental conditions in the unaltered estuary. Although the impact of exotic species in

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[1]The term *accretion* refers to the build-up or accumulation of sediments.

the Columbia River estuary has not been examined, exotic species introduced into other ecosystems have had significant and often detrimental effects on the native flora and fauna.

With regard to salmonids (*Oncorhynchus* spp.), the effects of altered salinity regimes in the estuary are expected to be minor, or within the limits of natural variation. Juvenile salmonids are highly mobile, and should be able adjust to the altered regimes. Juvenile salmonids in the Columbia River (many of which are of hatchery origin) generally spend only a few days in the estuary as they migrate downstream, so the altered salinity patterns would have little impact on them. For salmonids residing in the estuary for longer periods, such as some subyearling chinook salmon (*O. tshawytscha*), altered salinity regimes may have a greater impact on their less-mobile prey than on the fish themselves. However, a primary prey, the amphipod *Corophium salmonis*, is fairly mobile, and has remained highly prolific in the present-day, altered estuary.

In addition to their affects on flow and salinity, dams impact estuarine water quality, although the degree of impact and subsequent biological effects are unknown, and high water volumes in the river strongly moderate water quality impacts. Activities dependent on dams, such as irrigation and industries requiring large amounts of electricity, contribute to water-quality degradation by introducing contaminants into the river. These activities and the dams themselves, with their extensive reservoir storage, may also affect water temperature. Both water-quality degradation and water-temperature changes potentially affect the growth rates and health of fishes in the estuary, including those juvenile salmonids (subyearling chinook salmon) with longer residence times in the estuary, for which estuarine growth is critical to long-term survival. Dams are thought to affect water quality both directly and indirectly;

however, the specific contribution of dams to present-day water-quality conditions in the estuary and the ecological significance of that contribution are not documented.

Dams may be beneficial to some invertebrate and fish species in the Columbia River estuary because of the increased amount of plankton produced in reservoirs, which enters the estuary from upstream. Plankton has increased approximately two and a half times since the 1870s (Sherwood et al. 1990). This imported material consists primarily of phytoplankton, which lyses on contact with salt water in the estuary, providing an abundant source of microdetritus. Fauna that utilize the pelagic component of the estuarine food web, such as the exotic American shad (*Alosa sapidissima*), probably received the greatest benefit from increased phytoplankton importation from the trophic linkage between phytoplankton and zooplankton. However, species that utilize the benthic, macrodetritus-based, component of the food web, such as juvenile salmonids, likely do not benefit from the increased importation of phytoplankton.

There appears to be no apparent effects of regulated river flow, water quality alterations, or other alterations associated with dams on the offshore portion of the estuary in the plume. This lack of observed impact is primarily due to factors independent of river flow which dominate physical and biological processes offshore. These factors produce sufficiently high variability in the plume to mask the relatively subtle effects expected from dams-related impacts.

In conclusion, the largest impacts of dams on the inland portion of the Columbia River estuary appear to result from riverflow regulation, which has altered sediment transportation rates and salinity intrusion, both of which affect the estuarine community. High production of plankton in reservoirs has benefitted the pelagic portion of the food web in the estuary.

Dams are expected to have little impact on salmonids in the estuary; juvenile salmonids, many of which are of hatchery origin and spend little time in the estuary, should be able to adapt to the resulting physical changes in the estuary. Dams may also impact water quality, although the relative degree of impact has not been documented. Without further studies of present-day physical processes and biotic interactions in the Columbia River estuary, and without a large and accurate historic database, the true impact of dams on the Columbia River estuary will remain largely unsubstantiated and unquantifiable.

INTRODUCTION

This report analyzes existing information to determine the effects of dams[2] on the physical and biological characteristics of the Columbia River estuary, especially with regard to anadromous salmonids (Fig. 1). The Columbia River estuary is defined as the region of the Columbia River from the east end of Puget Island (approximately River Km [RKm] 75) to, and including, the plume offshore. To accomplish this objective, we first ascertained the physical and biological changes which have been documented or suggested for the estuary, and then attempted to determine the role of dams in effecting those changes. This effort was confounded by several factors. First, there is a lack of historical physical and biological data collected before the river and its estuary were altered by man. For example, water-temperature records for the estuary are good since 1977 (U.S. Geological Survey [USGS] 1978), but prior to 1977, especially prior to the completion of the first mainstem dam on the river in 1933, the records are difficult to find or non-existent. The paucity of data makes it especially difficult to evaluate the biological changes in the estuary suggested by the literature. Aside from catch records for commercially important species such as chinook salmon (*Oncorhynchus tshawytscha*), which began in the 1860s (Craig and Hacker 1940), there were no ecological investigations of the estuary until after nine dams had already been completed, and other extensive anthropogenic alterations such as navigational improvements and extensive diking occurred in the estuary (Haertel and Osterberg 1967). Although the long-term catch records have been extremely useful for estimating historic runs of commercially important species (Chapman 1986, Oregon Department of Fish and Wildlife

[2]The term *dams* includes the physical presence of mainstem dams, the operation of the hydropower system, reservoir storage, and water withdrawal associated with dams.

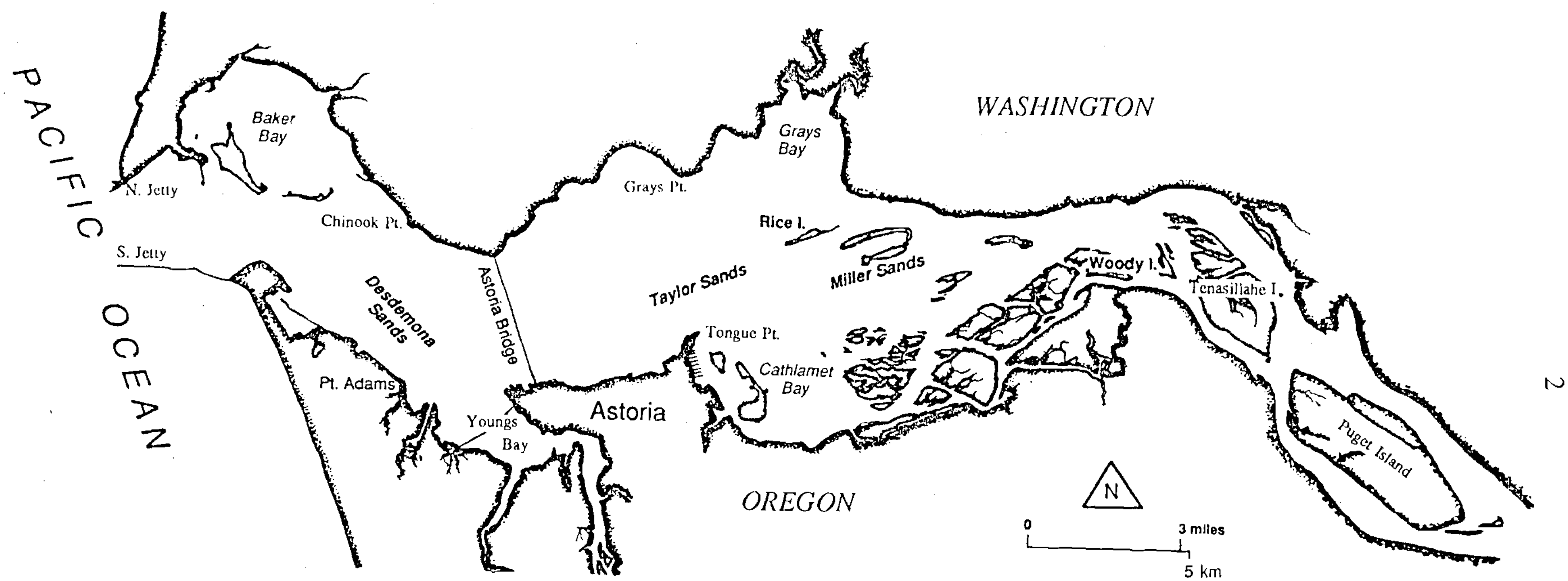

Figure 1.--The Columbia River estuary, showing major landforms and place names.

[ODFW] and Washington Department of Fisheries [WDF] 1991), any discussion of historical ecological changes in the estuary is completely speculative (Sherwood et al. 1990).

A second hindrance to determining the role of dams on the estuary is that the estuary is physically and biologically an extremely complex and dynamic system. Water is constantly moving about the estuary in response to daily tidal cycles and variations in seasonal river flow, while organisms are constantly entering and exiting the estuary from both the river and ocean. These organisms may remain in the estuary from a few hours to a lifetime. Most physical attributes and processes in the estuary function over a wide range of spatial and temporal scales and interact to create a complex, constantly changing environment that strongly determines the biotic community. Most physical and biological processes in the estuary result from the interaction of several attributes and processes, as opposed to a single attribute or process. For example, circulation in the estuary results from the interaction of tide, river flow, and bathymetry (or bottom topography), and varies with hourly, daily, monthly, and seasonal cycles. Circulation in turn affects bathymetry and sedimentation, which strongly influence the distribution of organisms in the estuary. Because of this high degree of interaction in the estuary, it is difficult to single out the impacts of individual factors or processes or the relative importance of a single factor. Consequently, it is difficult to determine the effects of dams on physical and biological characteristics of the estuary because dams produce effects that are often similar to the effects of other factors or natural forces.

A third difficulty in determining the role of dams in effecting changes in the Columbia River estuary is that dams are only one of several changes that have occurred in the Columbia River basin. Dredging and filling for navigational improvements, the rapid growth in human population, overfishing, and natural climatic variation are a few of the other factors which have also impacted the estuary. Because these other factors often cause changes very similar to those produced by dams, it is difficult to tell whether dams had a role in an observed or suggested change, and to what degree the change is attributable to dams. This is especially true for cumulative effects, since the net result of numerous effects can be greater than the sum of its individual components.

Finally, dams have both direct and indirect impacts on the Columbia River estuary, and the indirect impacts of some factors exceed those of the direct effects (Sylvester 1958). For example, the current, extensive irrigation in the Columbia River basin would not have been possible without dams to provide water storage. Similarly, some industries in the basin might have been located elsewhere were it not for the electricity and plentiful water provided by the hydropower system. The demographic consequences (i.e., increases in human population and activities) of these agricultural and industrial activities, in addition to the activities themselves, have had a large impact on water quality; potentially as much as the dams themselves. However, it is nearly impossible to determine how much of the observed water-quality degradation is due to activities dependent on dams, and how much is due to human activities not dependent on dams.

This report is presented in four sections. The first section reviews the major sources of change, including dams, which may have impacted the Columbia River estuary. The second discusses the physical characteristics of the estuary: how it has changed in historical times, the sources of change, and the role of dams in effecting change. The third section examines the biological characteristics of the estuary, including the major taxonomic groups, ecological processes and food webs, and how dams may have affected the ecosystems of the estuary. The last section is a synopsis of the effects of dams on the Columbia River estuary. References not cited in this report but which may provide additional information are included as supplemental references.

SOURCES OF CHANGE IN THE COLUMBIA RIVER ESTUARY

Anthropogenic Changes

Dams

Fourteen hydropower dams have been constructed on the mainstem Columbia River, while seven large hydropower dams have been constructed on the mainstem Snake River (Fig. 2). In addition, over 150 dams have been constructed on the tributaries throughout the Columbia River basin (Columbia River Water Management Group [CRWMG] 1982). The first tributary dam was completed in 1890; the mainstem dams were completed between 1933 and 1983 (CRWMG 1982). Dams and their reservoirs have been implicated in causing numerous, often deleterious, environmental effects such as reducing water quality, altering water temperatures and river-flows, blocking upstream and downstream fish passage, and delaying downstream migration of juvenile fishes (Ebel and Raymond 1976, Stober et al. 1979, Ward and Stanford 1987).

Irrigation

Irrigation in the Columbia River basin began in the 1840s and has grown dramatically. The area of land under irrigation increased from 2,000 square kilometers (km^2) in 1900 to 31,600 km^2 in 1980 (Sylvester 1958, Sherwood et al. 1990), with a total annual water withdraw of 13,300 million cubic meters (m^3) in 1985 (CRWMG 1986). The majority of irrigated land in the Columbia River basin is in the Snake River and Yakima River watersheds (Stober et al. 1979). Irrigation depletes river water, and irrigation return flows decrease water quality by increasing chemical constituents and raising the river temperature (Stober et al. 1979).

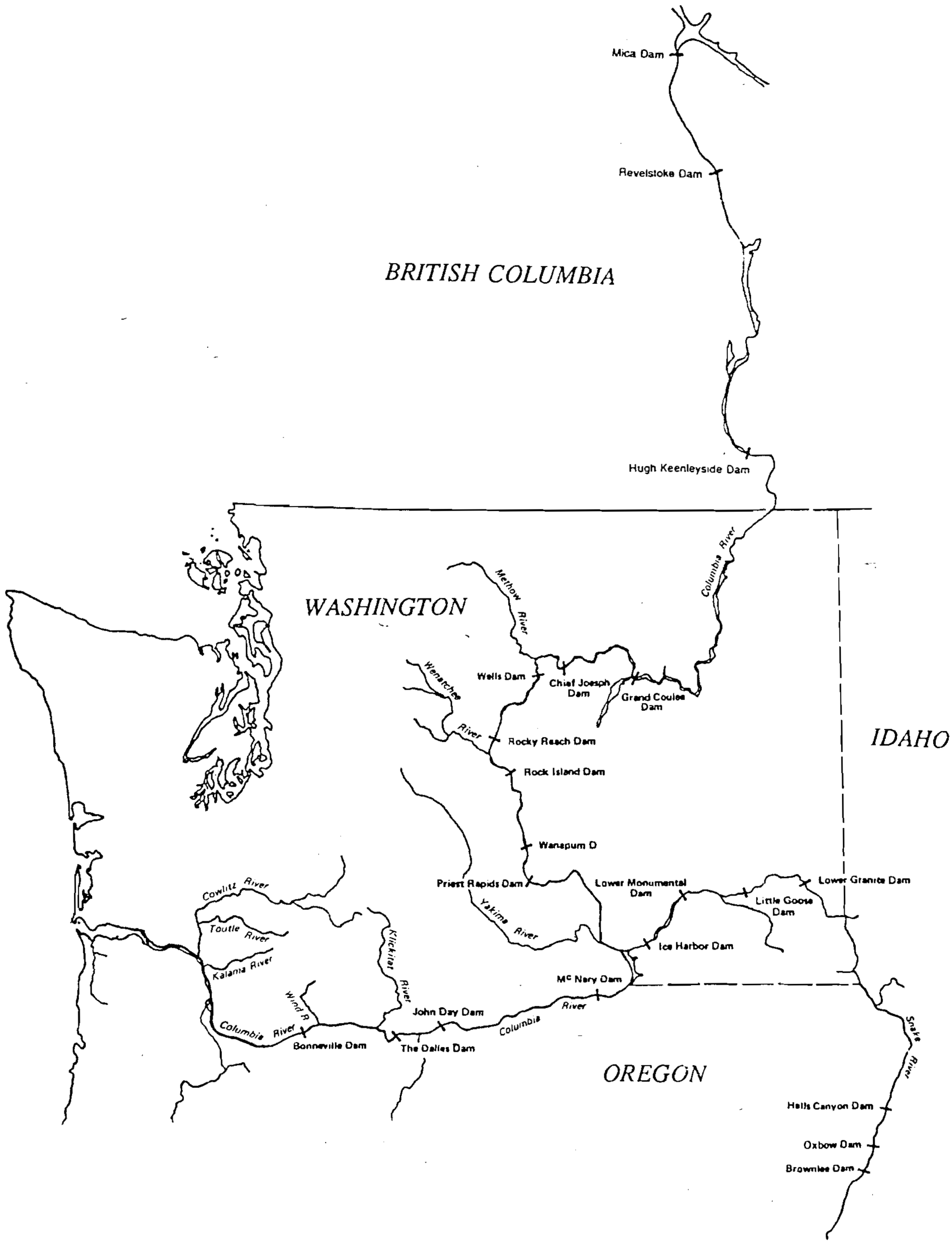

Figure 2.--Locations of the mainstem Columbia and large Snake River Dams.

Human Population

One of the most dramatic changes in the Columbia River basin in the last 150 years has been the increase in human population. The aboriginal population of the Columbia River basin was estimated at approximately 50,000 people (Craig and Hacker 1940), with 2,000 members of the Chinook tribe residing around the estuary (Simenstad et al. 1990). The introduction of European diseases, however, reduced this population to about one-sixth its former size between the 1820s and 1840s, to about 8,000 people in the Columbia River basin by 1851 (Craig and Hacker 1940).

Beginning in the 1840s, settlers began to migrate into the Columbia River basin, and the human population has continued to increase ever since. For example, the combined populations of the six largest cities in the Columbia River basin (Portland, Boise, Tri-Cities [Richland, Kennewick, and Pasco], Yakima, Vancouver, and Wenatchee) increased from a few thousand in 1850 to 700,000[3] in 1980 (Androit 1983). State populations have increased from 1,201 (1850) to 4.9 million (1990) in Washington, from 12,093 (1850) to 2.6 million (1980) in Oregon, and from 14,999 (1870) to 1.0 million (1990) in Idaho (Androit 1983, Cenarrusa 1992, Washington State Office of Financial Management [WSOFM] 1992).

The human population surrounding the estuary has also increased, although not as rapidly as in other areas of the Columbia River basin. With the exception of Wahkiakum County in Washington whose population has remained at about 3,500 since 1910, the populations in the three other counties surrounding the estuary (Pacific County in Washington, Clatsop and Columbia Counties in Oregon) have increased from a few hundred in the 1850s and 1860s to 18,000-35,000 in the 1980s and 1990s (Androit 1983, WSOFM 1992). The

[3]This value refers only to populations within the respective city limits.

dramatic increase in human population both around the estuary and in the entire basin has had a marked effect on many aspects of the Columbia River.

Industry

Industries along the Columbia River have contributed substantially to water-quality degradation. Industries introduce organic and inorganic contaminants, petroleum products, radioactive elements, and heat into the water (Karrick and Gruger 1976). Levels of most pollutants have increased concomitantly with the increased human population and industrialization of the Columbia River basin, although better water quality standards in recent years have moderated this increase (Buchman 1989).

Navigational Improvements

The Columbia River estuary and lower river have been highly modified for navigation. These modifications, which have had a substantial impact on the estuarine ecosystem, included the construction of jetties on both sides of the entrance, extensive dredging, and the installation of numerous pile dikes to force river flow into a main channel (Simenstad et al. 1984). Dredging began in the estuary in 1873, the construction of the South Jetty began in 1885, and the first pile dikes were constructed in the 1890s (Simenstad et al. 1984). By 1976, a 12.2-m deep channel extended from the river mouth to Portland, Oregon, and between 5 and 10 million (m^3) of material was dredged annually (Simenstad et al. 1984, Sherwood et al. 1990). These navigational improvements have forced the river flow into a single channel and altered the entrance bar, thus affecting the circulation, salinity intrusion, and bathymetry of the estuary.

Diking

Diking of tidal swamps for farmland has been extensive in the Columbia River estuary, and over 29,700 hectares (ha) (80%) of swampland has been impacted (Thomas 1983). The first dikes in Youngs Bay were constructed in 1868 (Simenstad et al. 1984), and most existing dikes were completed by 1926 (Blanchard 1977). Tidal swamps are an integral part of most estuaries, and their loss has had significant biological implications (Simenstad et al. 1990).

Fishing

Fishing has been a way of life on the Columbia River since Native Americans first occupied the area. Columbia River basin tribes relied extensively on salmon as their main dietary staple and as a trading commodity, and harvested an estimated 8.2 million kilograms (kg) annually within the basin (Craig and Hacker 1940). Canneries were first established in the 1850s, and their numbers peaked in 1887 with 39 canneries in operation. They packed approximately 14 million kg of chinook salmon per year (Craig and Hacker 1940, Pruter 1972). Harvests for other salmon species increased in later years, as heavily-harvested species, such as chinook salmon, declined in number. Current annual harvests (1990) of salmonids from the Columbia River are 256,700 fish or 1.8 million kg (ODFW and WDF 1991). Overfishing is cited as one of the main causes for the decline in salmonid runs originating in and returning to the Columbia River (Craig and Hacker 1940, Chapman 1986).

Logging

Logging, like fishing, is one of the oldest industries in the Northwest. Log and lumber exports began in the 1840s and continue today. For example, 4.1 billion board feet were cut in Washington State in 1910 (Van Winkle 1914), while 5.8-7.8 billion board feet were cut annually between 1973 and 1980 (WSOFM 1982). Logging has numerous environmental effects such as increased runoff, erosion that increases stream sediment loads, and altered nutrient regimes, all of which negatively affect fish habitats (Dunford 1960). Splash dams and related activities, used from the 1880s to the 1930s to move logs downstream, were particularly destructive (Sherwood et al. 1990). They completely blocked upstream access to migratory fishes, destroyed stream beds, and directly or indirectly caused considerable fish mortality (Wendler and Deschamps 1955).

Salmon Hatcheries

The first salmon hatcheries were established on the Columbia River in 1876, soon after the first canneries were in operation, in an effort to increase salmon runs. Twenty-four hatcheries were in operation by 1907 (Stober et al. 1979), and 57 hatcheries currently operate in the Columbia River basin (ODFW and WDF 1991). Hatcheries continue to play a major role in maintaining runs of spring, summer, and fall chinook and coho (*O. kisutch*) salmon, and steelhead (*O. mykiss*) for harvest (ODFW and WDF 1991). For example, in 1992, 24.4 million chinook salmon, 43.9 million coho salmon, and 16.5 million steelhead smolts were released throughout the Columbia River basin[4]. However, the merits of hatcheries have been debated considerably in recent years because of their high cost and maintenance requirements, low adult returns, and negative impacts on wild stocks. Hatcheries are also criticized for their failure to increase runs toward sustainability, and because they do not address the causes of declining runs, such as habitat degradation, overfishing, and passage obstruction (Meffe 1992).

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[4]Stan Allen, unpubl. data. Pacific States Marine Fisheries Commission, 45 S.E. 82nd Drive, Suite 100, Gladstone, OR 97027-2522.

Naturally Occurring Changes

Climate Variation

Historical climatic records indicate that periods of unusually wet, dry, hot, or cold weather have occurred throughout the Columbia River basin (National Oceanic and Atmospheric Administration [NOAA] 1990). For example, average annual air temperatures and precipitation at Yakima, Washington, fluctuated widely from 1937 to 1985 (Fig. 3) (NOAA 1976, 1986). Although these annual variations are on the scale of a few degrees of temperature or centimeters of precipitation, they may have a large effect on winter snowfall and melting of the mountain snowpack, which is the primary source of Columbia River water. This type of climatic variation has had a large effect on both Columbia River flows and water temperatures. For example, 15-year averages of river flow were 25-30% higher in the 1870s and 1880s than in the 1930s (Sherwood et al. 1990). The naturally occurring changes in river temperature and flow rates make it exceedingly difficult to detect anthropogenic changes contributing to either factor, as well as the fact that variable climatic conditions have their own biological impacts (Haertel and Osterberg 1967).

The Eruption of Mount St. Helens

On 18 May 1980, Mount St. Helens erupted, sending ash over 280,000 km^2, and mud and debris down the Toutle, Lewis, and Cowlitz Rivers (Whitman et al. 1982). Thirty-four million m^3 of mud arrived at the confluence of the Cowlitz and Columbia Rivers, with tons of ash and mud transported downstream into the estuary (CRWMG 1982). The eruption had a large impact on estuarine organisms and food webs, as well as anadromous fishes from the Toutle and Cowlitz Rivers (Emmett 1982, Whitman et al. 1982, Brzezinski and Holton 1983). The eruption occurred during the 1980-1981 Columbia River Estuary Data Development

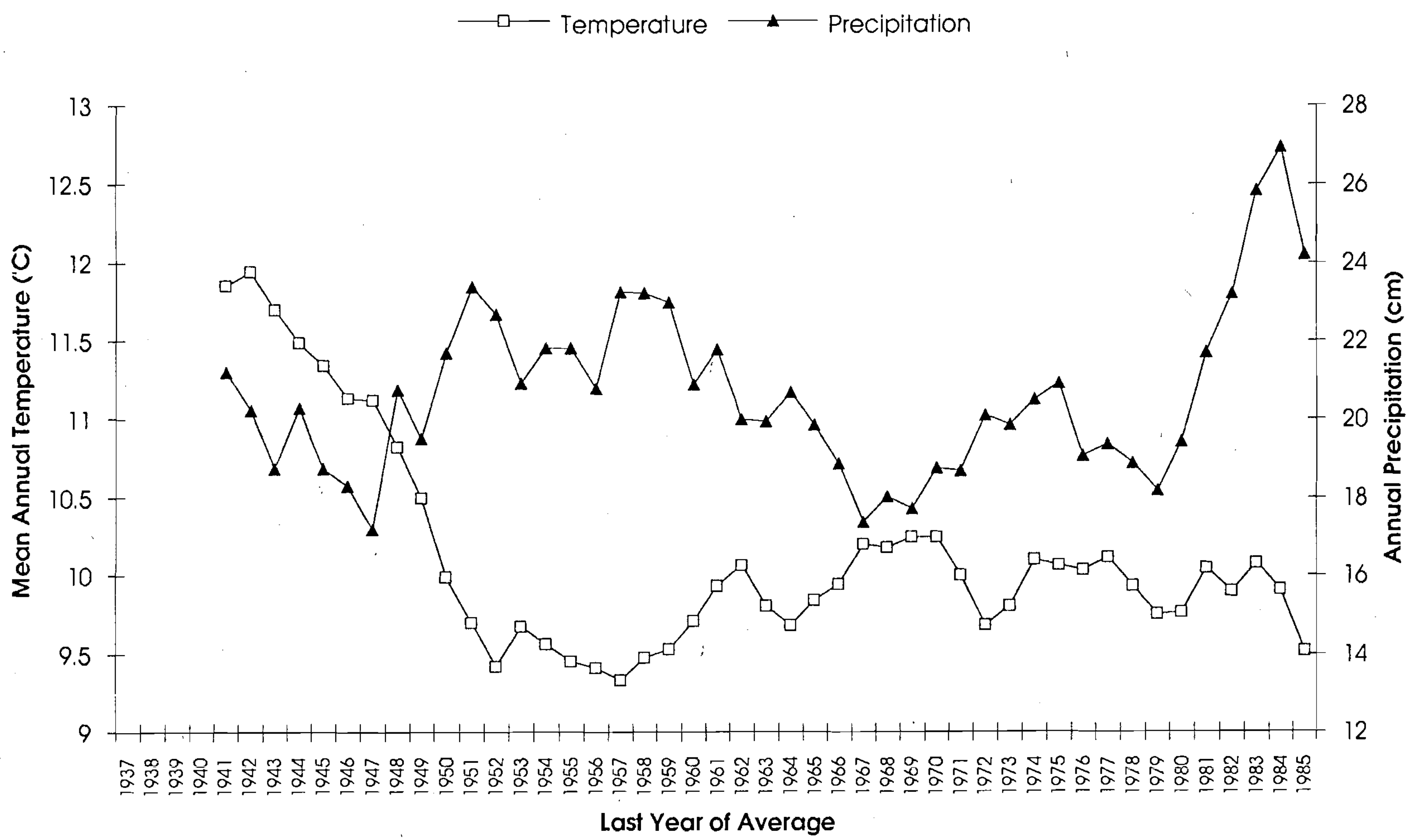

Figure 3.--Five year averages of mean annual air temperature and annual precipitation for Yakima, Washington, from 1937 to 1985. The value for the year indicated is for a five-year average ending in that year. Data from NOAA (1976, 1986).

Program (CREDDP), a study that was attempting to quantify physical and biological processes in the estuary. The eruption of Mount St. Helens during the study likely affected the physical and biological processes in the estuary, and likewise affected the conclusions drawn from the CREDDP study.

Natural Land Rise

The land on which the Columbia River estuary is located has been rising at a rate of 0.01-0.10 mm per year (Chelton and Davis 1982). Although amounting to a fairly small change over several years, this land rise would have significant effects over longer time periods, such as the 140 years since the first maps of the estuary were made (Simenstad et al. 1984). This rise could have affected salinity intrusion and circulation, with resultant biological impacts. However, compared to bathymetric changes due to navigation improvements, the effect of natural land rise on estuarine bathymetry is probably insignificant.

Summary of the Sources of Change in the Columbia River Estuary

As detailed in the above discussion, the Columbia River and its estuary have been subjected to numerous changes, both natural and anthropogenic, over the last 150 years. While the impacts of these changes vary in magnitude and form, they all contribute to the condition of the present-day estuary, which is physically and biologically different from the unaltered estuary. In many cases, various sources of change have contributed to similar impacts, while other sources cause fairly unique impacts. For example, many of the factors listed above degrade water quality and reduce habitat quantity and quality, while only dams regulate river flow on a system-wide macroscale; although irrigation and natural climatic variations also influence river flow. Consequently, determining the factors responsible for some changes is fairly straightforward, while determining the effects of other factors is quite complex, requiring additional, detailed investigations beyond the scope of this report.

PHYSICAL CHANGES IN THE COLUMBIA RIVER ESTUARY

Introduction

The physical environment of the Columbia River estuary results from the interaction of numerous physical factors. The most important of these are river flow, tides, sediment input, bathymetry (bottom topography), circulation and salinity intrusion, and, to a lesser extent, atmospheric conditions. These factors also affect each other, while themselves undergoing change, and operate on different temporal and spatial scales. Processes are important on hourly, daily, monthly, and annual time scales, and affect spatial scales ranging from microhabitats to the entire river basin and nearshore ocean. Some of the physical characteristics in the estuary have undergone substantial human modification, while others have not. For example, the bathymetry of the estuary has been extensively modified by navigational improvements and diking, dramatically affecting the movement of fresh and salt water in the estuary, while the tides have remained unchanged. Consequently, it is extremely difficult to determine the effects of Columbia River dams on the physical environment of the estuary because no single factor is solely responsible for observed or suspected changes.

The rate at which the physical environment of the estuary responds to modifications also varies with the processes involved. Petts (1987) predicted that water temperature would respond almost immediately to modifications in river flow caused by dams, but river bottom topography would require over 100 years to fully adjust. Consequently, the Columbia River estuary is still undergoing physical changes in response to modifications which occurred decades ago (Sherwood et al. 1990). Because physical characteristics influence the biological community to a large extent, parts of the estuarine assemblage are still adapting to previous system modifications.

Numerous studies on the physical characteristics of the Columbia River estuary have been conducted. The U.S. Coast Survey (later called the U.S. Coast and Geodetic Survey, and now the National Ocean Service) and the U.S. Army Corps of Engineers have made extensive surveys of the estuary as the navigation channel was planned and modified. These surveys are still being conducted. Water quality surveys were first conducted by Van Winkle (1914) from 1909 to 1912, with later surveys by Sylvester (1958) and the U.S. Geological Survey (USGS). The most extensive research effort of the estuary was made in 1980 and 1981 as part of CREDDP, and although affected by the eruption of Mt. St. Helens, resulted in a published series that included biological and physical inventories, modeling of present and former patterns of circulation, and an assessment of ecosystem-level dynamics.

This section summarizes specific physical changes that have occurred in the Columbia River estuary and cites the causes of those changes, insofar as possible. How the physical changes may have affected the ecological assemblage in the estuary are suggested, and the role of dams in effecting specific changes is assessed. Although many of the changes are not directly caused by Columbia River dams, the ways in which dams have contributed indirectly to observed changes are discussed.

River Flow

A common characteristic of all dammed systems is flow regulation, which occurs as water passes through or over dams (Baxter 1977, Ward and Stanford 1987). The pattern of flow regulation depends on the type of water usage and natural river flow, and varies temporally. For example, domestic electrical power production has morning and evening peaks of demand, requiring higher flows through the turbines during these periods, while seasonal changes in demand occur as people heat or cool their houses. The net results of the dams on river flow in the Columbia River are higher variability in hourly and daily flow rates, and dampening of annual maximum and minimum flows. River flow has an important influence on the physical characteristics of the estuary, which in turn affects the biotic estuarine community.

Columbia River Flow

Prior to dam construction, the flow regime of the Columbia River was characterized by late summer and fall low flows, small winter freshets from tributaries west of the Cascade Mountains (hereafter Cascades), and large spring freshets caused by mountain and eastern-basin snow melt (Sherwood et al. 1990). Before regulation, spring flows at the mouth were typically 18,690 cubic meters per second (m^3s^{-1}) (Ebel et al. 1989), with a maximum recorded flow of 35,100 m^3s^{-1} in 1894 (Sherwood et al. 1990). Fall low flows at the mouth averaged 1,980 m^3s^{-1} (Ebel et al. 1989), with a minimum recorded flow of 990 m^3s^{-1} in 1937 (Sherwood et al. 1990). Although dams have been present on the Columbia River mainstem since the completion of Rock Island Dam in 1933 (CRWMG 1982), river flow was not significantly regulated until 1969 (Sherwood et al. 1990), when John Day (1968), Wells (1967) and Hugh Keenleyside (1968), Hells Canyon (1968), and Lower Monument (1969) Dams became operational, in addition to 12 existing mainstem and lower/middle Snake River Dams. Peak spring flows have been reduced by approximately 21 to 28% (3,900-5,200 m^3s^{-1}) to 13,500-14,800 m^3s^{-1}, while fall flows have increased by 50% (990 m^3s^{-1}) to 2,970 m^3s^{-1} (Sherwood et al. 1990). Much greater regulation of peak flows is possible if needed.

Figure 4 illustrates the average annual hydrographs at the Columbia River mouth between 1928 and 1984, divided into five time periods based on the increased active storage capacity of mainstem and lower/middle Snake River (Brownlee and below) dams, compiled from USGS (1973-76, 1977a-1985a) data. Average monthly flow during the first two periods, before dams (1928-32) and with less than 0.2 km^3 of active storage (1933-41), is fairly similar, the largest differences occurring from December through February. Given that the first period represents only five years of data, this difference may result more from interannual variability than from the effects river flow regulation. Differences between the first two periods and subsequent periods are much more pronounced, as the increasing levels of active storage provide greater ability to manipulate and modify river flow. For example, increased flows in winter are apparent between the first/second (1928-1941) and third (1942-1967) periods, although peak spring flows do not appear to be affected. The active storage capacity between these periods increases from less than 0.2 cubic kilometers (km^3) to 6.6-8.7 km^3. However, the changes between the earlier periods (1928-67) and the fourth (1968-1972) and especially the fifth (1973-1984) periods are much greater as the storage capacity increased to 17.6-20.2 km^3 (fourth period) and 35-42 km^3 (fifth period). Winter and minimum annual (September/October) flows increase while peak spring flows decrease such that by the fifth period, there is little difference in average monthly flow from December to June.

Because the total water-storage capacity of the mainstem dams ($77.7 \times 10^9\ m^3$) is approximately one-third of total annual river flow ($223 \times 10^9\ m^3$), significant interannual variation in storage can occur (Sherwood et al. 1990, Simenstad et al. 1992). This has the potential to significantly vary the annual flow rate, because water that would naturally flow one year is not released until the following year. Another source of annual flow variation is irrigation withdrawal. Currently 31,600 (km^2) of land are irrigated in the Columbia River basin, requiring 360 to 560 m^3s^{-1} of water, with most withdrawal in May and June (Sherwood

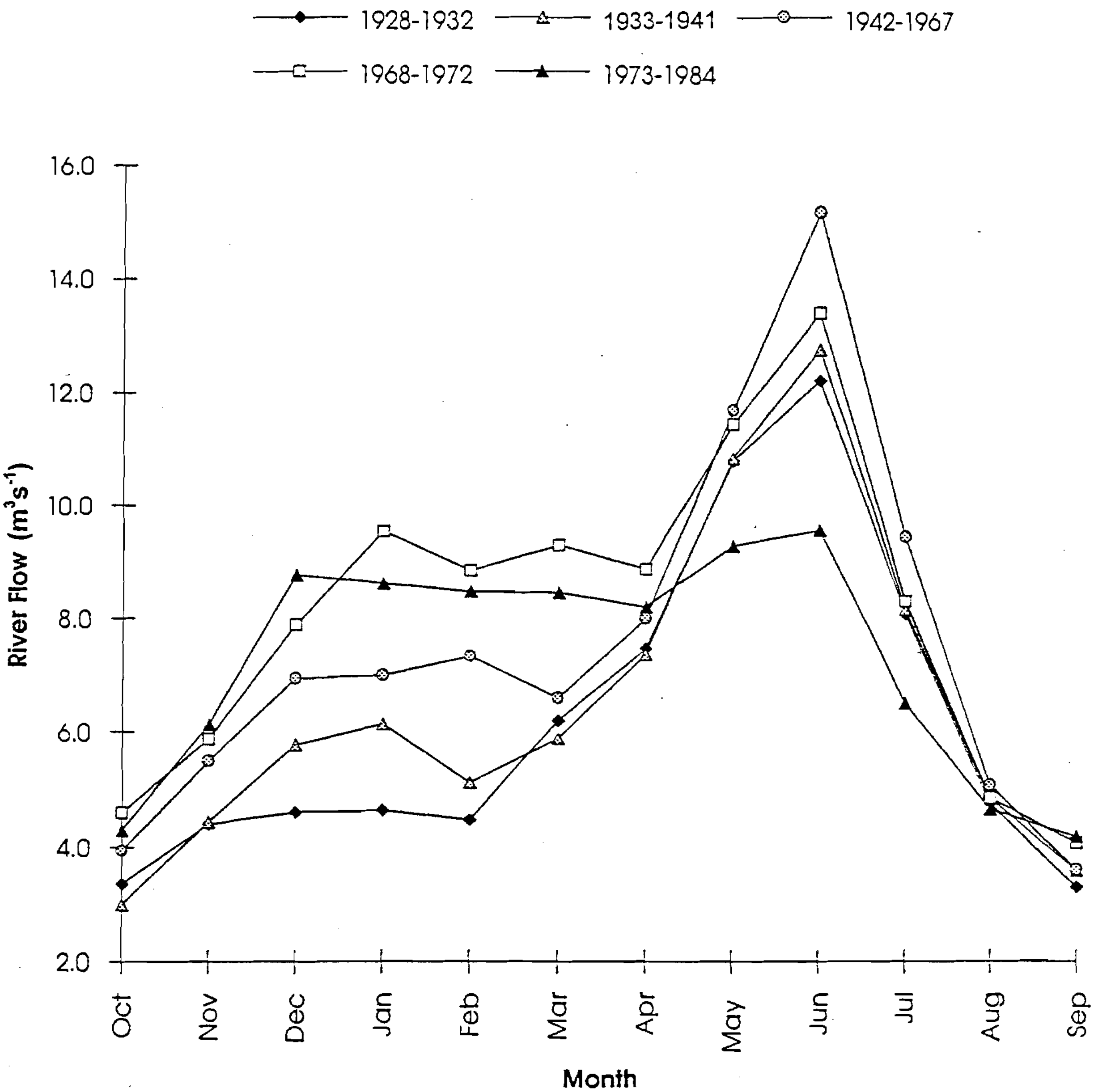

Figure 4.--Mean monthly flow at the mouth of the Columbia River 1928-84, divided into five time periods to reflect the increasing active storage capacity of Columbia and lower Snake River reservoirs. The active storage and dams completed for each periods are as follows: 1928-1932--no active storage or dams on the Columbia River mainstem or lower Snake River; 1933-1941--less than 0.2 km^3 active storage; Rock Island (1933) and Bonneville (1938) Dams completed; 1942-1967--6.6-8.7 km^3 active storage; Grand Coulee (1942), McNary (1953), The Dalles (1957), Chief Joseph (1958), Brownlee (1959), Priest Rapids (1961), Oxbow (1961), Ice Harbor (1961), Rock Reach (1962), Wanapum (1964), and Wells (1967) Dams completed; 1968-1972--17.6-20.2 km^3 storage; Hugh Keenleyside (1968), Lower Monumental (1969), Little Goose (1970), and Dworshak (1971) Dams completed; 1973-1984--35.0-42.0 km^3 storage; Mica (1973), Lower Granite (1975), and Revelstoke (1983) Dams completed. Data from Orem (1968), USGS (1973-77, 1978a-85a).

et al. 1990). Although irrigation depletes 7 to 10% (223-241 x 10^9 m^3) of the river flow annually and interannual storage may be high, this amount of human-derived water usage is considerably less than the range of natural annual flow variation. For example, flows between 1920 and 1940 declined 25% because of climatic factors, compared to 7-10% flow alteration because of irrigation (Sherwood et al. 1990).

The mainstem of the Columbia River has 14 dams (Fig. 2), 13 of which are on the east side of the Cascades (hereafter east side), an area characterized by an extremely dry climate with much of the winter precipitation falling as snow. West of the Cascades, average precipitation is much higher and rainfall occurs year round at lower elevations. Flow originating east of the Cascades is regulated by mainstem dams, while most flow originating west of the Cascades is not, further complicating the present-day Columbia River hydrograph. Because east-side spring freshets have been restricted, and west-side winter freshets have not, west-side winter freshets now approach the size of east-side spring freshets, although the duration of winter freshets is shorter (Sherwood et al. 1990).

Physical Effects of Altered River Flow

Sediment transport--Columbia River dams have a significant impact on sediment transportation in the estuary (Simenstad et al. 1992). Because dams suppress peak river flows, which transport the most sediment (primarily sand), there has been a 50% decrease in sediment, including associated nutrient constituents, transport by the river at Vancouver, Washington, from 14.9 million tons (t) per year from 1868 to 1934, to 7.6 million t per year from 1958 to 1981, as well as a reduction in nutrients associated with sediment (Sherwood et al. 1990, Simenstad et al. 1992). The decreased energy available from suppressed floods, paired with a decline of tidal energy caused by bathymetric changes, has decreased the ability of the estuary to move sediment seaward, and has accelerated the rate of natural sediment deposition in the estuary, resulting in the increased need for dredging (Thomas 1983, Sherwood et al. 1990, Simenstad et al. 1992).

The effects of altered river flow are not apparent in the offshore portion of the estuary in the plume. There has been no significant alteration in nutrient or carbon levels in the plume or deposition in offshore sediments between 1960-1985 (Carpenter 1987), nor any changes to plume productivity (Fielder and Laurs 1990). However, relatively subtle changes such as those caused by regulated Columbia River flow would be difficult to detect. The plume is in the very dynamic offshore zone, an area with high interannual variability, including extremes caused by El Niño-Southern Oscillation events (Johnson 1988, Landry and Hickey 1989). Consequently, variations in the relatively minor contribution of the Columbia River to offshore processes would be difficult to detect (Landry et al. 1989).

Circulation and salinity intrusion--Although bathymetric changes have had the greatest impact on circulation and salinity intrusion in the estuary, river flow regulation has also been important (Hamilton 1984). However, determining the specific effects of changing river flow on circulation and salinity intrusion is complicated because of the anthropogenic bathymetric changes. The degree of impact of flow also depends on estuarine location: above river kilometer (RKm) 30, river flow is the important source of energy for water movement, while below RKm 30, tidal energy has increasing importance (Sherwood et al. 1990).

Circulation of water through the estuary has been affected by river flow in several ways. One general measure of estuarine water movement is flushing time: the rate at which water passes through the estuary. Because maximum spring flows have been reduced and minimum fall flows have increased in magnitude and duration, and tidal exchange has been reduced by bathymetric changes, the annual flushing time of the estuary has increased (Sherwood et al. 1990). In addition, the variability in flushing time has decreased as extreme minimum and maximum flows have been modified (Sherwood et al. 1990).

River flow controls salinity intrusion because denser saltwater in the estuary must flow against and underneath less dense fresh river water; the more freshwater moving downstream, the less saltwater is able to intrude upstream. It is speculated that in 1868, maximum salinity intrusion was greater than at present because of lower minimum flows and different bathymetry, and salinity intrusion was more variable because of higher seasonal variability in flow (Hamilton 1984). For example, it was estimated that in the unaltered estuary, salinity intrusion could reach as far as RKm 65 during minimum river flows, while maximum salinity intrusion currently reaches only RKm 50 (Simenstad et al. 1992). The present day estuary is less saline, especially during fall low flows, with lower variability in salinity intrusion. Similar alterations in salinity intrusion have been recorded in the Sacramento/San Joaquin (California) estuary in response to altered river flows (Monroe et al. 1992).

Biological Effects of Altered River Flow

Flushing rate and salinity intrusion--Changes in the flushing rate and salinity intrusion in the Columbia River estuary are thought to affect the estuarine biological community. The altered flushing rate may have affected the ability of organisms to maintain themselves in the estuary. This is particularly crucial to weak swimmers, such as zooplankton and larval fishes. Many weak swimmers utilize upstream bottom currents to maintain their position in the estuary. Such organisms are carried downstream during ebb tides, and return upstream during flood tides in the upstream bottom flow of salt water. Such behavior is clearly apparent in detailed investigations of zooplankters in the Columbia River estuary (Haertel et al. 1969, Cordell et al. 1992). During extremely high flows with high flushing rates, downstream ebb flow far exceeds upstream flood flow and organisms may be transported out of the estuary, with no means of returning. For example, Haertel et al. (1969) noted depressed *Eurytemora* densities in 1965 following very high river flows (>20,000 m^3s^{-1}) in June and December 1964. The present reduction of extreme flows may allow organisms to better maintain their position in the estuary, and reduce their chance of being swept from the estuary. However, this process may also benefit exotic species in the estuary. For example, Cordell et al. (1992) suggest that lack of high flows may have allowed the Asian copepod *Pseudodiaptomus inopinus* to become established and prosper in the estuary.

Floods are generally considered to be extremely important in riverine systems for the existence, productivity, and interactions of biological communities (Junk et al. 1989, Ward and Stanford 1989). Predictable seasonal floods provide evolutionary pressure to adapt to and utilize recycled nutrients, high productivity, and new habitats produced during the flood (Junk et al. 1989). When floods are controlled by dams, this expected burst of productivity is constrained, and the aquatic community may suffer (Aleem 1972, Ward and Stanford 1989).

On a smaller scale, anthropogenic daily and monthly fluctuations in river flow decrease species diversity and abundance and dramatically alter community composition because most species are not adapted to such short-term fluctuations (Baxter 1977, Gaschignard and Berly 1987, Saltveit et al. 1987). However, because of the considerable tidal range in the estuary, many benthic organisms, particularly in the lower estuary, are adapted to withstand flow reversals and dewatering for extended periods of time. These adaptations potentially reduce the impact of short-term flow variability. The effects of diminished floods on the estuarine community are difficult to assess because much of the floodplain in the Columbia River estuary has been diked and filled, and is no longer subjected to flooding (Blanchard 1977, Thomas 1983). Because of the absence of baseline data from the unaltered estuary, the biological effects of reduced annual flow variability in the Columbia River estuary remains unknown.

However, it is certain that altered salinity intrusion impacts the biotic community because most aquatic organisms have specific salinity tolerances. Salinity tolerance is determined by an organism's ability to maintain osmotic balance with its surroundings. If the Columbia River estuary has become less variable and less saline, especially during traditional low flow periods, then the geographic range of organisms should have shifted downstream to remain within their preferred salinity ranges. Ward and Stanford (1989) found upstream or downstream shifts in aquatic communities to be a common consequence of dams, with the direction of the shift dependent on the dam location.

Organisms in the Columbia River estuary may have moved seaward because of modified river flow and bathymetric changes, although there is no documentation of this movement. Downstream movements of many organisms already occur seasonally in the Columbia River estuary in accordance with seasonal changes in river flow. For example, during spring peak flows, the freshwater species *Bosmina* and *Daphnia* are found farther downstream in the estuary than at other times, presumably because of decreased salinity (Jones et al. 1990). Because extreme low flows are modified, Jones et al. (1990) suggest fewer marine species enter the Columbia River estuary than historically.

Increased occurrence of marine or exotic species because of decreased river flow have been recorded in other estuaries. For example, Zalumi (1970) documented increases in marine and brackish water species in the Dnieper River estuary (former U.S.S.R.) after considerable decrease in river flow because of dams. Carlton et al. (1990) suggest that the Asian clam *Potamocorbula amurensis* was able to become established in the Sacramento River estuary, California, because several years of exceptionally low river flow and consequent salinity changes had weakened the native fauna. Whether any of the exotic species in the Columbia River estuary have benefitted from the altered salinity regime due to river regulation remains to be determined.

Sediment transport--The reduction of sediment transport through the estuary may affect both the shallow inshore estuary, and the portion of the estuary extending offshore in the plume. Inshore, increased accretion in the estuary may limit the distribution of species requiring deeper areas, while benthic species may be buried. Newly deposited sediments may be unconsolidated and unstable, providing undesirable habitat for many species. However, the increased rate of deposition is hypothesized to benefit organisms associated with the estuarine turbidity maximum (ETM) (Sherwood et al. 1990). The ETM is a transitory area of sediment deposition, characterized by high turbidity, concentrations of organic material, productivity, and high zooplankton densities (Gelfenbaum 1983). Decreased input of macrodetritus from greatly reduced tidal marshes and swamps of the Columbia River estuary, and increased input of lysed phytoplankton (microdetritus) and sediment from upriver which become entrained in the ETM has increased the importance of the ETM as a major pathway for estuary energy transfer (Sherwood et al. 1990). Organisms utilizing the ETM include planktonic copepods (primarily *Eurytemora* and *Scottolana*), and their planktivorous predators, including pelagic fishes (American shad, Pacific herring, and smelts) and motile macroinvertebrates (California bay shrimp, mysids). Consequently, it is hypothesized that there has been a shift in estuarine production from shallow water benthic detritivores, such as *Corophium* spp. and other gammarid amphipods, native clams, and polychaetes, which are preyed upon by juvenile salmon, sculpins, and flatfishes, to planktonic detritivores (*Eurytemora* and *Scottolana*) and their predators (Sherwood et al. 1990). However, there is no direct evidence to prove that this occurred (Sherwood et al. 1990).

The importance of riverine sediment and nutrient inputs to the coastal portions of estuaries has been well established in other large river systems. For example, primary and secondary productivity in the unregulated Fraser River (British Columbia) plume is much higher than either inland or offshore, and the high productivity supports a concentration of juvenile salmonids (St. John et al. 1992). In a more extreme case, the annual flood of sediment and nutrient-rich water in the Nile River (Egypt) produced huge phytoplankton and zooplankton blooms in the south-east Mediterranean Sea, supporting an extensive sardine fishery (Aleem 1972, Sharaf El Din 1977). With the suppression of the annual flood by the Aswan High Dam, however, blooms were completely eliminated, and the sardine fishery was drastically reduced within 2 years (Aleem 1972).

It is difficult to assess whether a similar situation has occurred in the offshore region of the Columbia River estuary since the construction of the numerous dams. One primary difference between the Columbia River and other rivers is that the Columbia River does not play a large a role in offshore production (Carpenter 1987, Landry and Hickey 1989). The primary source of nutrients in coastal waters off Washington and Oregon result from wind-driven upwelling, supporting high levels of productivity (Landry and Hickey 1989, Landry and Lorenzen 1989). Although the Columbia River contributes significant amounts of sediment (Kachel and Smith 1989), and some silica and phosphate to the coastal region, the contribution of nitrogen input is not significant (Park et al. 1972, Carpenter 1987, Landry et al. 1989). Chlorophyll *a* concentrations are not necessarily higher in the Columbia River plume than in the surrounding oceanic waters (Small and Curl 1972, Landry et al. 1989), nor are zooplankton (Peterson 1972, Landry and Lorenzen 1989) or fish densities (Alverson 1972, Fisher and Pearcy 1988). In addition, the offshore region is highly dynamic, and the plume itself often undergoes rapid changes in orientation and shape because of brief reversal in wind direction (Fielder and Laurs 1990).

Only two fish species are consistently concentrated in the Columbia River plume: cutthroat trout (*Oncorhynchus clarki*) (Miller et al. 1983, Loch and Miller 1988, Pearcy et al. 1990), and the northern anchovy, *Engraulis mordax* (Richardson 1981), which can be an important prey for chinook and coho salmon (Emmett et al. 1986b, Brodeur and Pearcy 1990). Cutthroat trout appear to use the plume as an offshore extension of their otherwise coastal distribution, while the northern anchovy northern subpopulation appears to spawn exclusively in the Columbia River plume, possibly because of the warmer temperatures and more stable environment than in other oceanic areas (Richardson 1981). Although chinook and coho salmon and steelhead can be found in the plume, they are not concentrated there during their coastal residence (Brodeur and Pearcy 1990).

The ecology of the plume prior to the construction of dams on the Columbia River is unknown. However, given the currently minor role the plume plays in the productivity of the coastal region, its limited exclusive utilization by fishes, and its occasionally transitory nature, it is expected that changes to the plume caused by river flow regulation are minor compared to the effects of offshore processes on the plume, such as anomalies associated with strong El Niño-Southern Oscillation (ENSO) events.

Summary of the Effects of Dams from River Flow Alteration

Although the effects of regulated river flow on the estuarine assemblage may be considerable, there has been no documentation. Decreased variability of maximum and minimum flows in the modern estuary may have provided less environmental pressure to which organisms must adapt. The historic wide range of salinities, water height, and velocities experienced over the course of a year has narrowed, potentially allowing organisms which may not have been well adapted to the extreme fluctuations to multiply. Exotic as well as native species may have benefitted from these changes. In general, regulation of river flow has contributed to decreased physical variability in estuarine habitats, potentially altering the evolutionary pressures to which estuarine inhabitants must adapt. Compared to the highly dynamic and variable conditions offshore, it is expected that changes in the river plume caused by river flow alteration are minor compared to changes caused by offshore processes, or changes in the inland portion of the estuary.

Bathymetry

Changes in the bathymetry, or bottom topography, have been the greatest anthropogenic physical change to the Columbia River estuary (Sherwood et al. 1990). These changes were made primarily to improve navigation, and to convert the floodplain into farm land (Thomas 1983). Bathymetric changes for navigation resulted from efforts to widen, deepen and stabilize the shipping channel, beginning in 1885 with the construction of the South Jetty at the mouth of the river. In addition to the construction of a second major jetty and numerous pile dikes, the estuary has been subjected to extensive dredging, with the removal of 5-10 million m^3 of material annually (Sherwood et al. 1990). Navigational improvements have concentrated water from a system of complex channels into a single deep channel. Flood control and agricultural expansion have extensively impacted intertidal and floodplain habitats via diking. Sixty-eight percent of the lower Columbia River floodplain has been diked and leveed, mostly prior to 1926 (Blanchard 1977, Thomas 1983).

Although dams were not the major cause of bathymetric changes in the estuary, it is nonetheless important to describe the bathymetric changes that occurred and their consequences. Because bathymetry plays a major role in determining the biotic community and is highly interactive with other physical processes in the estuary, it is impossible to fully understand the changes caused by dams without understanding changes in the bathymetry.

Changes in the Columbia River Estuary Bathymetry

In their comparisons of changes to the Columbia River estuary bathymetry between 1868 and the present, Thomas (1983) and Sherwood et al. (1984) list several causes of the observed bathymetry changes. These include increased sediment deposition, removal of estuarine habitat from the estuary, and conversion from one depth to another. Estuaries are places of sediment deposition, because sediment carried by the river settles out of the water column as water velocity slows or comes in contact with salt water. Anthropogenic activities on the Columbia River have greatly accelerated the rate of sediment deposition. Approximately 370-485 million m^3 have been deposited in the estuary since 1868, at a rate of 0.5 cm per year (excluding the entrance), resulting in an annual net gain of 43 million m^3 (Sherwood et al. 1990).

This increased sedimentation resulted from altered circulation, allowing increased accretion in stagnant backwaters, and a decrease in the number and magnitude of floods, which would normally transport large amounts of sediment from the estuary (Thomas 1983).

The increase in sedimentation may also be due to locally increased sediment loads in the river and estuary tributaries (Thomas 1983). Certain land-use practices such as logging and road building increase sediment input into streams, which may be especially important in tributaries in the lower reaches of the river (Thomas 1983, Sherwood et al. 1984, Bisson et al. 1992). Upriver dredging with flow-lane disposal may also increase the amount of sediment entering the estuary (Thomas 1983, Sherwood et al. 1984). However, Sherwood et al. (1990) estimate the total sediment load of the river has been reduced by one half since the construction of dams on the Columbia River.

Much of the land formerly within the estuary has been removed from estuarine influence by diking and filling. The estuary in the 1870s covered 63,200 ha, while the current estuary covers only 48,250 ha, for a loss of 14,950 ha (Thomas 1983). Of this 14,950 ha loss, 9,700 ha were diked for agriculture, 2,300 ha were filled, and approximately 2,800 ha were converted into non-estuarine wetlands (Thomas 1983).

The conversion of area from one depth to another has been a large source of change in the Columbia River estuary bathymetry. This conversion has occurred in all parts of the estuary, and is multidirectional and asymmetric: shallow areas may become deeper in one location, deep areas may become shallower in another part of the estuary, while other areas remain unchanged (Thomas 1983). One cause of depth conversion has been the natural filling of side channels because of pile dikes (Sherwood et al. 1984). These permeable dikes restrict water flow and concentrate it in the main channel, which encourages sediment deposition in low water-velocity areas and scouring in the main channel. Many previously important channels, such as Cordell Channel, Prairie Channel, Cathlamet Channel, and the channels of eastern Grays Bay have become isolated from the main river flow by pile dikes and have experienced considerable accretion (Sherwood et al. 1984).

Total estuary--Sherwood et al. (1984, 1990) and Thomas (1983) have made extensive calculations of changes in the morphology and habitat types in the Columbia River estuary from the time of the earliest complete estuarine survey (1868) and the most recent available survey information (1958). In both 1868 and 1958, the largest depth category was the area above +0.9 m mean lower low water (MLLW); the contribution by each depth category decreased with depth (Fig. 5). Although the percent change between 1868 and 1958 for this shallowest depth interval is not the greatest (Fig. 6), the change is significant because of the large area the category represents (Table 1). The greatest change between 1868 and 1958 was an increase in very shallow (2.1 to 0 m) and medium-deep (-9.1 to -24.4 m) areas, with a decrease in shallow (0 to -9.1 m) and very deep (>-24.4 m) intervals (Table 1, Fig. 6). This pattern reflects, in part, the formation of dredge spoil islands from shallow flats, filling of medium-deep side channels, and deepening of the main channel (Thomas 1983).

Lower estuary--The lower estuary (below Tongue Point, RKm 29) was formerly extremely dynamic with shifting sands and channels caused by high tidal, wave, and fluvial energy. With the construction of jetties and pile dikes and dredging, the entrance channel has deepened, while the sides have accreted. Medium-shallow water has been converted to deep or shallow water, resulting in a 28% decrease in water between 1.8 and 5.5 m deep, while water deeper than 5.5 m has increased by 1% and water less than 1.8 m deep has increased by 8% (Table 1).

Approximately 3,500 ha of shallow flats were converted to uplands, adding significantly to Clatsop and Peacock Spits. Sand Island was stabilized in its current position

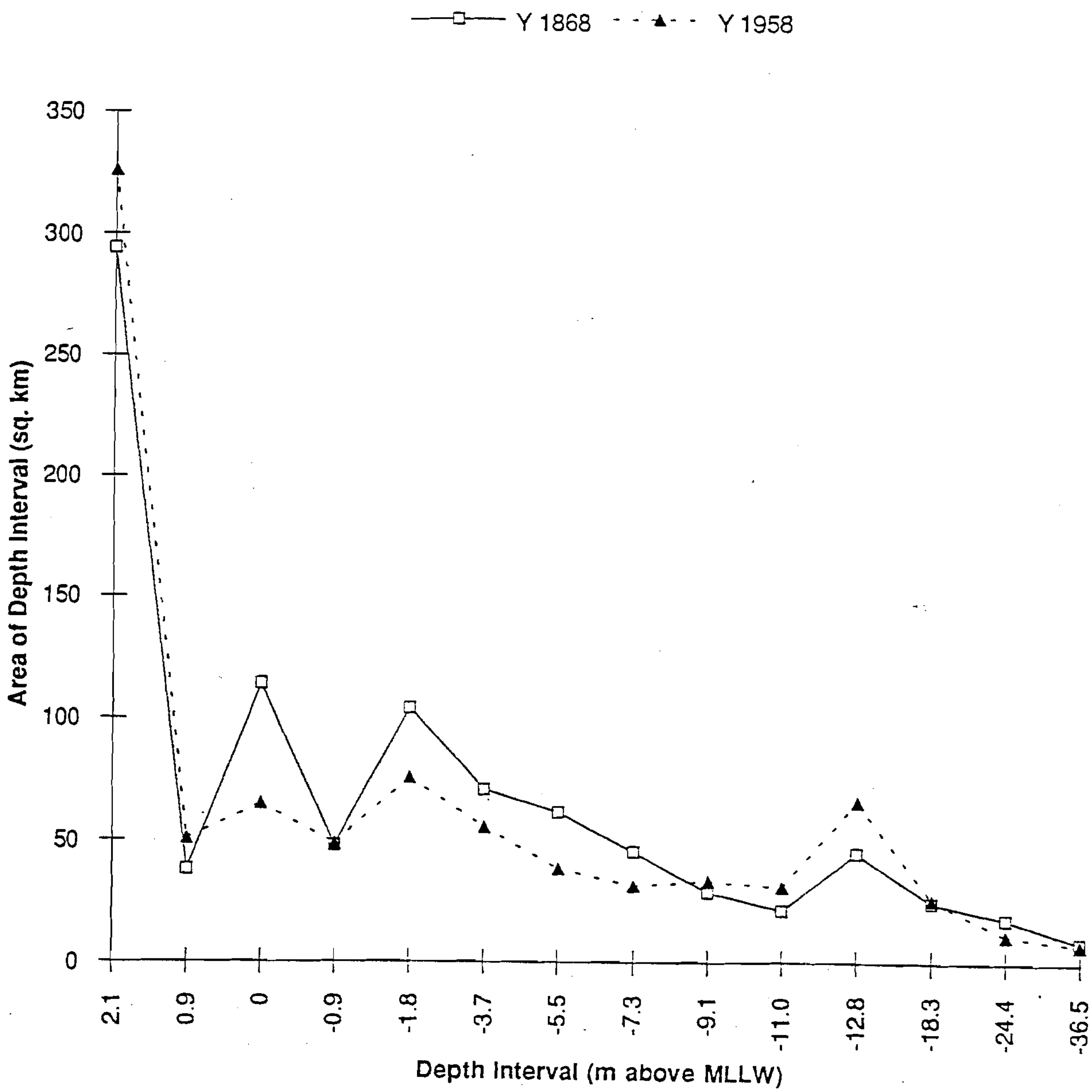

Figure 5.--Changes in the distribution of depth intervals in the Columbia River estuary between 1868 and 1958. Depth intervals are from the depth indicated by the data point and the next lower interval (i.e., the data for depth interval -7.3 represents an interval ranging from -7.3 to -9.1 m). All depth are in meters below mean lower low water (MLLW). Data from Sherwood et al. (1990).

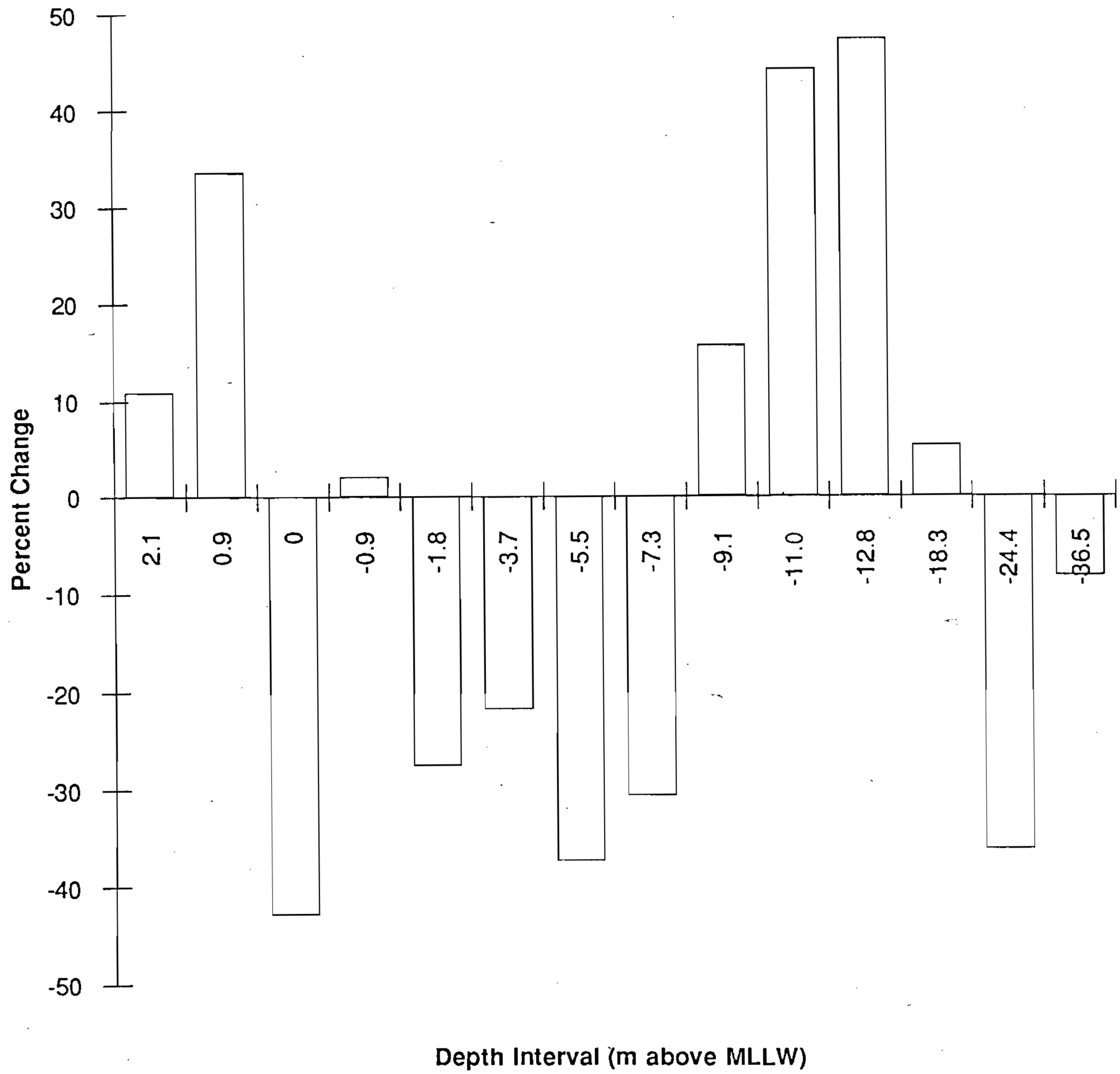

Figure 6.--Percent change in the area represented by each depth interval shown in Fig. 5 between 1868 and 1958. Depth intervals are defined as in Fig. 5. Data from Sherwood et al. (1990).

Table 1.--Past and present areas (ha) in the Columbia River estuary by depth interval and habitat type. The upper estuary is defined as upstream from Tongue Point (RKm 29), the lower estuary is downstream from Tongue Point. Habitat types are defined as (depths above mean lower low water [mllw]): deep water--area deeper than -5.5 m; medium water--area between -5.5 and -1.8 m; shallows/flats--area between -1.8 m and mean higher high water (mhhw); tidal marshes--vegetated areas between mllw and mhhw; tidal swamps--vegetated area above mhhw. After Thomas (1983).

Habitat	1870 area	Present area	Change in area	Percent change
Total Estuary				
Deep water	14,221	13,185	-1,036	-7
Medium water	13,845	10,409	-3,436	-25
Shallows/flats	16,447	18,118	1,671	10
Tidal marshes	6,548	3,723	-2,825	-43
Tidal swamps	12,149	2,813	-9,336	-77
Lower Estuary				
Deep water	8,078	8,191	113	1
Medium water	8,531	6,147	-2,384	-28
Shallows/flats	8,802	9,502	700	8
Tidal marshes	3,586	797	-2,788	-78
Tidal swamps	2,622	53	-2,570	-98
Upper Estuary				
Deep water	6,143	4,994	-1,149	-19
Medium water	5,314	4,261	-1,052	-20
Shallows/flats	7,645	8,616	971	13
Tidal marshes	2,962	2,926	-36	-1
Tidal swamps	9,527	2,760	-6,767	-71

at the entrance of Baker Bay, restricting circulation in Baker Bay. This caused increased deposition in the bay, resulting in a 72% loss of medium-depth water and a 75% increase in shallow flats. Wave energy reduced by the position of the jetties and Sand Island has allowed the establishment of tidal marshes in the entrance and Baker Bay in areas which previously were too exposed to allow successful vegetative colonization (Thomas 1983). Extensive tidal marshes and swamps in the lower estuary were diked and filled, reducing the area of tidal swamps by 98% and of marshes by 78% for a net loss of 5,358 ha (Table 1) (Thomas 1983).

Extensive dredging has been conducted in the lower estuary to maintain the shipping channel. Approximately 50 million m^3 of material was removed below RKm 40 before 1939, 1.7 million m^3 was removed annually from the entrance from 1958 to 1975, and 4.5 million m^3 is annually dredged from the entrance today (Sherwood et al. 1990).

Between the entrance and Tongue Point there has been little net change in area represented by each depth interval, although the geographic location of area in each interval has undergone considerable modification. The South Channel has been deepened for navigation while the North Channel has accreted (Thomas 1983), allowing greater downstream flow in the deeper South Channel, and less downstream flow in the shallower North Channel (Sherwood et al. 1984).

Upper estuary--The primary changes to the upper estuary (upstream of Tongue Point) have resulted from river channelization, by use of dredging, pile dikes, and dikes. Medium and deep water (> 1.8 m) area has decreased 19% while the area with water shallower than 1.8 m has increased 13% (Table 1) as the many side channels, especially in Cathlamet Bay, have filled. Dredge spoils have been used to create islands throughout the upper estuary, such as Rice Island, which covers 115 ha (Thomas 1983).

Dikes and levees are responsible for a 71% decrease in tidal swamps in the upper estuary, a loss of 6,767 ha (Table 1). Tidal marshes have only experienced a 1% loss (Thomas 1983), probably because their lower elevation makes them difficult to dike.

Physical Effects of Bathymetric Changes

The greatest effect of altered bathymetry in the Columbia River estuary has been a change in circulation and salinity intrusion (Sherwood et al. 1990). Conversely, because navigational improvements such as pile dikes alter the bathymetry by altering the current, the circulation of the estuary has altered the bathymetry. The flow of water has been concentrated in the navigation channel, with reduced flow in peripheral bays and channels (Hamilton 1984). This has radically altered the pattern of saltwater intrusion and river flow, creating asymmetrical fresh and saltwater flows. Incoming salt water must flow against outgoing fresh water; consequently, salt water is able to intrude farther in areas with minimal river flow. In the unaltered estuary, saltwater intrusion was approximately equal in the North and South Channels, since river flow was roughly equal in both channels. Now that river flow has been concentrated in the South Channel, saltwater intrusion is much greater in the North Channel, and the South Channel remains much less saline (Hamilton 1984). Because many of the peripheral bays have become shallower, the importance of wind for circulation and mixing in the bays has increased (Jay 1984).

One effect of filling in the estuary has been to significantly reduce the tidal prism. The loss of deep-water areas, coupled with reduction in intertidal areas, has resulted in a 10 to 15% reduction of the tidal prism of the estuary (Sherwood et al. 1984). In addition, the entrance shoal is shallower, with a single deep channel (Sherwood et al. 1990), that alters the flow of salt water across the entrance bar. The net effect has been a decrease of tidal energy in the estuary, resulting in decreased salinity intrusion for any given river flow (Hamilton 1984).

The Columbia River estuary has generally become more favorable for sediment deposition due to the concentration of river flow into a relatively narrow channel, with reduced flow in side channels and peripheral areas (Sherwood et al. 1990). Reduced peak flows have also favored sediment deposition, as previously discussed. Reduced flow in peripheral areas has also reduced shear stress, decreasing the likelihood of sediment particle movement, and increasing the settlement rate of suspended particles (Sherwood et al. 1984). Because of the shallower bar, the modern estuary also has less tidal energy to move sediment. The net result of these effects is increased residence time of sediment particles in the estuary (Sherwood et al. 1990).

Biological Effects of Bathymetric Changes

Because much of the Columbia River estuary was converted from one depth to another as a result of navigational improvements and other anthropogenic activities, the biota in the estuary have been impacted by changes in habitat types. Based on area alone, the 1,671 ha increase in shallows and flats represents a considerable expansion of available habitat for intertidal and shallow subtidal benthic species. Species requiring deeper or shallower habitats have not benefitted from these changes. Forty-three percent of the area at depths between -1 and -5.5 m MLLW has been lost, although the area deeper than -12.8 m MLLW has increased by 15%, but this area is restricted to the entrance (Sherwood et al. 1990). In addition, because of altered salinity intrusion and circulation, other habitat attributes, such as substrate, flow, and salinity range have also been altered. Concentrated flow in the main channel and decreased flow in side channels have decreased the amount of medium-deep, medium-velocity habitat, while the area of shallow, low-velocity, soft-bottom habitat has increased. With a reduced tidal prism, habitats in the estuary have also become less saline seasonally.

Marsh and swamp habitats have received the greatest impact because of their huge loss of area, primarily from the peripheral bays (Thomas 1983). Over 9,000 ha (77%) of tidal swamps and 2,800 ha (43%) of tidal marshes have been lost from the estuary (Table 1), although the loss of tidal marshes has been slightly moderated by the formation of new marsh in previously inhospitable habitats (Thomas 1983). Losses of tidal marsh have been particularly high in Youngs Bay (86%), Baker Bay (56%), and upstream from Aldrich Point (64%), while considerable loss of tidal swamp has occurred in Youngs Bay (96%), Baker Bay (100%), Grays Bay (88%), Cathlamet Bay (49%), and upstream from Aldrich Point (80%) (Thomas 1983).

Emergent marsh vegetation is extremely important as a primary food source in estuarine food webs (Thomas 1983, Small et al. 1990). Decaying vegetation is especially important as detritus, and many detritivores such as bivalves, gammarid amphipods, and polychaete annelids live or feed in sloughs or flats adjacent to marshes (Jones et al. 1990, Thomas 1983). Marshes and their associated tidal channels also provide important habitat for birds, mammals, fish, and insects. Simenstad et al. (1990) calculated that because of the reduction in tidal marshes and consequent reduction in detrital export, the present estuary supports 12 times fewer infaunal detritivores than it did prior to 1870, assuming detrital production is the factor controlling detritivore populations. This decreased production results in decreased available food for detritivore predators such as juvenile fishes, including salmonids, and shorebirds, whose populations may have also decreased (Sherwood et al. 1990).

Because tidal swamps occur at a higher elevations than tidal marshes, they have limited value as habitat for aquatic estuarine species. However, they play a significant role as a source of habitat for insects that serve as food for aquatic organisms, especially juvenile fishes such as salmonids[5]. In addition, these swamps provide habitat for upland-oriented species such as mammals, birds, and amphibians (Thomas 1983).

One unique impact of the altered bathymetry concerns the islands created from dredge spoils. Seven islands (Rice Island, Mott Island, Lois Island, Miller Sands, Jim Crow Sands, unnamed sandbar near Tenasillahe Island, and unnamed sandbar near Puget Island), covering over 420 ha, were created from formerly deep to shallow areas by filling with dredge spoils (Thomas 1983). The intertidal and subtidal habitats associated with some of the islands are among the most productive in the estuary, and support a diverse and abundant fish assemblage, including juvenile salmonids (Hinton et al. 1990).

In addition, the supratidal area provided by the islands may also benefit fish predators and competitors. The islands serve as nesting and resting sites for piscivorous gulls (*Larus* spp.), terns (*Sterna* spp.), mergansers (*Mergus* spp.), cormorants (*Phalacrocorax* spp.) and grebes (*Aechmophorus* spp., *Podiceps* spp.), and large flocks have been observed feeding on fish adjacent to the river islands (Hazel 1984, Hinton et al. 1990). Double-breasted cormorants (*Phalacrocorax auritus*) also nest on navigation markers between Miller Sands and Rice Island, possibly to prey on fish congregating at the islands (Hazel 1984). Scoters (*Melanitta* spp.) and peeps (*Actitis* spp., *Calidris* spp.), which may compete with fishes for bivalves and the gammarid amphipod *Corophium*, also nest on the islands (Hazel 1984).

[5]Greg Hood, Doctoral Candidate, Fisheries Research Institute, University of Washington, Seattle, WA 98195. Pers. commun., October 1992.

Harbor seals (*Phoca vitulina*) and occasionally California sea lions (*Zalophus californianus*) also use the dredge-spoil islands as haulouts during low tide, from which they have easy access to numerous estuarine prey (Everitt et al. 1981, Fox et al. 1984).

Summary of the Effects of Dams from Bathymetry Alterations

Although most bathymetric changes in the Columbia River estuary result from navigational improvements and agricultural conversion of floodplains, dams have contributed to some of the changes through flow regulation. Decreased fluvial energy resulting from the suppression of peak floods has decreased sediment transport through the estuary, which has contributed to the increased deposition and accretion in the estuary. Increased deposition and accretion may have limited the availability and amount of depth-specific habitats, while recently deposited sediments may be unconsolidated and easily resuspended. Consequently, dams may have contributed to a shift from deeper habitats with firm substrates, to shallower, soft-bottom habitats.

Water Quality

The major sources of alterations in water quality[6] in the Columbia River estuary are dams, reservoirs, irrigation and other land-use practices, and industrial and domestic pollution. Because many substances remain in suspension as water travels downstream, water quality alterations occurring great distances upstream may still be detectable when the water reaches the estuary. Consequently, it is important to consider factors which potentially alter water

[6]The term *water quality* refers to measurable parameters of water, such as temperature, dissolved oxygen, pH, concentrations of various ions and nutrients, dissolved solids, hardness, etc.

quality, regardless of where they occur in the system, in order to understand water quality in the estuary.

Despite a data gap from 1912 to the 1950s, water-quality records for the Columbia River are reasonably good. Van Winkle (1914) conducted water-quality surveys of the Columbia River and its major tributaries in 1910-12. More extensive surveys were conducted in the 1950s by the University of Washington Department of Engineering (Sylvester 1957, 1958, 1961; Sylvester and Ruggles 1957; Sylvester and Seabloom 1962). Since 1978, the USGS has conducted extensive annual water-quality surveys as part of its water resource program (USGS 1978-84, McGavock et al. 1985-88, Miles et al. 1989-92).

Although dams have some direct impact on water quality, anthropogenic activities resulting from the presence of dams may have an equal or greater impact (Sylvester 1958). For example, extensive irrigation in the Columbia River basin would not have been possible without the diversion and storage of water made possible by dams. Irrigation affects water quality through water depletion and by concentrating nutrients and increasing temperatures of return flows. Construction of hydroelectric dams has allowed the establishment of industries requiring large amounts of electricity, and these industries may degrade water quality through near-continuous effluent discharge.

To determine how dams directly or indirectly affect water quality, the major sources of water quality alterations on the Columbia River are detailed in this section, and Columbia River water-quality changes over time are reviewed. The effects resulting from dams that may cause or contribute to observed changes in Columbia River water quality are then summarized.

Sources of Water Quality Alterations

Dams--The primary effects of dams on water quality are temporal alterations in downstream water temperature due to reservoirs and gas supersaturation caused by water spilling over dams. Dams have allowed extensive irrigation and industrial development, which have also impacted water quality.

Reservoirs--The cumulative effects of run-of-the-river reservoirs on the Columbia River are expected to be a 1-2 °C increase in annual maximum water temperature and a 1-2 °C decrease in annual minimum water temperature below the dams. However, there is no expected change in mean annual water temperature (Jaske and Goebel 1967). In contrast, large reservoirs dampen water temperature extremes, resulting in decreased temperature amplitude, and delays in timing of annual temperature rise and fall, although the annual average temperature is generally not altered (Sylvester 1958). For example, Lake Roosevelt, above Grand Coulee Dam, is estimated to lower water temperature below the lake 0.6 to 1.7 °C in summer and raise water temperature up to 3.9 °C in winter (Sylvester 1958, Davidson 1969), while the annual temperature cycle has been delayed by approximately 1 month (Sylvester 1958, Jaske and Goebel 1967). These changes are characteristic of large reservoirs in general, and have been documented in other regions (Crisp 1987).

Reservoirs also affect the chemical characteristics of water quality. Local impacts on water chemistry may occur because of high biological oxygen demand (BOD) in newly filled reservoirs (Sylvester 1958) or increased sedimentation from reservoir bank erosion (Davidson 1967). Reservoirs also appear to moderate fluctuations in water quality between periods of high and low flows (Sylvester 1958, Park et al. 1969). In this respect, reservoirs may serve as as a buffer for otherwise harmful water quality conditions by diluting and slowing the passage of contaminants.

Because water velocity is reduced in reservoirs, they allow the accumulation of nutrients and sediments which would otherwise be transported downstream (Robeck et al. 1954, Puig et al. 1987), resulting in increased downstream water clarity (Stober et al. 1979, Stober and Nakatani 1992). Nutrient retention has contributed to a proliferation of phytoplankton and aquatic vegetation in the reservoirs, which can support a diverse food web (Davidson 1967, Stober et al. 1979, Bristow et al. 1985). Phytoplankton are able to remove large quantities of nutrients from the water during periods of peak primary production; an estimated four- to sevenfold reduction in phosphate and nitrate occurs between Pasco, Washington, and Clatskanie, Oregon, in late spring and summer, compared to a 10% reduction of these nutrients in winter (Park et al. 1970). Some of this productivity is transported downstream and used in the estuary (Sherwood et al. 1990).

Dissolved gas supersaturation--Another impact of dams on water quality has been dissolved gas supersaturation caused by water plunging into the tailrace after passing through spillways (Ebel and Raymond 1976). Water and entrained air passing through spillways plunges into the tailrace, resulting in water supersaturated with atmospheric gases. High levels of dissolved gas supersaturation (>110%) can cause gas bubble trauma in fish and invertebrates, characterized by the presence of gas bubbles accumulating in various tissues. The disease may impair swimming ability and reproduction, alter behavior and blood chemistry, and cause blindness or even death (Ebel and Raymond 1976, Weitkamp and Katz 1980). Elevated levels of dissolved gas supersaturation are not restricted to water directly below dams: dissolved gas supersaturation levels only decrease slightly between their source and the river mouth. For example, between 1966 and 1968, nitrogen saturation near the mouth of the Columbia River at Astoria, Oregon, (RKm 22), some 213 km below the nearest dam, exceeded 110% from May to August, reaching a maximum saturation of nearly 125% (Beiningen and Ebel 1970, Ebel 1971).

Attempts to minimize dissolved gas supersaturation resulting from spill with the use of spillway deflectors ("fliplips") have been quite effective in decreasing deep plunging of the water entering the tailrace (Johnsen and Dawley 1974). Spillway deflectors have been installed on most dams on the Columbia and Snake Rivers. In addition, the increasing number of turbines in dams throughout the river has decreased the volume of water passing through spillways, thereby decreasing the potential for high concentration of dissolved atmospheric gas. However, high spill rates continue to increase gas saturation levels (Weitkamp and Katz 1980).

Irrigation--Because of the arid conditions and the ability to pool the abundant river water throughout most of the Columbia River basin, irrigation has been extensive, especially in the Snake and Yakima River basins (Sylvester 1958). The area under irrigation has increased from 2,000 km^2 in 1900 to over 31,600 km^2 in 1980 (Sylvester 1958, Sherwood et al. 1990), with a total annual water withdraw of 13 million m^3 in 1985 (CRWMG 1986).

Irrigation return flows contain constituents absorbed from soils, including fertilizers, pesticides, and other chemicals applied to crops (Stober et al. 1979). Return flows impact both surface- and subsurface-water quality (Walker 1960, Sylvester and Seabloom 1962). Irrigation water is lost to evaporation and plant transpiration, concentrating the remaining substances (Stober et al. 1979). Many of the pesticides applied to crops in the Columbia River basin, including those applied in the past that contained DDT, have been transported to the estuary, where they can occur at high levels in sediments (NOAA 1987, 1989). High nutrient loads in irrigation return flows are readily used by phytoplankton in the mainstem reservoirs, providing some basis for productive food webs which export organic material downstream (Sherwood et al. 1990).

In addition to absorbing and concentrating nutrients and pesticides, irrigation water also absorbs heat as it passes through shallow ditches and furrows. Concurrently, low volumes of water remaining in the river decrease in velocity and can therefore absorb more heat, resulting in further increases in river temperature (Sylvester 1958). Although no basin-wide estimates of irrigation-caused river warming have been calculated, Sylvester and Seabloom (1962) estimated that at 1960 levels of irrigation, water temperature over a 115 km section of heavily irrigated Yakima River increased 4.0 °C, primarily due to irrigation.

In the Columbia River basin, much of the irrigation return flow occurs during the period of high river flow, which increases the dilution of nutrients and pesticides (Sylvester 1958, Wilcox 1960, Stober et al. 1979). Although irrigation depletion from the Columbia River represent 7-10% of the annual flow (Sherwood et al. 1990), the relative amount of irrigation depletion from the Columbia River compared to other heavily irrigated systems is minor. For example, as the Rio Grande crosses Texas, 80% of the water is removed for irrigation with a concomitant tenfold increase in dissolved solids (Wilcox 1960). Similarly, approximately half of the Sacramento/San Joaquin River flow is currently diverted for irrigation and other uses, with over 70% of the flow diverted during dry years (Monroe et al. 1992).

Pollution--Pollutants in the Columbia River come from many sources, the two primary sources being agriculture (as discussed above) and industrial and domestic wastes (Karrick and Gruger 1976). These pollutants may contain toxic compounds such as metals, organic and inorganic contaminants, radioactive isotopes, and petroleum products. Effluents may require huge amounts of oxygen for organic decomposition, and are also often sources of thermal pollution (Karrick and Gruger 1976). Of the over 300 facilities discharging into the Columbia River and its tributaries below Bonneville Dam (Jay 1977), 27 discharge over 3,800 m^3 of effluent daily (Buchman 1989). Effluent released directly into the estuary raises the biological oxygen demand (BOD) in the estuary to more than 14,500 kg O_2 per day (Jay 1977).

Historically, one of the most visible pollution problems in the lower Columbia River was effluent from pulp and paper mills in Camas and Longview, Washington. Large quantities of sulfite liquor released from the mills promoted the growth of the slime *Sphaerotilus*, which reached sufficient densities to clog nets and consequently halt fishing in the estuary some 70 km downstream from the mills (Lincoln and Foster 1943). Glues and preservatives used in the manufacture of plywood have also been introduced into the Columbia River (Spies 1960).

The Willamette River, Oregon, was a major source of pollution in the Columbia River basin because of discharged effluent with high BOD and contaminant levels (Fish 1950, Gleeson 1972). However, since water quality improvement efforts in the 1960s, water quality has improved dramatically (Gleeson 1972). The Willamette River still remains a large source of river-born metals, primarily zinc (Buchman 1989). Willamette River pollution is greatest locally, and has much less impact on the mainstem Columbia River where contaminants are highly diluted (Fish 1950, Buchman 1989).

Prior to 1970, the Columbia River was the most radioactive river in the United States (Forster 1972). Radioactivity was released into the Columbia River in contaminated water used to cool Hanford nuclear reactors (Foster 1972). Although many of the radioactive isotopes have short half-lives, longer-lived radionuclides released by Hanford, such as ^{51}Cr, ^{54}Mn and ^{65}Zn, were detected throughout the river and off the coasts of Washington and Oregon (Forster 1972, Osterberg and Perkins 1972). Since six of eight reactors at Hanford were decommissioned by 1970, radioactive contamination of the Columbia River decreased, resulting in a rapid decrease in radioactivity in riverine and estuarine sediments and biota (Foster 1972, Renfro et al. 1972).

Effluents released from industries on the Columbia River have resulted in localized concentrations of metals in the water and sediments. These metals entered riverine and estuarine food webs (Damkaer and Dey 1986, Buchman 1989, Stober and Nakatani 1992). Concentrations of these metals appear greatest at sites of release, around dams where low water velocities in reservoirs fail to dissipate the metals, and in areas of high sediment deposition, such as Baker Bay. Elevated concentrations of cadmium, chromium, copper, lead, mercury, and zinc have been recorded from the sediments and waters of the Columbia River estuary (Buchman 1989).

Releases of oil into the Columbia River occur constantly at an estimated rate of 68 kg per day (Buchman 1989). Oil enters the river in runoff and storm sewers, in domestic and industrial wastes, and from marine lubricants, incompletely burned fuels, and spills (Clark 1976, Blahm et al. 1980). Although some components of oil evaporate and others decompose naturally, rendering it harmless to aquatic life, large quantities may have a significant impact on aquatic organisms (Clark 1976). Many organisms may ingest oil without absorbing it;

however, oil may also cause lethal and sub-lethal effects, such as decreased productivity, increased susceptibility to infection and predation, direct mortality, and disruption of habitats and food webs (Clark 1976, Seymour and Geyer 1992).

In 1980, from 113 to 220 m^3 of Bunker C fuel oil were spilled at RKm 164, resulting in contamination of sediments and fish from the site of the spill to the estuary (Blahm et al. 1980). Although the spill resulted in downstream decreases in fish and benthic invertebrate densities, no dead fish were reported and fish and invertebrate assemblages appeared to have recovered within a year (Blahm et al. 1980). The surprising resilience of some aquatic communities to large oil spills (Blahm et al. 1980), including the *Exxon Valdez* spill (Sturdevant et al. 1992), suggests that some biotic systems may be more resilient to small, chronic oil introductions than anticipated (Seymour and Geyer 1992).

Thermal pollution, the release of effluent or water at a different temperature than the river, is widespread throughout the Columbia River basin (Craddock 1976), although basin-wide estimates of water temperature change from the various sources are not available. The largest source of thermal pollution on the Columbia River is irrigation (discussed above), although there are many other sources (Craddock 1976). Hanford has been a source of thermal pollution, because water used to cool its nuclear reactors was returned directly to the river; often at 2 to 10 °C warmer than the mainstem (Craddock 1976). However, dilution by the river and eventual restrictions on the temperature of returning water resulted in actual river temperature increases of approximately 0.4-1.0 °C at the confluence of the Snake and Columbia Rivers (Jaske and Synoground 1970, U.S. Environmental Protection Agency [USEPA], Atomic Energy Commission [AEC], and National Marine Fisheries Service [NMFS] 1971). Other large sources of thermal pollution include industrial effluent, such as from pulp and paper mills, which may locally raise water temperature. Thermal pollution from outfalls between Bonneville Dam and Astoria are estimated to raise the river temperature 0.3 °C (Craddock 1976).

With better treatment of discharged industrial waste and domestic sewage, effluents entering the Columbia River have a less serious effect on water quality. However, the steady increase in human population and industry in the Columbia River basin has resulted in increased total effluent.

Land-use practices--Because rivers drain large areas of land, the activities occurring on land affect the quality of river water. As mentioned earlier, irrigation practices can have a great impact on water quality. Other land-use practices impacting water quality include logging and in-water log storage, grazing, and mining. Logging on tributary watersheds that drain directly into the Columbia River estuary has probably had the greatest impact on estuarine water quality. Both modern and historical logging practices alter the hydraulic characteristics of streams and dramatically increase the amount of sediment entering them (Wendler and Deschamps 1955, Dunford 1960, Moring 1982). The historical use of "splash" dams on tributaries, which facilitated floating of log rafts downstream, were particularly destructive to salmon runs because of the devastation of the stream bed and blockage to upstream migrants (Wendler and Deschamps 1955).

In-water log storage has been practiced within the Columbia River estuary as long as there has been logging (Envirosphere 1981). Principal areas used for log storage include the channels in and upstream from Cathlamet Bay, Elochoman Slough, and Youngs Bay and its tributaries (Envirosphere 1981). Log storage has been implicated in the alteration of estuarine habitats through several means. Benthic and emergent plant habitats underneath storage areas are subjected to compaction from logs that ground at low tides in intertidal areas, decreased light penetration from shading, and these storage areas are zones of high bark deposition. The logs themselves can leach potentially toxic compounds, while bark deposits may cause high BOD and can produce large quantities of sulfide (Envirosphere 1981). The log rafts may, however, provide habitat for birds and mammals, and potentially serve as cover for juvenile fishes (Envirosphere 1981).

Other land-use practices, such as grazing and mining are also responsible for increased sedimentation and turbidity in streams, and potentially toxic leachates (Karrick and Gruger 1976). The impact of these activities on estuarine water quality depends on their distance from the estuary, and the degree of local water-quality degradation involved.

Mt. St. Helens--On May 18, 1980, Mt. St. Helens erupted, covering over 280,000 km^2 with ash and sending tons of mud down the North and South Forks of the Toutle River and tributaries of the Lewis River (Whitman et al. 1982). One hundred million m^3 of sediment arrived at the confluence of the Cowlitz and Toutle Rivers after the eruption, while 34 million m^3 of sediment was deposited in the Columbia River, requiring extensive dredging of the shipping channel (CRWMG 1982, Flaherty 1983). High sediment loads in the Cowlitz River continued for several years after the eruption (Flaherty 1983, Sherwood et al. 1990). Turbidities in the Columbia River estuary reached 1,120 NTU for 3 days following the eruption, a level over 100 times greater than normal (McCabe et al. 1983).

In the estuary, the eruption of Mt. St. Helens appeared to impact both benthic production and fish feeding. Benthic invertebrate populations were reduced in areas covered by ash, although abundances were not reduced where ash was buried by or mixed with non-ash sediments (Brzezinski and Holton 1983). Following the eruption, some fish such as juvenile salmon and American shad (*Alosa sapidissima*), consumed a wider variety of prey and less *Corophium*, and had more empty stomachs than in the year preceding the eruption, although one year after the eruption, the diets had returned to "normal" (Emmett 1982). Low catches of adult salmon and steelhead in 1981 and 1983 were possibly related to the eruption; however, record catches occurred in 1985-87 (ODFW and WDF 1991). In general, although the eruption of Mt. St. Helens dramatically impacted Columbia River water quality, dramatic physical and biological effects of the eruption were relatively short lived, and the estuarine assemblage appeared highly resilient to the disturbance.

Former and Present Water Quality

Chemical constituents--Values of selected water quality parameters near Quincy, Oregon (RKm 87), the USGS recording site nearest the Columbia River estuary, are given in Table 2 for five dates in water year 1991 (October 1990 to September 1991). Included are the permissible levels above which each water-quality parameter is expected to cause detrimental biological effects (USEPA 1976). Although most concentrations for Quincy, Oregon, reported in Table 2 are below permissible levels, elevated concentrations of trace metals and chemicals have been recorded from the sediments and waters within the estuary, such as Baker Bay, Astoria docks and Skipanon River channel in the estuary (Buchman 1989). Levels of copper, mercury, zinc, DDT, and PCBs in the water at these locations occasionally exceeded EPA water quality criteria for safe levels, while aqueous lead concentrations consistently exceeded the criteria (Buchman 1989). In the sediments of these areas, levels of cadmium, copper, mercury, and zinc exceeded several federal sediment quality criteria (Buchman 1989).

Table 2.--Selected water quality parameters near Quincy, Oregon (RKm 87), water year 1991 (October 1990 to September 1991), and U.S. EPA water quality standards for freshwater aquatic life (USEPA 1976). Values in boldface exceed U.S. EPA water quality standards. All units are mg/L unless specified. Data from USGS (1992).

Parameter	Nov	Mar	May	Jun	Aug	U.S. EPA standards
Temp. (°C)	10.5	6.0	12.5	15.0	20.0	
pH (standard units)	7.7	7.9	8.3	8.1	8.1	6.5-9.0
Turbidity (NTU)	3.5	7.5	7.6	5.5	2	
Suspended sediment	12		37	21	10	
Dissolved oxygen	10.6	12.6	11.7	10.8	8.7	
Dissolved oxygen (%)	95	101	110	107	96	110
Hardness (as $CaCO_3$)	54	54	57	50	59	
Alkalinity	51	51	52	45	54	
Sulfate (as SO_4)	10.0	9.9	8.9	8.4	8.2	
Chloride	6.1	3.9	3.9	3.3	1.3	
Fluoride	0.2	<0.1	0.2	0.1	0.2	
Silica (as SiO_2)	10.0	11.0	10.1	10.2	5.8	
Dissolved solids	95.7	81.0	77.3	77.3	73.6	250[a]
Total nitrogen	0.2	0.2	0.7	0.3	0.4	10 mg/L nitrate[a]
Total phosphorus	0.03	0.04	0.04	0.04	0.04	
Potassium	1.2	1.1	1.2	0.9	0.8	
Sodium	6.8	5.9	6.2	4.6	3.3	
Aluminum (μg/L)	30	30		<10	<10	
Copper (μg/L)	**3**	**3**		**4**	**3**	1.0[a]
Iron (μg/L)	30	31		17	6	1,000
Zinc	<3	5		5	4	

Other metals with levels less than or equal to 1 μg/L (U.S. EPA standards (μg/L) in parentheses): arsenic (50), cadmium (0.4), chromium (100), lead (50[a]).

[a] Limits for domestic water supply. Limits for aquatic life not specified.

On a regional basis, total contaminant levels in Columbia River estuary sediments consistently exceed those of other Oregon estuaries (Buchman 1989). Nationally, sediment concentrations of several chlorinated pesticides in the Columbia River estuary rank in the top 23 of 200 sites sampled throughout coastal United States (NOAA 1989). Although the Columbia River estuary is generally considered relatively unimpacted by human development compared to heavily industrialized estuaries of the east coast and southern California (Stober and Nakatani 1992), it too, shows the effects of modern industry and agriculture.

The chemical composition of Columbia River water has undergone alterations over time. Table 3 gives monthly values for several parameters at Maryhill/Cascade Locks/Warrendale (RKm 227-338) in water years 1912, 1953 and 1991, and Table 4 displays numerous water quality parameters in August of 1909, 1955, and 1991 lower in the river at RKm 48-87. These limited comparisons serve as indicators of water quality trends over time. Although levels of many parameters at Maryhill/Cascade Locks/Warrendale increased between water years 1912 and 1953, current parameter levels are generally similar to, or lower than, those of 1912 (Table 3). Stober et al. (1979) reported decreased turbidity in the Columbia River mainstem between 1954-57 and 1975-78, which they attributed to the increased number of dams. Decreased nutrient (nitrate and phosphate) levels between 1952-53 and 1990-91 may be due, in part, to phytoplankton production in reservoirs, which were not as abundant in 1952-53. Although water quality appears to have been degraded between the 1910s and the 1950s, current Columbia River water quality, based on limited comparisons, appears to have improved since the 1950s.

Water temperature--Determining the effects of dams on downstream water temperature is complicated because rivers naturally gain and release heat as they flow

Table 3.--Water quality attributes at Maryhill/Cascade Locks/Warrendale (RKm 227-338), in water years 1912, 1953 and 1991. All values are in ppm. Data from Van Winkle (1914), Sylvester (1958), and USGS (1992).

Parameter	Year	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Total alkalinity	1912	63	51	61	45	39	44	51	61	59	64	66	64
	1953	71	82	78	68	62	73	66	80	75	86	90	89
	1991			64		53			55			63	
Total hardness	1912	67	53	60	50	49	47	55	59	71	71	73	68
	1953	83	93	89	74	63	64	71	76	80	95	98	99
	1991			71		57			57			66	
Dissolved solids	1912	110	110	113	98	78	72	79	91	92	105	111	111
	1953	146	149	149	135	133	135	116	100	111	92	120	118
	1991			94					78			97	
Sulfate	1912	14	10	13	10	8	9	10	12	13	13	14	15
	1953	22	21	17	14	14	14	17	20	26	27	26	20
	1991			11		8.9			8.3			13	
Silica	1912	16	20	20	19	12	12	10	11	10	14	12	14
	1953	15	17	15	11	6.5	7.5	8	8.6	11	11	13	16
	1991			9		9.8			6.8			8.9	
Iron	1912	0.04	0.04	0.05	0.24	0.11	0.1	0.06	0.04	0.04	0.01	0.02	0.01
	1953		0.08	0.1	0.17	0.07	0.03	0.02	0.04	0.07	0.02	0.02	0.02
	1991			0.02		0.01			0.01			0.01	
Chloride	1912	4.6	3.6	6	3.5	2.2	1.5	1.5	1.9	2.0	3.6	4.7	4.1
	1953	5.3	5.2	5.3	4.0	2.9	2.8	2.2	4.3	4.2	6.5	6.9	6.1
	1991			2.8		1.6			3.6			4.7	
Nitrate	1912	0.4	0.6	0.6	0.5	0.3	0.3	0.3	0.6	0.5	0.5	0.8	0.6
	1953	1.9	1.1	1.2	0.9	0.5	0.5	0.3	0.4	1.0	0.9	1.1	1.4
	1991			0.3		0.08			0.04			0.2	

Table 4.--Comparison of selected water quality parameters at RKm 48-87 in August of 1909, 1955 and 1991. All units are ppm unless specified, T = trace. Data from Van Winkle (1914), Sylvester (1958), USGS (1992).

	Year		
Parameter	1909	1955	1991
Temperature (°C)		18.9	20
pH (standard units)		7.95	8.1
Turbidity (NTU)		6	2
Dissolved oxygen		9.6	8.7
Dissolved oxygen (%)		104	96
Hardness (as $CaCO_3$)	33	65	59
Sulfate (as SO_4)	5.5	14.4	8.2
Chloride	5.5		1.3
Silica (as SiO_2)	2.4		5.8
Dissolved solids		78	73.6
Total phosphorus	0.08		0.04
Potassium	4.1	1.5	0.8
Sodium	4.0	2.7	3.3
Aluminum (µg/L)		500	<1
Copper (µg/L)		0	3
Iron (µg/L)		150	6
Magnesium		1.1	4.1
Calcium		17	17
Zinc		0	4
Ammonia	0.42	T	0.02
Solids at 180 °C	52		76

downstream. The rate and direction of heat transfer depends primarily on air and tributary temperatures, and the tributary size (Sylvester 1958). Water temperature is further impacted by anthropogenic and natural sources as it flows downstream.

One method to differentiate between sources of temperature alteration is to examine the expected and observed patterns of temperature change. For example, large storage dams delay the cycle of seasonal maximum and minimum temperatures, and moderate the temperature extremes; run-of-the-river dams increase temperature extremes; irrigation increases water temperature in late spring and summer; and industrial sources of thermal pollution are expected to increase water temperature throughout the year. In contrast, climatic conditions may cause large interannual variations with no distinct pattern.

Water temperatures from Sylvester (1958) and USGS (1978b-85b, 1986-1992) were compiled to detect any relationship between temperature and the storage capacity of the hydroelectric system. This comparison was hampered by a paucity of data and sampling locations. Figure 7 illustrates five-year averages of annual maximum (August and September) and minimum (January and February) water temperatures at Bonneville Dam/Warrendale (RKm 227-235) for water years 1938-91. (Data for 1956-76 were not available.) Averages were used to minimize annual variability and allow the examination of general trends. Prior to 1942, there was less than 0.2 km^3 of active water storage potential on the Columbia and lower Snake Rivers; between 1942 and 1967 active storage increased to 6.6-8.7 km^3; and since 1973 active storage potential has increased from 35.0 to 42.0 km^3 (CRWMG 1986). Despite these dramatic changes in the storage capacity of the Columbia River Basin, minimum temperatures show no consistent difference over time for February, while January temperatures show an erratic increase from exceptionally low temperatures before 1945, to

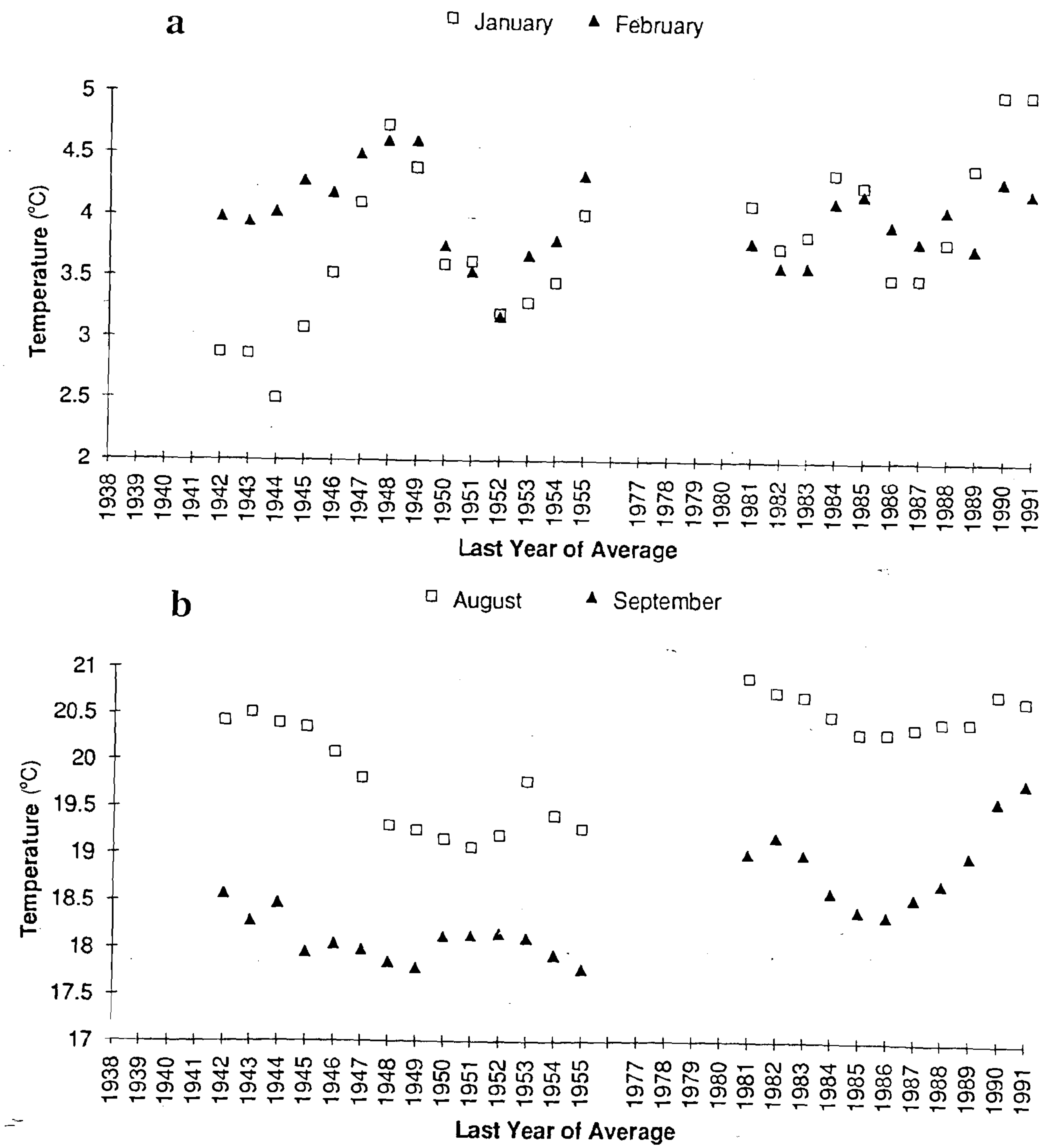

Figure 7.--Five year averages of mean monthly water temperature at Bonneville Dam/Warrendale during annual minimum temperatures (January and February) (a) and annual maximum temperatures (August and September) (b) for water years 1938-91. Data for 1956-76 were not available. The year given for each data point is the last year computed in the average. Data from Sylvester (1958), USGS (1978b-92b).

exceptionally high temperatures in 1990 and 1991. However, the timing of minimum temperature appears to have changed: prior to 1955, with less than 8.7 km^3 of active storage, minimum temperatures primarily occurred in January, while since 1980, with over 35 km^3 of active storage, minimum temperatures primarily occurred in February. Both the increase of minimum temperatures and the delay suggest dam-induced temperature effects. However, the increase in minimum temperatures could also be caused by climatic variation, especially since such short-term variability is typical of climatic variability. In contrast, the delay of minimum temperatures may be caused, in part, by the amount of reservoir storage.

Using 5-year averages, maximum temperatures in August and September generally decreased between 1942 and 1949 and then increased by 1977 (Fig. 7). This pattern does not appear to follow the increasing storage potential of the Columbia and lower Snake Rivers during those times, as described above. Furthermore, August temperatures exceed September temperatures for all years, suggesting no lag in annual timing of maximum temperatures. However, the direct effects of dams on these temperatures may be masked by other factors affecting water temperature, such as irrigation and other sources of thermal pollution. Like the minimum temperatures of January and February, high variability in maximum temperatures suggests considerable climatic variation.

Based on this limited examination of water temperature changes, it appears that the only distinctive, potentially dam-induced temperature cycle alteration at Bonneville Dam between 1938 and 1991 is the change in the timing of minimum temperature from January to February. The high variability and recent increase of five-year-averaged maximum and minimum water temperatures suggests climate-induced causes for the observed trend in water temperatures. Water temperature is in part a function of air temperature (Sylvester 1958), and air temperatures in the Columbia River basin have been slightly higher than average since 1985 (NOAA 1990). These warmer air temperatures may have contributed to the warmer water temperatures, although correcting water temperatures for air temperature for purposes of comparison is beyond the scope of this paper. Coincident to the construction of dams and increased water storage capacity of the Columbia River during this period, there have also been dramatic increases in other factors which may also affect water temperature, such as the amount of water withdrawn for and returned from irrigation, and the amount of thermal pollution entering the Columbia River (see above). However, it appears that climatic conditions may cause greater alterations in water temperature in the lower Columbia River than anthropogenic influences (Simenstad et al. 1992).

Biological Effects of Altered Water Quality

The increased levels of chemical contaminants in the Columbia River estuary may impact the biota. Although levels of most contaminants given in Table 2 are below acceptable levels, trace metals and compounds are concentrated in sediments and overlying water in several locations in the estuary (Buchman 1989). Effects of the observed levels are largely unknown, but subacute levels of toxins may result in long-term lethal effects (Buchman 1989). Sublethal exposure to contaminants may impact reproduction, mobility, and reaction times, which result in diminished fitness of the individual and potentially of the population, and may have far-reaching effects within the food web (Buchman 1989). Determining specific effects on individual species is complicated because of widely diverse species-specific tolerances to contaminants (Buchman 1989), although salmon are relatively sensitive (USEPA 1976). The effects of trace metal levels on biota is further complicated because of the metals' complex chemistries in water, and the differential toxicity of various metal forms (Buchman 1989). Although contaminant levels in the Columbia River estuary are below acute levels, they are sufficiently high to stress organisms, and potentially cause significant long-term effects (Buchman 1989).

Alterations in water temperature have been associated with numerous biological impacts to fish and invertebrates. Altered temperatures may directly affect fish by altering cellular metabolism, growth, reproduction, behavior such as predator-prey interactions, mobility, and susceptibility to and prevalence of disease. Temperature indirectly affects fish by altering prey and predator abundance, and both direct and indirect affects may increase mortality rates (Sylvester 1958, Blahm and Snyder 1975, Stober et al. 1979, Crisp 1987, Jensen 1987). Exotic species, adapted to warmer or colder temperatures than indigenous species in the undisturbed river often benefit from altered temperature regimes at the expense of native species (Holden and Stalnaker 1975, Achieng 1990, Neiland et al. 1990, Neves and Angermeier 1990, Ogutu-Ohwayo 1990). Upstream temperature alterations may partially explain the proliferation of exotic species in the Columbia River (Brege 1981, Maule and Horton 1985, Stober and Nakatani 1992) because exotic species may have a competitive advantage in the temperature-modified niche. The slight increase in maximum and minimum water temperatures in the Columbia River may have some impact on biological processes in the estuary, although this has not been documented.

Like other changes in the physical environment of the Columbia River estuary, biological changes in the estuary caused by the effects of dams on water quality may have been considerable. However, the lack of documentation of prior conditions makes such changes nearly impossible to verify.

BIOLOGICAL CHARACTERISTICS OF THE COLUMBIA RIVER ESTUARY

Introduction

Compared to the physical environment of the Columbia River estuary, the biological characteristics have received very little study. The only biological data collected in the estuary prior to the 1960s were in relation to commercially important species. This long-term data set consists primarily of catch data for salmonids (*Oncorhynchus* spp.), eulachon (*Thaleichthys pacificus*), white sturgeon (*Acipenser transmontanus*), and American shad, with only anecdotal observations on species ecology (Craig and Hacker 1940). Catch data have been extremely useful in estimating run sizes during the 1800s and early 1900s (Chapman 1986) and identifying long-term trends in fish and run abundances (ODFW and WDF 1991). However, catch data provide little information on the ecology of the estuary. The first ecological investigation of the estuary was conducted in 1963-65 (Haertel and Osterberg 1967) after the completion of nine mainstem dams on the river: Rock Island, Bonneville, Grand Coulee, McNary, Chief Joseph, The Dalles, Priest Rapids, Rocky Reach, and Wanapum Dams, in addition to extensive anthropogenic alterations in the estuary, such as navigational improvements and extensive diking. Consequently, aside from the catch records and estimated run sizes of commercially important species, there was no documentation of estuarine organisms before the construction of dams.

Since the 1960s, numerous biological investigations have been undertaken in the estuary. Many of these studies concern limited areas in the estuary over short time periods, and are consequently of limited utility for this report. However, the intensive, comprehensive study of the biology of the estuary as part of the CREDDP study provided a wealth of recent information on Columbia River estuarine ecology and suggested changes which have occurred historically. Like the physical environment, the estuarine community is extremely dynamic, with processes occurring over a wide range of spatial and temporal scales.

In this section, what is known regarding the biological environment of the Columbia River estuary is reviewed, how it may have changed over time, and the effect of dams on the observed or predicted changes. The review is divided by functional groups of organisms (primary producers, non-fish consumers, and fishes), with each group considered separately, followed by a discussion of their interactions in the estuarine food web. The long-term data sets and the current status of commercially important species are also discussed. The effects of exotic species in the estuary are also reviewed: what is known about them and their potential impacts on the estuarine community. Finally, the potential role of dams in effecting the changes in the estuarine community is discussed.

Primary Production

Water Column Primary Production

Primary production in the Columbia River estuary occurs in the water column and in the sediment. *In situ* primary production in the water column is limited to marine phytoplankton in the lower estuary, which represent only 25% of the total phytoplankton in the total estuary. The water-column diatoms *Melosira* spp., *Asterionella formosa*, and *Syndra ulna* are the most numerous phytoplankton species and essentially form an estuarine extension of riverine plankton as they are transported through the estuary to the ocean (Haertel et al. 1969). Phytoplankton are most abundant in the estuary from April to September, when light levels are highest, and reach maximum densities in May and June (Haertel et al. 1969; Lara-Lara et al. 1990a, 1990b). The majority of phytoplankton originating upstream lyse on contact with the saltwater in the estuary; surviving phytoplankton are swept to sea (Frey et al.

1984). This results in decreasing phytoplankton abundance downstream, and a huge input of microdetritus into the estuary from the lysed cells (Haertel et al. 1969, Lara-Lara et al. 1990a). In comparison, grazing by zooplankton is thought to remove 1% of the phytoplankton biomass per day (Frey et al. 1984).

Phytoplankton production in the Columbia River estuary appears to be limited by light, and possibly by nitrogen in late spring and summer (Frey et al. 1984). The effect of light limitation on phytoplankton production was clearly demonstrated during the highly turbid period following the eruption of Mount St. Helens, which coincided with a 75% decrease in primary productivity (Frey et al. 1983).

The water column primary production rate in the Columbia River estuary (90 grams carbon per meter squared [g C m^{-2}] per year) is lower than either upstream or offshore production rates, and lower than that of other estuaries (Small and Frey 1984, Lara-Lara et al. 1990a). This low productivity is thought to result from the relatively fast (1-5 day) flushing rate of the estuary, which prohibits extended residence time and reduces the potential to maintain reproductive phytoplankton populations in peripheral bays (Frey et al. 1984, Lara-Lara et al. 1990a). The amount of phytoplankton imported into the estuary is thought to have increased from 9,000 tons carbon [t C] per year prior to the 1870s, to 36,000 t C per year at present (Sherwood et al. 1990). This increased rate of importation has resulted from increased abundances of phytoplankton upstream. Upstream phytoplankton have become more abundant because of increased concentrations and accessibility of nutrients and increased water temperature and water clarity (Sherwood et al. 1990). Most of these factors have increased as a direct or indirect result of dams.

Benthic Primary Production

Benthic primary production results from production by both benthic diatoms and rooted macrophytes. Although benthic diatoms may be consumed directly, macrophytes are an important source of macrodetritus and provide habitats for fish and invertebrates. Benthic diatom production is highest in the lower peripheral bays in the Columbia River estuary, such as Baker and Youngs Bays (McIntire and Amspoker 1984). This high benthic production is thought to result from the stability and particle size of sediments in these bays: benthic diatom production appears to be highest where sediment disturbance is minimal (McIntire and Amspoker 1984). The distribution of various benthic diatoms reflects the salinity gradient in the estuary: diatoms in the upper estuary are freshwater species, while those in Baker Bay are brackish-water species (McIntire and Amspoker 1984). Primary production rates for benthic diatoms in the estuary vary from 60-80 milligrams [mg] C m^{-2} per year in Baker and Youngs Bays to 20-40 mg C m^{-2} per year in the upper area, resulting in an annual production of 2,175 t C per year from benthic diatoms for the estuary (McIntire and Amspoker 1984).

Dredging is thought to have a large impact on benthic diatom production (McIntire and Amspoker 1984). The altered water chemistry and changes in the physical environment caused by extensive dredging are thought to have decreased benthic diatom primary production (McIntire and Amspoker 1984).

The major source of macrophyte-derived benthic primary production in the Columbia River estuary are tidal marshes, which are dominated by *Carex lyngbyei*, with limited contribution from eelgrass (*Zostera* spp.) beds (McIntire and Amspoker 1984). Like benthic diatoms, macrophyte assemblages are distributed according to salinity within the estuary, with freshwater species in the upper parts of the estuary and brackish-water species in the lower estuary. Above-ground biomass reaches its peak from June to August, and gross production is estimated at 964 g dry weight [wt] m^{-2} per year (McIntire and Amspoker 1984). Of this gross production, a minimum of 460 g dry wt m^{-2} per year is thought to be exported from the marsh as macrodetritus (McIntire and Amspoker 1984).

Macrophyte benthic primary production in the Columbia River estuary has decreased dramatically because of the decreased area presently occupied by macrophytes. The area covered by tidal swamps has decreased by 77%, while tidal marshes have declined 62% (Thomas 1983). This has resulted in a sixfold decrease in macrodetritus production, estimated to be 19,938 million t C per year in the unimpacted estuary (1870s), compared to 3,605 million t C per year at present (Sherwood et al. 1990).

Nonfish Consumers

The primary nonfish consumers in the Columbia River estuary include zooplankton and benthic and epibenthic invertebrates. The most comprehensive information on nonfish consumers comes from Haertel and Osterberg (1967) and the CREDDP research (Holton et al. 1984, Jones and Bottom 1984).

Zooplankton

Zooplankton in the Columbia River estuary are represented by both indigenous (i.e., their populations constitute reproductive units) and exogenous (i.e., taxa passively transported into the estuary from the river or the ocean) groups (Simenstad et al. 1984). The most abundant zooplankton groups include calanoid, cyclopoid and harpacticoid copepods, cladocerans, rotifers, and mysids (Haertel and Osterberg 1967, Jones and Bottom 1984, Simenstad and Cordell 1985).

Zooplankton are generally distributed by salinity range into three groups within the estuary: a freshwater group originating upstream, found from RKm 29 to 37; a brackish-water group, found in the mixing zone from RKm 16 to 29; and a marine group, found below RKm 16 (Haertel and Osterberg 1967, Jones and Bottom 1984). The freshwater group is dominated by the cladocerans *Bosmina longirostris* and *Daphnia* spp. and the cyclopoid copepods *Cyclops* spp. The copepods *Eurytemora affinis*, *Pseudodiaptomus inopinus*, and *Scottolana canadensis* are abundant in the brackish-water group, while the marine group is represented by the copepods *Acartia clausi*, *Centropages abdominalis*, *Pseudocalanus elongatus*, and *Paracalanus parvus* (Haertel and Osterberg 1967, Jones and Bottom 1984).

Zooplankton densities are generally highest in late spring and summer, and lowest in winter. The timing of peak abundance varies by location in the estuary: peak abundances occur in late spring in the marine region of the estuary, in early summer in the mixing zone, and in late summer in the freshwater region of the estuary.

The dominant zooplankters in the estuary, *E. affinis*, *P. inopinus* and *S. canadensis*, are associated with the estuarine turbidity maximum (ETM). The ETM is a zone of high turbidity caused by sediment entrainment by circulation and tidal resuspension of sediments where marine and riverine waters meet (Simenstad et al. 1984). The ETM supports concentrated primary productivity and provides a detrital food source for zooplankters (Simenstad et al. 1984). The ETM, associated with the null zone of the salt wedge, moves with daily tidal cycles, and is also affected by seasonal changes in river flow. Consequently, zooplankters associated with the ETM show daily and seasonal patterns of distribution, and are generally farther downstream during low tides and high river flow (HRF) or fluctuating river flow (FRF), and farther upstream during high tides and low river flows (LRF)

(Simenstad et al. 1984). Other zooplankters not associated with the ETM show similar seasonal movements within the estuary in response to seasonal salinity changes (Haertel and Osterberg 1967, Jones and Bottom 1984). The upstream, near-bottom currents associated with ETM also serve as a mechanism by which zooplankters can remain in the estuary despite net downstream flows (Haertel and Osterberg 1967, Simenstad et al. 1984).

The segregation of zooplankton into freshwater, brackish-water, and marine groups is altered seasonally due to changes in river flow and the consequent salinity intrusion and water-column mixing. During spring HRF and winter FRF, zooplankton assemblages are distributed by salinity tolerance: freshwater species (cyclopoids and cladacerans) are concentrated above RKm 25, and brackish-water (*E. affinis*, *P. inopinus*, and *S. canadensis*) and marine species (*A. clausi*, *P. parvus*) dominate below RKm 25 (Simenstad and Cordell 1985). During summer and fall LRF, however, the pattern of distribution becomes much more complex, and fresh, brackish-water, and marine species are found intermixed throughout the estuary. Simenstad and Cordell (1985) suggest that this distribution pattern results from the degree of mixing in the estuary. During FRF or HRF in winter and spring, the estuary becomes relatively stratified, and segregation by salinity preference occurs. During summer and fall, however, the LRF and relatively high tidal energy increase mixing of the water column, and allow transportation and mixing of fresh, brackish-water, and marine species.

Seasonal changes in riverflow may also determine estuarine zooplankton assemblages by influencing the influx of marine species, the contribution and distribution of food resources from upriver, water temperature and reproductive rates, and flushing rate (Jones and Bottom 1984). The direction of offshore currents and reservoir production also affects the composition of marine and freshwater zooplankton assemblages, respectively (Jones and Bottom 1984).

Haertel and Osterberg (1967) document the effects of a large flood in 1965, the magnitude of which no longer occurs because of flow regulation by dams. They observed three different responses to the flood by each of the three (freshwater, brackish-water, and marine) zooplankton assemblages. In the spring following the January flood, the freshwater assemblage decreased in abundance by two orders of magnitude compared to the previous year's densities, but recovered by the summer. The brackish-water assemblage, dominated by *E. affinis*, decreased by three orders of magnitude in the spring following the flood, compared to the previous spring, and was still two orders of magnitude lower by summer. In contrast, the marine assemblage showed no measurable effects of the flood by the following spring (Haertel and Osterberg 1967). This study clearly shows the influence of less-regulated river flow on the estuarine zooplankton community and its potential to dramatically alter zooplankton assemblages.

Benthic and Epibenthic Invertebrates

Much like the zooplankton, benthic and epibenthic invertebrate assemblages in the Columbia River estuary are strongly influenced by the seasonal salinity regime of the estuary, although sediment type and stability also influence benthic and epibenthic assemblage structure (Jones et al. 1990, Furota and Emmett 1993). The upper reaches of the estuary are dominated by freshwater species, such as the snail *Hydrobia*, the Asiatic clam *Corbicula fluminea*, numerous worms, the amphipod *Corophium* spp., and insect (chironomid) larvae (Haertel and Osterberg 1967). The assemblage in the brackish-water zone of the estuary includes the California bay shrimp (*Crangon franciscorum*), the amphipods *Eogammarus* spp.

and *Corophium* spp., and the mysid *Neomysis mercedis* (Haertel and Osterberg 1967, Jones et al. 1990). The marine region of the estuary is dominated by California bay shrimp, Dungeness crab (*Cancer magister*), the mysid *Archaeomysis grebnitzkii*, and the clam *Macoma balthica* (Haertel and Osterberg 1967, Jones et al. 1990).

The habitat type within each salinity zone also affects benthic and epibenthic invertebrate assemblage composition. For example, Dungeness crabs are restricted to channel bottoms, while *Corophium* spp. are typically found on tidal and shallow subtidal flats (Simenstad et al. 1984, Hinton et al. 1992a). The highest densities of benthic invertebrates occur on protected flats, while the lowest concentrations are in large channels (Jones et al. 1990). Benthic invertebrates display dramatic increases in abundance in spring and summer, although the species composition and diversity remain fairly stable (Holton et al. 1984).

Like some zooplankton, some mobile benthic and epibenthic invertebrates, such as *Corophium* and California bay shrimp also make seasonal migrations within the estuary in response to seasonal changes in river flow and salinity intrusion, and are found lower in the estuary during HRF and FRF, and higher in the estuary during LRF (Holton et al. 1984). Sessile species are also impacted by seasonal salinity changes, and salinity extremes may limit population growth (Furota and Emmett 1993).

Corophium in the Columbia River estuary are widely distributed, extremely abundant, and form the basis of many food chains (Jones et al. 1990). They produce two generations per year: the first generation in May from overwintering adults, and the second in July and August. This second generation then overwinters in the estuary (Holton et al. 1984). *Corophium* are absent from the lower estuary in winter, and are thought to migrate upstream to areas of lower salinity (Holton et al. 1984). For example, *Corophium* densities peak from

June through October in the lower estuary in Baker Bay as the result of both migration and reproduction (Furota and Emmett 1993). Farther up the estuary at Rice Island, mean densities of *Corophium salmonis* exceeded 88,000 m^{-2} during winter 1991 and 1992, while mean densities between 8,500 and 31,500 m^{-2} occurred there during summer 1991 (Hinton et al. 1992a, 1992b).

The relatively low species diversity and simple structure of the benthic invertebrate assemblage in the Columbia River estuary results from the highly dynamic physical environment (Holton et al. 1984). The strong currents, active sediments, and high tidal and seasonal salinity variability prohibit the establishment of more complex benthic assemblages (Holton et al. 1984). Instead, the benthos is dominated by a few opportunistic species, such as *Corophium*, which dominate the assemblage biomass and production and are the basis for estuarine food chains (Holton et al. 1984).

Fishes in the Columbia River Estuary

A wide variety of fishes can be found in the Columbia River estuary, ranging from newly hatched 4-mm-long eulachon to decades-old, 500-kg white sturgeon. They are adapted to marine, brackish, and freshwater habitats, and reside in the estuary for varying amounts of time; there are permanent residents and anadromous species that briefly pass through the estuary twice in their lives.

Like other biological attributes in the Columbia River estuary, most fishes, especially noncommercial species, were not studied until the 1960s. However, because of their commercial value, long-term catch records and some estimated run-size data are available for many of the anadromous species such as salmon and steelhead, sturgeon (*Acipenser* spp.), American shad, and eulachon. In this section, the distribution of resident[7] fish assemblages, the life cycle patterns of anadromous fishes, and finally long-term records of commercially important species in the Columbia River estuary are discussed.

Resident Fishes

Numerous surveys of noncommercial fish fauna have been conducted in various locations in the Columbia River estuary since 1963 (Haertel and Osterberg 1967). The largest and most comprehensive survey was conducted in 1980 and 1981 as part of the CREDDP (Bottom et al. 1984, Fox et al. 1984, Bottom and Jones 1990). The results of the CREDDP study generally substantiated prior studies, were consistent with subsequent work, and will form the basis of the following discussion. The occurrence, distribution, and food habits described in the following discussion represent general patterns, and individual fish may deviate from the described patterns.

Over 95 fish species, representing fish families from the evolutionarily primitive Petromyzontidae (lampreys) to the more advanced Pleuronectidae (flatfishes), have been recorded in the Columbia River estuary (Durkin 1980, Bottom et al. 1984). These fishes represent a wide range of residence times, distributions, food habits, and ages within the estuary, although the most abundant fishes are juveniles (Bottom et al. 1984). The distribution and seasonal abundance of fishes in the estuary are determined by fishes' patterns of migration and life history, preference for specific salinity regimes (marine, brackish-water, or freshwater), habitat types (i.e., bays, channel, pelagic, demersal), and prey (Simenstad et al. 1984).

[7]The term *resident* refers to fish species which are generally found in the estuary throughout the year, although individual fish may repeatedly enter or exit the estuary.

Seasonal occurrence--Of the most commonly caught fish species, most were present in the estuary throughout the year (Bottom et al. 1984). Fish species diversity was highest during spring, summer, and early fall (April through October), and lowest during late fall and winter (November through March) (Bottom et al. 1984). The largest group of fishes that were absent in winter but present during other seasons were subyearling American shad, Pacific herring (*Clupea harengus pallasi*), shiner perch (*Cymatogaster aggregata*), and starry flounder (*Platichthys stellatus*). These fishes, in addition to subyearling longfin smelt (*Spirinchus thaleichthys*), English sole (*Parophrys vetulus*), and juvenile chinook salmon, were highly abundant in summer (Bottom et al. 1984). Locally abundant fishes with the most restricted seasonal occurrences in the estuary included juvenile chum and sockeye (*O. nerka*) salmon, eulachon, northern anchovy, and Pacific lamprey (*Lampetra tridentata*) (Bottom et al. 1984).

Distribution--The distribution of resident fishes in the Columbia River estuary is best described by assemblage groups, which are generally distributed by salinity tolerance and habitat preference (Table 5) (Bottom et al. 1984). These groups show seasonal variation in distribution and assemblage structure, presenting a temporally and spatially complex pattern of fish distributions and associations.

Most of the common fishes in the estuary tolerate a wide range of salinities, and consequently their ranges within the estuary commonly encompass two, if not three salinity zones (marine, brackish water, or freshwater) (Table 5) (Bottom et al. 1984). The few exceptions to this distribution pattern include the freshwater restriction of white sturgeon during fluctuating river flow, and the freshwater restriction of largescale sucker (*Catostomus macrocheilus*), prickly sculpin (*Cottus asper*), and subyearling American shad during low

Table 5.--Common species assemblages, most frequent habitats, and range within the estuary for the three hydrologic seasons (fluctuating, high, and low river-flow). Numbers in parentheses are fish ages in years. Habitat types are: wc--water column; cb--channel bottom; ns--nearshore; sh--shoal; by--bay. Ranges are: mar--marine, below Chinook Point and Point Adams; brck--brackish, between the marine zone, and Tongue and Portugese Points; frsh--fresh, above the brackish water zone. From Bottom et al. (1984).

Assemblage	Habitat	Range
Fluctuating river-flow (November-March)		
American shad (2), surf smelt	wc	brck-frsh
American shad (1), eulachon, longfin smelt (1), threespine stickleback	wc, cb	mar-frsh
chinook salmon (0)	ns, sh, by	brck-frsh
Pacific staghorn sculpin, prickly sculpin, starry flounder (1 and 2)	cb, sh, ns	brck-frsh
Pacific tomcod, English sole, snake prickleback, butter sole	cb	brck-frsh
northern anchovy (1), sand sole, whitebait smelt	cb	mar-brck
High river-flow (April-June)		
American shad (1 and 2)	wc	mar-frsh
chinook salmon (0 and 1), steelhead, threespine stickleback, coho and sockeye salmon, cutthroat trout	wc, ns	mar-frsh
northern anchovy (1), longfin smelt (1), Pacific herring (1), surf smelt (1)	wc, cb	mar-brck
Pacific staghorn sculpin, English sole (0), starry flounder (1 and 2), shiner perch (1)	cb, by	mar-frsh
white sturgeon	cb	frsh
prickly sculpin, peamouth	cb	brck-frsh
Pacific tomcod, snake prickleback, English sole (1), Pacific sand lance, shiner perch (0), Pacific herring (0), butter sole, speckled sanddab	cb	mar-brck

Table 5.--Continued.

Assemblage	Habitat	Range
Low river-flow (July-October)		
American shad (1 and 2), surf smelt, Pacific herring (0 and 1)	wc	mar-frsh
chinook (0), peamouth, threespine stickleback	wc, ns	mar-frsh
river lamprey, white sturgeon	wc, cb	brck-frsh
northern anchovy, spiny dogfish, whitebait smelt, English sole (1)	cb	mar-brck
longfin smelt (0 and 1), sand sole, Pacific tomcod, English sole (0), snake prickleback, starry flounder (1 and 2)	cb	mar-frsh
butter sole	cb	mar
shiner perch (0 and 1), starry flounder (0), Pacific staghorn sculpin	cb, ns, by	mar-frsh
largescale sucker, prickly sculpin, American shad (0)	cb, ns	frsh

river flows. Butter sole (*Isopsetta isolepis*) is primarily caught in the marine region of the estuary during low river flows. Juvenile salmon are generally found in all three salinity regions in the estuary, although subyearling chinook are most common in brackish and freshwater regions during the winter (Bottom et al. 1984).

Distribution of fish assemblages within the different habitat types remained fairly consistent throughout the year (Table 5). For example, assemblages consisting of the planktivores, American shad, and various smelt (Osmeridae) were most common in the water column, while flatfishes (Pleuronectidae), white sturgeon, Pacific tomcod (*Microgadus proximus*), and snake prickleback (*Lumpensus sagitta*) were common in channel bottoms (Bottom et al. 1984). Some exceptions to these general trends include the use of channel bottoms by northern anchovy and whitebait smelt (*Allomerus elongatus*) during fluctuating and low river flows, and by Pacific sand lance (*Ammodytes hexapterus*), subyearling Pacific herring, and subyearling and yearling longfin smelt during high and low river flows, respectively.

Haertel and Osterberg (1967) reported that distributions varied on the much smaller time-scale of tidal cycle. At a site adjacent to Astoria, Oregon, freshwater fishes were only present during low tides, and marine fishes were only present during high tides, while brackish-water species were present during all tidal stages.

Distribution of fishes in the Columbia River estuary is believed to be influenced, in part, by prey availability and distribution. For example, many planktivorous fishes consume calanoid and harpacticoid copepods, which are associated with the ETM (Simenstad et al. 1984). The ETM moves up and down the estuary in response to tidal cycles and river flow, as do the copepods (Simenstad and Cordell 1985) and their fish predators (Haertel and

Osterberg 1967, Bottom and Jones 1990). Similarly, Hinton et al. (1990) found the highest fish densities in areas with the highest benthic invertebrate densities.

Diet--Despite the relatively high diversity and abundance of potential prey in the estuary, most fishes primarily consumed only a few taxa; prominently calanoid copepods and *Corophium salmonis* (Table 6) (Bottom et al. 1984). Other important but less-used prey included the gammarid amphipod *Eogammarus* sp., harpacticoid copepods, the mysid *Archaeomysis grebnitzkii*, *Daphnia*, and insects. The most common calanoid copepod consumers included American shad, various smelt species, Pacific herring, Pacific sand lance, snake prickleback, shiner perch, and sockeye and chum salmon (Table 6) (Bottom et al. 1984). The primary consumers of *Corophium salmonis* included sculpins (Cottidae), starry flounder, threespine stickleback (*Gasterosteus aculeatus*), chinook and coho salmon, and cutthroat trout (Bottom et al. 1984).

Larval fishes and spawning--Larval fishes are abundant in the Columbia River estuary (Misitano 1977, Jones and Bottom 1984). Larval osmerids, especially the longfin smelt, dominate, with lower densities of larval eulachon and the cottid prickly sculpin (Misitano 1977, Jones and Bottom 1984). Pacific herring larvae, adults of which are occasionally consumed by salmon in offshore areas, were notably absent from the ichthyofauna, yet they constitute over 80% of larval fish catches in other coastal estuaries in the region (Haertel and Osterberg 1967, Misitano 1977). Larval fishes were most abundant January through May; species diversity was greatest nearest the mouth, where salinities were highest, and decreased upstream (Misitano 1977, Jones and Bottom 1984).

Spawning in the estuary was observed for Pacific herring from April through July, and for an unidentified snailfish (Cyclopteridae) in February; longfin smelt are also thought to

Table 6.--Primary food items of common Columbia River estuary fishes during the three hydrologic seasons (fluctuating, high, and low river-flows). Numbers in parentheses are fish age in years. From Bottom et al. (1984).

Calanoid copepods	***Corophium salmonis***	**Other prey**
	Fluctuating river-flow (November-March)	
American shad (1)	longfin smelt (1)	***Eogammarus* sp.**
longfin smelt (1)	threespine stickleback	Pacific tomcod
	Pacific staghorn sculpin	
	starry flounder (1, 2)	**Harpacticoida**
	prickly sculpin	snake prickleback
	chinook salmon (0)	
	High river-flow (April-June)	
English sole (0)	Pacific staghorn sculpin	***Eogammarus* sp.**
snake prickleback	starry flounder (1 & 2)	shiner perch (1)
Pacific sand lance	prickly sculpin	
Pacific herring (0, 1)	chinook salmon (0, 1)	***Archaeomysis***
American shad (1)	coho salmon	butter sole
sockeye salmon	cutthroat trout	English sole (1)
longfin smelt		
surf smelt		**Insects**
chum salmon		cutthroat trout
		chum salmon
	Low river-flow (July-October)	
longfin smelt (0, 1)	starry flounder (0, 1)	***Daphnia* sp.**
Pacific tomcod	Pacific staghorn sculpin	Pacific herring (0)
shiner perch (0)	white sturgeon	chinook salmon (0)
whitebait smelt	prickly sculpin	surf smelt
Pacific herring (0, 1)		threespine stickleback
American shad (0, 1, 2)		
surf smelt		**Harpacticoida**
		shiner perch (1)
		northern anchovy

spawn in the estuary (Misitano 1977). Gravid females of Pacific staghorn sculpin (*Leptocottus armatus*) and prickly sculpin were also caught from February to April (Misitano 1977).

English sole (Haertel and Osterberg 1967) and butter sole (Misitano 1977) are thought to use the estuary as a nursery. English sole, however, appear to metamorphose before entering the estuary, since no larvae or postlarvae were found. In contrast, eulachon larvae appear to pass directly through the estuary on their way to the ocean soon after hatching (Misitano 1977).

Anadromous Fishes

Numerous species of anadromous fish use the Columbia River for some part of their life cycle. Anadromous fishes include the lampreys (*Lampetra* spp.); various smelt species; chinook, sockeye, coho, and chum salmon; steelhead and cutthroat trout; white and green sturgeon (*Acipenser medirostris*); and American shad. Historically, runs of anadromous fishes in the Columbia River were extremely large (Craig and Hacker 1940), and the Columbia River had the largest chinook salmon run in the world (Fulton 1968).

Seasonal abundance--The adults of anadromous species move through the estuary at all times of the year. For example, winter steelhead and eulachon adults are present in the estuary in winter; spring chinook salmon and summer steelhead adults are present in spring; American shad, summer chinook and sockeye salmon, and summer steelhead adults are present in summer; and fall chinook, coho, and chum salmon adults are present in the fall.

Few juvenile salmon are present in the estuary in January or February, but an initial small influx of subyearling and yearling chinook salmon occurs in March (Dawley et al. 1986). This early peak in abundance declines through mid-April, when increasing numbers of chinook and coho salmon and steelhead juveniles begin to enter the estuary. Most juvenile salmon reach peak abundances in May and early June, then rapidly decline in late June. Subyearling chinook salmon continue to be present in the estuary after the departure of other salmonid species, often with a second, larger peak in abundance in late July or early August; abundance declines rapidly after September (Dawley et al. 1986).

The length of time juvenile salmonids spend in the estuary is highly variable, but is generally quite brief compared to residence times in other Pacific Northwest estuaries, based on mark-and-recapture experiments with hatchery fish (Dawley et al. 1986). Estuarine residence times of wild juvenile salmon are not known. Juvenile coho, yearling chinook salmon and steelhead move through the estuary at approximately the same rate (average 3-36 km per day) as they migrate downstream (Dawley et al. 1986). Subyearling chinook salmon may slow to about 70% of their riverine migration rate when they enter the estuary, but generally they pass through the estuary within 6 days (Dawley et al. 1986). The only juvenile salmon to remain in the estuary for extended periods are subyearling chinook salmon originating in streams below Jones Beach (Rkm 75); they may remain in the estuary for several months (Dawley et al. 1986). For those fish that remain in the estuary for an extended time, growth is quite rapid (Dawley et al. 1986).

Most migrational movement for juvenile salmonids occurs during daylight hours, with little movement at night (Dawley et al. 1986). The rate of seaward movement in the estuary for both juvenile coho and chinook salmon is influenced by distance from the spawning ground and river flow: the farther upstream they hatched, or the higher the river flow, the faster they move through the estuary (Dawley et al. 1986). During the migration period, larger fish also move through the estuary first, followed by smaller individuals (Dawley et al. 1986).

Distribution--Subyearling chinook and coho salmon are usually concentrated in shallow nearshore habitats, while yearling chinook and coho salmon and steelhead are most abundant in channel areas. The exceptions to this include early yearling chinook salmon, which are often abundant in shallow nearshore habitat, and large subyearling chinook salmon, which may be abundant in channel areas (Dawley et al. 1986).

Fish age--The age of anadromous fish passing both downstream and upstream through the estuary also varies between and within species. Some anadromous fishes, such as American shad, chum and chinook salmon, and eulachon are less than a year old when they move downstream through the estuary (ODFW and WDF 1991). Coho and chinook salmon generally spend one year in fresh water, sockeye salmon spend one or two years in fresh water, while steelhead may go to sea as 2- to 3-year-olds (ODFW and WDF 1991). Most anadromous fishes (sockeye, and chum salmon, summer steelhead, eulachon, and American shad) are 3-5 years old when they return to their natal streams, the upper mainstem, or hatcheries; chinook salmon are 2-6 years old; coho salmon are 2-3 years old; and winter steelhead are 4-5 years old (ODFW and WDF 1991).

Diet--Diets of the several species of juvenile salmonids in the estuary are similar to and overlap considerably with those of other fishes associated with them (McCabe et al. 1983, Bottom et al. 1984). In spring, *Corophium* spp. are the primary prey of subyearling and yearling chinook salmon, juvenile coho salmon, and juvenile steelhead, with lesser numbers of adult and larval insects consumed. *Daphnia* are the primary prey of subyearling chinook salmon in the summer (McCabe et al. 1983, Bottom et al. 1984). In spring, other fishes caught along with juvenile salmon such as peamouth, threespine stickleback, Pacific staghorn sculpin, and starry flounder, also consume *Corophium*, while in summer, *Daphnia* are also consumed by American shad, Pacific herring, longfin smelt, surf smelt, and shiner perch. Consequently, diet overlaps can be quite high between sympatric salmonid and non-salmonid fishes (McCabe et al. 1983). For example, subyearling and yearling chinook salmon have over 60% diet similarity with juvenile coho salmon, juvenile steelhead, threespine stickleback, American shad, and each other; both juvenile coho salmon and steelhead share over 60% diet similarity with subyearling and yearling chinook salmon, American shad, and each other (McCabe et al. 1983).

Historical Catches and Fish Runs

Historical catch data are available for commercially important fishes in the Columbia River, primarily salmon and sturgeon (Craig and Hacker 1940). Commercial catch record-keeping began in 1866 for chinook salmon, and from 1866 to 1890 for other species. Record keeping of sport catches was not initiated until 1969 (ODFW and WDF 1991), and estimates of minimum run sizes, calculated from commercial and sport catches, hatchery returns, tributary runs, and dam counts, were not available until 1938 for most species, and 1960 for coho salmon (Fish Commission of Oregon [FCO] and WDF 1971). Fishes in the Columbia River are currently harvested in limited commercial, treaty, and recreational fisheries. Treaty catches have been increasing to approximately 54% of commercial catches, while the recreational fishery has increased dramatically since 1984 (ODFW and WDF 1991). Between 1960 and 1990, annual salmon and steelhead catches reached a high of 985,300 fish in 1988, and a low of 85,500 fish in 1983, while 1990 catches were the lowest since 1983, at 256,700 fish (ODFW and WDF 1991).

Current runs of most native anadromous fishes are much lower than historical runs due to dam construction and operation, inaccessible spawning areas, over fishing, and habitat degradation and destruction (Fulton 1970, Chapman 1986, ODFW and WDF 1991). For example, predevelopment runs of all salmonids in the Columbia River are estimated to be between 7.5 and 16 million adults (Chapman 1986, Northwest Power Planning Council [NPPC] 1986) compared to 2.5 million adults at present (Chapman 1986), and several runs are listed as threatened or endangered (NMFS 1992).

Chinook salmon--Chinook salmon was the most heavily harvested fish in the Columbia River. The peak catch occurred in 1883 when 19 million kg of fish were landed, producing 629,000 cases of canned chinook salmon worth $3.2 million dollars (in 1883) (Craig and Hacker 1940, Chapman 1986). Commercial chinook salmon catches have declined ever since, to an average annual commercial catch of 2.7 million kg of fish at present, although recreational catches have increased in recent years (Fig. 8) (Pruter 1972, ODFW and WDF 1991). Prior to 1928, the chinook salmon catch consisted primarily of spring and summer runs, which predominately spawned in the Salmon River (Idaho) and upper Columbia River above Grand Coulee Dam, respectively (Fulton 1968, ODFW and WDF 1991). After 1928, the majority of the catch was composed of fall runs (Fulton 1968), which spawn in the lower and middle mainstem, and remains so at present (ODFW and WDF 1991). Chinook salmon runs since 1938 have displayed no consistent long-term decrease or increase in abundance (Fig. 9) (FCO and WDF 1971, ODFW and WDF 1991). This relative stability in the population is due in part to hatchery production (Pruter 1972, ODFW and WDF 1991), with 92-160 million chinook salmon produced annually in the Columbia River basin since 1988 (Fig. 10) (Footnote 4) (Wahle and Smith 1979, Wahle and Pearson 1987). However,

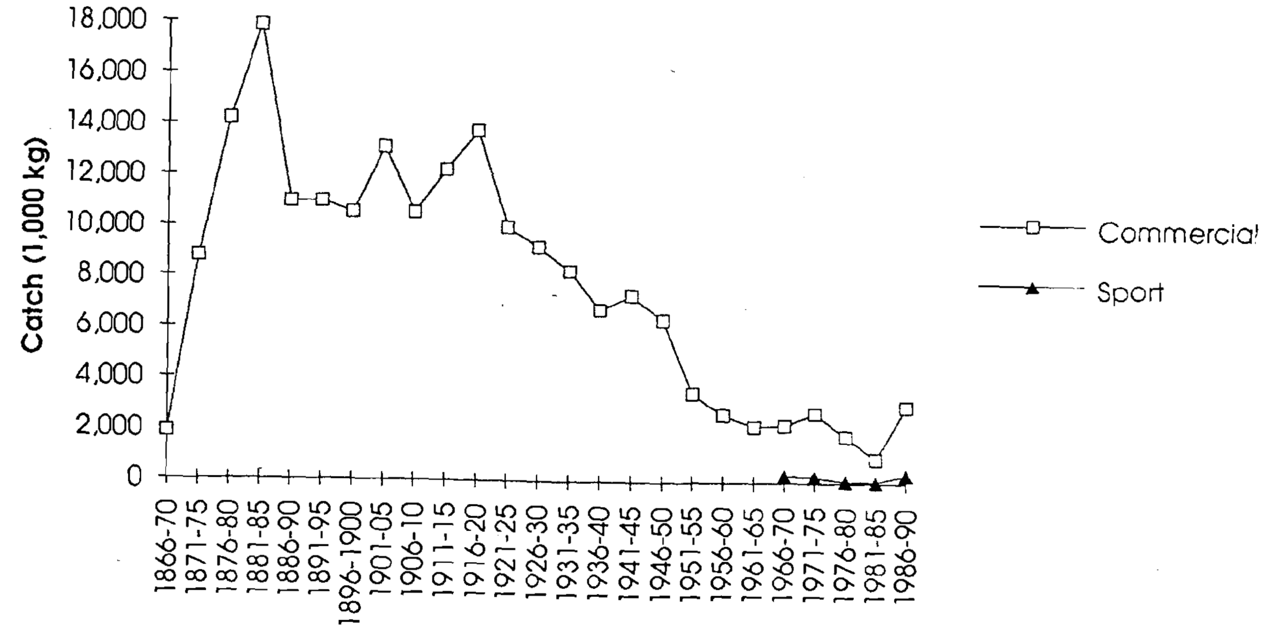

Figure 8.--Annual commercial and recreational catches of chinook salmon in the Columbia River, averaged over five year periods, from 1866 to 1990 (Craig and Hacker 1940, Pruter 1972, ODFW and WDF 1991).

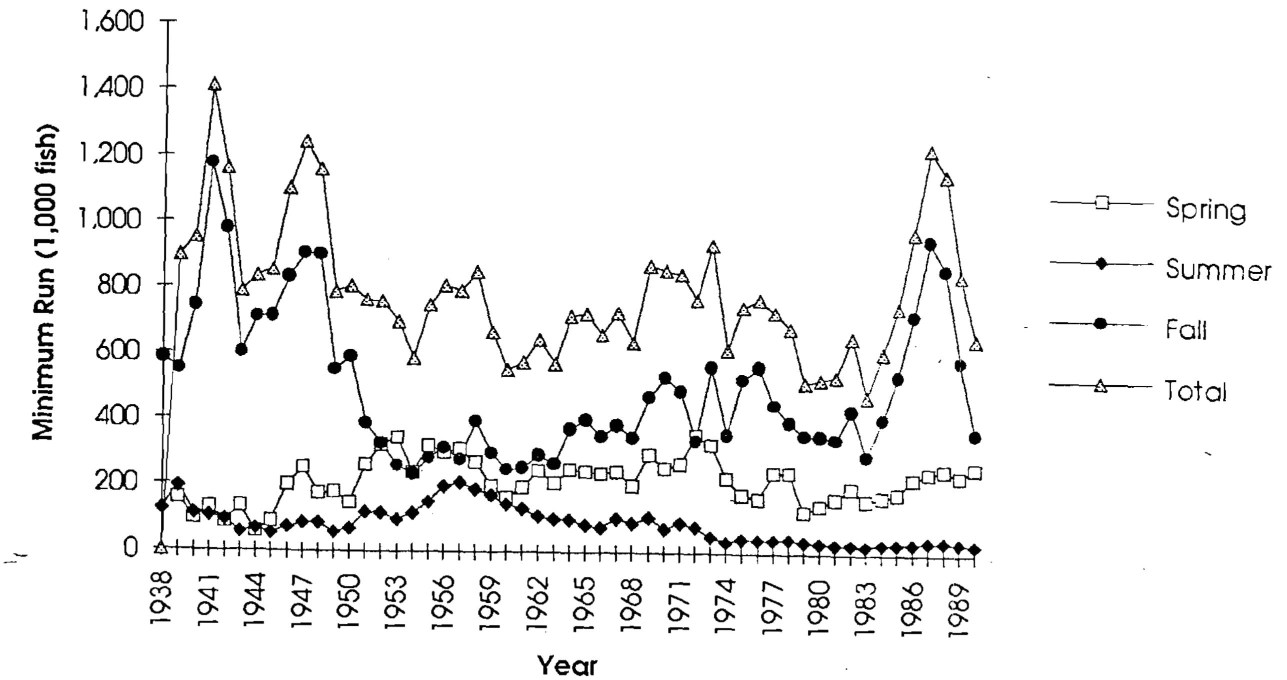

Figure 9.--Estimated minimum run sizes for spring, summer, fall, and total chinook salmon in the Columbia River from 1938 to 1990. Minimum run sizes were calculated from total landings below Bonneville Dam, hatchery returns, tributary escapement, and Bonneville Dam fish counts (FCO and WDF 1971, ODFW and WDF 1991).

the number of spring, summer, and fall chinook salmon do not uniformly reflect the relatively constant level of chinook salmon runs. While the fall run of chinook salmon has been increasing since the early 1960s, the spring run has remained fairly stable, and the summer run has been decreasing (Fig. 9) (FCO and WDF 1971, ODFW and WDF 1991). During this period, hatchery releases of all three chinook salmon races have been increasing, with 60-117 million fall, 24-42 million spring and 3-5 million summer chinook salmon released annually since 1988 (Fig. 10) (Footnote 4) (Wahle and Smith 1979, Wahle and Pearson 1987). The Snake River runs of all three stocks of chinook salmon are listed as threatened species (NMFS 1992).

Coho salmon--Coho salmon was most heavily fished between 1921 and 1935, when 2.8 million kg of fish were caught annually (Fig. 11) (Craig and Hacker 1940, Pruter 1972, ODFW and WDF 1991). Current annual commercial catch is 1.3 million kg, with fish taken both offshore and in the river, while sport catch has dramatically increased since 1980 to approximately 0.25 million kg annually in 1986-1990 (ODFW and WDF 1991). Hatchery production of coho salmon has been gradually increasing since 1960, with 43.9 million coho salmon smolts released in 1992 (Fig. 10) (Footnote 4) (Wahle and Smith 1979, Wahle and Pearson 1987). This has led to a fairly constant run since the early 1960s (Fig. 12) (OFC and WDF 1971, ODFW and WDF 1991).

Sockeye and chum salmon and steelhead--Annual commercial catches of steelhead (Fig. 13) and sockeye salmon (Fig. 14) peaked in the 1890s at about 1.1 and 1.7 million kg, respectively, while commercial catches of chum salmon peaked in the late 1920s at about 1.8 million kg (Fig. 14) (Craig and Hacker 1940, Pruter 1972, ODFW and WDF 1991). All annual commercial catches have since declined. With a few exceptions, commercial fisheries

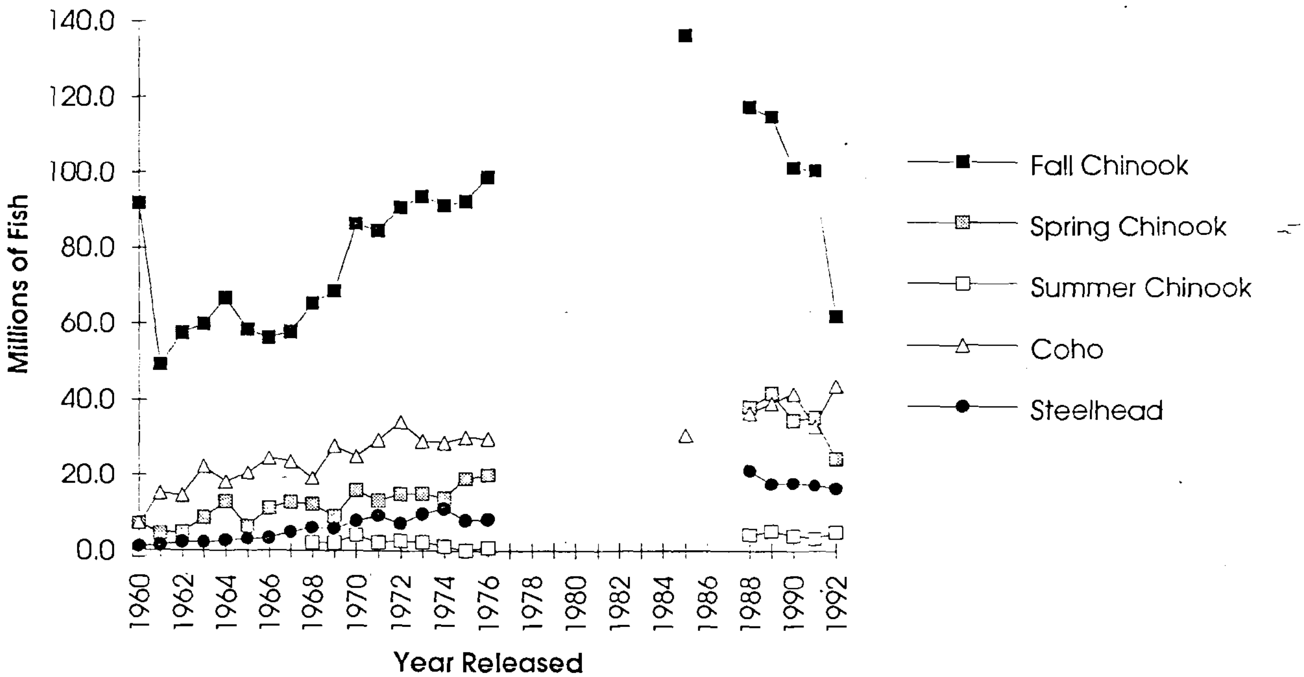

Figure 10.--Columbia River basin-wide hatchery production of fall, spring and summer chinook salmon, coho salmon, and steelhead, 1960-1992. Data from 1977-1987 have not been compiled at this time. Data from Wahle and Smith (1979), Wahle and Pearson (1987). (S. Allen, unpubl. data. Pacific States Marine Fisheries Commission, 45 S.E. 82nd Drive, Suite 100, Gladstone, OR 97027-2522.)

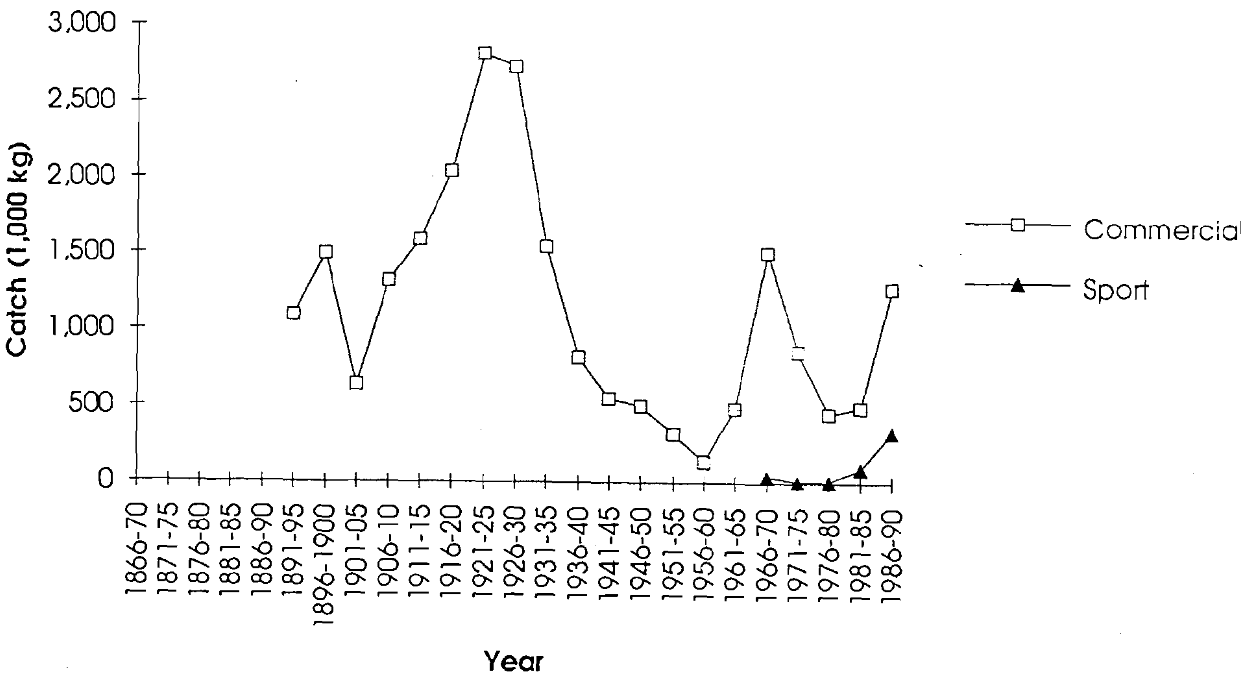

Figure 11.--Annual commercial and recreational catches of coho salmon in the Columbia River, averaged over five year periods, from 1866 to 1990 (Craig and Hacker 1940, Pruter 1972, ODFW and WDF 1991).

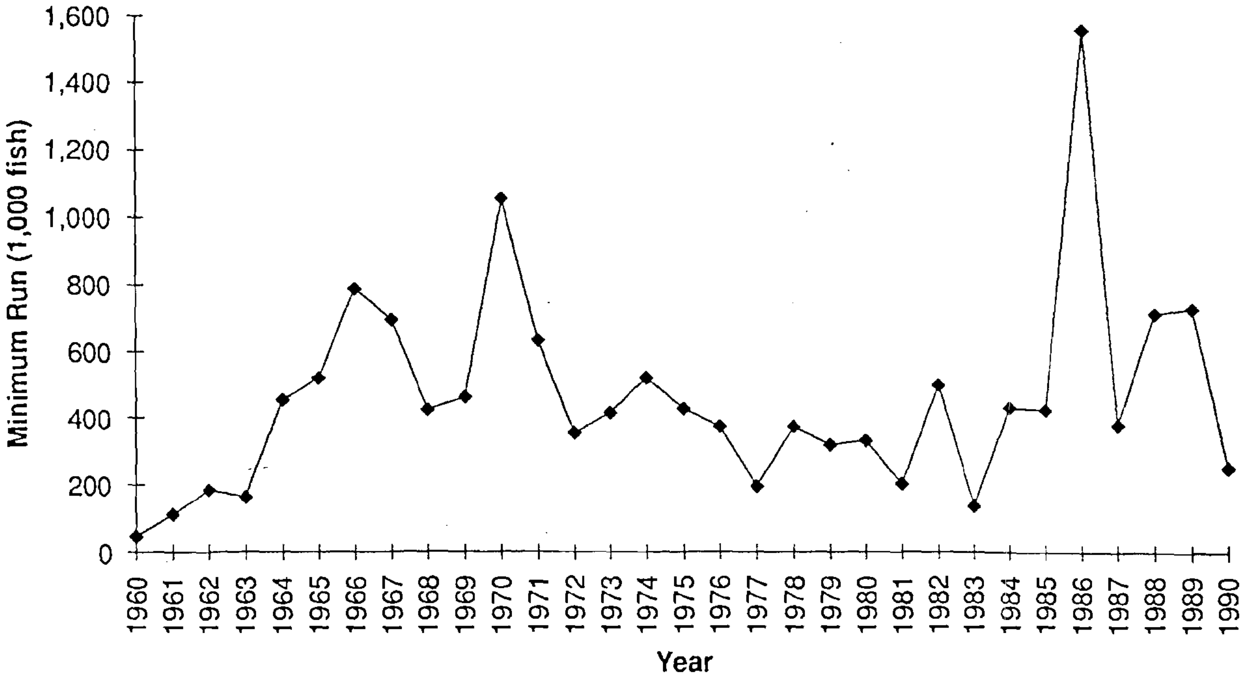

Figure 12.--Estimated minimum run sizes for coho salmon in the Columbia River from 1960 to 1990. Minimum run sizes were calculated from total landings below Bonneville Dam, hatchery returns, tributary escapement, and Bonneville Dam fish counts (ODFW and WDF 1991).

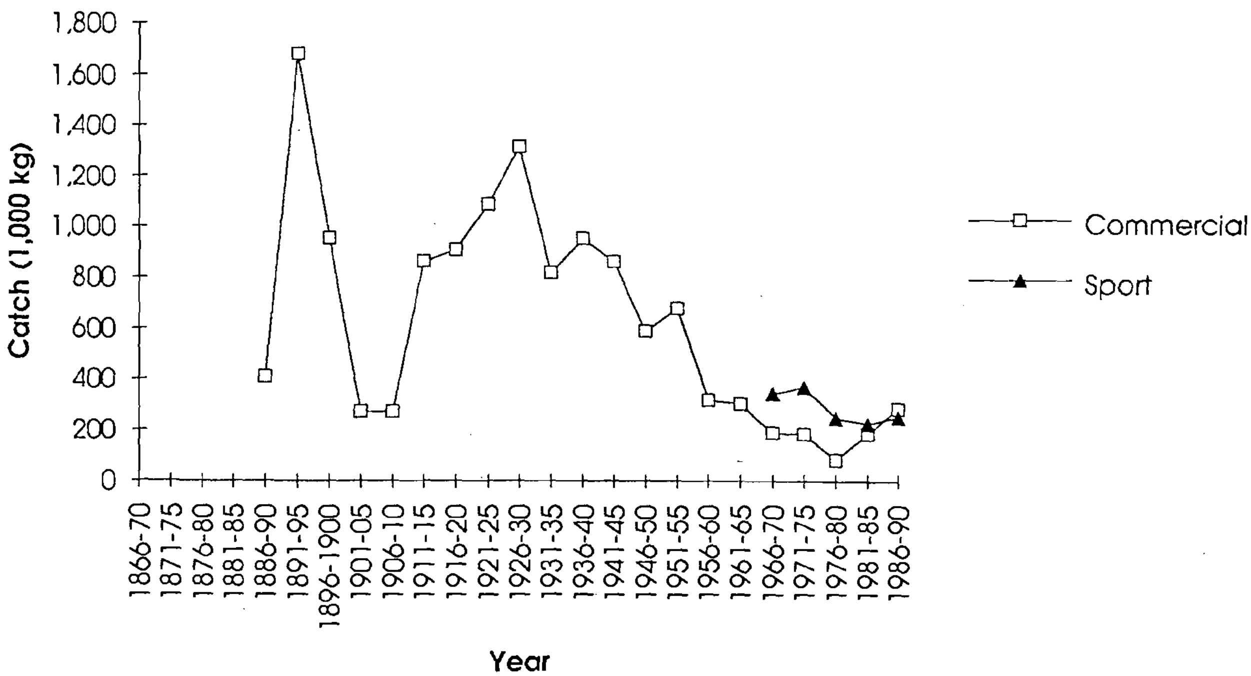

Figure 13.--Annual commercial and recreational catches of steelhead in the Columbia River, averaged over five year periods, from 1866 to 1990 (Craig and Hacker 1940, Pruter 1972, ODFW and WDF 1991).

have been prohibited from harvesting sockeye salmon and steelhead since the mid-1970s, and chum salmon commercial catches are less than 1,000 fish annually (ODFW and WDF 1991). Sportfishing for steelhead in the lower river is quite extensive, and 145,000 kg of summer steelhead were taken in 1990 (Fig. 13) (ODFW and WDF 1991). The estimated run size of steelhead has been slowly increasing since 1938, while Columbia River sockeye salmon stocks were much higher in the 1950s than they are at present (Fig. 15) (OFC and WDF 1971, ODFW and WDF 1991), and Snake River sockeye salmon runs are listed as endangered (NMFS 1992).

Hatchery production of steelhead has been steadily increasing since the 1960s, with 16.5 million smolts released in 1992 (Fig. 10), the majority of which presently occur in Idaho (Footnote 4). In contrast, hatchery production of sockeye salmon has historically been very low (Wahle and Smith 1979), although releases in 1991 and 1992 were quite high compared to previous years, with 0.3 and 0.2 million smolts released, respectively (Footnote 4).

White sturgeon--Commercial catches of Columbia River white sturgeon were enormous in the 1880s and 1890s (Fig. 16), with 2.5 million kg caught in 1892 (Craig and Hacker 1940, Pruter 1972). Sturgeon was considered a pest species because of its abundance, size, initially limited market value, and tendency to destroy salmon gill nets. Consequently, sturgeon were often left to rot, and many sturgeon under 23 kg were systematically destroyed (Craig and Hacker 1940). White sturgeon was nearly exterminated from the river by 1895, but has since increased in the lower Columbia River and currently 255,000 kg are commercially harvested annually (ODFW and WDF 1991). Current sport fishing harvests far exceed commercial catches (Fig. 16) (ODFW and WDF 1991). White sturgeon populations

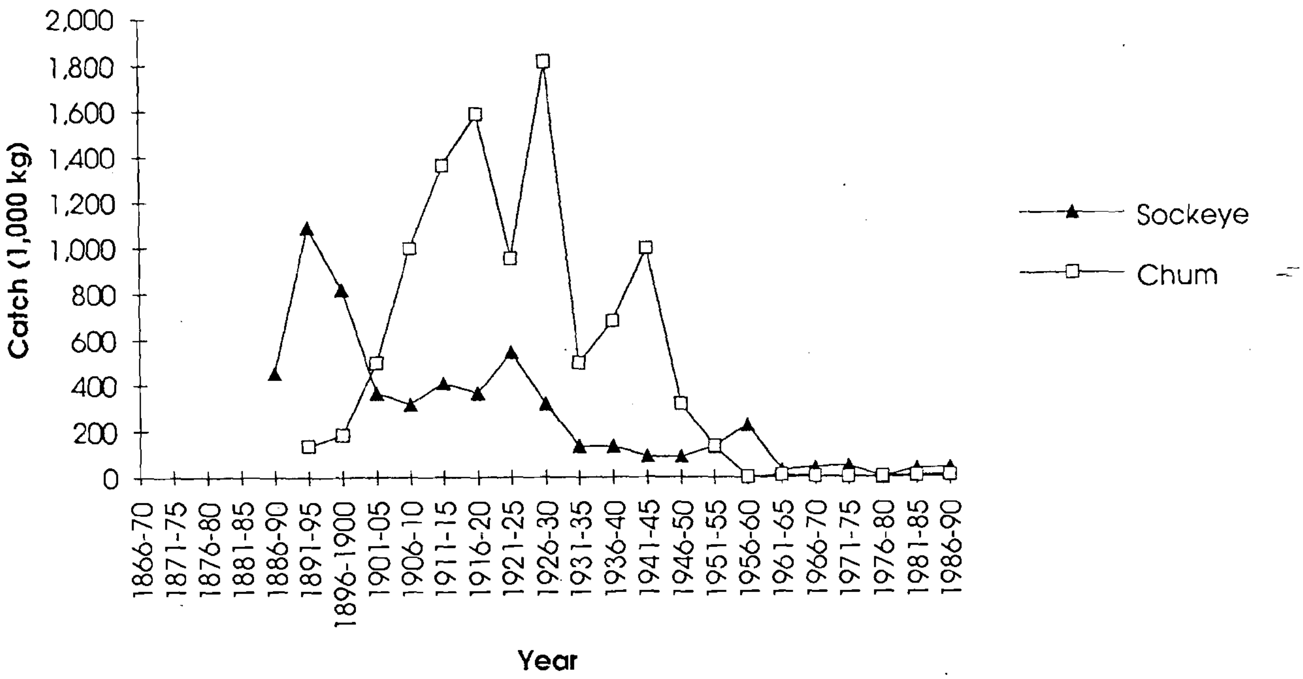

Figure 14.--Annual commercial catches of sockeye and chum salmon in the Columbia River, averaged over five year periods, from 1866 to 1990 (Craig and Hacker 1940, Pruter 1972, ODFW and WDF 1991).

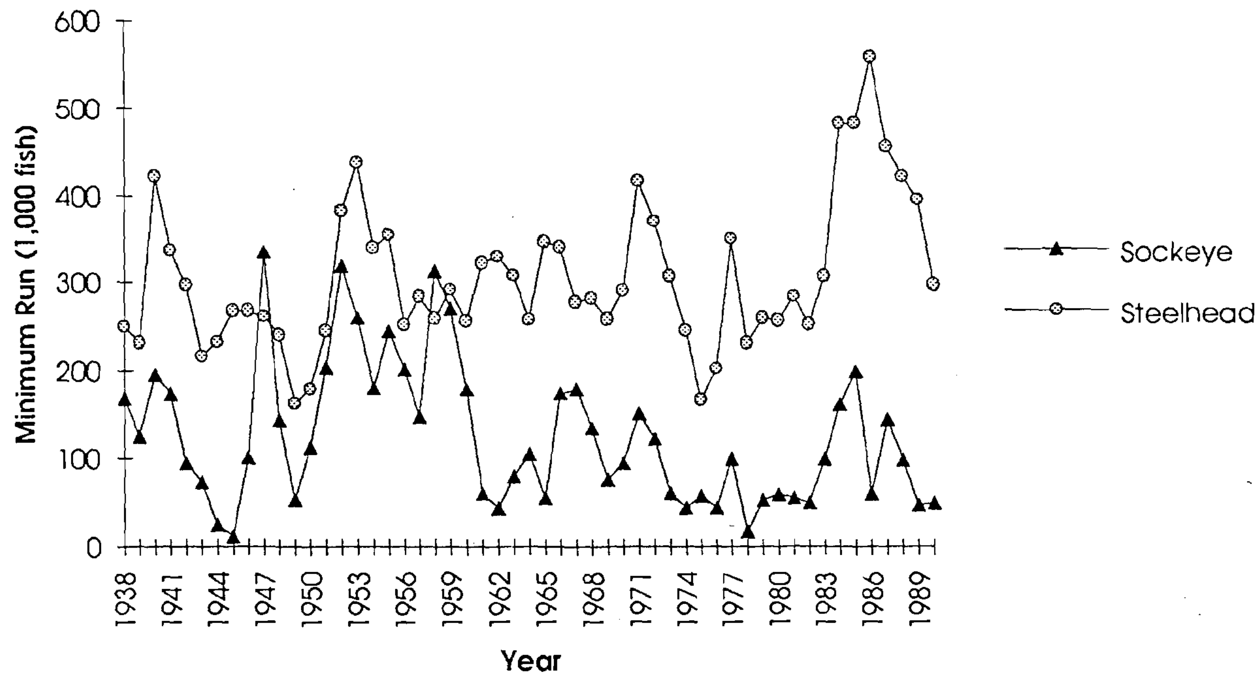

Figure 15.--Estimated minimum run sizes for sockeye salmon and steelhead in the Columbia River from 1938 to 1990. Minimum run sizes were calculated from total landings below Bonneville Dam, hatchery returns, tributary escapement, and Bonneville Dam fish counts (FCO and WDF 1971, ODFW and WDF 1991).

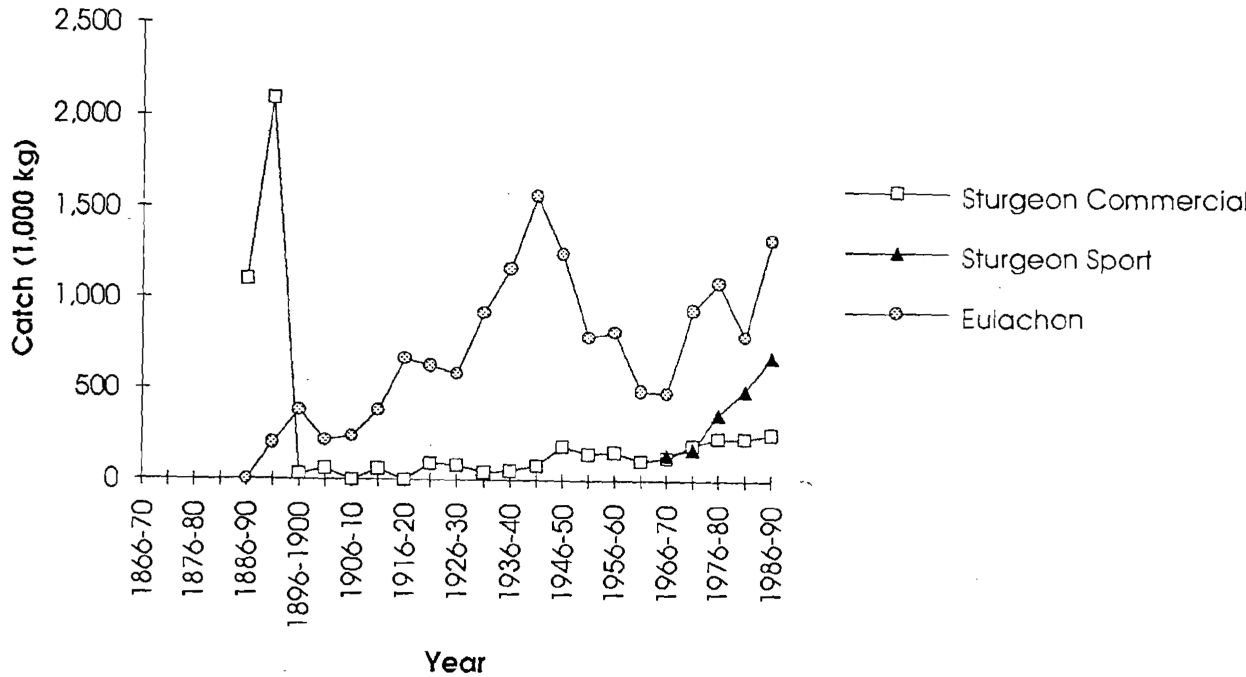

Figure 16.--Annual commercial and sport catches of white sturgeon and commercial catches of eulachon in the Columbia River, averaged over five year periods, from 1866 to 1990. Data for recreational catches of eulachon are not available (Craig and Hacker 1940, Pruter 1972, ODFW and WDF 1991).

are now relatively stable, with numerous fishing restrictions in effect to further enhance the runs (ODFW and WDF 1991).

Eulachon--Eulachon is one of the few commercially important, native, anadromous fishes in the Columbia River that has not suffered a massive decline in abundance, and it is the second largest fishery in the Columbia River after chinook salmon (Smith and Saalfeld 1955, Pruter 1972). Commercial catches of eulachon peaked in the 1940s and are currently high, with 1.3 million kg caught annually (Fig. 16). The recreational eulachon catch is not recorded, but is estimated to be equivalent to, if not exceed, the commercial catch (ODFW and WDF 1991).

American shad--American shad, a species introduced from eastern North America, has produced a very successful anadromous fish run in the Columbia River. American shad was introduced into the Columbia River in 1885-86; the first commercial catch was recorded in 1889, and by 1990 an estimated 4 million adult American shad entered the Columbia River (Fig. 17) (Craig and Hacker 1940, ODFW and WDF 1991). Peak catches of 450,000-680,000 kg of shad were recorded in 1926-30 and 1946-47 (Pruter 1972). The present commercial fishery catch is about 75,000 kg annually and has been increasing, but bycatch of summer chinook salmon and a limited market have kept fisheries for American shad extremely low compared to the substantial increase in its population (Fig. 17) (ODFW and WDF 1991).

Anadromous fishes in the estuary--In general, anadromous fishes in the Columbia River have been severely impacted since European settlement of the Northwest. Despite extensive hatchery production and fishing regulations intended to protect runs, the run sizes of most anadromous fishes have never returned to pre-commercial fishing levels. The cause of

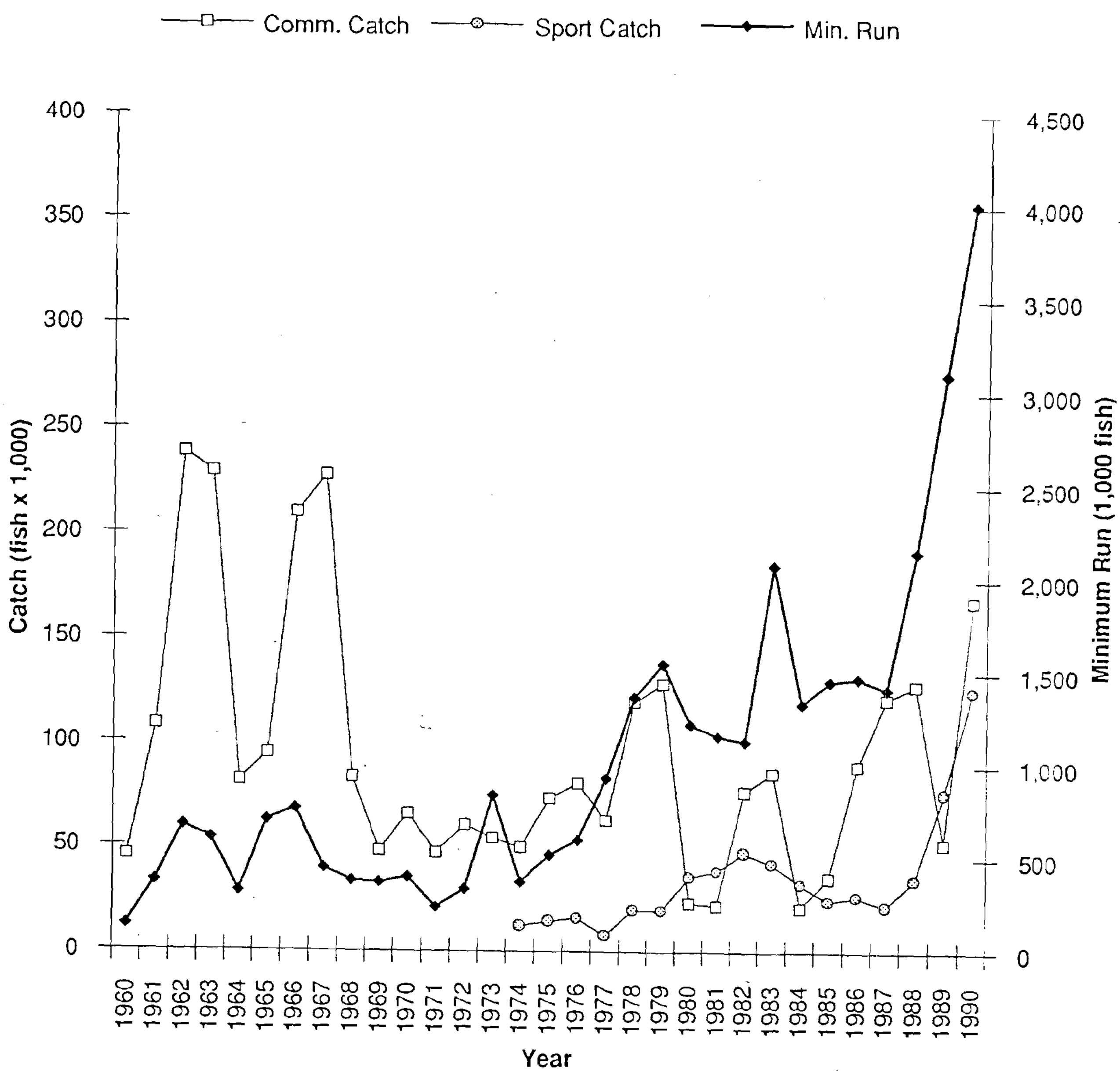

Figure 17.--Annual commercial and sport catches, and estimated minimum run sizes for American shad in the Columbia River, averaged over five year periods, from 1866 to 1990. Minimum run sizes were calculated from total landings below Bonneville Dam, hatchery returns, tributary escapement, and Bonneville Dam fish counts (Craig and Hacker 1940, Pruter 1972, ODFW and WDF 1991).

this decline can be attributed in part to overfishing, habitat degradation, inaccessible spawning areas, delayed migrations, direct mortality from dam passage, and pollution (Fulton 1970, Northcote and Larkin 1989). Runs that originate lower in the basin have generally fared better than those that originate higher in the basin, as evidenced by endangered and threatened listings of several Snake River salmonid stocks (NMFS 1992). The two species that are exceptions to this decline are American shad and eulachon. Eulachon spawn primarily in the lower tributaries of the Columbia River and consequently has not been extensively impacted by inaccessible spawning grounds. The exotic American shad does not appear to have been impacted by many of the changes that have reduced native anadromous fish populations, because the altered Columbia River ecosystem apparently provides it with high-quality habitat. A thorough understanding of environmental factors leading to the recent explosive growth in the American shad population is lacking.

Food Web Structure in the Columbia River Estuary

As in most estuaries, the food web in the Columbia River estuary is based on both primary production and detritus (Simenstad et al. 1990), and is driven, in large part, by the physical dynamics of the estuary. The physical processes of mixing and stratification, sediment accretion and erosion, and salinity intrusion determine the distribution and amount of primary production and detritus (Simenstad et al. 1990). Many portions of the food web converge in the ETM, the null zone in the central region of the estuary (Simenstad et al. 1990). A simplified representation of the major linkages between trophic groups in the Columbia River estuary is presented in Figure 18.

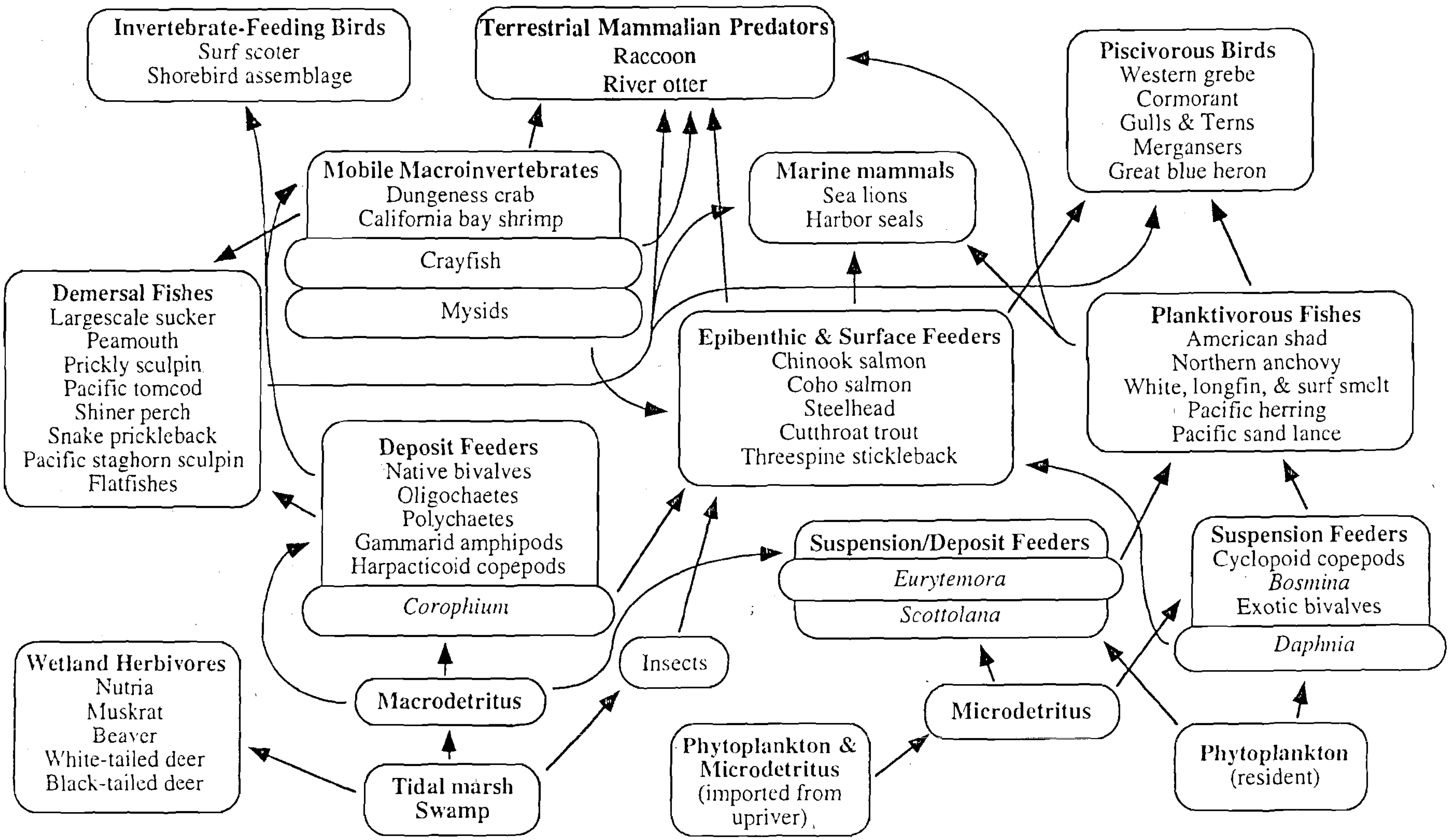

Figure 18.--A simplified representation of the major linkages between trophic group in the Columbia River estuary. Arrow indicates direction of energy flow. The relative amount of energy transferred through the various pathways should not be implied from this diagram. Derived from Sherwood et al. (1990).

No investigations of the estuarine food web were undertaken prior to dam construction. Consequently, any changes which have occurred in response to the operation of the hydropower system are unknown. However, some potential effects of impoundments on the structure of the food web have been suggested based on the observed physical and biological changes associated with impoundments (Sherwood et al. 1990, Simenstad et al. 1990). In this section, the food web structure in the estuary and the ways in which it may have changed in response to dams are reviewed.

Primary Production and Detritus

The principal primary producers in the estuary are phytoplankton, benthic diatoms, and emergent vegetation. However, contributions to the food web from these sources are minor compared to the input of phytoplankton and microdetritus from upstream. Large quantities ofphytoplankton are transported downstream into the estuary. As the cells contact the slightly higher salinity, they lyse, and the plankton becomes suspended microdetritus. Plant litter produced by emergent marshes and swamps is transformed into macrodetritus by biological degradation. The estimated amount of detritus imported into the estuary from upriver is about 50 times (146,000 t C per year) that produced by emergent vegetation in the estuary (3,600 t C per year) (Small et al. 1990).

Secondary Consumers and Detritivores

Primary producers are consumed directly by zooplankton and wetland herbivores. Zooplankton planktivores such as *Eurytemora*, *Cyclops*, *Bosmina*, and *Daphnia*, consume approximately 1% of the total phytoplankton in the estuary per day, although the bulk (85%) of their diet is microdetritus. The wetland herbivores, primarily muskrat (*Ondatra zibethica*), nutria (*Myocaster coypus*), beaver (*Castor canadensis*), and white- (*Odocoileus virginianus* *leucurus*) and black-tailed (*Odocoileus hemionus columbianus*) deer, feed on marsh and swamp vegetation, with the largest amount consumed by the introduced nutria (Simenstad et al. 1990).

Both deposit- and suspension-feeding detritivores are present in the Columbia River estuary. Deposit feeders generally consume macrodetritus and deposited microdetritus, while suspension feeders consume suspended microdetritus (Fig. 18) (Holton et al. 1984). Most infauna (native bivalves, oligochaetes, polychaetes, and gammarid amphipods including *Corophium salmonis*) and some epibenthic crustaceans (the harpacticoid copepods Ectinosoma sp., *Microarthridion littorale, Tachidius triangularis, Attheyella* sp.) are deposit-feeding detritivores. Suspension-feeding detritivores are represented by the two exotic clams (*Corbicula fluminea* and *Mya arenaria*); cyclopoid copepods such as *Corycaeus anglicus and Cyclops* spp.; and cladocerans including *Daphnia* spp. and *Bosmina longirostris* (Simenstad et al. 1990). Two epibenthic copepods, the calanoid *Eurytemora affinis* and the harpacticoid *Scottolana canadensis* are considered to be capable of both suspension and deposit feeding (Simenstad et al. 1990).

On an estuarine-wide basis, benthic infauna consume approximately 11 times more detritus (1,943 t C per year) than epibenthic organisms (167 t C per year). In contrast, suspension-feeding benthic infauna consume about 2% (282 t C per year) of the suspended detritus consumed by pelagic and epibenthic zooplankton (12,585 t C per year) (Simenstad et al. 1990). Total consumption by deposit feeders (2,110 t C per year) is about one sixth that of suspension feeders (12,867 t C per year) (Simenstad et al. 1990).

Predators

Predators in the Columbia River estuary include a wide variety of animals such as benthic infauna; zooplankton; mobile macroinvertebrates; fish; avifauna; and terrestrial, aquatic, and marine mammals (Fig. 18). In the upper estuary, the errant polychaetes and amphipods are the primary infaunal predators. The mobile predatory macroinvertebrates include Dungeness crab, California bay shrimp, and mysids (*Archaeomysis grebnitzkii* and *Neomysis mercedis)*. Dungeness crabs have a spatially and temporally limited distribution, while California bay shrimp and mysids are more widely distributed in the central estuary and prey extensively on epibenthic crustaceans such as the harpacticoid copepod *Scottolana canadensis*. Predatory benthic infauna consume two to three times the carbon equivalents (306 t C per year) of mobile macroinvertebrates (110-147 t C per year) (Simenstad et al. 1990).

Fishes--Fishes exhibit a wide range of feeding types, but can generally be classified as planktonic, demersal, and epibenthic and water-surface feeding (Fig. 18). The zooplanktivores include American shad, which often inhabits the brackish-water and freshwater portions of the estuary, and northern anchovy, whitebait, longfin and surf (*Hypomesus pretiosus*) smelt, Pacific herring, and Pacific sand lance, which are most common in the marine and brackish water zones of the estuary. Planktivorous fishes are generally pelagic and consume primarily harpacticoid and calanoid copepods, with seasonal contributions by *Daphnia* and *Corophium salmonis*. This fish group is the main consumer of the abundant *Eurytemora.*

The demersal group of predators can also be divided into species which usually inhabit the upper (brackish and freshwater) or lower (marine and brackish-water) portions of the estuary. Largescale sucker, peamouth (*Mylocheilus caurinus*), and prickly sculpins are demersal predators, generally found in the brackish and freshwater zones of the estuary, while Pacific tomcod, shiner perch, snake prickleback, Pacific staghorn sculpin and pleuronectids (speckled sanddab; butter, sand (*Psettichthys melanostictus*) and English sole; and starry flounder) are generally found in the marine and brackish water regions of the estuary with some exceptions (Bottom et al. 1984). Demersal predators consume a wide variety of prey; primarily *Corophium salmonis*, other gammarid amphipods, mysids, and copepods.

The epibenthic and surface-feeding group consists of most juvenile salmonids (chinook and coho salmon, steelhead, and cutthroat trout) and, seasonally, threespine sticklebacks. This group primarily consumes *Corophium salmonis* in addition to insects, *Daphnia*, and fish. The total consumption of prey by all fishes (422 t C per year) is approximately equivalent to the consumption by benthic infauna and motile macroinvertebrates combined (416-453 t C per year) (Simenstad et al. 1990).

Nonfish vertebrate predators--Nonfish vertebrate predators in the Columbia River estuary include fish- and invertebrate-eating birds and terrestrial, aquatic, and marine mammals (Fig. 18). Piscivorous birds include the western grebe (*Aechmophorus occidentalis*); the double-breasted cormorant; various gulls and terns; the common crow (*Corvus brachyrhynchos*); the common merganser (*Mergus merganser*); and the great blue heron (*Ardea herodias*). The surf scoter (*Melanitta perspicillata*) and a multi-species shorebird assemblage including sanderling (*Calidris alba*), dunlin (*C. alpina*), and western sandpiper (*C. mauri*) prey on invertebrates (Hazel 1984).

The terrestrial raccoon (*Procyon lotor*) consumes crayfish (*Pacificus trowbridgii*), clams, and fish, while the aquatic river otter (*Lutra canadensis*) feeds primarily on fish and crayfish (Dunn et al. 1984). Marine sea lions (northern [*Eumetopias jubatus*] and California) and harbor seals consume a variety of fish, such as northern anchovy, eulachon, Pacific staghorn sculpin, longfin smelt, lamprey, Pacific tomcod, Pacific herring, sand sole, and rockfish (*Sebastes* spp.) (Jeffries et al. 1984). They also consume adult salmon caught in gillnets, much to the displeasure of fishermen. Of the nonfish vertebrate consumers, avifauna consume the least carbon equivalents (1 t C per year), terrestrial and aquatic mammals consume about 10 times that of the avifauna (10-11 t C per year), while sea lions and especially harbor seals consume (314 t C per year) about 30 times that of other terrestrial and aquatic mammals (Simenstad et al. 1990).

Comparison Between Pre-1870s and Present-Day Food Webs

In the present-day estuary, as described above, microdetritus is imported from upriver in the form of phytoplankton, which is then utilized by zooplankton, such as *Eurytemora*. Zooplankton are consumed by pelagic fish, which are in turn consumed by avifauna and mammals; the remaining detritus is exported from the estuary into the ocean. A second pathway consists of macrodetritus derived from emergent vegetation, which is consumed by benthic infaunal deposit feeders, including *Corophium salmonis*. These bottom feeders are then consumed by demersal and epibenthic and surface-feeding fish, including juvenile salmonids, and avifaunal predators.

This present-day food web is thought to differ from the unimpacted (pre-1870s) estuarine food web in several ways (Table 7). First, the unimpacted estuary is thought to have had much less importation of phytoplankton and microdetritus from upstream. Estimates indicate that the unimpacted estuary received inputs of 9,000 million t C per year phytoplankton and 73.000 million t C per year detritus from upstream, compared to the

Table 7. Comparison of estuarine characteristics at present and pre-1870s. From Sherwood et al. (1990).

Character	Present	Pre-1870s
Inputs from upriver (million t C $year^{-1}$)		
Phytoplankton	61,400	9,000
Detritus	147,000	73,000
Zooplankton	102	25
Area covered by tidal swamps and marshes (ha)		
Tidal marshes	3,723	6,548
Tidal swamps	2,813	12,149
Carbon equivalents of tidal marshes (million t C $year^{-1}$)	11,300	62,600
Macrodetritus production from marshes and swamps (million t C $year^{-1}$)	3,600	20,000
Estimated populations of primary consumers pre-1870s		
Wetland herbivores	12-138 times present levels	
Infaunal detritivores	12 times present levels	
Relative importance of foodweb pathways		
Pelagic microdetritus-based pathway	High	Low
Benthic macrodetritus-based pathway	Low	High
Detritus export to the ocean (million t C $year^{-1}$)	159,200	80,000

present-day estuary, which receives nearly seven times as much phytoplankton (61,400 million t C per year) and twice the input of detritus (147,000 million t C per year) from upriver (Sherwood et al. 1990). Secondly, the unimpacted estuary had a much greater array of emergent marshes and swamps that, along with tidal flat microalgal production, have been lost to diking, filling, and bathymetry changes. The production of macrodetritus from the emergent marshes, swamps, and microalgae in the unimpacted estuary was estimated to be 20,000 t C per year, compared to 3,600 t C per year at present (Table 7) (Sherwood et al. 1990).

These changes are thought to have had a large impact on estuarine food webs, although estimates of actual values have not been calculated. Benthic infauna consume a considerable portion of the macrodetritus produced in the estuary (Sherwood et al. 1990). Sherwood et al. (1990) estimated that the unimpacted estuary could support 12 times the benthic infauna of the present-day estuary because of this difference in macrodetritus production. Since benthic infauna, especially *Corophium salmonis*, are consumed by both demersal fish predators, such as juvenile salmonids, flatfishes and sculpins, and avifauna, such as wading shorebirds and surf scoters, the reduced infaunal abundances between the unimpacted and present-day estuary may have impacted higher trophic levels as well (Sherwood et al. 1990), assuming benthic infauna are detritus-limited.

Similarly, the increased input of phytoplankton and microdetritus into the present-day estuary is thought to have caused a shift in food web pathways. Microdetritus-consuming calanoid (*Eurytemora affinis*) and harpacticoid (*Scottolana canadensis*) copepods in the ETM are thought to reach extremely high densities in response to the concentration of imported microdetritus in that area (Sherwood et al. 1990). These copepods are consumed by pelagic fishes, such as Pacific herring, American shad, and various smelts, and some motile macroinvertebrates. Consequently, it is suggested that the estuarine food web has shifted from a system largely supported by macrodetritus with demersal fish predators, to a system largely supported by microdetritus with pelagic predators (Sherwood et al. 1990). Although American shad may have benefitted from the expansion of its pelagic zooplankton-based portion of the food web, other commercially important fishes, such as salmonids, may have experienced a decrease in food resources.

Exotic Species in the Columbia River Estuary

Sixteen species of exotic fishes and four species of exotic invertebrates have been recorded from the Columbia River estuary (Table 8) (Ingram 1948, Stober et al. 1979, Wydoski and Whitney 1979, Lee et al. 1980, Bernard 1983, Bottom et al. 1984, Holton et al. 1984, Dawley et al. 1986, Cordell et al. 1992, Farr and Ward 1993). All the invertebrates are common in the estuary, while only American shad, common carp (*Cyprinus carpio*), banded killifish (*Fundulus diaphanus*), and yellow perch (*Perca flavenscens*), are regularly caught (Dawley et al. 1978, 1985; Hinton et al. 1990). The following section briefly describes the history of the most abundant exotic species in the Columbia River estuary, their potential impacts on the estuarine ecosystem, and how the presence of dams may have facilitated their success.

Exotic Fishes

American shad--American shad is the most abundant exotic fish in the Columbia River basin and estuary, with a 1990 run of over 4 million adults (ODFW and WDF 1991). Much is known about American shad on the East Coast because of its former economic importance

Table 8.--Exotic fishes and invertebrates known to be present in the Columbia River estuary, at or below Jones Beach (RKm 75). Area of origin of exotic species are: As--Asia; e NA--eastern North America (east of the Rocky Mountains); A c--Atlantic coast. (Ingram 1948, Stober et al. 1979, Wydoski and Whitney 1979, Lee et al. 1980, Bernard 1983, Bottom et al. 1984, Holton et al. 1984, Dawley et al. 1986, Northcote and Larkin 1989, Cordell et al. 1992, Farr and Ward 1993).

Scientific name	Common name	Year of introduction/ First record	Origin
FISHES			
Clupeidae			
Alosa sapidissima	American shad	1885	A c
Cyprinidae			
Cyprinus carpio	Common carp	1882	As
Carassius auratus	Goldfish	unknown	As
Ictaluridae			
Ictalurus punctatus	Channel catfish	1874: Calif.	e NA
Ictalurus nebulosus	Brown bullhead	1882	e NA
Ictalurus natalis	Yellow bullhead	1942: Calif.	e NA
Cyprinodontidae			
Fundulus diaphanus	Banded killifish	1970s-80s	e NA
Percichthyidae			
Morone saxatilis	Striped bass	1879: Calif.	A c
Centrarchidae			
Micropterus salmoides	Largemouth bass	1890-95	e NA
Pomoxis nigromaculatus	Black crappie	1890	e NA
Pomoxis annularis	White crappie	1890-92	e NA
Lepomis gibbosus	Pumpkinseed	1893	e NA
Lepomis gulosus	Warmouth	unknown	e NA
Lepomis macrochirus	Bluegill	1890	e NA
Percidae			
Perca flavescens	Yellow perch	1890-92	e NA
Stizostedion vitreum	Walleye	1940s-50s	e NA

Table 8.--Continued.

Scientific name	Common name	Year of introduction/ First record	Origin
INVERTEBRATES			
Annelida, Polychaeta			
Hobsonia florida	----	unknown	A c
Arthropoda, Copepoda			
Pseudodiaptomus inopinus	----	~1980	As
Mollusca, Bivalvia			
Corbicula fluminea (= *C. manilensis*)	----	1938	As
Mya arenaria	----	1870s	A c

and the dramatic declines in East Coast runs (Walburg and Nichols 1967). The American shad is native to the Atlantic coast of North America, and was first introduced in the Columbia River and other major west coast rivers in the late 1800s (Walburg and Nichols 1967). It now ranges from Baja, Mexico, to the Kamchatka Peninsula on the western side of the North Pacific (Parks 1978, Miller 1993). Adult American shad enter the Columbia River estuary in May (Parks 1978), and spawn in slow-moving sloughs and the mainstem (Kujala 1976, Hammann et al. unpubl.). Juveniles migrate downstream in the fall (O'Leary and Kynard 1986), and overwinter in the estuary (Hammann et al. unpubl.). American shad of all ages feed on calanoid copepods, *Corophium*, *Neomysis*, *Daphnia*, chironomids, and other insects (Hammann et al. unpubl., Bottom et al. 1984). The estuary is an important rearing area for subyearling and yearling American shad and are found there all months of the year in densities of up to 125 fish ha^{-1} (Hammann et al. unpubl., Hinton et al. 1990).

American shad were established in the Columbia River over 40 years before the first mainstem dam on the Columbia River was completed. However, the dams may be partially responsible for its rapid population growth. American shad appear to successfully navigate some dams (Miller and Sims 1983, Hammann et al. unpubl.), and the completion of the Dalles Dam in 1956, and the subsequent inundation of Celilo Falls allowed extensive expansion of its range in the Columbia and Snake Rivers (Stober et al. 1979). Ironically, the decimation of East Coast stocks of American shad has been blamed in part on the construction of dams on American shad-producing streams (Walburg and Nichols 1967). Sherwood et al. (1990) suggest that the increased importation of plankton into the estuary from upstream reservoirs has benefitted zooplanktivores, such as American shad.

American shad may have considerable impact on the estuarine ecosystem because of its large numbers, year-round presence in the estuary, and high degree of dietary overlap with other planktivorous fishes, such as longfin and surf smelt, threespine stickleback, and Pacific herring (Bottom et al. 1984). In comparisons of Columbia River fishes in the estuary, American shad had growth rates comparable to longfin smelt and Pacific herring. Yearling American shad had fewer empty stomachs than surf and longfin smelt, threespine stickleback and Pacific herring (Bottom et al. 1984), suggesting some competitive advantage exists for American shad. More detailed studies are required to determine the ecological effects of American shad on the estuarine community.

Common carp--Common carp of European origin was introduced into the Columbia River system in the 1880s as a food fish (Wydoski and Whitney 1979). Common carp is present in spring, summer, and fall in the brackish and freshwater regions of the estuary (Bottom et al. 1984), and has been recorded at densities of 20 fish ha^{-1} at Miller Sands (Hinton et al. 1990). Common carp is generally found in shallow habitats with slow-moving water; it aggregates to spawn in spring when water temperatures exceed 15 °C (Wydoski and Whitney 1979). Common carp are highly fecund, and a single female may contain as many as 2 million eggs. Juvenile common carp consume zooplankton, primarily cladocerans and copepods, while older fish are omnivores and consume aquatic vegetation, insects, and clams (Wydoski and Whitney 1979). Common carp are very tolerant of adverse environmental conditions, and can withstand low dissolved oxygen levels, fluctuating temperatures, turbidity, and pollution (Wydoski and Whitney 1979).

The effects of dams on the success of common carp are unknown, but its tolerance to adverse environmental conditions, coupled with its high reproductive potential, may have allowed it to successfully withstand the human-induced changes in the Columbia River estuary. The impact of common carp on the estuarine ecosystem is also unknown, although its relatively low abundance decreases the likelihood that it may significantly impact the estuarine assemblage via competitive or predatory interactions.

Banded killifish--Banded killifish was presumably introduced into the Columbia River basin illegally (Farr and Ward 1993). It was first recorded from the upper estuary at Jones Beach in 1971 (Misitano and Sims 1974), but was not consistently caught in excess of one fish per beach- or purse-seine haul until the late 1980s (Hinton et al. 1990). Densities of 375 fish ha^{-1} have been observed at Miller Sands, where it is present at least in the summer and fall (winter sampling has not been conducted) (Hinton et al. 1990, 1992a). Banded killifish is native to eastern North America, where it inhabits shallow areas of lakes, streams and sloughs, in fresh and brackish-water areas (Smith 1985). In its native range, the banded killifish is primarily a surface and midwater feeder, consuming cladocerans and ostracods, but also consumes molluscs and flatworms (Smith 1985).

The diet of banded killifish in the Columbia River estuary, its impact on native fish fauna, its impact on the estuarine ecosystem, the extent of its range, and the role of dams in fostering its survival are unknown. Estuarine environmental conditions resulting from the operation of the hydropower system may have facilitated the survival and range extension of banded killifish.

Yellow perch--Yellow perch was planted in the Columbia River basin in 1890-92 for food (Wydoski and Whitney 1979), and is commonly present in the upper Columbia River estuary in densities of 10 fish ha^{-1} (Bottom et al. 1984, Hinton et al. 1990). Yellow perch is fairly fecund and spawns in April and May. A 35-cm long female may contain as many as

140,000 eggs. Young yellow perch consume zooplankton and insects, while older fish are generalized bottom and midwater feeders, and consume a variety of insects, crustaceans, and fish (Wydoski and Whitney 1979, Smith 1985).

The role of dams in the success of yellow perch and/or the ecological effects of yellow perch on the estuarine ecosystem are unknown. Yellow perch has been shown to have a negative impact on resident fishes in closed systems: yellow perch introduced into a lake populated by brown and rainbow trout caused dramatic diet shifts and decreased productivity of the resident trout (Smith 1985). Its ecological impact in open systems, such as the Columbia River estuary, is unknown. Like common carp, the relatively low abundance of yellow perch suggests that its impact on other fishes would be less than that of the more abundant exotic fishes.

Exotic Invertebrates

Corbicula fluminea (=***C. manilensis***)--The Asian freshwater clam, *C. fluminea*, was first discovered in North America in the Columbia River estuary in 1938 (Ingram 1948). It was probably introduced from ship fouling, although the introduction may have been deliberate because *C. fluminea* is valued for food and bait in Asia (Ingram 1948, 1959). *C. fluminea* has since spread throughout the southern U.S. and northern Mexico, and by 1970 was found in 21 states and Baja and Sonora Mexico (Fast 1971). *Corbicula fluminea* has received considerable attention elsewhere in North America because of its rapid spread, high reproductive capability, transport of larvae, and its tendency to clog water pipes (Ingram et al. 1964, Sinclair 1971, Gardner et al. 1976, McMahon 1977). The clam also has considerable feeding plasticity (Way et al. 1990), and has been shown to outcompete native bivalves; it caused the near-extinction of some endemic bivalve assemblages in the southeastern U.S.

(Gardner et al. 1976). Like common carp, *C. fluminea* is very tolerant of adverse environmental conditions, and can withstand wide ranges and fluctuations in temperature, dissolved oxygen, flow velocity, water level, and pollution (Sinclair 1971, Gardner et al. 1976).

In the Columbia River estuary, *C. fluminea* is found from Tansy Point in the brackish zone, upstream throughout the estuary, and in the river basin (Blahm and McConnell 1979; Durkin et al. 1979, 1981; Holton et al. 1984; McCabe and Hinton 1990). Densities exceeding 10,000 clams m^{-2} have been recorded from Cathlamet Bay (Emmett et al. 1986a), although densities of 100 to 3,000 m^{-2} are more typical in the estuary (Emmett et al. 1986a; Hinton et al. 1990, 1992a).

The role of dams in the spread of *C. fluminea*, or its effects on estuarine food webs and benthic infauna are unknown. Sherwood et al. (1990) suggest that *C. fluminea* may have benefitted from the altered circulation and deposition patterns in the present-day estuary because more detritus is transferred to peripheral bays where high *C. fluminea* densities are found. *Corbicula fluminea* in the Columbia River estuary is consumed by many species, including white sturgeon (Kujala 1976, McCabe et al. 1989), juvenile coho salmon (Durkin et al. 1981), steelhead (Bottom et al. 1984), American shad (Bottom et al. 1984), surf scoters (Hazel 1984) and raccoons (Dunn et al. 1984). It has also been consumed by common carp, suckers (*Minytrema* spp.), channel catfish (*Ictalurus punctatus*) and other fishes in other river systems (Rinne 1974, Britton and Murphy 1977). Given the locally high densities of *C. fluminea* in the estuary, its resilience to adverse conditions, and its feeding plasticity, it should be expected that some ecological effects are occurring. The effects of *C. fluminea* on native bivalves in the estuary is unknown. Like other abundant exotic species in the Columbia River estuary, *C. fluminea* warrants further research because of its high potential to impact the estuarine community.

Mya arenaria--The bivalve *M. arenaria* is thought to have originated in the North Pacific Ocean, spread to the Atlantic Ocean, and then become extinct in the North Pacific in the late Tertiary (Bernard 1983, Carlton 1992). It was incidentally reintroduced into the North Pacific in the 1870s with the introduction of the Atlantic oyster (*Crassostrea virginica*) for aquaculture (Bernard 1983, Carlton 1992).

Mya arenaria is restricted to tidal flats in the lower, more saline portions of the Columbia River estuary such as Baker Bay (Holton et al. 1984, Simenstad et al. 1984, Furota and Emmett 1993). Densities are relatively low (1-40 m^{-2}), and typically much lower than those of the coexisting native bivalve, *Macoma balthica* (Holton et al. 1984). The lower densities may partially reflect sampling efficiency, since *M. arenaria* is often located more than 20 cm below the sediment surface, and is consequently difficult to sample effectively (Furota and Emmett 1993). Little is know about *M. arenaria* in the Columbia River estuary except that it is consumed by juvenile starry flounder and English sole (Bottom et al. 1984). Based on densities alone, one would expect the ecological effects to be relatively minor compared to other more abundant exotic invertebrates. The role of dams in the success of *M. arenaria* is also unknown. However, its restriction to the lower estuary suggests less impact from dam-related fluvial changes than other anthropogenic changes occurring within the estuary, such as navigational improvements.

Pseudodiaptomus inopinus--The calanoid copepod *Pseudodiaptomus inopinus* is a recent introduction into the Columbia River estuary, which was first recorded in 1990 (Cordell et al. 1992). The species is thought to have been introduced from the Indo-Pacific region in ballast water from cargo ships. Its successful establishment in the estuary is thought to have resulted, in part, from moderated peak flows and warmer minimum water temperatures caused by anthropogenic activity, including the operation of dams (Cordell et al. 1992). This species is now the third most abundant zooplankter in the estuary and has been recorded at densities of 17,000 m^{-3} (Cordell et al. 1992). The role of *P. inopinus* in the estuarine ecosystem is largely unknown, especially with respect to interactions with other species, and its usefulness as prey for fish.

Pseudodiaptomus inopinus, like the other two abundant zooplankters in the estuary (*Eurytemora affinis* and *Scottolana canadensis*), is associated with the ETM. However, frequent, intensive, multiparameter sampling of the ETM has shown that *P. inopinus* is associated with different physical attributes of the ETM than the other two copepods, suggesting a reduced potential for competitive interactions between the exotic and resident zooplankton (Cordell et al. 1992). Like *C. fluminea*, the high densities of *P. inopinus* suggest that it may be having a significant ecological impact, and this organism deserves further study.

Hobsonia florida--*Hobsonia florida* is an Atlantic Ocean surface deposit-feeding Annelid polychaete, now widely distributed and abundant along the west coast of North America (Holton et al. 1984). Densities of 3,000 to 6,000 m^{-2} are common in the Columbia River estuary and peak abundances in June and July may reach 30,000 m^{-2} in Baker Bay (Holton et al. 1984), making *H. florida* a major member of the infaunal assemblage in the bay (Furota and Emmett 1993). *Hobsonia florida* is consumed by fishes such as juvenile pleuronectids (Durkin et al. 1981), although it is not a common food item for fishes (Bottom et al. 1984). Like *M. arenaria*, the role of dams in the success of *H. florida* is unknown, but dams may also be of limited importance to *H. florida* because of its restriction to the lower estuary.

Summary of Exotic Species in the Columbia River Estuary

Of the 29 exotic fish and 4 exotic invertebrate species known to be in the Columbia River basin, 16 fish species and all of the invertebrates are found in the estuary (Table 8). The ecological effects of these exotic species on the estuarine community are mostly unknown, but the potential for measurable impacts may be restricted to a few of the more abundant species. Abundant species have the greatest potential to impact the species assemblage via competitive or predatory interactions because of the greater number of individuals available for interaction. Of the four most common exotic fishes found in the estuary (American shad, banded killifish, yellow perch, and common carp), only the American shad is abundant. Three of the four invertebrates are locally abundant, although all have limited distributions. Consequently, of the 20 known exotic species in the estuary, only 4 have the potential at this time to impact the estuarine community, and only the American shad is distributed throughout the estuary. Banded killifish could impact the estuarine food web in the future, but its distribution, which is limited to shallow habitats, and its small size may limit its ecological impact on the estuary. However, if its population continues to grow, the banded killifish may warrant detailed study with respect to its role in the ecology of the estuary.

The attributes of exotic species and the history of their interactions with native species can be used to indicate the potential for further interactions, and to determine which species warrant further investigation. For example, *C. fluminea* is highly resilient to adverse environmental conditions, has considerable feeding plasticity, high filtration and reproductive rates, and can outcompete native bivalves (Gardner et al. 1976). Based on these attributes, its potential for impacting the infaunal community is expected to be greater than exotic species without such attributes. Similarly, American shad has considerable dietary overlap with other fishes in the estuary, and therefore may be a significant competitor for available food resources both in the estuary and upstream. Unfortunately, aside from *C. fluminea* and American shad, little is known about many of the exotic species in the estuary and their potential for effecting ecological change.

The role of dams in the survival and proliferation of exotic species in the estuary is largely unknown. Some exotic species are thought to have benefitted from altered hydrologic conditions and temperatures associated with dams (Stober et al. 1979; Maule and Horton 1984, 1985; Li et al. 1987), although such facilitation would be difficult to prove. Numerous exotic species were introduced into the Columbia River system before dams were constructed but with the exception of American shad, their presence in the estuary was not monitored until after mainstem dams were completed. Dams, by altering riverine environments, have decidedly promoted the successful proliferation of exotic fishes in other river systems (Holden and Stalnaker 1975, Achieng 1990, Nieland et al. 1990, Ogutu-Ohwayo 1990, Carlton 1992). However, the impact of dams on exotic species in the estuary may be less than that of strictly freshwater exotic species farther upstream.

Summary of the Effects of Dams on the Estuarine Community

Dams are thought to have impacted the Columbia River estuarine community in several ways, ranging from minor biotic adjustments to major changes in the structure of the food web. Because no scientific investigations of the estuary were conducted prior to dam construction, almost all the predicted impacts are based on speculation rather than on the results of scientific investigations. Without more detailed analyses, such as the carbon budget calculated by Simenstad et al. (1990), there is little information with which to quantify the relative magnitude of dam-induced impacts and their true ecological significance.

One effect of dams on the estuary has been to reduce the variability in the environment, primarily because of flow regulation. Prior to present-day flow regulation, the difference between seasonal maximum and minimum flows was greater, causing greater seasonal variation in salinity in any given location in the estuary. As discussed above, many organisms in the estuary are distributed according to their salinity tolerance, and many exhibit seasonal changes in distribution in response to seasonally changing salinities. A decrease in the variability and extent of seasonal salinities would allow range extensions and establishment of populations in areas no longer subjected to highly variable and often intolerable salinity levels. Seasonal distributional changes in response to changing salinities would be damped, allowing both marine and freshwater species to extend their ranges farther into the estuary. Such range alterations in response to natural- and anthropogenic-induced variable freshwater flow have been observed for fish and invertebrates in the Sacramento-San Joaquin estuary (Armor and Herrgesell 1985, Moyle et al. 1986). For example, freshwater organisms in that system are found lower in the estuary during wet years, while marine species are found higher in the estuary during dry years. In addition, the long-term distribution of fish and invertebrates, such as striped bass, have been changing as annual river flow is altered (Armor and Herrgesell 1985).

Decreased variation in river flow would also allow pelagic organisms, especially weak-swimming forms, a better chance to remain in the estuary rather than being swept out to sea. Haertel and Osterberg (1967) reported a large and extended decrease in brackish-water pelagic estuarine copepods following large regulated floods. These floods no longer occur, whereas in the unregulated Columbia River, floods of large magnitude were much more common (Orem 1963), and may have caused periodic depletions of pelagic brackish-water zooplankton in the estuary. Such depletions likely affect zooplanktivores, perhaps to the extent of causing dietary shifts and altered distribution. In this respect, dams may have stabilized the resident brackish-water plankton community.

American shad and other planktivorous fishes are thought to have benefitted from the increased amount of phyto- and zooplankton imported into the estuary from upstream (Simenstad et al. 1990). The slow-moving, warmer water in reservoirs; the increased nutrient levels from irrigation return flows; and the improved light penetration are likely responsible for elevated plankton production in the river (Sherwood et al. 1990). This production decreases with distance upstream (Bristow et al. 1985); however, the importation of available energy from upstream to the estuary is thought to have increased from 82,000 t C per year prior to 1870, to 208,000 t C per year in 1980 (Sherwood et al. 1990). Assuming these estimated values are representative of historical and present-day conditions, the increased importation of plankton from upriver has substantially benefitted the pelagic portion of the food web and organisms utilizing suspended organic material such as the suspension-feeding exotic clams, *Corbicula fluminea* and *Mya arenaria.*

The putative impacts of dams on the biological community in the Columbia River estuary appear to have a fairly neutral effect on salmonids in the estuary. Flow regulation caused by dams has altered the salinity regime in the estuary, but highly mobile and migratory salmonids move through the estuary relatively quickly and are not likely to be influenced by the altered salinity regime. The generally transient residents of the estuary can probably adjust to salinity changes without severe negative effects. The effects of altered salinity regimes on juvenile salmonid prey are largely unknown, although *Corophium salmonis*, one of their primary prey, are also fairly mobile as indicated by their seasonal migrations. Because juvenile salmonids are generally not planktivorous in the estuary, they would probably not benefit from the increased plankton importation from upstream. Therefore, the diet of non-planktivorous juvenile salmonids may minimize the potential for competitive interactions with American shad. Most migrating juvenile salmonids travel directly through the estuary without pausing, so impacts from their exposure to altered conditions in the estuary may be minimal. However, juvenile chinook salmon which reside in the estuary for extended periods undergo rapid growth during residence, suggesting that the altered estuary provides suitable habitat for juvenile salmonids.

The exotic species in the estuary may have benefitted from dams on the Columbia River. Changes caused by dams, such as the reduced variability in salinity, may have removed the competitive advantage of native species over exotic competitors. Similarly, changes caused by dams may have transformed a habitat that was formerly inhospitable to exotic species into one in which exotic species can exist and even thrive. Studies of exotic species in other systems suggest that exotic species may negatively impact native species through competitive or predatory interactions, although such interactions have received limited attention in the Columbia River and estuary. For anadromous fishes that pass through the estuary, predation by exotic species upstream has been shown to be significant, as in the case of predation on juvenile salmonids by walleye (*Stizostedion vitreum*) and channel catfish (Beamesderfer et al. 1987, Beamesderfer and Reiman 1988). In this respect, dams contribute to the mortality of anadromous species that use the estuary by providing suitable habitat for piscivorous exotic species in the upstream reservoirs (Li et al. 1987, Stober and Nakatani 1992).

There does not appear to be any prominent effect of river flow regulation on the ecology of the estuary offshore in the plume. This area is highly dynamic and subjected to physical and biological influences much greater than river regulation, such as variations in nutrient-rich upwelling and anomalies produced by ENSO events. Examinations of biological attributes of the plume over time have indicated little or no changes attributable to river flow regulation, detectable above the considerable annual variability.

In conclusion, it does appear that dams have had an impact on the biotic community in the land-confined portion of the Columbia River estuary, primarily through flow alteration and increased importation of plankton. Dams may have also aided in the establishment of exotic species in the estuary, which may negatively impact native species. However, at present the impact of dams on salmonids in the estuary appears to be limited. The lack of data prior to the construction of dams on the river, and the lack of understanding of interactions between exotic and native species have greatly hindered an accurate evaluation of the effects of dams on the Columbia River estuary.

CONCLUSIONS

Based on the previous discussions, the following conclusions can be made about the effects of dams on the Columbia River on the estuary:

1. Numerous factors in addition to dams have affected the estuary, most importantly navigational improvements, diking, and increases in the human population. These other factors have often had effects similar to or greater than the effects of dams, making it difficult to isolate effects related to dams. The lack of biological data collected in the estuary prior to the completion of any mainstem dams has further hampered the effort to determine the effects of dams on the estuary.

2. One of the largest environmental effects of Columbia River dams on the estuary has been river flow regulation. The suppression of maximum and minimum river flows has decreased variability in both estuarine circulation and salinity intrusion, and has possibly increased accretion in the estuary. Decreased variability in circulation and salinity intrusion affects the biological community by decreasing the selective pressures to which organisms must adapt. This potentially allows the proliferation of species that were unable to withstand the unmodified estuary. River flow regulation does not appear to have impacted the portion of the estuary offshore in the plume.

3. Columbia River dams have altered water quality primarily through activities dependent on dams, such as irrigation and the development of industries requiring hydroelectricity. Both activities introduce contaminants into the river and cause thermal pollution. However, the relative impact on water quality caused by activities dependent on dams, as compared to other sources of water quality degradation, is unknown.

4. Columbia River reservoirs and nutrient-rich irrigation return flows are thought to be largely responsible for dramatically increased levels of plankton imported into the estuary from upriver. It has been suggested that this input of plankton benefits the pelagic portion of the estuarine food web, including the exotic American shad (*Alosa sapidissima*). Species dependent on the *Corophium*-based portion of the food web, such as juvenile salmonids, may have not benefitted from the increased input.

5. Most juvenile salmon in the Columbia River are of hatchery origin and spend little time in the estuary as they migrate directly through the estuary to the ocean. Consequently, they might not be exposed to altered conditions in the estuary for a sufficient length of time to cause measurable impacts. However, for subyearling fall chinook salmon residing in the estuary for extended periods of time, their high mobility should allow them to withstand dam-induced alterations to the estuary, such as changes in salinity, circulation patterns, and prey distribution, without severe effects.

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APPENDIX A

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