APPENDIX C. ESTUARINE HABITAT EVALUATION

I. INTRODUCTION

Since 1991, 12 different ESUs (Evolutionarily Significant Units) of anadromous salmonids that reproduce in the Columbia River Basin have been listed as threatened or endangered under the Endangered Species Act (ESA) of the United States (McClure et al. 2003). In recent years, there has been increasing emphasis on the role of the estuary and plume in the decline and recovery of these salmonids. The estuary and plume are the connection between freshwater and marine habitats and are used by all anadromous species to some degree for feeding, refugia from predators, and physiological transition (Simenstad et al. 1982; McCabe et al. 1983, 1986; Bottom and Jones 1990). Including the estuary as an element in salmon recovery represents a significant departure from previous management efforts in the system and recognizes that effects of hydroelectric development and other upriver alterations are not localized to "between the dams".

Here, we evaluate the effects of a number of factors associated with the Columbia River estuary (which includes the plume) on the viability of listed, anadromous ESUs in the Columbia River Basin. We only consider those ESUs spawning above Bonneville Dam and Columbia River chum salmon. Our analysis is aimed primarily at addressing the issue of whether there is potential to improve anadromous salmon population status through improvement in conditions in the estuarine environment. It was conducted in support of analyses of the potential to improve anadromous salmon population status through improvement in conditions in tributary environments.

II. THE COLUMBIA RIVER ESTUARY

We defined the Columbia River estuary to encompass the entire habitat continuum downstream of Bonneville Dam where tidal forces and river flows interact, regardless of the extent of saltwater intrusion. Thus, the upstream boundary of the estuary is Bonneville Dam, which is the extent of tidal influence, while the "downstream" boundary includes the plume.

The estuary can be divided into different zones based upon a variety of attributes, such as geomorphic features, ecological functions, tidal conditions, salinity regimes, and physical characteristics. A number of approaches have been employed to describe and classify the different zones of the estuary (e.g., Johnson et al. 2003). For purposes of this report, we consider only two zones of the estuary, primarily because our information on use of the estuary by juvenile salmon does not yet allow us to discriminate use at a finer scale. The first zone extends from the river mouth upstream to Bonneville Dam and includes conditions ranging from tidally influenced freshwater in the upper estuary to higher salinities and higher wave energies near the river's mouth.   The second zone is the river plume which is generally defined by a reduced-salinity contour near the ocean surface of 31 parts per thousand. Its geographic position varies greatly with seasonal changes in river discharge, prevailing nearshore winds and ocean currents. During summer months, the plume extends far to the south and offshore along the Oregon coast; during the winter it shifts northward and inshore along the Washington coast.

Throughout the estuary is a mix of habitats that the juvenile salmon can potentially occupy. Habitat is the physical, biological, and chemical characteristics of a specific unit of the environment occupied by a specific plant or animal. Thus, habitat is unique to specific organisms and basically encompasses all the physiochemical and biological requirements of that organism within a spatial unit. The function of any estuarine habitat for juvenile salmon depends upon site specific or patch scale attributes such as vegetation type, substrate type, and salinity regime (Simenstad and Cordell 2000). In addition, habitat functions depend upon larger scale attributes or the landscape context of that habitat (Simenstad 2000). Landscape context refers to the spatial arrangement of habitat, including its size and shape; location of the habitat within the estuary; the composition of surrounding habitat; and connectivity with other habitats (Turner 1989).

III. APPROACH

A. Defining Life history Type and Life History Strategy

Our overall purpose here is to evaluate and rank selected factors in the estuary and plume with respect to their potential to improve viability of listed populations. Ideally, we would like to link factors in the estuary to their potential to affect the viability of each listed population. However, because we do not have specific, empirical information describing estuarine habitat use by anadromous populations in the Columbia River estuary and plume, we used an alternate approach where effects of candidate factors were linked to viability of an ESU. As each ESU is comprised of a group of populations, we can then infer responses of populations based upon what we predict will occur for the ESU.

We first defined each ESU as either stream type or ocean type based upon characteristics of the juvenile outmigrants. Here, we use the terms stream and ocean type to separate ESUs into two groups based strictly upon certain characteristics exhibited by juveniles during their first year of life, including how long they rear in freshwater, when they outmigrate and how long they spend in estuarine habitats. Ocean type populations are generally (but not exclusively) composed of individuals that migrate to sea early in their first year of life after spending only a short period (or no time) rearing in freshwater. Stream type fish generally migrate to sea after rearing for at least a year in freshwater. Thus, ocean type fish tend to spend longer periods in ocean habitats compared to stream type populations. Information used to define life history types came primarily from the species status reviews: chinook salmon (Myers et al. 1998), chum salmon (Johnson et al. 1997), and sockeye and steelhead (Busby et al. 1996).

Each life history type is comprised of individual members who employ a variety of alternative spatial and temporal strategies to use available habitats. We defined a life history strategy as an approach to the use of available habitats, including the estuary. To define alternate strategies of estuarine habitat use, we used the size at estuarine entry and the time when they arrive in the estuary. Numerous studies suggest there are strong linkages between fish size, habitat use, and residence time (Healey 1980, 1982; Levy and Northcote 1981, 1982; Simenstad et al. 1982; Carl and Healey 1984; Levings et al. 1986; Bottom et al. 2001; Miller and Sadro 2003). Juvenile salmon are generally distributed along a habitat continuum based upon water depth with the depth of the water occupied by the fish generally increasing as the size of the fish increases. Smaller fish are most closely associated with shallow water areas while larger fish (e.g., yearlings) are more associated with deeper water habitats such as larger channels.

Based upon patterns of size and time of estuarine entry, we identified six life history strategies based upon historic use: 1) early fry, 2) late fry, 3) early fingerling, 4) late fingerling, 5) subyearling, and 6) yearling. Fry are defined as fish that enter the estuary at a size < 60 mm with early fry entering in approximately March and April and late fry from May to June. Fingerlings are those fish that enter the estuary at a larger size than fry, which implies there was some period of freshwater rearing, but have yet to begin the physiological transition associated with smolting. Subyearlings rear primarily in freshwater with relatively little time spent in the estuary, and smolt as they outmigrate during their first year of life. Yearlings rear for at least one year in freshwater and then emigrate; these fish generally spend less time in the estuary than fry, fingerlings, or subyearlings. Some differences between populations within an ESU in the relative proportions life history strategies can be expected but we could not discriminate such differences. Therefore, we assumed that all populations within a life history type/ESU produce the same characteristic mix of these strategies when viewed over long time scales.

B. Factors Included in the Analyses

The major estuarine related factors that we believe can potentially affect salmonid population viability include climate and climate change (which control other factors), water flow, access to and quality of habitats, sediment, salinity, temperature, toxics, predators (e.g. terns, cormorants, northern pikeminnow), and hatchery and harvest practices. Although it would be useful to evaluate the role of each of these factors, we only selected factors where: 1) a significant change was evident between current and historical conditions, 2) the factor could potentially affect population viability, and 3) there was quantitative data available that could be used to analyze the effect of the factor within the time we had been allotted. We wish to emphasize that factors were not selected based upon whether or not we believed they had a significant affect on salmonid population viability. Some factors that were not selected for consideration in this report may have a significant affect on viability. The factors that were included in this analysis are water flow, availability of salmon habitats, toxics, and predation by Caspian terns. For each of these factors we provide a brief analysis as to how this factor could affect population viability. From these analyses, we developed a series of hypotheses that helped guide how we rated their relative importance for ESUs.

C. Analyzing and Rating the Relative Importance of Limiting Factors

To describe the importance of each factor, we developed a rating system that ranked each factor as having a high, medium, or low ability to improve the status of anadromous salmon populations. We drew inferences about how a factor affects an ESU based upon the life history type of that ESU and how we believed the factor would affect the life history strategies that characterized that life history type. Thus, all stream type ESUs were rated similarly while all ocean type ESUs were rated similarly. Ratings were developed by considering each factor relative to other estuarine factors within an ESU; ratings were not considered with the context of other non-estuarine factors such as tributary habitat.

We defined improvement in population status to mean improvement in population viability (McElhaney et al. 2000) as defined by the four VSP performance criteria: abundance, population growth rate, spatial structure, and diversity (McElhaney et al. 2000). The rating system consisted of two levels. The level 1 screens evaluated ***if*** the factor was likely a concern for an ESU based upon its effects on VSP and change in the factor from historic conditions. The level 2 screens asked ***how*** the factor affected an ESU based upon where the effects occurred.

Level 1

Question 1: What is the effect on each VSP parameter? Each factor can potentially have some effect on each VSP parameter. We assumed if the factor affected large numbers of individuals in the ESU (again relative to the other factors) that there was a significant effect on abundance and productivity. Because most populations in threatened or endangered status are at low levels of abundance, we doubled the score of any factor that affected abundance or productivity. We reasoned that these depressed populations needed short term increases in abundance before long term benefits resulting from increased diversity and spatial structure would be useful. If a factor affected particular life history types or affected specific habitat types more than others, we assumed the primary affect was on spatial structure and diversity.

Question 2: Has the factor changed from historic conditions and could it be improved relative to the other factors? We considered whether each factor had changed significantly from historical conditions. Because we intentionally selected factors that we believed had changed significantly from historical conditions, this criteria did not result in much difference between factors. We also considered from a practical perspective how much change in each factor was possible. A factor could be significantly changed from historical levels but be relatively difficult to modify relative to other factors.

LEVEL 2

Question 1: Does the factor have a significant effect on the abundance of the dominant life history strategy? For the dominant life history strategy, we asked how the factor affected the abundance of juveniles of that life history type in estuarine shallow water, estuarine deep water and plume habitats. Because it is consistent with available information on salmon use, we considered the estuary from Bonneville Dam to the mouth as one zone and the plume as a second major zone. Within the portion of the estuary from the mouth to Bonneville Dam, shallow, low velocity habitats (e.g., swamps, emergent marshes, and shallow flats) were distinguished from medium and deep, higher velocity channel habitats in the analysis; there is strong evidence that habitat use varies between these habitat types. The plume was considered as one habitat unit.

Question 2: For the dominant life history strategy, does the factor affect habitat quality, quantity, and opportunity? For the dominant life history strategy, we asked what type of effect the factor had in estuarine shallow water, estuarine deep water and plume habitats. We considered effects of the factor on habitat quantity, quality, and opportunity. The concepts of opportunity and quality (or capacity) were proposed by Simenstad and Cordell (2000) and adopted by Bottom et al. (2001) for the Columbia River estuary. Opportunity attributes relate to the accessibility of habitat to juvenile salmon and are largely physical and chemical in nature such as tidal elevation and location of habitat. In general, capacity measures primarily relate to the biotic and ecological functions (i.e., acquiring food and avoiding being eaten) of habitat. In addition to capacity and opportunity, we also included quantity of habitat as a separate metric. For toxics, we rated effects separately in shallow water and deep water estuarine habitat for water borne and sediment borne contaminants. For example, if there were risks to the main life history type from both types of contaminants in shallow water, then the score was doubled.

Each of the four questions listed above was evaluated for each factor for each ESU based upon whether they were an ocean or stream life history type. Scoring was done using guidance from the principles/hypotheses developed in the following discussion of limiting factors. Each cell in the matrix was either scored as a yes (+1) or no (0) with two exceptions: 1) abundance and productivity which were given a 2 score, and 2) toxics in deep and shallow water which each could be scored a 2 if there was effects from both water borne and sediment associated toxics. Thus, for flow, habitat and predation, the maximum possible score was 20 whereas the maximum possible toxics score was 28. The final rating was computed as the ratio between the assigned score and maximum possible.

IV. LIMITING FACTORS

A. Flow

Water, interacting with the land, forms the habitat that juvenile salmon occupy. The estuarine habitat features to which salmon have adapted are largely the result of riverine and tidal processes. However, the shaping of estuarine habitats is also controlled by several "external" factors which help establish the physical template for the entire estuary. First, characteristics of the watershed affect such factors as the amount and timing of water arriving in the estuary. For example, because most of the western sub-basin is at too low an elevation to accumulate a large seasonal snow pack, the highest flows are observed in this region during and shortly after winter storms between December and March. In contrast, most of the flow in the interior sub-basin occurs as the result of melting of a seasonal snow pack between April and June.

Second, natural variations in Columbia River flows occur as a result of both short and long term fluctuations in climate. Because of the vast extent of the Columbia River basin, the effects of climate vary considerably within the basin. Climate-induced variations in Columbia River flow occur on time scales from months to centuries (Chatters and Hoover 1986, 1992). One example of this is the Pacific Decadel Oscillation, commonly known as the PDO (Mantua et al.1997), which alternates between cold and warm phases at approximately 30-year time scales. The cold phases of the PDO (e.g., the 1945-1976 period) are generally considered to benefit salmonid production in the Pacific Northwest.

Changes in flow attributes, such as when and how much water arrives in the estuary, are an integral measure of changes in a river system. In a recent analysis and review, Jay (as reported in Bottom et al 2001) concluded that there has been approximately a 16.4% reduction in annual flow over the last approximately 100 years. Seasonal changes, particularly those in spring freshet timing and magnitude, have been much greater than changes in annual average flow. Spring freshets are extremely important for juvenile salmonids in that high flows (especially overbank flows) provide habitat, limit predation by increasing turbidity, maintain favorable water temperatures, and supply organic matter to the detritus-based food web, centered in the estuarine turbidity maximum (ETM). Jay (2001) found that the timing of the freshet flows is now about a month earlier than historically. And, the maximum daily spring freshet flow is now about two weeks earlier that historically (Water Year Day 242 vs 256).

Flow regulation is clearly the source of the largest reduction in spring flow, with climate change having little effect (Bottom et al. 2001). The present decrease in freshet seasonal flow due to water withdrawal was an estimated 10.5% (a reduction of 5.7% for May, 12.5% for June, and 20.8% for July, respectively). In contrast, the estimated freshet seasonal flow decrease due to flow regulation was overall 33.1% (a reduction of 31.6% for May, 32.4% for June, and 19.8% for July, respectively). The effect of climate was relatively small.

A feature of water flow that is significant to juvenile salmon is the occurrence of overbank flows. Historical bankfull levels exceeding 18,000 $m^3 s^{-1}$ now rarely occur due to effects of flood control measures and irrigation depletion (Bottom et al. 2001). Prior to 1900, some overbank flow occurred in many years both in winter and in spring. The season when overbank flow typically occurs has also shifted from spring to winter (Bottom et al. 2001). Flood protection, diking, flow regulation, and water withdrawal largely eliminated climate influence on overbank flow (Bottom et al. 2001).

A significant consequence of altering flow regimes is changes in movements of sediment through the Columbia River system. It is not possible to precisely apportion the reduction in sediment transport between climate change, water withdrawal, and flow regulation. The largest single factor is, however, reduction in spring freshet flow. This is demonstrated when evaluating the impact of flow on sediment volumes under recent conditions compared to a more historical period when the spring freshet was less impacted. For example, Jay (2001) reported that the difference between annual average sediment transports at Vancouver for the 1858-1899 virgin flow and 1945-1999 observed flow was 10.8 million x 10^6 metric or 52% of the 19th-century sediment transport. The difference between annual average sediment transports for the 1879-1899 virgin flow and 1970-1999 observed flow is ~12.5 x 10^6 t or 61.5% of the 19th-century sediment transport.

Alterations in the amount and timing of water delivery to the estuary has significantly affected availability and quality of the habitat needed in the estuary to sustain the diverse life history strategies for the various source populations of salmon and steelhead. flow changes in the basin into perspective. Using a hydrologic model developed specifically for the Columbia River, Baptista (2001) found the estuary during the historic period (late 1800s) was able to sustain habitat features defined to be important to salmon (characterized as water velocities less than 30 cm/sec-- important to smaller juvenile salmon) to a greater degree in the face of ever increasing water flows than is evident now. In the tidal freshwater zone of the Columbia River estuarine system (RM 50 to 90), Kukulka and Jay (2003) demonstrated that there was approximately a 62% loss of shallow water habitat (defined by depth between 10cm and 2 m) that was attributable to diking (physically removing access of water to the tidal floodplains) and the reduction of peak flows. Diking and flow reductions have reduced shallow water habitat in the freshwater tidally influenced region of the Columbia River estuary by 52% and 29%, respectively.

Major departures from the historical template of an ecosystem can thus potentially alter the ability of the estuary to support juvenile salmon. For example, diversity of salmonid rearing and migration behaviors are linked to various habitats and environmental conditions that can support each developmental stage (e.g., egg, fry, smolt, etc.). Alteration of the physical environment of the estuary can create mismatches between established salmon behaviors and the physical environment or, similarly, prevent the expression of potential behaviors by eliminating habitat opportunity. The loss of these behaviors can reduce the life history diversity of populations and increase the vulnerability of the population to environmental variability. One especially significant change is the reduction of the spring freshet to which the timing of downstream migrations and patterns of habitat use of some subyearling and yearling life-history types may have been linked. One potential result of dampening flow variations in the Columbia River could be a greater uniformity of migration patterns with potential consequences in the timing and sizes of salmon arrival in the estuary and/or ocean. The nearly complete elimination of overbank flow may pose significant consequences for Columbia River salmonids. Access to off-channel floodplain habitats during high flow events has been greatly reduced by reductions in overbank floods. If, as we suspect, patterns of extended estuary use by small subyearling migrants are directly linked to the availability of shallow-water habitat, the loss of these habitats has significant implications for these estuarine dependent strategies.

Flow regulation, in conjunction with floodplain diking, may also influence the productive capacity of the estuary by regulating "ecological processes" such as food production, competition, and predation. Elimination of overbank flooding can prevent the pulsed delivery of structural and energetic components to the rest of the estuary, including large wood, sediments, detritus, and prey organisms produced in adjacent riparian and floodplain habitats. Floodplain inundation can greatly increase the surface area of tidal estuarine and riverine habitats available to salmonids, allowing fish to expand their distribution into potentially more productive off-channel areas.

The effect of flow changes is not restricted to the area traditionally considered the estuary in the Columbia River system (i.e., upstream of the river mouth). In addition, flow from the Columbia River, as well as other physical changes in the estuary (e.g., dredging), can modify the features that define habitat in the dynamic plume environment (Barnes et al. 1972). Attributes of the plume affected by flow changes that define habitat important to salmon include surface area of the plume, the volume of the plume waters, the extent and intensity of frontal features, and the extent and distance offshore of plume waters.

Clearly, any such changes in plume habitat are only relevant if the plume has a role that influences how juveniles make the transition from a freshwater to marine environment. Evidence collected to date suggests the plume serves salmon in multiple ways, such as facilitating primary production during the spring freshet period (Thomas et al. 2003), distributing juvenile salmon in the coastal environment, concentrating food resources such as zooplankton, and providing a refuge from predators in the more turbid low salinity plume waters. Effects of changing plume attributes on juvenile salmon are largely speculative. It seems reasonable to hypothesize that a change in plume size would reduce the amount of foraging habitat potentially available to the juvenile salmon. A significant enough reduction in plume size could result in density dependent interactions that diminish growth. Another effect of reducing the size of the plume would be to increase predation on yearlings occupying this habitat since the fish are occupying a smaller space. If one function of the plume is to transport fish offshore, then having the plume closer to shore could affect the ability of fish to migrate towards oceanic feeding grounds.

In summary, flow is a fundamental factor affecting characteristics of salmon and their habitat in the estuary and plume. Large scale effects on flow occur as a result of spatially explicit interactions of short and long term climate cycles (ENSO and PDO, respectively) with the watershed. The generation of electricity, flood control, and irrigation have had significant effects on attributes of flow. These include a reduction in the mean annual flow, reductions in the size of the spring freshets, an almost complete loss of overbank flows, and changes in timing of ecologically important flow events. The hydrological changes, along with floodplain diking, represent a fundamental shift in the physical state of the Columbia River ecosystem. Such changes potentially have significant consequences for both expression of salmonid diversity and productivity of the populations by affecting quality of habitat available, its accessibility and quantity. In particular, because the changes in habitat are most pronounced in shallow water areas, effects on the ESUs and life history strategies (the fry and fingerling strategies) that use these and depend upon these shallow water areas is most significant.

B. Habitat

The estuary contains an extensive and diverse array of habitats that are shaped by the interactions of flow and tides with the land. Although quantitative descriptions of habitat attributes important to salmon are limited in the Columbia River estuary, research in estuarine systems throughout the Pacific Northwest has demonstrated that fish size is one of the major factors defining use of estuarine habitats (Healey 1980, 1982; Levy and Northcote 1981, 1982; Simenstad et al. 1982; Levings et al. 1986; Miller and Sadro 2003). As the size of salmon increases (due either to recruitment of larger individuals or growth in the estuary), juvenile salmon shift to a broader array of habitats. Yearlings (which are often hatchery origin fish) are more prevalent in deeper water habitats within the estuary, located more centrally to mainstem channels whereas smaller (e.g., fry and fingerlings that are typically naturally produced) juvenile salmon use the more peripheral side channel areas associated with the more shallow water habitats (McCabe et al. 1986). Salmon representing most of the endangered ESUs have been found to use these shallow, peripheral habitats of the Columbia River estuary based upon recent genetic analysis (Paul Moran, NOAA Fisheries, personal communication). In addition, spring chinook that express both yearling and subyearling strategies have been identified in the plume environment (Paul Moran, NOAA Fisheries, personal communication).

Although the abundance of juveniles in the estuary fluctuates, recent evidence indicates that juvenile salmon currently use the estuary during the entire year (D. Bottom, NOAA Fisheries, personal communication). This characteristic year long presence is consistent with the historical record. Burke (2001) reconstructed the presence of juveniles from research conducted by Willis Rich in the early 1900s and compared it to more recent data sets. Over the year, juvenile salmon representing different cohorts expressing varying life history strategies were historically using the Columbia River estuary. The diversity of juvenile salmon sizes now present in the estuary and when they are present has changed from these historical conditions (Bottom et al. 2001).

The major anthropogenic factors affecting the amount and location of estuarine habitat are flow alterations and diking. Dikes are built to prevent over-bank flow and are used for flood control and conversion of aquatic to terrestrial land (e.g. for farming). The construction of dikes is not a direct result of the operations of the hydropower system, although dikes must be built to accommodate the timing and magnitude of flows that pass below Bonneville Dam. Because dikes affect the connectivity of the river and floodplain (Tetra Tech 1996), the diked floodplain is higher than the historic floodplain and inundation of floodplain habitats only occurs during times of extremely high river discharge (Kukulka and Jay 2003). Given modern bathymetry and the altered flow regime scenario that we described in the previous section, the critical river discharge level in which significant shallow water habitats become available through floodplain inundation is relatively high. Because the frequency of occurrence of this river discharge is rare, floodplain inundation is uncommon and availability of these shallow water habitats is now more limited than it was historically (Kukulka and Jay 2003).

Several analyses demonstrate the dramatic changes in the amount and location of shallow water habitat (such as emergent marsh and forested wetland habitat) that have occurred (Thomas 1983; Sherwood et al. 1990). Kukulka and Jay (2003) indicated that diking removed nearly 52% of the shallow water flood plain habitat in the tidally influenced freshwater zone of the estuary. Thomas (1983) and Sherwood et al. (1990) calculated that approximately 121.6 km^2 of tidal marshes (77% decline) and swamps (62% decline) that existed prior to 1870 have been lost. In addition, the historic surface area of the estuary has decreased by approximately 20% as a result of diking or filling of tidal marshes and swamps. The largest increase of non-estuarine habitat from 1870 to 1983 was that of developed floodplain habitat. Of the 36,970 total acres of lost estuarine habitat, 64.8% was converted to developed floodplain (Thomas 1983).

The loss of estuarine wetlands has clearly affected the opportunity of salmon to use this type of habitat. Emerging research in the Columbia River and else where demonstrates that these shallow vegetated habitats are important to non-yearling life history strategies, especially fry and fingerlings (D. Bottom, NWFSC, personal communication, Shrefler et al. 1990, 1992; Gray et al. 2002). The degree to which estuary habitat types have been affected by diking is directly proportional to elevation; thus, the highest elevation habitat type (i.e. tidal swamp) has been impacted by diking the most (Thomas 1983).

In addition to the lost opportunity to use shallow water habitats, estuarine wetland loss has altered the magnitude and character of habitat capacity by reducing wetland primary production. Approximately 15,800 mt carbon $year^{-1}$ (84%) of macrodetritus that historically supported estuarine food webs has been eliminated. However, these losses were accompanied by an increase of approximately 31,000 t carbon $year^{-1}$ of microdetritus from upriver sources, originating principally from increased phytoplankton production in the reservoirs behind the mainstem dams (Sherwood et al. 1990). The implications of this shift in detrital sources are unclear. For example, whereas the macrodetrital food web was historically distributed throughout the lower river and estuary, the contemporary microdetrital food web is concentrated within the localized mid-estuary region of the estuarine turbidity maximum (ETM).

In summary, the location and types of habitats present in the Columbia River estuary have been substantially changed from historic conditions. Although the entire estuary has not yet been surveyed, the main changes that have been quantified in the estuary have been a loss of emergent marsh, tidal swamp, and forested wetlands. Shallow water dependent life history strategies (fry and fingerlings) have been most affected by the loss of these vegetated habitat types. Alterations in attributes of flow and diking have caused these changes. Diking is a significant change primarily because it completely isolates habitat from the river and eliminates it from use by juvenile salmon. Further, it has altered estuarine food webs from macrodetrital to microdetrital based, with unknown consequences. Clearly, restoration of shallow water vegetated habitat by removing dikes is a tactic that can benefit those populations that have large numbers of shallow water dependent members.

C. Toxics

Sources of contaminants in the estuary are numerous and include such activities as agriculture, logging, mining, industrial discharges, and stormwater runoff began to degrade water quality in the Columbia Estuary. Currently, the section from Bonneville Dam to the estuary mouth is the most urbanized section of the river, receiving contaminants from over 100 point sources (Fuhrer et al. 1996), as well as urban and agricultural non-point sources. Contaminants may also be transported to estuary from areas of above Bonneville Dam such as the Yakima River (Fuhrer et al. 1996; Rinella et al. 2000), Lake Roosevelt (Bortleson et al. 1994) and other tributaries (Fuhrer 1989; Roy F. Weston Inc. 1998).

Potentially toxic water-soluble contaminants that have been detected in the Columbia River estuary include a wide range of current-use organophosphate pesticides (OPs; e.g., simazine, atrazine, chlorpyrifos, metolachlor, diazinon, and carbaryl) and trace metals (Fuhrer et al. 1996; Hooper et al. 1997). Contaminants that have been documented in Columbia River estuary bed sediments and suspended sediments include trace metals (cadmium, copper, and zinc), dioxins, furans, chlorinated pesticides and other chlorinated compounds (e.g., dieldrin, lindane, chlordane, PCBs, and DDT and its metabolites), and polycylic aromatic hydrocarbons (PAHs) (Fuhrer and Rinella 1983; Fuhrer 1986; Harrison et al. 1995; Fuhrer et al. 1996; Tetra Tech Inc 1996; US Army Corps of Engineers 1998; Roy F. Weston, Inc 1999; McCarthy and Gale 2001).

Exposure to these contaminants in the estuary likely varies by life history type and ESU. Stream type populations (e.g, Snake River sockeye), are not likely to accumulate high body burdens of bioaccumulative, sediment-associated contaminants such as PCBs and DDTs because of their short residence time in the estuary. However, they may be affected by short-term exposure to waterborne contaminants such as OPs and dissolved metals. Ocean-type populations (e.g., lower river chum salmon), are also at risk for exposure to current use pesticides and dissolved metals. At the same time, they are more likely than stream type fish to be affected by bioaccumulative toxicants (DDTs, PCBs) that they may absorb through their diet during estuarine residence. Ocean-type populations may also be more at risk because of their greater use of shallow-water habitats and tendency to rear for longer periods in the estuary. Fine-grained sediments to which toxics adsorb are most likely to be deposited in areas with slower water velocities, including backwater areas in side channels and along the river's margins (Tetra-Tech 1994). These areas are heavily utilized by ocean type fish.

Although data on contaminant concentrations in listed salmon from the Columbia River estuary are limited, available data indicate that bioaccumulative contaminants are present in prey and tissues of juvenile salmon from the Columbia River Estuary. Contaminant concentrations were measured in juvenile fall Chinook salmon from several sites in the Columbia River estuary such as near the confluence of the Columbia and Willamette rivers, near Longview, and at several sites within the Lower Columbia Estuary such as White Island and West Sand Island). The primary contaminants found in whole body samples of Chinook salmon from the estuary were PCBs and DDTs. Average concentrations of PCBs at estuarine sampling sites ranged from 23 to 90 ng/g wet wt), while average DDT concentrations ranged from 32 to 115 ng/g wet wt). In individual fish, DDT levels as high as 270 ng/g wet wt and PCB levels as high as 340 ng/g wet wt were measured. These contaminants were also detected in stomach contents of juvenile fall Chinook salmon from sites within the estuary, indicating they were absorbing some contamination from prey during estuarine residence.

For some contaminants, exposure levels in juvenile salmon from the Columbia River estuary are approaching concentrations that could affect their health and survival. For PCBs, Meador et al. (2002) estimated a critical body residue of 2400 ng/g lipid for protection against 95% of effects ranging from enzyme induction to mortality in a fish with 2% lipid, based on a range of sublethal effects observed in salmonids in peer-reviewed studies conducted by NMFS and other researchers. Mean PCB body burdens in juvenile salmon analyzed by the NWFSC were at or above these thresholds at several sites in the estuary. Of individual fish analyzed from sites within the estuary, ~35% were above the effects threshold. Moreover, in field studies in Puget Sound, at estuarine sites contaminated with PAHs, PCBs, and other OCs also present in the Lower Columbia, juvenile salmon showed immunosuppression, reduced disease resistance, and reduced growth rates, (Arkoosh et al. 1991, 1994, 1998; Varanasi et al. 1993; Casillas et al. 1995a,b, 1998a). Similar results were observed in growth and disease challenge studies with juvenile salmon exposed in the laboratory to PCBs and PAHs (Arkoosh et al. 1994; 1998; 2001; Casillas et al. 1995a,b; 1998a, b).

The likely impact of DDTs on listed salmon are less clear. Most reported effects of are associated with whole body tissue concentrations above those typically found in juvenile salmon captured in the estuary(≥500 ng/g wet wt) (Allison et al. 1962;; Johnson and Pecor 1969; Peterson 1973; Poels et al. 1980; Hose et al. 1989). However, they may represent a hazard to salmon bioaccumulation and bioconcentration in estuarine food webs (Anthony et al. 1993; Henny et al. 2003; Thomas and Anthony 1999, 2003).

In addition to bioaccumulative contaminants, waterborne contaminants such as dissolved metals and current use pesticides may pose a threat to listed salmon. Various OPs such as diazinon, carbofuran, and chlorpyriphos at concentrations of 1-10 ug/L, as well as copper at concentrations of 3-6 ug/L, can disrupt olfactory function in salmon after exposures of as little as 30 minutes (Moore and Waring 1996; Waring and Moore 1997; Scholz et al. 2000; Baldwin et al. 2003). In these studies, affected fish could no longer respond normally to test odorants, so predator avoidance, feeding responses, homing, pheromone-triggered sexual behavior were impaired (Moore and Waring 1996; Waring and Moore 1997; Scholz et al. 2000). Concentrations of diazanon in the 1-10 ug/L range have been reported in NASQAN sampling in the Lower Columbia, and other OPs with similar modes of action (e.g., chlorpyriphos, malathion, aldicarb, carbaryl, carbofuran) are detected even more frequently and at higher concentrations. Dissolved copper concentrations at estuary sites sampled in the USGS NAQAN survey were within this range (Fuhrer et al. 1996), and copper in suspended sediments was substantially higher (45-120 ug/L).

Available data show that environmental concentrations and tissue burdens of several classes of contaminants are within the range where they could potentially affect two important VSP parameters, abundance and population growth rate, in listed stocks. The true magnitude of the effect is uncertain, but a recent modeling study suggests it could be significant for at least some ESUs. Spromberg and Meador (2004) used life cycle models to examine the impacts of low-level toxic effects (10-25% response level for mortality, immune suppression, and growth) on the population dynamics of fall run chinook salmon. The results of indicate that after 20 years of continued reductions at the 10% level, population abundance was severely depressed (up to 2 - 3 times lower than non impacted populations) for several of the endpoints. When the 25% toxicity response was modeled for 20 years, population abundance was between 3 and 20 times lower, depending on the endpoint.

In summary, exposure to chemical contaminants has the potential to affect survival and productivity of both ocean and stream-type stocks in the estuary. Stream-type ESUs are most likely to be affected most by short-term exposure to waterborne contaminants such as current use pesticides and dissolved metals, that may disrupt olfactory function and interfere with associated behaviors, such as capturing prey, avoiding predators, and imprinting and homing. Ocean-type ESUs may also be exposed to these types of contaminants, but will also be affected by persistent, bioaccumulative toxicants such as PCBs and DDTs, which they may absorb during their more extended estuarine residence. Consequently, the impact on ESUs exhibiting the ocean life history type may be higher.

D. Caspian Tern Predation of Juvenile Salmon

The potential for changes in predation on juvenile salmon throughout the Columbia River Basin are significant due to habitat changes and introductions of exotic species. Here, we consider here Caspian Tern predation in the Columbia River estuary. In the early 1990s, a substantial increase in the size of newly established Caspian tern nesting colonies on man-made islands in the Columbia River estuary was observed. Caspian terns arrive in the Columbia River estuary in April and begin nesting at the end of the month (Roby et al. 1998). The timing of courtship, nesting and chick rearing corresponds with the outmigration of many of the salmonid stocks in the basin (Collis et al. 2002). Terns are piscivorous (Harrison 1984), requiring about 220 grams (roughly one-third of their body weight) of fish per day during the nesting season. Salmon and steelhead constitute a major portion of tern diets, particularly when the birds nested on Rice Island. Based on diet analyses conducted in 1997-1998, juvenile salmonids constituted 77.1% of prey items and 76.7% of prey diet mass of Caspian terns nesting on Rice Island (Collis et al. 2002). During the May peak in the smolt out-migration of steelhead, yearling chinook salmon, and coho salmon through the estuary, the diet of Caspian terns on Rice Island was over 80% juvenile salmonids (Collis et al. 2002). In early May, steelhead were the primary salmon eaten while coho salmon were the primary salmonid prey eaten in late May.

Two approaches to evaluate the impact of Caspian tern predation on juvenile salmon were conducted by Good et al. (2003). One approach using bioenergetics modeling, estimated that smolt consumption from 1999 to 2002 ranged from 5.9 to 11.7 million. A second approach used detections of passive integrated transponders (PIT) tags on Caspian tern colonies to estimate salmonid predation rates overall as well as by ESU (Collis et al. 2001a, b; Ryan et al. 2003). Ryan et al. (2003) analyzed PIT tag data from 1998 to 2000 on Rice Island and East Sand Island and determined that steelhead experienced higher predation rates (0.6% to 8.1% on East Sand Island and 1.3% to 9.4% on Rice Island) than chinook salmon (0.2% to 2.0% on East Sand Island and 0.6% to 1.6% on Rice Island). Overall, Caspian terns consumed approximately 6% to 14% of the estimated outmigrating population of juvenile salmonids originating from the Columbia River basin.

In a recent analysis of the impact of Caspian tern predation on salmon recovery, a linear response of predation rate on all salmon to the number of Caspian terns nesting on East Sand Island during the breeding seasons of 1999-2002 was noted. The per capita consumption rate in 1999 (mean = 437.5) was equivalent to that of 2000 (mean = 431.1), even though there was an almost five-fold difference in colony size.

Using matrix life cycle model (e.g., Kaervia et al. 2001), Good et al. (2003) estimated the impact of Caspian tern predation on the population growth rate (λ) of all steelhead and spring Chinook salmon in the basin using predation rate estimates derived from bioenergetics modeling and PIT tag detections. Because of the similarity in the results between the two approaches, we present information only from estimates derived from PIT tag detections, as ESU specific impacts can ultimately be derived.

The predation rate for 20,000 Caspian terns on all steelhead and spring Chinook salmon was estimated using the regression equations generated using PIT tag detections. This number of terns represents the maximum number observed to date on East Sand Island. Reductions in predation rate corresponding to reduced tern population sizes were used to model the potential increase in λ (population growth rate), assuming all steelhead or spring Chinook salmon mortality attributable to terns is not compensated for by mortality due to other sources. The maximum proportional increase in λ corresponding to complete elimination of mortality due to tern predation (i.e. removal of all terns from the estuary) was 1.9% and 0.8% for steelhead and spring Chinook salmon, respectively, using the PIT-tag estimate of predation rate. Predation rates for 20,000 Caspian terns on four of the five ESA-listed steelhead and spring Chinook salmon ESUs were also estimated using linear regression. The maximum proportional increase in λ corresponding to complete elimination of mortality due to tern predation ranged from 1.9% to 4.9% for steelhead ESUs. These results may not be as easy to achieve as they are to calculate. There is no compensatory mortality assumed to occur later in the life cycle, and any reduction in tern predation is assumed to be fully realized. It is also important to recognize that other factors such as ocean conditions may also influence population growth rate to a greater degree than the potential gains that may be realized from reducing predation by one species of avian predator on one island located in the lower estuary of the Columbia River basin.

Overall, it is evident that Caspian tern predation effects primarily salmon and steelhead that exhibit a stream type life history rather than an ocean type life history as they move and utilize the Columbia River estuary. This is primarily because salmon from this life history type move in great numbers at a time when Caspian terns begin nesting (May through June) and have the greatest energetic needs for chick production. Although there are some impacts to juvenile salmon exhibiting an ocean type life history, the impact is less than for the stream type salmonids (Roby et al. 2003). Good et al. (2003) concluded that gains in λ for steelhead ESUs were comparable to gains that could be derived from additional improvements to the FCRPS to increase survival, but much less than can be achieved by harvest modifications. Because steelhead ESUs were most strongly affected by Caspian tern predation, improvements to λ by managing terns were considered to benefit other salmon ESUs in the basin, albeit to a much lesser degree.

In summary, Caspian tern predation has significantly increased due to a recent change in nesting habits of the birds. The main impact of tern predation is on ESUs with stream type life history types, especially steelhead. This is a result of the dominant migratory periods employed by salmonids with a stream type life history. Improvements to λ by managing terns would be expected to benefit these ESUs especially, although benefits to other salmon ESUs in the basin should be evident, albeit to a much lesser degree.

V. IMPACT OF FACTORS ON RELEVANT ESUS AND POTENTIAL FOR IMPROVEMENT IN ESU CONDITION

A summary of scoring for Level 1 and Level 2 questions for each life history type/ESU are provided in Table C-1 with detailed scoring provided in Tables C-2 to C-5. To help guide our scoring, we used the following hypotheses about the effects of specific limiting factors which were developed during our analyses of each factor. Cumulative impacts were not considered in the analysis.

1. Tern predation differentially affects the larger yearling strategies, especially steelhead, more than smaller life history strategies such as fingerling chinook (Ryan et al. 2003). Tern predation is assumed to occur in the estuary zone but primarily in medium and deep water channel habitat rather than shallow water area. Tern predation is assumed to be minimal in the plume.
2. Although mortality of juvenile salmon in the estuary is occurring due to predation by species other than terns, we did not consider these other predators in this analysis.
3. The main effect of flow reductions is to affect amount of shallow water habitat available to fish and opportunity for the fish to use the habitat; the main effect of habitat changes is on distribution, quantity and quality of habitat; the main effect of toxics is on habitat quality (capacity).

4. Any reduction in quality or quantity of shallow water habitat affects smaller juvenile salmonids employing strategies such as fry and fingerlings significantly more than subyearlings and yearlings. From the perspective of ocean type populations in the estuary, changes in the quantity and quality of shallow water habitats most impacts viability of these populations.
5. Subyearling and yearlings primarily use medium and deep channel habitats.
6. Fry and early fingerling life history strategies do not move into the plume, but more likely utilize the surf zone when they exit the estuary proper.
7. Reductions in flow above Bonneville affect the size and shape of the plume. Primarily as a result of flow but also know doubt also a result of physical changes to the estuary (eg., dredging and diking), the shape, behavior, size, and composition of the plume has been changed.
8. Toxics impact the quality of habitat but consequences of toxics can occur downstream of where the burden was acquired. The impact, though, is assumed to be associated with the habitat where the exposure occurs.
9. Flow and habitat changes due to diking in the estuary are interrelated.

For stream-type ESUs (e.g., Snake River spring/summer Chinook salmon and mid Columbia River steelhead), the primary estuarine factors affecting population viability are tern predation and flow (Tables C-1, C-2 and C-4). Tern predation, which is not affected by operation of the Federal Hydropower System, was ranked in the medium category for several reasons. First, tern predation is primarily directed at subyearling and yearling size fish which are the dominant strategies in stream type ESUs such as Snake River steelhead. Second, these larger fish occur in habitats (deeper water channel habitats) where they are most vulnerable to the terns. Third, these larger fish migration at a time when they are most susceptible to tern predation. Fourth, tern predation significantly affects abundance and productivity; scores for these parameters were doubled if we assumed there was an affect. Based upon anecdotal observations of NOAA Fisheries working in the Columbia River plume, we assumed that most tern predation occurred upstream of the river mouth. If significant predation did occur in the plume, then the score for this factor would increase.

Flow changes were also ranked medium for stream type ESUs because of effects of flow changes on plume habitat. These flow related changes in attributes of plume habitat are due in large part to operation of dams in the Columbia River basin (i.e., water withdrawal and flow regulation). The main life history strategies of stream type ESUs are abundant in the plume. We concluded both abundance and productivity are the main VSP parameters affected by flow related changes in the plume. Toxics and habitat were ranked low for stream type ESUs because the main life history strategies associated with this ESU do not occupy the habitat where the main effects occur. There may be some risk associated with short term exposure to waterborne contaminants for these fish.

For ocean type ESUs (e.g, Lower Columbia River fall Chinook salmon), flow and habitat were rated as having a high ability to affect population viability (Tables C-1, C-3 and C-5) for the following reasons. First, their effects are most significant in vegetated shallow water areas such as wetlands and emergent marshes and less pronounced in deeper water areas. The dominant life history strategies using these shallow water areas are fry and fingerling strategies which are the primary life history strategies associated with ocean type ESUs. A major function of these shallow water habitats for small size classes is to support feeding and growth; high growth rates experienced here can help population members avoid some of the high predation mortality that these small fish experience (Simenstad et al. 1982). Second, both factors affect the quantity of habitat and the opportunity to use this habitat. Third, both flow and habitat affect all VSP parameters for ocean type populations. The loss of shallow water habitat and changes in its distribution and quality will reduce the capacity of estuarine habitats to support ocean type populations; this will reduce abundance and productivity of these populations. Because of the loss of shallow water, estuarine dependent strategies (i.e., fry and fingerlings), the number and quality of the spatial and temporal trajectories expressed by these populations will decline.

Changes in shallow water habitat can be a result of both flow related effects of hydropower operation and diking, although the effects of these two factors are interrelated. Clearly, restoration of shallow water vegetated habitat by removing dikes is a tactic that can benefit those populations that have large numbers of shallow water dependent members.

Effects of toxic contamination on ocean type ESUs was rated medium. Both water borne and sediment contaminants can affect these life history strategies in shallow water areas where the dominant life history strategies are most abundant. Based upon our analyses of toxics, we concluded that toxics impact the quality of habitat upstream of the river mouth and that there was not significant affects in the plume. The consequences of the uptake of toxics can occur downstream of where the burden was acquired including if the exposure occurred above Bonneville Dam. However, we assumed the impact was associated with the habitat where the exposure occurred. Tern predation has a low affect on this ESU because terns do not target fry and fingerling strategies (the dominant ones associated with this life history type).

VI. REFERENCES

Allison, D., B.J. Kallman, O.B. Cope, and C. Van Valin. 1963. Insecticides: Effects on cutthroat trout of repeated exposure to DDT. Science 142:958-961.

Anthony, R. G., M. G. Garrett, and C. Schuler. 1993. Environmental contaminants in bald eagles in the Columbia River Estuary. Journal of Wildlife Management 57:10-19.

Arkoosh, M. R., E. Casillas, E. Clemons, P. Huffman, A. N. Kagley, N. Adams, H.R. Sanborn, T. K. Collier, and J. E. Stein. 2001. Increased susceptibility of juvenile chinook salmon (*Oncorhynchus tshawytscha*) to vibriosis after exposure to chlorinated and aromatic compounds found in contaminated urban estuaries. Journal of Aquatic Animal Health 13:257-268.

Arkoosh, M. R., E. Casillas, E. Clemons, B. McCain, and U. Varanasi. 1991. Suppression of immunological memory in juvenile chinook salmon (*Oncorhynchus tshawytscha*) from an urban estuary. Fish & Shellfish Immunology 1:261-277.

Arkoosh, M. R, E. Casillas, P. Huffman, E. Clemons, J. Evered, J.E. Stein, and U. Varanasi. 1998. Increased susceptibility of juvenile chinook salmon (*Oncorhynchus tshawytscha*) from a contaminated estuary to the pathogen *Vibrio anguillarum*. Transactions of the American Fisheries Society. 127:360-374.

Arkoosh, M. R., E. Clemons, M. Myers, and E. Casillas. 1994. Suppression of B-cell mediated immunity in juvenile chinook salmon (*Oncorhynchus tshawytscha*) after exposure to either a polycyclic aromatic hydrocarbon or to polychlorinated biphenyls. Immunopharmacology and Immunotoxicology 16(2):293-314.

Baldwin, D. H., J. F. Sandhal, J. S. Labenia, and N. L. Scholz. 2003. Sublethal effects of copper on coho salmon: Impacts on non-overlapping receptor pathways in the peripheral olfactory nervous system, Environmental Toxicology and Chemistry 22:2266-2274.

Barnes, C., A. C. Duxbury, and B .A. Morse. 1972. Circulation and selected properties of the Columbia River plume at sea, pp 41-80. In: A.T. Pruter and D.L. Alverson editors. The Columbia River Estuary and Adjacent Ocean Waters. University of Washington Press, Seattle.

Bortleson, G. C., S. E. Cox, M. D. Munn, R. J. Schumaker, E. K. Block, L. R. Bucy, and S. B. Cornelis. 1994. Sediment quality assessment of Franklin D. Roosevelt Lake and the upstream reach of the Columbia River, Washington, 1992. U. S. Geological Survey Open-File report 94-315. 130 pp.

Bottom, D. L., and K. K. Jones. 1990. Species composition, distribution and invertebrate prey of fish assemblages in the Columbia River estuary. Progress in Oceanography 25:243-270.

Bottom, D.L., C. A. Simenstad, A. M. Baptista, D. A. Jay, J. Burke, K.K. Jones, E. Casillas, and M. H. Schiewe. 2001. Salmon at River's end: The role of the estuary in the decline and recovery of Columbia River Salmon. Draft Report, National Marine Fisheries Service.

Busby, P. J., T. C. Wainwright, G. J. Bryant, L. J. Lierheimer, R. S. Waples, F. W. Waknitz, and I. V. Lagomarsino. 1996. Status review of west coast steelhead from Washington, Idaho, and California. U.S. Dep. Commerce., NOAA Tech. Memo. NMFS- NWFSC-27, 261 p.

Carl, C.M., and M.C. Healey. 1984. Differences in enzyme frequency and body morphology among three juvenile life history types of chinook salmon (*Oncorhynchus tshawytscha*) in the Nanaimo River, British Columbia. Canadian Journal of Fisheries and Aquatic Sciences 41:1070-1077.

Casillas E., M. R. Arkoosh, E. Clemons, T. Hom, D. Misitano, T. K. Collier, J.E. Stein, and U. Varanasi. 1995a. Chemical contaminant exposure and physiological effects in out-migrant juvenile chinook salmon from urban estuaries of Puget Sound, Washington. In Puget Sound Research 95; Proceedings. Puget Sound Water Quality Authority, PO Box 40900, Olympia, WA 98504. Pp. 657-665.

Casillas E., M. R. Arkoosh, E. Clemons, T. Hom, D. Misitano, T. K. Collier, J. E. Stein, and U. Varanasi. 1995b. Chemical contaminant exposure and physiological effects in out-migrant juvenile chinook salmon from selected urban estuaries of Puget Sound, Washington. In Salmon Ecosystem Restoration: Myth and Reality: Proceedings of the 1994 Northeast Pacific Chinook and Coho Salmon Workshop, M. Keefe (ed.), American Fisheries Society, Oregon Chapter, Corvallis, OR. Pp. 85-102.

Casillas, E., B-T. L. Eberhart, T. K. Collier, M. M. Krahn, and J. E. Stein. 1998a. Exposure of juvenile chinook salmon to chemical contaminants specific to the Hylebos Waterway: Tissue concentrations and biochemical responses. Interpretive Report prepared for NOAA Damage Assessment Center.

Casillas, E., B-T. L. Eberhart, F. C. Sommers, T. K. Collier, M. M. Krahn, and J. E. Stein. 1998b. Effects of Chemical contaminants from the Hylebos Waterway on growth of juvenile chinook salmon. Interpretive Report prepared for NOAA Damage Assessment Center.

Chatters, J.C., and K.A. Hoover. 1986. Changing late Holocene flooding frequencies on the Columbia River, Washington. Quaternary Research 26:309-320.

Chatters, J.C., and K.A. Hoover. 1992. Response of the Columbia River fluvial system to holocene climate change. Quaternary Research 37:42-59.

Collis, K., D. D. Roby, D. E. Lyons, R. M. Suryan, M. Antolos, S. K. Anderson, A. M. Meyers, and M. Hawbecker. 2001a. Caspian Tern Research on the Lower Columbia River, Final 2001 Summary. Columbia Bird Research, www.columbiabirdresearch.org.

Collis, K., D. D. Roby, D. P. Craig, B. A. Ryan, and R. D. Ledgerwood. 2001b. Colonial waterbird predation on juvenile salmonids tagged with passive integrated transponders in theCcolumbia River estuary: vulnerability of different salmonid species, stocks and rearing types. Transactions of the American Fisheries Society 130:385-396.

Collis, K., D. D. Roby, D. P. Craig, S. L. Adamany, J. Y. Adkins, and D. E. Lyons. 2002. Colony size and diet composition of piscivorous waterbirds on the lower Columbia River: Implications for losses of juvenile salmonids to avian predation. Transactions of the American Fisheries Society 131:537-550.

Fuhrer, G. J. 1986. Extractable cadmium, mercury, copper, lead, and zinc in the Lower Columbia River Estuary, Oregon and Washington. U.S. Geological Survey Water Resource Investigations Report 98-4052, 67 pp.

Fuhrer G. J. 1989. Quality of bottom material and elutriates in the Lower Willamette River, Portland Harbor, Oregon. U.S. Geological Survey Water Resources Investigations Report 89-4005, 30 pp.

Fuhrer, G. J., and F. A. Rinella. 1983. Analysis of elutriates, native water, and bottom material in selected rivers and estuaries in Western Oregon and Washington. U.S. Geological Survey open-file report 82-922, 147 pp.

Fuhrer, G. J., D. Q. Tanner, J. L. Morace, S. W. McKenzie, and K. A. Skach. 1996. Water quality of the Lower Columbia River Basin – Analysis of current and historical water quality data through 1994: U.S. Geological Survey Water Resources Investigations Report 95-4294, 157 pp.

Good, T. P., K. Barnas, D. M. Marsh, B. A. Ryan, B. Meyers and E. Casillas. In Preparation. Caspian Tern Predation on Juvenile Salmonid Outmigrants in the Columbia River Estuary. Conservation Biology.

Gray, A. C. A. Simenstad, D. L. Bottom, and T. J. Cornwell. 2002. Contrasting functional performance of juvenile salmon habitat in recovering wetlands of the Salmon River Estuary, Oregon, U.S.A. Restoration Ecology 10:514-526.

Harrison, C. S. 1984. Terns: Family Laridae, pp 146-160. In: Seabirds of Eastern North Pacific and Arctic Waters. D. Haley editor, pp. 146-160. Pacific Search Press, Seattle. WA. 214 pp.

Harrison, H. E., C. W. Anderson, F. A. Rinella, T. M. Gasser, and T. R. Pogue, Jr. 1995. Analytical data form phases I and II of the Willamette River Basin Water Quality study 1992-1994. US Geological Survey Open-File Report 95-373. 176 pp.

Healey, M.C. 1980. Utilization of the Nanaimo River estuary by juvenile chinook salmon, *Oncorhynchus tshawytscha*. Fisheries Bulletin 77:653-668.

Healey, M. C. 1982. Juvenile Pacific salmon in estuaries: the life support system, pp. 315-341. In: V.S. Kennedy, editor. Estuarine Comparisons. Academic Press, New York.

Henny, C. J., J. L. Kaiser, R. A. Grove, V. R. Bentley, and J. E. Elliott. 2003. Biomagnification factors (fish to Osprey eggs from Willamette River, Oregon, U. S. A.) for PCDDs, PCDFs, PCBs and OC pesticides. Environmental Monitoring and Assessment 84:275-315.

Hooper, P. R., D. A. Goolsby, D. A. Rickert, and S. W. McKenzie. 1997. NASQUAN – A program to monitor water quality of the Nation's large rivers. U. S. Geological Survey Fact Sheet FS-055-97, 6 pp.

Johnson, G. E., R. M. Thom, A. H Whiting, G. B. Sutherland, J. X. Southard, B. D. Ebberts, and J. D. Wilcox. 2003. DRAFT. An ecosystem-based restoration plan with emphasis on Sslmonid habitats in the lower Columbia River and Estuary. Pacific Northwest National

Laboratory and Cooperators.

Johnson, O. W., W. S. Grant, R. G. Kope, K. Neely, F. W. Waknitz, and R. S. Waples. 1997. Status review of chum salmon from Washington, Oregon, and California.. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-NWFSC-32, 280 p.

Johnson, H.E. and C. Pecor. 1969. Coho salmon mortality and DDT in Lake Michigan. Transactions of the 34th North American Wildlife Conference.

Kareiva, P., M. Marvier, and M. McClure. 2000. Recovery and management options for spring/summer chinook salmon in the Columbia River Basin Science 290:977-979.

Kukulka, T. and D. A. Jay. 2003. Impacts of Columbia River discharge on salmonid habitat; changes in shallow-water habitat. Journal of Geophysical Research

Levings, C. D., C. D. McAllister, and B. D. Chang. 1986. Differential use of the Campbell River estuary, British Columbia, by wild and hatchery-reared juvenile chinook salmon (*Oncorhynchus tshawytscha*). Canadian Journal of Fisheries and Aquatic Sciences 43:1386-1397.

Levy, D. A. and T.G . Northcote. 1981. The distribution and abundance of juvenile salmon in marsh habitats of the Fraser River estuary. Westwater Research Center University of British Columbia Technical Report 25:117pp.

Levy, D. A. and T.G. Northcote. 1982. Juvenile salmon residency in a marsh area of the Fraser River Estuary. Canadian Journal of Fisheries and Aquatic Sciences 39:270-276.

Mantua, N. J., S. R. Hare, Y. Zhang, J. M Wallace, and R. C. Francis. 1997. A Pacific interdecadal climate oscillation with impacts on salmon production. Bulletin American Meteorological Society 78:1069-1079.

McCabe Jr., G. T., R. L. Emmett, W. D. Muir, and T. H. Blahm. 1986. Utilization of the Columbia River estuary by subyearling chinook salmon. Northwest Science 60:113-124.

McCabe Jr., G. T., W. D. Muir, R. L. Emmett, and J. T. Durkin. 1983. Interrelationships between juvenile salmonids and nonsalmonid fish in the Colombia River estuary. Fishery Bulletin 81:815-826.

McCarthy, K. A. and R. W. Gale. 2001. Evaluation of persistent hydrophobic organic compounds in the Columbia River Basin using semipermeable-membrane devices. Hydrological Processes 15:1271-1283.

McClure, M. M., E. E. Holmes, B. L. Sanderson, and C. E. Jordan. 2003. A large-scale, multi-species status assessment: anadromous salmonids in the Columbia River Basin. Ecological Applications 13:964-989.

McElhaney, P., M. H. Ruckelhaus, M. J. Ford, T. C. Wainwright, and E. P. Bjorkstedt. 2000. Viable salmon populations and the recovery of evolutionary significant units. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-NWFSX-42, Seattle, WA 156 pp

Meador, J. P., T. K. Collier, and J. E. Stein. 2002. Use of tissue and sediment-based threshold concentrations of polychlorinated biphenyls (PCBs) to protect juveniles salmonids listed under the US Endangered Species Act. Aquatic Conservation, Marine and Freshwater Ecosystems 12:493-516.

Miller, B.A. and S. Sadro. 2003. Residence time and seasonal movements of juvenile coho salmon in the ecotone and lower estuary of Winchester Creek, South Slough, Oregon. Transactions of the American Fisheries Society 132:546-559.

Moore, A., and C. P. Waring. 1996. Sublethal effects of the pesticide Diazinon on olfactory function in mature male Atlantic salmon parr. Journal of Fish Biology 48:758–775.

Myers, J. M., R. G. Kope, G. J. Bryant, D. Teel, L. J. Lierheimer, T. C. Wainwright, W. S. Grand, F. W. Waknitz, K. Neely, S. T. Lindley, and R. S. Waples. 1998. Status review of chinook salmon from Washington, Idaho, Oregon, and California. U.S. Department of Commerce, NOAA Technical Memorandum. NMFS-NWFSC-35, 443 pp.

Peterson, R. H. 1973. Temperature selection of Atlantic salmon (*Salmo salar*) and brook trout (*Salvelinus fontinalis*) as influenced by various chlorinated hydrocarbons. Journal of the Fisheries Research Board Canada 30:1091-1100.

Poels, C.L.M., M.A. van Der Gaag, and J.F. J. van de Kerkhoff. 1980. An investigation into the long-term effect of Rhine water on rainbow trout. Water Res. 14:1029-1033.

Rinella, J. F, S. W. McKenzie, K. J. Crawford, W. T. Foreman, G. J. Fuhrer, and J. L. Morace. 2000. Surface-water quality assessment of the Yakima river Basin, Washington—Distribution of pesticides and other organic compounds in water, sediments, and aquatic biota, 1987-91. U.S. Geological Survey Water Supply Paper 2354-B.

Roby, D. D., D. E. Lyons, D. P. Craig, K. Collis, and G. H. Visser. 2003. Quantifying the effects of predators on endangered species using a bioenergetics approach: Caspian terns and juvenile salmonids in the Columbia River estuary. Canadian Journal of Zoology 81: 250-265

Roby, D. D., D. P. Craig, K.Collis, and S.L . Adamany. 1998. Avian predation of juvenile salmonids in the lower Columbia River. 1997 Annual Report, Oregon State University and Columbia River Intertribal Fish Commission, Corvallis and Portland, OR

Roy F., Weston, Inc. 1998. Portland Harbor sediment Investigation Report: Portland, Oregon,, prepared for US EPA, Region 10, contract no. 68-W9-0046.

Ryan, B.A., S. G. Smith, J. M. Butzerin, and J. W. Ferguson. 2003. Relative vulnerability to vulnerability to avian predation of juvenile salmonids tagged with Passive Integrated Transponders in the Columbia River Estuary, 1998-2000. Transactions of the American Fisheries Society 132:275-288.

Scholz N. L., N. K. Truelove, B. L. French, B. A. Berejikian, T. P. Quinn, E. Casillas, and T. K. Collier. 2000. Diazinon disrupts antipredator and homing behaviors in chinook salmon (*Oncorhynchus tshawytscha*). Canadian Journal of Fisheries and Aquatic Sciences 57: 1911-1918.

Sherwood, C.R., D.A. Jay, R. B. Harvey, P. Hamilton, and C.A. Simenstad. 1990. Historical changes in the Columbia River estuary. Progress in Oceanography 25:299-357.

Shreffler, D.K., Simenstad, C.A., and R.M. Thom. 1990. Temporary residence by juvenile salmon of a restored estuarine wetland. Canadian Journal of Fisheries and Aquatic Sciences 47:2079-2084.

Shreffler, D.K., Simenstad, C.A., and R.M. Thom. 1992. Juvenile salmon foraging in a restored estuarine wetland. Estuaries 15:204-213.

Simenstad, C. A. 2000. Commencement Bay aquatic ecosystem assessment. Ecosystem-scale restoration for juvenile salmon recovery. University of Washington, School of Fisheries, Sof-UW-2003. 25pp.

Simenstad, C. A. and J. R. Cordell. 2000. Ecological assessment criteria for restoring anadromous salmon habitat in Pacific Northwest estuaries. Ecological Engineering 15:283-302.

Simenstad, C. A., K. L. Fresh, and E. O. Salo. 1982. The role of Puget Sound and Washington coastal estuaries in the life history of Pacific salmon: an unappreciated function, pp. 343-364. In: V.S. Kennedy, editor. Estuarine Comparisons. Academic Press, New York.

TetraTech. 1994. Lower Columbia River backwater reconnaissance survey, Reconnaissance report TC 9405-01, Redmond, WA. Prepared for Lower Columbia River Bi-State Water Quality Program.

Tetra Tech. 1996. The health of the river 1990-1996. Integrated Technical Report 0253-01. Prepared for the Lower Columbia River Bi-State Water Quality Program.

Thomas, A. C., P. T. Strub, and P. Brickley. 2003. Anomalous satellite-measured chlorophyll concentrations in the Northern California Current in 2001-2002. Geophysical Research Letters 10.1029/2003GL017409.

Thomas, C. M., and R. G. Anthony. 1999. Environmental contaminants in great blue herons (*Ardea herodias*) from the lower Columbia and Willamette Rivers, Oregon and Washington, USA. Environmental Toxicology and Chemistry18: 2804-2816.

Thomas, C. M., and R. G. Anthony. 2003. Environmental contaminants and nesting behavior of great blue herons from the Columbia and Willamette Rivers, Oregon and Washington. Northwest Science 77:320-330.

Thomas, D. W. 1983. Changes in the Columbia Rive estuary habitat types over the past century. Columbia River Estuary Data Development Program. Astoria, OR 51 pp.

Turner, M.G. 1989. Landscape ecosystem: the effect of pattern on process. Annual Review of Ecology and Systematics. 20:171-197.

USACE (United States Army Corps of Engineers). 1998. Dredged Material Evalutaion Framework. Lower Columbia River Management Area. Prepared by the U. S. Army Corps of Engineers, Northwest Division, EPA Region 10, the Oregon Department of Natural Resources, and the Oregon Department of Environmental Quality. November 1998.

Varanasi, U., E. Casillas, M. R. Arkoosh, T. Hom, D. A. Misitano, D. W. Brown, S-L Chan, T. L. Collier, B. B. McCain, and J. E. Stein. 1993. Contaminant exposure and associated biological effects in juvenile chinook salmon (*Oncorhynchus tshawytscha*) from urban and nonurban estuaries of Puget Sound, U.S. Department of Commerce, NOAA Technical Memorandum, NMFS-NWFSC-8, p. 112.

Waring, C. P., and A. P. Moore. 1997. Sublethal effects of a carbamate pesticide on pheromonal mediated endocrine function in mature male Atlantic salmon (*Salmo salar*) parr. Fish Physiology and Biochemistry 17: 203–211.

TABLES

Table C-1. Summary rating table for listed Columbia River Basin ESUs for estuary factors. Ranks were assigned based upon the following ranges: low (0-0.32), medium (0.33-0.66) and high (0.67-1.00).

Life History Type	Stream Type				Ocean Type			
ESUs	Snake River Spring/Summer Chinook Upper Columbia River Chinook Snake River Steelhead Upper Columbia River Steelhead Middle Columbia River Steelhead Lower Columbia River Steelhead Upper Willamette Steelhead Upper Snake River Sockeye Lower Columbia River Coho				CR Chum Salmon Snake River Fall Chinook Upper Willamette Chinook Lower Columbia River Fall Chinook			
Rating Level	Factor				Factor			
	Tern Predation	Toxics	Habitat	Flow	Tern Predation	Toxics	Habitat	Flow
Level 1	6	5	4	7	3	5	8	7
Level 2	6	2	0	3	2	6	6	7
TOTAL SCORE	12	7	4	10	5	11	14	14
TOTAL POSSIBLE	20	28	20	20	20	28	20	20
RATIO	**0.60**	**0.25**	**0.20**	**0.50**	**0.25**	**0.39**	**0.70**	**0.70**
RANK	**Medium**	**Low**	**Low**	**Medium**	**Low**	**Medium**	**High**	**High**

Table C-2. Level 1 ratings of estuary factors for stream-type ESUs (Snake River spring/summer chinook, Upper Columbia River chinook, Snake River Steelhead, Upper Columbia River Steelhead, Middle Columbia River Steelhead, Lower Columbia River steelhead, Upper Willamette Steelhead, Upper Snake River sockeye, and Lower Columbia River coho). An answer to a question of yes equals a 1 other than for the VSP criteria of productivity and abundance which are scored a 2 for yes. An answer of no equals a 0.

Screening Criteria	Factor			
	Tern Predation	Toxics	Habitat	Flow
LEVEL 1- IS THE FACTOR OF CONCERN FOR THE ESU?				
What is the relevance of the factor to the ESU?				
Are there large numbers of fish affected (2x)	2	2		2
Is there a significant affect on productivity (2x)	2	2		2
Is there a significant affect on LH Diversity			1	1
Is there a significant affect on spatial structure			1	1
What is the level of change possible in factor?				
Is there a significant change from historic levels	1	1	1	1
Is the amount of Improvement possible substantial	1		1	
Score	6	5	4	7
Max Possible Score	8	8	8	8

Table C-3. Level 1 ratings of estuary factors for ocean-type ESUs (Columbia River chum salmon, Upper Willamette Chinook, Lower Columbia River fall chinook, and Snake River fall chinook). An answer to a question of yes equals a 1 other than productivity and abundance which are scored a 2 for a yes. An answer of no equals a 0.

Screening Criteria	Factor			
	Tern Predation	Toxics	Habitat	Flow
LEVEL 1- IS THE FACTOR OF CONCERN FOR THE ESU?				
What is the relevance of the factor to the ESU?				
Are there large numbers of fish affected (2x)		2	2	2
Is there a significant affect on productivity (2x)		2	2	2
Is there a significant affect on LH Diversity	1		1	1
Is there a significant affect on spatial structure			1	1
What is the level of change possible in factor?				
Is there a significant change from historic levels	1	1	1	1
Is the amount of Improvement possible substantial	1		1	
Score	3	5	8	7
Max Possible Score	8	8	8	8

Table C-4. Level 2 ratings of estuary factors for stream-type ESUs (Snake River spring/summer chinook, Upper Columbia River chinook, Snake River Steelhead, Upper Columbia River Steelhead, Middle Columbia River Steelhead, Lower Columbia River steelhead, Upper Willamette Steelhead, Upper Snake River sockeye, and Lower Columbia River coho). With the exception of toxics (see footnote), an answer to a question with a yes equals a 1. An answer of no equals a 0.

Screening Criteria	Terns			Toxics[a]			Habitat			Flow		
	SW[b]	DW	Pl	SW	DW	Pl	SW	DW	Pl	SW	DW	Pl
LEVEL 2- SIGNIFICANCE OF FACTOR												
For the dominate LHS, is the relative impact on numbers by habitat type significant?	1	1	1		1							1
For the dominate LHS, does the factor significantly affect habitat--												
1. Quantity												
2. Quality					1							1
3. Opportunity	1	1	1									1
Score	2	2	2	0	2	0	0	0	0	0	0	3
Total Factor Score		6			2			0			3	
Max Possible Score		12			20			12			12	

a- Scores for toxics include a value for sediment and water in estuary (ie, the sw quality score can be a 2) and water in the plume.
b- SW=Shallow water from the river mouth to Bonneville, DW=Deep water from the river mouth to Bonneville, Pl=Plume

Table C-5. Level 2 ratings of estuary factors for ocean-type ESUs (Columbia River chum salmon, Upper Willamette Chinook, Lower Columbia River fall chinook, and Snake River fall chinook). With the exception of toxics (see footnote), an answer to a question with a yes equals a 1. An answer of no equals a 0.

Screening Criteria	Terns			Toxics[a]			Habitat			Flow		
	SW[b]	DW	Pl	SW	DW	Pl	SW	DW	Pl	SW	DW	Pl
LEVEL 2- SIGNIFICANCE OF FACTOR												
For the dominate LHS, is the relative impact on numbers significant by habitat type?	1			2	1		1			1		1
For the dominate LHS, does the factor significantly affect habitat--												
1. Quantity							1		1	1		1
2. Quality	1			2	1		1		1	1		
3. Opportunity							1			1		1
Score	2	0	0	4	2	0	4	0	2	4	0	3
Total Factor Score		2			6			6			7	
Max Possible Score		12			20			12			12	

a- Scores for toxics include a value for both sediment and water in estuary (ie, the sw quality score can be a 2) and water only in the plume.
b- SW=Shallow water from the river mouth to Bonneville, DW=Deep water from the river mouth to Bonneville, Pl=Plume