Role of the Estuary in the Recovery of Columbia River Basin Salmon and Steelhead: An Evaluation of Limiting Factors

Kurt L. Fresh, Edmundo Casillas, Lyndal Johnson, and Daniel L. Bottom

NOAA Technical Memorandum

Northwest Fisheries Science Center
National Marine Fisheries Service
National Oceanographic and Atmospheric Administration
U.S. Department of Commerce
2725 Montlake Blvd E.
Seattle, WA  98112

December 2003
CONTENTS

INTRODUCTION .................................................................................................................... 7

APPROACH .......................................................................................................................... 9
  Broadening Perspectives: Population Viability ............................................................. 9
  Defining Life History Type and Life History Strategy .............................................. 10

THE COLUMBIA RIVER ESTUARY .................................................................................. 13

ROLE OF THE ESTUARY IN VIABILITY OF SALMON POPULATIONS .......... 14
  Production vs. Population Perspectives of Salmon in Estuaries: A Shift in Paradigms .. 15
  Spatial Structure and Life History Diversity: Conceptual Framework ................... 17

USE OF ESTUARINE HABITATS BY COLUMBIA RIVER SALMONIDS .......... 20

FACTORS LIMITING SALMON USE OF THE COLUMBIA RIVER ESTUARY .... 21
  Water Flow..................................................................................................................... 22
  Interior Basin Spring Freshet Magnitude and Timing .............................................. 25
  Historical Changes in Overbank Flow........................................................................ 26
  Habitat............................................................................................................................... 29
  Toxics............................................................................................................................... 36
  Caspian Tern Predation of Juvenile Salmon............................................................... 44

LIMITING FACTORS ANALYSIS SYNTHESIS...................................................... 50

ACKNOWLEDGEMENTS ................................................................................................. 56

REFERENCES .................................................................................................................... 57

TABLES ............................................................................................................................. 81

FIGURES ........................................................................................................................... 84
LIST OF FIGURES

Figure 1. Map of the Columbia River estuary…………………………………84

Figure 2. Historical and contemporary early life history types for one brood-year of chinook salmon in the Columbia River estuary………….85

Figure 3. Comparison of the monthly averaged Columbia River interior sub-basin virgin, adjusted, and observed river-flow…………………..86

Figure 4. Changes in the annual flow cycle of Columbia River flow………..87

Figure 5. Peak freshet day vs. year……………………………………………….88

Figure 6. The incidence of flows above the pre-1900 estimated bankfull flow level and above the present bankfull flow level…………………..89

Figure 7. Monthly average flows at Beaver, present and historical bankfull flow levels, under warm and cold-PDO cycles…………………..90

Figure 8. Relationship between average river flow and the maximum distance offshore juvenile chinook and coho salmon were captured….91

Figure 9. Biomass of Cancer magister megalopae captured in May 2001 and 2002 in the ocean, front and plume habitats…………………..92

Figure 10. Abundance of salmonids captured in the ocean, front and plume habitats………………………………………….93

Figure 11. Habitat opportunity in Cathlamet Bay as a function of river discharge…………………………………………………………..94

Figure 12. The change in availability of shallow water habitat in the tidally influenced region of the Columbia River under unmodified and modified flow conditions…………………………..95

Figure 13. Catch per unit effort for juvenile chinook salmon at several sites in the mouth of the Columbia River estuary…………………..96

Figure 14. Mean fork length of juvenile chinook salmon at several sites in the mouth of the Columbia River estuary………………………….97
Figure 15. Box plots of size of juvenile salmon captured in peripheral habitats and near the main channels of the Columbia River estuary.............................. 98

Figure 16. Proportion of Chinook salmon ESU’s originating from various parts of the basin taken from peripheral habitats of the Columbia River estuary................................. 99

Figure 17. Stock composition of subyearling chinook salmon in the Columbia River plume.......................................................... 100

Figure 18. Temporal presence of naturally and hatchery produced juvenile chinook salmon in Skagit Bay, WA................................. 101

Figure 19. Movement of juvenile Chinook salmon in the Skagit River estuary system ............................................................... 102

Figure 20. The relationship between freshwater wild chinook smolt population size and density of juvenile wild Skagit Chinook in Skagit River delta habitat.................................................. 103

Figure 21. Change in acreage of various habitat types used by salmon in the Columbia River estuary................................................. 104

Figure 22. Daily Shallow-Water habitat (SWH) Area from for virgin and observed river flows with and without dikes......................... 105

Figure 23. Mean concentrations of total PCBs in whole bodies of juvenile fall Chinook salmon...................................................... 106

Figure 24. Mean concentrations of DDTs in whole bodies of juvenile fall Chinook salmon...................................................... 107

Figure 25. Mean concentrations of PCBs and DDTs in stomach contents of juvenile fall Chinook salmon...................................... 108

Figure 26. Mean concentrations of high molecular weight polycyclic aromatic hydrocarbons and metabolites of HAHs in fall Chinook salmon...................................................... 109

Figure 27. The leukocyte primary and secondary in vitro plaque forming cell response per culture (PFC/culture) against an antigen .......... 110

Figure 28. Cumulative mortality of juvenile chinook salmon collected from an urban and non-urban estuary following disease challenge with the marine bacterium Vibrio anguillarum.............. 111
Figure 29. Foraging activity in the presence of olfactory signals of a potential predator in juvenile Chinook salmon exposed to diazinon................................................................. 112

Figure 30. Dose–response curves and threshold determinations for sublethal copper neurotoxicity.......................................................... 113

Figure 31. Estimated predation rates on all steelhead in the Columbia River estuary by Caspian terns....................................................... 114

Figure 32. Per capita smolt consumption by Columbia River estuary Caspian terns on all salmonids..........................................................115
LIST OF TABLES

Table 1. A summary of the characteristics of stream-type and ocean-type salmonids………………………………………………………….. 76

Table 2. Description of life history types and selected attributes associated with Columbia River anadromous salmonid populations based upon historic use of the system………………….. 77

Table 3. Linkage between salmonid ESU, life history type (ocean or stream type), and dominant life history strategies of juvenile salmon in the Columbia River…………….. 78

Table 4. Major types of estuarine habitats and some of their important attributes in the Columbia River below RM 46…………….. 80

Table 5. Linkages between limiting factors associated with the estuary, the expression of life history strategies, life history type (ocean type and stream-type) and ESU……………………………. 81

Table 6. Linkages between limiting factors associated with the plume, the expression of life history strategies, life history type (ocean type and stream-type) and ESU……………………………. 82

Table 7. Overall relative rating of each limiting factor for estuary and plume based upon their potential effect on each VSP parameter……………………………………………………………………. 83
INTRODUCTION

Since 1991, 12 different populations 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. These populations include steelhead, chum, chinook, and sockeye populations that spawn from the upper Snake River Basin to tributaries of the lower river below Bonneville Dam. Every subbasin of the Columbia that is currently accessible to anadromous salmonids contains at least one threatened or endangered population. The Federal Columbia River Power System (FCRPS) has had a variety of well documented impacts on anadromous salmonids in the basin including the loss and degradation of spawning and rearing habitat and increased mortality of upstream and downstream migrating fish during passage at hydroelectric facilities (Independent Science Group - ISG 2000). As a result, efforts to recover these populations at risk have almost exclusively focused on identifying and modifying risk factors directly associated with the large hydroelectric dams constructed throughout the basin.

Increasing attention is now being focused on indirect impacts of the hydropower system and non-hydropower related issues in the decline and recovery of salmonids in the basin. One area that encompasses both of these factors is the condition and availability of habitats in the estuary. The increasing emphasis on the role of the estuary in the recovery of Columbia Basin salmonids as a contributing element in salmon recovery represents a significant departure from previous management efforts in the system. There are several developments which appear to be responsible for this shift. First, legislation passed in 1996 by Congress that amended the Power Act requiring the Northwest Power Planning Council to consider the effect of ocean conditions on fish and wildlife populations when recommending hydropower mitigation projects for the Columbia River Basin. This legislation focused new attention on the estuary, plume and coastal ocean habitats.

Second, life stage risk and sensitivity modeling analyses of Columbia River salmon populations by Kareiva et al. (2000) and McClure et al. (2003) suggest that additional actions beyond passage improvements are needed to recover salmonid populations. Two life stages identified as sensitive to perturbations included the first years spent rearing in the river, estuary and ocean. Kareiva et al. (2000) concluded that maximum potential to contribute to recovery were associated with these life stages.

Third, scientific perspectives of the life history and ecology of anadromous salmonids have shifted in recent years. Previously, habitats and life stages important to salmon were considered in isolation with efforts directed to identify single limiting factors restricting salmon production. We now recognize that marine, estuarine and riverine environments are each components of an extended salmon ecosystem that cannot be treated independently (ISG 2000; Bisbal and McConnaha 1998). Thus, the estuary and Columbia River plume are part of the continuum of landscapes all juvenile and adult anadromous salmonids use that originate from the Columbia River Basin. The estuary and plume are the connection between freshwater and marine habitats and are used by all
life stages to some degree for feeding, refugia from predators, and physiological transition (McCabe et al. 1983, 1986; Bottom and Jones 1990). Moreover, the effects of hydroelectric development and other upriver alternations are not localized but instead are far reaching, directly affecting estuarine and coastal ocean habitats (e.g., availability of essential habitat in the estuary, size of the plume) of salmon. In addition, recent research shows that decadal scale regime shifts in climatic and oceanic conditions can produce long term changes in salmon production across the entire North Pacific Ocean (Francis and Sibley 1991; Beamish and Bouillon 1993; Mantua et al. 1997). Such shifts in the production regime along with natural variability must be taken into account when developing appropriate recovery goals, actions and expectations for Columbia River salmonids.

Finally, our understanding of the functions of habitats in the persistence of salmon populations has evolved in recent years. We now recognize that habitats cannot be valued simply on the basis of their role in producing fish biomass or abundance (Bottom 1997). Instead, diverse, high quality habitats and the expression of life history strategies based upon these habitats are directly linked to salmon population viability (persistence) over long time scales (McElhany et al. 2000). These linkages were explicitly recognized by the ISG for the NWPPC who concluded that estuary/ocean dynamics helps to control salmon productivity and that salmon biodiversity (including the diversity of estuarine life histories) minimizes the effects of fluctuations in ocean and presumably freshwater conditions (ISG 2000). Loss of estuarine habitats implies the loss of estuarine dependent strategies, which has implication for the viability of populations that depend on these strategies.

The challenges of identifying, designing, implementing and evaluating recovery actions in the estuary are significant, in part because we know little about the estuary and the salmon that use the estuary. While ongoing research efforts will significantly upgrade our knowledge base in upcoming years, much of what we now know is conceptual or based on research in other areas such as Puget Sound. Estuary restoration at any scale is a challenge and in the Columbia River estuary it is an especially daunting challenge because of the massive size of this system. Further, the estuary is among the most heavily modified portions of the basin (Thomas 1983) due to the long history of coastal development and the cumulative effects of flow regulation, habitat modification in the estuary and other changes upriver which have altered sediment transport and salinity regimes in the system (Simenstad et al. 1992; Weitkamp 1994). In the last 100 years, these and other changes have decreased the amount of some types of wetland habitats in this region by as much as 70% from historical levels (LCREP 1999). To help guide the development and implementation of estuarine related recovery activities in the Columbia River Basin, this review presents our understanding of the role of the Columbia River estuary in salmon life history and evaluates the role that various limiting factors might play in affecting viability of listed populations in the basin.
APPROACH

Broadening Perspectives: Population Viability

The overall purpose of this document is to evaluate and rank candidate limiting factors in the estuary with respect to the potential of each factor to affect population status or suppress population specific recovery. Rankings will be characterized as having a low, moderate, or high potential to improve population status by changing or improving current habitat conditions. The key element of our approach is to evaluate limiting factors from the perspective of overall population viability status and not simply abundance or survival as has traditionally been done.

To define salmon recovery needs, NOAA Fisheries evaluates Viable Salmon Populations (VSP) based on four performance criteria: abundance, population growth rate, spatial structure, and diversity (McElhany et al. 2000). In general, levels of these criteria in aggregate define extinction risk or persistence of the population. Until now, these attributes have been applied primarily to the riverine environments of salmon, where spawner escapement, smolt production, breeding structure, and stream-habitat quality can be directly measured. These freshwater metrics incompletely describe population status because they do not distinguish the contribution of factors at other life stages to overall population trends. Because the progeny of many populations intermingle after they leave their natal streams, abundance and survival of fish from any particular stream or watershed is difficult to track in the estuary and ocean. In addition, the patterns of estuarine and ocean habitat use vary among source populations, so that the most appropriate recovery strategies may depend on the life-history traits of each population of interest. As salmon begin to disperse and distribute themselves throughout the landscape, it is unrealistic to presume that only abundance based metrics measured at a single point (e.g., the estuary) can adequately represent the role habitats serve in facilitating the ability of salmon to grow, survive, and return to their natal river or stream to reproduce.

To better understand the role of the Columbia River estuary in population viability, we emphasize the role of the estuary in contributing to the VSP attributes of diversity and spatial structure. We focused on these VSP attributes because as described in a subsequent section, our hypothesis is that a primary role of estuaries in salmon life history and ecology is to contribute to the persistence or viability of populations by contributing to the spatial structure and the range of life history strategies to use these places. The focus of this document is thus on defining potential for improvement in population status from the perspective of how limiting factors in the estuary primarily, but not exclusively, affect these two VSP attributes.
Defining Life History Type and Life History Strategy

In order to link limiting factors in the estuary to their potential to improve the viability or status of each listed ESU by changing current habitat conditions, it is necessary to define how each ESU uses the estuary. However, specific empirical information describing estuarine habitat use by salmonid ESU’s is largely absent in the Columbia River Basin, although ongoing research will make it possible for us to better characterize these relationships in the future. We can, however, link estuarine use by salmon ESU’s through expression of accepted life history types and strategies.

To define current and historic estuarine habitat use for each ESU, the life history type of each ESU was identified. The division of salmon populations into two major life history groupings, stream type and ocean type, was originally proposed by Gilbert (1912) to distinguish individual fish that emigrated to the ocean as subyearlings (ocean type) and those that remained in freshwater for at least a year prior to emigration (stream type). The separation of populations into stream and ocean types has generally been used to differentiate chinook salmon populations (Myers et al. 1998). Recent genetic analyses of chinook populations have found that that stream and ocean type populations are distinct evolutionary lineages that can be distinguished genetically (Carl and Healey 1984; Healey 1991; Teel et al 2000; Rasmussen et al. 2003). In recent years, the terms stream and ocean type have been increasingly applied to other species as well. Although stream and ocean type fish can be distinguished from each other by differences in ocean distribution patterns (Healey 1991), the major distinguishing feature used to separate these two life history types is characteristics of the juvenile outmigrants. While each life history type can potentially produce any life history strategy, ocean type populations are generally (but not exclusively) composed of members 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.

Each ESU was classified as either stream type or ocean type according to whether it is comprised primarily of ocean types or stream types of populations using information documented in the species status reviews: chinook salmon (Myers et al. 1998), chum salmon (Johnson et al. 1997), and sockeye and steelhead (Busby et al. 1996). Throughout this document, we use the terms ocean type and stream type to refer to life history type of an ESU (Table 1). We assumed that all populations in aggregate within an ESU fit a general life history type model, even though there will be differences between populations within an ESU in the relative proportion of each life history strategy that is expressed (see below).

Within each life history type, individual members employ a variety of alternative spatial and temporal strategies or approaches to using available habitat. We defined life history strategy as an approach (or a trajectory) to use available habitats, including the estuary. While in the extreme, each member of a population employs its own unique strategy, a number of general strategies can be defined for a population based upon
patterns of habitat use. There is no single unifying and accepted approach to identifying and defining life history strategy (e.g., Reimers 1973; Carl and Healey 1984; E. Beamer, SSC, personal communication), however, in general the strategies that have been employed capture some distinctive features of the life history of the fish. For example, Rich (1919) used scale pattern analyses of fish recovered from the Columbia River estuary to define life history strategies based upon size at estuarine entry, time of entry, and freshwater life history (where and how long the fish reared in freshwater) (summarized in Bottom et al. 2001). Carl and Healey (1984) defined life history strategy based upon how much time the fish spent rearing in various habitats, spawning area, and body morphology.

To define alternate strategies for estuarine use, we used the size at estuarine entry and the time when they arrive in the estuary as defining criteria that can be linked back to ESU. Size at entrance into the estuary can be used to classify life history strategy because there is a linkage between fish size, habitat use, and residence time (Healey 1980, 1982; Levy and Northcote 1981, 1982; Simenstad et al. 1982; Levings et al. 1986; Miller and Sadro 2003). In general, residence time in the estuary decreases with size at estuarine entry (with the exception of pink salmon). Juvenile salmon are generally distributed along a habitat continuum based upon water depth with the depth of the water occupied by the fish increasing as the size of the fish increases (McCabe 1995). Larger fish can result from growth either in estuarine or freshwater habitats.

The time the fish arrive in the estuary was also used to help define life history strategy because, depending upon species and population, arrival timing of salmon in an estuary can vary within a general size class of individuals (Carl and Healey 1984; Bottom et al. 2001). Because available resources and habitats can be different depending upon when a fish arrives in the estuary, arrival timing represents a reasonable way to define how the fish use habitats. The wide range in variability in size at estuarine entry and time of entry that can occur is illustrated by Figure 1 (reproduced from Bottom et al. 2001). For example, under historic conditions, fry (fish < 60mm at estuarine entry) arrived in the estuary nearly year round while yearlings could be present from February to June. The source populations were not identified in these analyses, but it is highly likely that all ESU’s contributed to the pattern of fry appearance throughout the year historically in the Columbia River estuary.

Based upon size and time of estuarine entry, we defined six life history strategies that used the estuary historically (Table 2): 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. Fingerlings rear in the estuary for some period with early fingerlings entering between January and July and late fingerling from August to December. Subyearlings are fish that rear in freshwater, rear little in the estuary, and smolt as they outmigrate during their first year of life. They may be larger than fingerlings entering the estuary and may reside in the estuary for less time...
than fry or fingerling salmon. 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.

All populations can potentially produce all life history strategies. We assumed that all populations in aggregate within a life history type/ESU generally produce a characteristic mix of these strategies when viewed over long time scales, even though there will be differences between populations within an ESU in the relative proportion of each life history strategy that is expressed. Salmon or steelhead ESU’s which express the stream life history types are dominated by the yearling life history strategy. However, from a historical perspective, other life history strategies may have been present as well to provide a way to reduce extinction risk. Similarly, non yearling strategies would be expected to dominate the characteristic approaches employed by ocean type ESUs; yearling strategies would probably be important components of some populations, particularly those ESU’s, which are some distance from the river mouth. Based upon the historical evidence that all life history strategies were evident for longer periods of time throughout the year (Rich 1919) compared to the more restrictive population structure now present in the estuary, we have extrapolated the likely mix of life history strategies historically employed by each life history type (Table 3). The likely mixes of life history strategies for each ESU under current conditions which provides an assessment of the likely reduction in life history expression by each life history type evident today (Table 3).
THE COLUMBIA RIVER ESTUARY

An estuary is generally defined as a semi-enclosed coastal body of water with a free connection to the open ocean in which salt water is diluted with runoff from the land (Pritchard 1967). Estuaries are among the most productive ecosystems on earth, supported by a constant input of organic matter and nutrients from both terrestrial and ocean sources. They are also extremely dynamic environments where the interplay of river and tidal forces continuously modifies chemical and biological gradients, reshapes habitat features, and alternately exposes and inundates shallow tidelands. The biota of estuaries therefore consists primarily of organisms with the physiological or behavioral means to tolerate rapidly changing physical conditions. Among these organisms are juvenile salmon that use estuaries as transitional environments for gradually adapting to salt water and as productive rearing areas where individuals can grow rapidly to large sizes, improving their prospects for surviving at sea.

For the purpose of this review, we defined the Columbia River estuary more broadly to encompass the entire habitat continuum (ecotone) upstream of the river mouth where tidal forces and river flows interact, regardless of the extent of saltwater intrusion. During low river flows, salt-water enters deeper channels only as far as River Mile 30 (RM-30) although tidal influence extends upstream all the way to Bonneville Dam (RM-145). The estuary can be divided into different zones based upon various attributes such as geomorphic features, ecological functions, and physical characteristics and each zone can be further subdivided into different habitat types and features (Figure 2). Johnson et al. (2003) using information gleaned from efforts by Thomas (1983), Graves et al. (1995), USACE (1996), and Garono et al. (2002) divided the estuary into eight zones. Here, we define three zones. First, from approximately RM-45 to RM-145 is a long tidal-freshwater zone (referred to as the tidal river zone) where the river is constrained to a simple deep channel and there is only narrow fringe of intertidal habitat. Second, between Tongue Point (RM-18) and upper Puget Island (RM-45) there is a large estuarine mixing zone (referred to as the estuarine mixing zone) where mean salinities range from 0-15 parts per thousand (in deep channels only). At this point, the estuary widens into a series of complex (based upon vegetation and shoreline geomorphology) islands, forested and emergent wetlands, and low-salinity bays (Grays Bay and Cathlamet Bay). Third, from Tongue Point to the river mouth is a high-energy zone from the river mouth to Tongue Point where the salinity gradient increases to more than 30 parts per thousand at the river entrance (referred to as the lower estuary). A major feature of this zone is the pair of shallow, peripheral bays (Baker Bay and Youngs Bay) with expansive intertidal flats that occur along either side of the lower estuary.

Within each of these three zones is a mix of habitats that the juvenile salmon can potentially occupy. The range, distribution, and diversity of habitats in each zone have been significantly altered by land use practices and hydropower operations (Bottom et al. 2001). Habitats can be classified based upon site scale (e.g. depth, temperature, vegetation type, and substrate type) and landscape scale (e.g., connectivity, shape, and size) attributes. The functions of these habitats for juvenile salmon and steelhead depend
upon how these attributes, in aggregate, affect the accessibility of the habitat to the fish and its quality (Simenstad and Cordell 2000). Table 4 presents a general classification of major habitat types within the estuary below RM 46, after Thomas (1983) and Johnson et al. (2003).

Beyond the semi-enclosed estuary is the final transitional zone, the Columbia River plume, the salmon must occupy before they are fully entrained in oceanic habitats (Figure 2). The Columbia River exerts substantial influence on the physical properties of the northeast Pacific Ocean, accounting for 60 to 90% of the total freshwater inflow into the ocean between San Francisco Bay and the Strait of Juan de Fuca (Fox et al. 1984). During high flows, the plume front is readily visible as a sharp interface between sediment-laden river water and the clear ocean. The river plume 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. Strong density gradients between ocean and plume waters create relatively stable habitat features where organic matter and organisms are concentrated. Recent evidence suggests that this offshore extension of the estuary may serve as another important habitat for outmigrating juvenile salmon.

ROLE OF THE ESTUARY IN VIABILITY OF SALMON POPULATIONS

Our view of the role of the estuary in salmon life history and ecology has evolved as our knowledge of salmon life history and ecology has increased. Early research on salmon biology and ecology focused on the freshwater phase of life, primarily because early salmon researchers were guided by the philosophy that numbers of returning adult salmon were a function of conditions occurring in freshwater habitats (e.g., Neave 1953; Walters et al. 1978). In the 1950’s and 1960’s, researchers began to pay increasing attention to the possibility that non-freshwater factors also had a role in determining numbers of returning adults (e.g., Manzer and Shepard 1962; Gilhousen 1962). The development and analysis of long-term data sets on salmon production suggested freshwater conditions could not by themselves adequately explain variability in numbers of returning adults (e.g., Salo and Bayliff 1958; Hunter 1959; Gilhousen 1962; Parker 1968; Peterman 1978). This stimulated research in the early 1970’s, primarily in Puget Sound, British Columbia, and Alaska, on the estuarine and early marine periods of life of juvenile salmon (e.g., Stober and Salo 1973; Kaczynski et al. 1973; Reimers 1973; Mason 1974; Bailey et al. 1975; Salo et al. 1980; Healey 1979, 1980).

Several studies were instrumental in directing attention at the estuarine and early marine periods of life. Parker’s analyses of pink salmon in a Central British Columbia river system, suggested that there was a period of very intense mortality that occurred during early marine life (Parker 1968). He concluded in an ensuing study that predation was the
primary factor causing this mortality (Parker 1971). In Puget Sound, the work by Kaczynski et al. (1973) on the feeding ecology of small pink and chum fry in littoral areas was critical because it demonstrated that these small fish were closely linked to shallow water habitats during their early marine life in a way that had not been previously appreciated. Studies in the Nanaimo River, B.C., confirmed this close linkage between shoreline areas and chum and pink fry with the discovery that prey items of these fish originated from organic matter produced within this zone (Healey 1979; Sibert et al. 1979). This represented a significant departure of traditional models of marine food webs where organic matter came from pelagic sources. For Oregon coastal and Columbia River coho salmon, Gonsolus (1978) presented evidence suggesting that conditions that the fish encountered soon after ocean entry were having a significant affect on their productivity.

During this period, the Columbia River was largely ignored by researchers. Although estuarine related research has occurred throughout the Pacific Northwest since the late 1960’s, investigations of use of the Columbia River estuary (and plume) by juvenile salmon did not begin until recently. There are a number of reasons for this such as the large size of the system (i.e., it may have been intimidating to study) and philosophical views of the role of the estuary in the life history and ecology of salmon (see below). It is remarkable that despite the importance of Columbia River salmon in the Pacific Northwest and the large size and diversity of this estuarine system, empirical knowledge base about how salmon use this estuary compared to other estuarine systems in the Pacific Northwest is lacking.

Production vs. Population Perspectives of Salmon in Estuaries: A Shift in Paradigms

One reason why estuarine related research on salmon evolved so late is that for decades salmon species were primarily assumed to be regulated by density dependent factors in freshwater (Bottom et al. 2001; Bottom 1997). Based upon this freshwater paradigm, more adults were expected simply as a result of an increase in the number of eggs and fry. Thus, the goal of much of early salmon research was to understand freshwater sources of mortality so that they could be more easily manipulated and more adults produced (Bottom 1997). Estuaries and the ocean were considered unimportant or irrelevant as opposed to a necessary part of the life cycle of the salmon. Such a freshwater centric philosophy was also convenient in that it avoided non-freshwater stages of life that were more difficult if not impossible to control, study, or even conceptualize (Bottom 1997).

This density-dependent, freshwater centric view of salmon life history and ecology was an outgrowth of the perspective that natural resources, such as salmon, were simply agricultural products to be managed for the benefit of all people (Bottom 1997). Such a production perspective defined salmon largely as a crop and the output of this crop was defined as short term changes in numbers of fish harvested or reproducing. Theoretical models of salmon population dynamics such as spawner recruit relationships (e.g., Simon and Larkin 1972) and the concept of maximum sustained yield evolved from the perspective that salmon species were controlled by density dependent factors.
The production view of salmon and their environment has been intertwined with our use of hatchery production. Hatcheries have been used for much of the last 100 years to increase numbers of returning adults to compensate for the fact there has never been enough salmon to go around due to habitat destruction, high demand for salmon, harvest, and expanding human population (Lichatowitch and McIntyre 1987). Hatcheries evolved from a density dependent philosophy that more adults would result in direct proportion to the additional number of eggs that survived (Lichatowitch 1999). They were a direct outgrowth of the production view because they focused on bypassing portions of freshwater life where the most significant sources of mortality were believed to occur. The goal of hatchery technology has always been to control more and more of the fish’s freshwater life in order to bypass as much mortality as possible. It was assumed that hatchery fish were no different than wild fish and so could replace lost production.

The continued failure of hatchery production to maintain or increase salmon returns raised new questions about whether passage through estuaries and the ocean might be critical to determining numbers of returning adults. Estuarine and early marine life rapidly came to be regarded as the “critical period” of high mortality that significantly affected overall survival rates and adult returns (Kaczynski et al. 1973; Ricker 1976; Peterman 1978; Healey 1980; Nickelson 1986; Pearcy 1992). Estuaries were regarded as bottlenecks to salmon productivity and a place where mortality was high (Parker 1968; Bax 1983).

In the 1970’s, the prospects that expansions in hatchery production would be used to meet continued high demand for salmon was a concern to many who worried that there was a limit or carrying capacity to estuarine environments. Simenstad et al. (1978) expressed concern that the carrying capacity of the Hood Canal estuarine environment for pink and chum salmon would be exceeded if enough hatchery fish were released. Bailey et al. (1975) suggested that releases of hatchery pink and chum in Alaska coastal areas would result in fewer adult returns than expected by exceeding carrying capacities of these habitats. Studies were initiated to estimate the quantities or carrying capacity of hatchery fish that could be supported by estuaries (e.g., Reimers et al. 1979) and to find the optimum conditions in the estuary that would maximize production. Some managers questioned if salmon were really estuarine dependent and so could be quickly moved or induced to move quickly through the estuary. This resulted in tests to ascertain if the estuary could be bypassed altogether and so render the whole issue of carrying capacity in this environment moot (Solazzi et al. 1991; Macdonald et al. 1988).

In contrast to the production perspective is the population perspective that was originally articulated in the last century, but has been recently reintroduced (Bottom et al. 2001). The population view of estuaries and other salmon habitats was an outgrowth of studies by Willis Rich on Columbia River salmon (e.g., Rich 1939), evaluating scale patterns of juveniles passing through the Columbia River estuary. He found that some fish were present nearly year round in the estuary and that these fish exhibited a wide variety in the time at which they arrived in the estuary, the amount of time they were spending there, and the size at which they arrived (Rich 1939).
To explain his results, Rich suggested that salmon species were comprised of a set of geographically discrete, self-perpetuating populations. Within each population, a range of behaviors was exhibited that were defined by the particular set of conditions found in the full range of spawning and nursery areas used. He concluded that there was not a unique or singular way for a salmon species or population to use the estuary or any other habitat but instead populations employed a diversity of approaches. Groups of salmon became locally adapted to the conditions they experienced. Rich proposed that use of the estuary or any other habitat the fish were occupying (e.g., when the fish arrived, size at arrival, how long they resided) depended upon which population the fish had come from. His view of the Columbia River estuary was that it was a mixing ground of fish from many different sources. The view that salmon are comprised of populations has become a fundamental tenant of salmon management, biology and research as it evolved into the stock concept of salmon management whereby salmon are managed as stocks or subunits of the species (e.g., Simon and Larkin 1972).

Spatial Structure and Life History Diversity: Conceptual Framework for the Role of the Estuary

We propose that the role of estuaries in salmon life history and ecology is primarily to affect the persistence or viability of populations by contributing to spatial structure of the populations (distribution of salmon in space- ie. habitats) and the life history strategies to use these places (referred to as life history diversity). This premise follows directly from the principle that salmon species are comprised of populations that vary in the spatial and temporal use of habitats. McElhany et al. (2000) defined viability of salmon populations as the risk of a population’s extinction over long time scales which was determined by four characteristics: abundance, productivity, spatial structure, and diversity. Abundance (numbers of fish) and productivity (rate of growth of the population) are straightforward concepts that suggest that populations that have lots of members and have a positive population growth rate are more likely to persist. Estuarine habitats clearly contribute to the abundance and productivity of salmon populations (e.g., MacDonald et al. 1988). For example, Reimers (1973) demonstrated that for the one brood year of chinook salmon that he studied in the Sixes River, most adult returns originated from fish that made the most extensive use of the estuary. Magnusson and Hilborn (2003) similarly concluded that survival to adult return of hatchery chinook salmon populations in coastal environments was directly correlated with the condition of the estuary.

How to include spatial structure and diversity and couple it to the role of the estuary in salmon recovery is a critical question. The spatial structure of a salmon population refers to the distribution of members in space and can be measured as the distribution, diversity and quality of habitats used by salmon. In general, salmon occupy a diverse array of habitats over their entire life cycles and have a complex geographic structure whereby members are widely distributed in the habitats they use over space and time (Bottom et al. 2001). For example, one year class of a chinook salmon population can occupy habitats ranging from the Gulf of Alaska to headwaters of spawning grounds (Healey 1991). Levin (1960) proposed principles of metapopulation dynamics to describe how groups of
populations (i.e., species) interacted with habitat; these principles have attracted considerable interest in salmonid conservation (Rieman and Dunham 2000; Issak et al. 2003). These same metapopulation principles can also be applied at a population scale and used to describe the relationships between the spatial geometry of habitats and the dynamics and long-term persistence of a population. According to metapopulation dynamics, persistence of a population in a variable environment will depend in part upon the spatial geometry of suitable habitat, including numbers, qualities, and quantities of habitat patches occupied, patterns in the use of patches, when patches are occupied, and the ability of population members to colonize and use habitat patches. At any one time, there are a wide variety of habitats occupied by members of a population, although not all suitable habitats that are available will be occupied.

Life history diversity consists of the variability in strategies or the approaches to how fish occupy and use habitats and represents the ways salmon species have evolved to cope with the environmental variability they experience (Healey 1991; Healey and Prince 1995). Such diversity is reflected in variability in a wide variety of traits including body size, fecundity, timing of life history events, location of spawning, residence time in various habitats, size at age, age at maturity, ocean distribution patterns and physiological characteristics (Healey and Heard 1984; Tallman and Healey 1991; Taylor 1991; NRC 1996; Beckman et al. 2003; Miller and Sadro 2003; Ramstad et al. 2003). In spatially and temporally varying environments, diversity allows a species to use a wide array of spawning and rearing places and so helps species and populations persist.

Diversity in life history approaches can occur both between and within populations. Between populations, diversity reflects the local adaptations that different salmon populations have evolved to cope with the specific conditions that they experience (Beachum and Murray 1987; Taylor 1990; Roni and Quinn 1995; Quinn and Unwin 1993). Local adaptations are defined as inherited traits that increase the fitness of populations that result from selective pressures. Because of differences in spawning, rearing, and migratory environments, populations become genetically and phenotypically distinct (Burger et al. 1985; Tallman and Healey 1991; Quinn et al. 2000; Hodgson and Quinn 2002; Ramstad et al. 2003). Different portions of a watershed often produce different life history types (Burger et al. 1985; Bottom et al. 2001) because these areas can be environmentally diverse enough that they produce local adaptations; further, the tendency of most salmon to home to their natal spawning grounds can mean that these adaptations produce divergence in genotypes over time. In general, the types of environmental differences that allow populations to differentiate must be “predictable” in the sense that they are in general consistently different across some spatial or temporal scale- e.g., difference in incubation water temperature.

Within populations, diversity represents a strategy to spread risks spatially and temporally in the face of large amounts of environmental variability. Healey (1991) termed this type of diversity tactical in nature. While the factors that produce diversity within a population are unclear, it is likely an interaction of the genotype of the animals, where the fish emerged from the gravel, incubation environment, environmental conditions during spawning and early freshwater residence, extreme environmental
events, micro habitat, and biological interactions (e.g., Taylor 1990; Healey and Prince 1995). A major factor affecting life history types present within a population is the spatial structure of habitats that can potentially be used by the salmon (NRC 1996). An important point here is that if the habitats do not exist because of either natural or anthropogenic factors, then population members cannot use them and potentially these life history types can be eliminated from the population. In order for a population to use diverse habitats requires that the right life history types exist and the existence of the right history types depends on existence of the appropriate habitats.

One way to conceptualize diversity is as a set of alternative life history pathways or trajectories (Wissmar and Simenstad 1998) that individual members of a salmon population can adopt. Each pathway or life history trajectory varies in how the fish spatially and temporally use the habitats that are available to them. The number and success (i.e., survival) of fish using different trajectories varies and depends upon processes operating over large scales. During any particular time period (e.g., ocean regime), there are trajectories that work best for a particular population while other trajectories will produce large number of recruits when conditions change. The number and quality of these trajectories provides a measure of the diversity of a population and their ability to persist over time. Clearly, estuaries are part of some of the alternate trajectories available for the fish to use.

The complex geographic distribution of populations and the alternate approaches to completing life cycles are not unique to salmon or anadromous species (e.g., Roughgarden et al. 1988; Sinclair 1988; Secor 1992; Able et al. 2003). Within the Alagnak River, Alaska, Meka et al. (2003) found that rainbow trout exhibited three life history patterns based upon their migratory movements within this river system. A similar diversity in movements within Yellowstone cutthroat trout was described by Gresswell et al. (1994). Although little genetic differentiation existed, they concluded that the range in variation in life history strategies within the cutthroat trout was an adequate basis for providing protection to each life history type. Curry et al. (2002) found a range of tactics related to the use of freshwater and estuarine habitats within one riverine brook trout population. Recent research has revealed that striped bass exhibit a variety of life history approaches that can vary within and between populations in use of freshwater and estuarine landscapes (Secor 1992; Secor and Piccoli 1996). Many populations of marine fish also exhibit complex approaches to how they distribute themselves in space and time that are similar to the tactics exhibited by anadromous species (e.g., Roughgarden et al. 1988; Able et al 2003).

In summary, a major role of estuaries in salmon population dynamics is to contribute to the persistence of salmon populations by contributing to the structural complexity of habitats available to the fish and the range of life history diversity that can be expressed by the population. Conservation of life history diversity and spatial structure of habitats is an emerging paradigm in recovery and management of both Pacific Salmon (e.g., McElhaney et al. 2000; Issak et al. 2003) and other fish species (e.g., Gresswell et al. 1994). Thus, we propose that a primary goal of salmon recovery actions in the Columbia River estuary should be to restore these ecosystems in such a way that it increases the spatial structure and diversity of target populations.
USE OF ESTUARINE HABITATS BY COLUMBIA RIVER SALMONIDS

As mentioned previously, there have been few studies of habitat use of the Columbia River estuary by wild juvenile salmon and steelhead. Much of the estuary, especially the portion from Bonneville Dam to Puget Island, has not been studied at all. Since 2001, work has been conducted by NOAA Fisheries in the middle and lower estuary that will significantly upgrade our knowledge about how juvenile salmon and specific ESU’s use the estuary. One factor that needs to be considered in any analysis of estuarine use by wild salmon and steelhead is the occurrence of hatchery fish. Because our ability to separate wild and hatchery fish captured in the estuary has been limited, and remains so even at present, many of the spatial and temporal patterns observed in historical data sets may apply to hatchery fish rather than wild fish (e.g., Dawley et al. 1985, 1986).

The only study that provides enough information to distinguish use of different estuarine habitats was by CREDDP (Columbia River Estuary Data Development Program). There are several limitations of this research. First, the work only occurred in the lower estuary. Second, because CREDDP infrequently sampled shallow water habitats off the main channel, potential use of these areas is largely unknown. In general, McCabe et al. (1986) found that subyearling chinook in shallow intertidal habitats of the Columbia River were smaller than subyearlings captured in deeper pelagic areas. Larger, yearling migrants spend little time in shallow estuarine habitats and more time in deeper channel sties (Bottom et al. 1984; McCabe et al. 1986).

Most of what is known about juvenile salmon use of the estuary concerns timing of fish passage through the estuary and was derived from seining studies conducted to recapture coded wire tagged (CWT) fish below Bonneville Dam. In the late 1970’s and early 1980’s, subyearling chinook salmon (these are all non-yearling fish combined) were present year round (Bottom et al. 1984; Dawley et al. 1986; McCabe et al. 1986). While in many of the years studied, overall peak abundance occurred from May to September, there were years when a bimodal distribution was observed (Dawley et al. 1985; 1986). There was also evidence that there were specific patterns in seasonal timing that particular populations exhibited that deviated from the general population- e.g., Lewis River (Dawley et al. 1985). Peak catches at Jones Beach, where much of the “estuary” timing work has been conducted, often were highly correlated with the timing of hatchery releases.

Although knowledge of estuarine habitat use in the Columbia River basin is limited, the information that does exist in combination with studies in other estuaries of the Northwest can be used to provide insight into how juvenile salmonids may use habitats in this large estuary. Estuarine research has demonstrated that juvenile salmon are generally distributed along a habitat continuum based upon water depth (Healey 1980; Levy and Northcote 1982; Simenstad et al. 1982; Levings et al. 1986; McCabe et
al. 1986; Bottom et al. 1984; Miller and Sadro 2003). As fish size increases, depth of the water used by the fish increases; fish size can change as a result of growth that occurs in the estuary, growth in freshwater, or some combination of rearing in the two environments. Based upon this size based model, the smallest juvenile salmon in the estuary (fry and fingerlings) will be primarily associated with the shallowest, peripheral, wetland type of habitat while the larger subyearlings and yearlings will be found in deeper pelagic areas. Coincident with the fish size/depth relationship, there is a general pattern for smaller salmon to spend longer in the estuary and for the larger, yearling migrants to spend less time.

**FACTORS LIMITING SALMON USE OF THE COLUMBIA RIVER ESTUARY**

To facilitate recovery of endangered salmon stocks in the Columbia River basin, it is important to identify factors that currently act to suppress their increased viability. If the factors are appropriately and correctly pinpointed, it is logical, although perhaps somewhat simplistic, to conclude that ‘fixing’ the factors (i.e., reducing their affect) should improve the recovery potential of targeted populations. Further, incorporating the influence of a factors’ impact on recovery of salmon populations should improve policy and management decisions. Although identifying single factor solutions to salmon survival problems in isolation has not historically been effective, we suggest that identification of limiting factors represents the first step that needs to be incorporated into a landscape scale assessment of strategies to improve the recovery potential for endangered salmon populations. With respect to the Columbia River estuary, this pertains to factors that directly affect survival and affect diversity, quality and quantity of habitat, which is consistent with the intent outlined in this report. Estuarine related factors that can potentially affect population viability include climate and climate change, water flow, access to and quality of habitats, sediment, salinity, temperature, toxics, predators (e.g. terns, cormorants, pike minnow), and hatchery and harvest practices.

Although it would be useful to evaluate the role of each of these factors, there are some constraints that limit the opportunity to critically consider each factor identified above. As a result, we have only considered a subset of the above factors in this review. The criteria used to consider the role of limiting factors affecting use of estuarine habitat included our ability to access empirical evidence acknowledging that the factor under review has changed significantly in recent times as a result of either natural or anthropogenic modifications. An additional criterion that was used was the potential contribution of hydropower operations by the FCRPS in affecting the diversity and quality of habitats. To this end, factors that satisfy these criteria and will be included in this review are an evaluation of the role of water flow, availability of salmon specific habitats, toxics affecting the quality of habitats, and predation by Caspian terns.

The logical construct employed in the evaluation is to ascertain the role each of the factors contributed to affecting estuarine habitat or mortality of juvenile salmon in the estuary during both the historical period and under current conditions. For the evaluation
to be relevant to recovery of salmon, a significant change had to be evident. For the purpose of this review, the change has to be significant and empirically based. Using our knowledge of changes throughout the Columbia River Basin included in this analysis, we draw inferences about what we believe the historic and current contribution of each factor included in the analysis with each life history strategy employed by salmon and steelhead. Recall that each strategy employed by salmon is subsequently related to each ESU based upon their classification as either ocean or stream type fish (Tables 1-2). We then conclude how the cumulative effect of all changes the fish would have experienced (loss of freshwater rearing areas, impacts to the estuary, harvest and so on) would have changed distribution of life history strategies to current conditions (Table 3). We do not wish to imply that estuary changes are the major affect on all ESU’s. Assessing the relative role of different limiting factors both within and outside the estuary will require additional analyses and is outside the scope of this review.

**Water Flow**

Water is the very essence and foundation for salmon use of any area. Water clearly allows the connection between the terrestrial oriented land mass (the freshwater zone) to the submerged land masses dominating the globe (the marine zone), fulfilling a major role of the estuary, a migratory corridor connecting these two principal zones. The freshwater, estuary, and marine zone represents the full landscape spectrum salmon occupy. However, more importantly, water interacting with land forms the habitat that salmon occupy. Variation in amount of water in relation to where it interacts with land provides the diversity of habitats that salmon utilize to complete their life cycle.

The estuarine features to which salmon life histories have adapted are largely the result of water associated riverine and tidal processes that transport various materials such as sediments, biota; establish salinity and temperature gradients; and regulate water levels and velocities. The highly productive nature of “pulsed” estuarine systems is a direct result of this dynamic interplay between river and tide. For this reason, the role of water flow in creating and shaping estuarine habitat in the Columbia River is clearly an important attribute to consider in salmon recovery in the basin. The shaping of estuarine habitats is, however, not controlled entirely within the estuary, but is determined by regional and basin-wide variations in climate affecting hydrology and ocean conditions. Because these “external” factors establish the physical template for the entire estuary, they also directly or indirectly affect each of the major attributes of salmonid performance, availability of estuarine habitat (habitat opportunity), the quality of estuarine habitat (habitat capacity), and salmon population structure and life history.

We start with the terrestrial backdrop influencing the delivery of water to the estuarine zone, i.e. the watershed. In general terms, the Columbia River has the largest average flow (~7,300 m$^3$s$^{-1}$) of any river on the Pacific coast of North America (Church and McLean 1992, Jay and Naik 2000). The Columbia River contributes some 60% (winter) to 90% (summer) of the total freshwater input into the Pacific Ocean between San Francisco and the Strait of Juan de Fuca and strongly affects regional seawater
properties of the Northeast Pacific Ocean (Barnes et al. 1972). The major geologic feature affecting flows through the basin is the Cascade mountain range. The Cascades of Oregon and Washington divide the Columbia River drainage basin into interior and western sub-basins. The moist and relatively warm western sub-basin contains only \( \sim 8\% \) of the total surface area of the 660,480-km\(^2\) basin, but contributes almost one-quarter of the total river flow (Orem 1968). Most of the western sub-basin is at too low an elevation to accumulate a large seasonal snow pack. Thus, the highest flows are observed 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. Much of the interior sub-basin is relatively arid, but its Canadian component experiences heavy winter snowfall and plays a major role in spring freshet flows.

Climate is another element that clearly affects the amount of water delivered to the estuary and the connection between the freshwater and marine environments. Climate-induced variations in Columbia River flow occur on time scales from months to centuries (Chatters and Hoover 1986, 1992). A singular measure of climate shown to relate to salmon survival is the Pacific Decadal Oscillation, commonly know as the PDO (Mantua et al.1997). The PDO has been shown to cycle in approximately 30-year time scales, alternating between a cold and warm phase. During the cold phase, more rainfall is typical in the Pacific Northwest, whereas in the warm phase, less rainfall occurs. The PDO primarily reflects sea surface water temperatures in the North Pacific Ocean and is associated with the positioning of the Aleutian Low Pressure Zone over the North Pacific (Beamish et al. ???). The cold phase of the PDO (e.g., the 1945-1976 period) was regarded as benefiting salmonid production in the Pacific Northwest while being less favorable for salmon originating in northern British Columbia and Alaska. The opposite circumstance prevails during the warm phase, characteristic of the recent period between 1977 to about 1998, when listing of salmon in the basin occurred. Another cold, wet phase seems to have commenced about 1998 (Peterson and Schwing 2003). These PDO-related fluctuations in salmonid survival have been linked to the degree of density stratification of the coastal ocean (Gargett 1997), but they are also likely influenced by conditions within the river and estuary (e.g., salinity, turbidity, and river flow itself).

Another climate related feature known to influence weather and conditions in the Pacific Northwest is the phenomena associated with the El Niño-Southern Oscillation (ENSO; typically 3-7 years in duration) index (Redmond and Koch 1991). ENSO cycles are shorter in duration compared to the PDO cycle, typically amplifying conditions associated with the cold or warm phase of the PDO. El Niño winters in the Pacific Northwest often bring high sea level, warm air temperature, low precipitation, low snow-pack, and weak subsequent spring freshet flows (Kathy and Dracup 1993, Dracup and Kathya 1994). La Niña winters (the contrast to El Niño) typically exhibit an opposing climate and hydrological response. As a consequence, the annual average flows of the Columbia and the Willamette Rivers during years with a strong El Niño winter are 91 and 92\% of the long-term annual average, while in case of strong La Niña winters, they are 110 and 111\%, respectively. Considering PDO cycles alone, average annual Columbia River flows at The Dalles and Willamette River flows, respectively, were 109 and 102\%
of the long-term average in the 1890-1921 cold phase, 86 and 87% of average in the 1922-1944 warm phase, 102 and 110% of average in the 1945-1976 cold phase, and 88 and 88% of average in the 1977-1995 warm phase (Jay 2001).

El Niño effects are intensified during a warm-PDO phase, while those of La Niña are enhanced during a cold-PDO phase (Gershunov et al. 1999). The net effect is that during an El Niño/warm-PDO combination, respective average annual Columbia River flows at The Dalles and Willamette River flows are 85 and 81% of the long-term annual average, while in case of a La Niña/cold-PDO combination, they are 111 and 119%, respectively (Figure 3a). (These differences are significant at the 95% confidence limit.) Conversely, El Niño effects are suppressed during the cold-PDO phase as are those of La Niña during the warm-PDO phase (Jay 2001).

The Columbia Basin's climate response is conditioned by its position between 41°30' and 54°40'N lat., within a latitudinal band of strong response to the ENSO cycle and to the PDO (Mantua et al. 1997). While the flow per unit area is much larger in the western than in the interior sub-basin, there are only modest variations across the basin in response to ENSO or PDO forcing. Still, the relatively large north-south extent of the basin brings about important differences in flow seasonality--the incidence of winter floods and timing of spring snowmelt--even within the interior sub-basin. Thus, climate introduces a wide degree of variation in water distributed throughout the basin and watershed, obviously affecting water flow. This is critical in that natural variations in Columbia River flows associated with El Nino and the PDO affect habitat conditions in the estuary. This helps determine what areas are wetted and potentially accessible to juvenile salmon, changes estuarine salinity gradients, influences sediment transport processes, and alters the distributions of marine and freshwater species with which salmon interact. It is against this backdrop of varying climatic conditions that we consider the role of water flow and its effect on habitat important to salmon in the Columbia River estuary.

Changes in annual average flow and when 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) provided a detailed estimate of changes in flows in the Columbia River for the past 100 years, thereby providing an accounting of flow conditions during the historical and current period. With respect to overall water delivery through the estuary, Jay concluded that there has been approximately a 16.4% reduction in flow during this period. Moreover, he made an assessment of the contribution of climate and human perturbation on the observed flow and concluded that approximately half of the change was due to climate (less rainfall) and half to human activities (water withdrawal for irrigation). In addition, a small percentage of the decrease was assigned to a combination of uncertainty (error) and evaporation from the impoundment of water in reservoirs due to increased surface area in the basin. Jay’s conclusions were based upon the record of observed flows for the past 100 years and an estimated adjusted flow provided by the USGS to account for reservoir manipulations; this was used estimate the virgin river flow (flow unadulterated due to hydropower operations and irrigation removal) for the past 100 years (Figure 3).
The above changes in annual average flow are only a small part of the total hydrological changes in the Columbia River basin. Seasonal changes, particularly those in spring freshet timing and magnitude, have been much larger than those 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, and maintain favorable water temperatures during the spring and early summer. Organic matter supplied by the river during the freshet season is also a major factor maintaining a detritus-based food web, centered in the estuarine turbidity maximum (ETM).

Very large freshets before modern flow regulation (i.e., before ~1970) lasted 30 to 60 days, with the sharpness of the peak largely governed by the relative timing of snowmelt throughout the basin. Flows in the Columbia River interior sub-basin (the flow measured at The Dalles) are primarily driven by spring snowmelt, although there are rain-on-snow freshets in some winters. Jay (2001) provided some critical analyses that documented the magnitude and timing of the freshet and subsequent change. Before 1900, the highest flows typically occurred during May-July. Monthly Columbia River virgin flows at The Dalles were 11,480 m$^3$s$^{-1}$ (for May), 16,760 m$^3$s$^{-1}$ (for June), and 12,600 m$^3$s$^{-1}$ (for July) during 1879-1899. The corresponding figures for 1945-1999 were 13,300 m$^3$s$^{-1}$, 15,840 m$^3$s$^{-1}$, and 9,420 m$^3$s$^{-1}$; these values represent changes of +15.9, -9.5, and -25.2%, respectively.

These flow changes were amplified throughout the basin leading Jay (2001) to conclude that flow regulation decreased spring freshet magnitude and increased flows during the rest of the year through winter draw-down of reservoirs, filling of the reservoirs during the freshet, and de-synchronization of flow peaks throughout the basin (Figure 4). More specifically, Jay apportioned the timing and magnitude of the freshet change to climate change, water withdrawal and flow regulation. Jay found the flow decrease in the freshet period resulting from climate was 5.6%. Furthermore, he noted that the January-July virgin flow average for 1879-1899 was 8,050 m$^3$s$^{-1}$, while for 1945-1989 it was 7,850 m$^3$s$^{-1}$, a decrease of only 2.5%. Thus, most of the loss of freshet flow represents flow that now occurs during winter, early spring, or late summer and fall. Similarly, the present decrease in freshet season 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. Finally, the present freshet season flow decrease due to flow regulation was overall estimated to be 33.1% (a reduction of 31.6% for May, 32.4% for June, and 19.8% for July, respectively. Flow regulation is clearly the source of the largest reduction in spring flow. The total reduction in freshet season flow (May-July) mean flow due to climate change, irrigation depletion, and flow regulation is 5,870 m$^3$s$^{-1}$ or 43% of the virgin flow for this period.

The timing of the maximum spring freshet flow has also changed (Figure 5). Maximum daily spring freshet flow now typically occurs at about water-year Day 242 (29 May), whereas maximum flow occurred in the 19th century at about water-year Day 256 (12 June), a change of about two weeks. In terms of the phase of the annual flow fluctuation, the freshet is about a month earlier. Part of this change is due to climate
warming, but a component is also due to pre-release of water for flood control before the spring freshet. Irrigation withdrawal usually peaks in June, which tends to further curtail the freshet.

Another feature of water flow significant to salmon is the occurrence of overbank flows. The historic bankfull flow level is estimated at about 18,000 m$^3$s$^{-1}$ for the main stem below Vancouver (Jay 2001). Modern bankfull level is set by the standard project flood level of ~24,000 m$^3$s$^{-1}$ for the lower river. According to Jay (2001) some overbank flow occurred in many years before 1900, both in winter and in spring (Figure 6) whereas substantial overbank flow (above 24,000 m$^3$s$^{-1}$) is now rare, with significant events occurring only five times during the last half century. Historical bankfull levels of 18,000 m$^3$s$^{-1}$ are rarely exceeded due to effects of flood control measures and irrigation depletion. The season when overbank flow typically occurs has also shifted from spring to winter, because western sub-basin winter floods (not interior sub-basin spring freshets) are now the major source of such flows (Jay 2001). Climate was found to be a secondary factor in the incidence of overbank flow (Jay 2001). Overbank flow events were clearly more common during the cold-PDO phase (1945-1977) than during the preceding warm-PDO phase (1921-1944), even though the degree of flow regulation and irrigation depletion grew over time (Figure 7). Nevertheless, Jay concluded flood protection, diking, flow regulation, and water withdrawal largely eliminated climate influence on overbank flow. Overbank flow was found to be rare during the more recent cold-PDO phases, and was totally absent during the last PDO warm phase (1977-1995).

The effect of flow changes is not restricted to the area traditionally considered the estuary in the Columbia River system. NOAA Fisheries has been investigating the role of the Columbia River plume as habitat important for salmon and steelhead since 1998. The primary issue being addressed is whether anthropogenic modification of the plume habitat has affected recovery of salmon populations from the Columbia River basin. Incumbent on this assessment is development of empirical evidence that the plume has a role that influences how juveniles make the transition from a freshwater to marine environment. The evidence to date suggests the plume serves salmon in multiple ways. For example, the plume appears to facilitate primary production during the spring freshet period. During low flow years, such as observed in 2001, the amount of chlorophyll evident off of the Oregon and Washington coast affiliated with the plume, as characterized with satellite observations by SeaWifs, was much lower than observed when more normal flows occurred, such as observed in 1999 (Thomas et al. 2003).

The plume also serves to distribute juvenile salmon in the coastal environment. In May and June when flows are higher, juveniles are found further offshore, in the low saline waters they appear to prefer, than when flows are lower (Figure 8). During the years when less flow out of the Columbia River is evident during the freshet period, salmon are more localized around the mouth of the Columbia River. Pearcy (1992) hypothesized that one function of the plume was to distribute juvenile salmon offshore, away from predation pressure closer to the shoreline. Our findings are consistent with this proposed role. Another feature of the interaction of the freshwater and saltwater characteristic of the plume is frontal features which are thought to concentrate food.
resources important for juvenile salmon. Findings to date support this function. Zooplankton biomass is in fact highly associated with frontal features at the plume margins (Figure 9) and less so either in the plume or oceanic zones. Our supposition that juvenile salmon would preferentially utilize frontal features was, however, not validated. Juvenile salmon were not higher in abundance exclusively around frontal features. We did conclude that smaller juvenile salmon showed a significant preference for the plume and front habitats as compared to the more marine, oceanic habitats (Figure 10). NOAA Fisheries is developing evidence that salmon continue their preference for the low saline environment of the plume, as they retain their orientation to the surface region (Emmett et al. 2003). The higher turbidity associated with the low salinity plume waters is considered to provide refugia from predators. A result of developing empirical linkages for a role for the plume to facilitate growth and survival of juvenile salmon is focusing on attributes of the plume that can be used to define habitat important to juvenile salmon. Features such as the 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 are now considered surrogate physical attributes defining habitat important to salmon. Obviously, flow from the Columbia River can modulate the features characterized as defining habitat in this dynamic estuarine zone. Factors, both natural and anthropogenic that modify flows can logically be considered to modify habitat used by salmon as they make the transition to a marine life. In this context, lowering flows beyond what would normally occur would be considered to diminish the availability of these habitats in the plume region for salmon.

Evaluating the impact of water flow on habitat utilized by salmon is a challenge, but remains the crux of the analysis needed to put flow changes in the basin into perspective. Several recent analyses provide some empirical evidence on the role of the altered flow regime described above on habitat in the Columbia River estuary. Baptista (2001) using a hydrologic model developed specifically for the Columbia River 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 (Figure 11). Physical changes in the lower Columbia River estuary (discussed in the next section) and the loss of resilience in maintaining estuarine habitat attributes important to salmon has altered the amount and timing of water delivery to the estuary. These changes will significantly affect availability of habitat needed in the estuary to sustain the diverse life history strategies for the various source population of salmon and steelhead.

Another recent analysis reveals the loss of habitat considered important to juvenile salmon that can be linked to reduced flow. The analysis is especially important because it considers the role of reduced flow on habitat in the tidal freshwater zone of the Columbia River estuarine system, an area seldom evaluated. Kukulka and Jay (2003) demonstrated that there was approximately a 62% loss of shallow water habitat (defined by depth between 10 cm and 2 m) that was attributable to diking (physically removing access of water to the tidal floodplains) and the reduction of peak flows by 40% (consistent with freshet flow reduction discussed earlier) for the region between river...
The diverse life histories of juvenile salmon are an adaptive response to longstanding features of the physical and biotic environments they experience. Diversity of salmonid rearing and migration behaviors thus are linked to various habitats and environmental conditions that can support each developmental stage (e.g., egg, fry, smolt, etc.). Within a suitable range of times and locations, individuals can therefore fully complete their life cycles and maintain membership in a population. Major departures from the historical template of an ecosystem can thus potentially create mismatches between established salmon behaviors and the physical environment or, similarly, prevent the expression of potential behaviors by eliminating habitat opportunity. The hydrological changes described above, particularly those associated with flow regulation and water withdrawals, along with floodplain diking discussed in the next section, represent a fundamental shift in the physical state of the Columbia River ecosystem. Such changes may have significant consequences for both salmonid diversity and production.

The effort to stabilize flows in the Columbia River basin ironically may create less stable conditions for salmonids whose migration and rearing behaviors have adapted to historical patterns of hydrologic variability. Of particular importance 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 flooding throughout the expansive tidal freshwater portion of the estuary may pose some of the most significant consequences for Columbia River salmonids. Flow regulation and diking effects together have largely eliminated access to off-channel floodplain habitats and refugia during high flow events. 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 the tidal floodplain could simplify salmon diversity and reduce rearing capacities of estuarine habitats.

In addition to the physical effects of reduced habitat opportunity on salmon diversity, flow regulation, in conjunction with floodplain diking, may influence the productive capacity of the estuary by regulating so-called “energetic processes” such as food production, competition, and predation. Floodplain inundation greatly increases the surface area of tidal estuarine and riverine habitats available to salmonids, allowing fish to expand their distribution into productive off-channel areas and may relax competitive interactions by reducing fish densities. For example, recent studies in a non-tidal portion...
of the lower Sacramento River found that tagged juvenile chinook salmon released in the seasonally-inundated floodplain had better growth, higher consumption rates, and improved survival compared with others released into the main river channel (Sommer et al. 2001). Elimination of overbank flooding also prevents 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. In summary, consistent with the desire to rank the contribution of water flow to habitat availability, spatial structure, and diversity of salmon populations, we conclude water flow has a likely medium to high impact on recovery potential for salmon populations, high for salmon life history strategies that utilize shallow water associated habitats, and medium to low for salmon expressing life history strategies that utilize deeper associated habitats.

Habitat

The role of habitat has been a defining and consistent objective in many studies focusing on salmon biology, particularly in the freshwater areas. Characterizing habitat and its role is critical in assessing needs of salmon using spawning and rearing habitat in the tributary and mainstem rivers. Defining habitat in these cases has largely been successful because the juveniles and adults are largely fixed in space for relatively long periods of time. However, once the juveniles begin moving from mainstem and tributary rearing habitats, defining habitat attributes using empirically defined specific criteria has been elusive. Although it is unclear why this is the case, the concept that salmon during migratory phases are using habitats simply as corridors to pass from one area to another has probably narrowed our expectations of how habitat functions during this phase.

With respect to the estuary, habitat is important, but exactly what constitutes salmon associated habitat in the Columbia River estuary is unclear. The need for more quantitative descriptions of habitat attributes important to salmon has intensified recently as the calls for habitat restoration actions have increased in response to mandates to reduce risks to endangered salmon populations. The recovery of ‘what’ represents the most common response to this call. In lieu of specific knowledge, restoring the entire estuary for all plants and animals becomes the primary option embraced (LCREP 1999). This type of broad scale approach is limiting, however, because estuaries are typically extensively urbanized environments and so any changes that restore historical attributes are costly, particularly those associated with land acquisition. Questions quickly erupt as to what specifically needs to be restored, once real costs for the broad brush approach become evident.

Although specific studies in the Columbia River estuary are limited, research in estuarine systems has demonstrated that estuaries are composed of a variety of habitats that are used differently by different salmon types that can be distinguished by their life history strategy (Healey 1980, 1982; Levy and Northcote 1981, 1982; Simenstad et al. 1982; Levings et al. 1986; Miller and Sadro 2003). One of the defining features of life history strategies employed by salmon that affects habitat specific use is juvenile size.
Larger juveniles typically use deeper water habitats within the estuary, located more centrally to mainstem channels whereas smaller juvenile salmon use the more peripheral side channel areas associated with the more shallow water habitats (McCabe 1986). Large and small, in these examples, are consistent with salmon that exhibit the yearling life history and to some extent, for juveniles exhibiting the subyearling strategy compared to the fry, fingerling, and subyearling strategies, respectively. Another distinguishing feature that salmon exhibit that influences their use of estuarine habitat is whether a juvenile is smolting compared to a juvenile that has yet to enter into the smolting process. Those animals having made the physiological transformation associated with the endocrine driven smolting process exhibit negative rheotactic behavior and typically use deeper water main channel habitats (associated with the stronger flow signals). Juvenile salmon that have not entered smoltification, but still are moving from natal rearing areas into the mainstem, estuary, and ocean habitats more frequently use side channel, shallow water habitats within the estuary. However, even for juvenile salmon that have smolted and have been shown to migrate through the estuarine habitat for short periods of time (days), a majority of them are found with prey items in their stomachs, suggesting they are extracting resources from estuarine habitats (Dawley 1989).

Although there are certain periods when large numbers of juveniles enter and utilize the estuary, current use patterns indicate that juvenile salmon use the estuary during the entire year (Figure 13). Obviously, existence of sufficient amount and subtypes of habitats allows all salmon species and steelhead in the Columbia River basin to express that appropriate spatial structure and diversity of life history strategies demanded by the environmental and biological conditions the juveniles encounter. 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 (Figure 14). It is apparent that over the year, juvenile salmon representing different cohorts expressing varying life history strategies were using the Columbia River estuary.

Information on the size of salmon in the estuary also reveals more specifically how and where salmon use the estuary. During year long surveys that are actively ongoing to characterize salmon and steelhead entry and exit in the Columbia River estuary, the size structure of chinook salmon, as an example, show several features of how they use the estuary (Figure 13). During the early winter months, the size of juveniles entering and leaving the estuary are nearly identical, consistent with salmon exhibiting the fry life history strategy. However, as the size of chinook salmon entering the estuary increases in length as the year proceeds, the corresponding length of juvenile chinook salmon exiting the estuary during the same period are significantly larger, suggesting growth and thus rearing is occurring in the estuary. The different sizes of salmon observed in these ongoing studies and thus representing the different strategies employed, provide evidence of the differential use of specific types of habitat that salmon utilize. Smaller, unmarked chinook salmon, characterized as likely to be largely composed of naturally produced wild salmon are associated with side channel, peripheral tidal marsh and forested marsh habitats, whereas larger chinook salmon (characterized as hatchery releases) dominate the more deeper-oriented mainstem channel habitats.
(Figure 15). Although it has been argued by Bottom et al (2001) that salmon occupying shallow water habitats express the range of strategies characteristic of ocean-type salmon, it is now evident that salmon representing most of the endangered ESUs are using the peripheral habitats of the Columbia River estuary. Using genetic analysis employing recently developed microsatellite DNA which allows segregation of mixed populations of captured juveniles revealed that both ocean and stream type chinook from upper and lower basin sources were found in these marsh and forested wetland habitats (Figure 16, Paul Moran pers. comm.). Although the predominant source of fish in marsh habitats were from the lower Columbia River Chinook ESU as predicted (Bottom et al. 2001), juvenile chinook from a variety of other source populations were evident. Confirmation that ESUs express a variety a strategies in estuarine habitats is gained from ongoing studies evaluating the role of the Columbia River plume as habitat for juvenile salmon. Spring chinook that express both yearling and subyearling strategies have been identified in the plume environment (Figure 17). Clearly, salmon expressing a variety of strategies from source populations originating throughout the basin use estuarine habitats throughout the year.

The characterization of how salmon employing varying strategies use the estuary is comparable to recent evidence developed regarding the role of the Skagit River estuary in Puget Sound, Washington, for salmon and salmon recovery. Casey Rice (NOAA Fisheries, pers. comm.) has developed empirical evidence that smaller, naturally developed wild salmon are present in the estuary for longer periods of time (Figure 18) and associated with the more peripheral oriented shallow water habitats than larger juvenile salmon either representing hatchery fish, earlier in the season, or wild fish growing larger and utilizing deeper water habitats later in the season (Figure 19). The value of this information stems from the similarity of both the Skagit and Columbia Rivers as large watershed systems with large historically floodplain dominated estuaries.

An important question to consider about estuarine habitat is whether habitat availability in the estuary can be a limiting factor to production and expression of a diversity of strategies. Although this information is forthcoming for the Columbia River estuary explicitly, studies in the Skagit River system have identified a density dependent limit to the number of juveniles in the estuary relative to the overall abundance of young salmon in the system (Figure 20). In the face of altered (i.e., reduced) habitat availability, it is clear that reducing the opportunity to access habitats at the appropriate time can be a limiting factor in production and recovery of depressed salmon and steelhead populations.

Finally, evidence that estuaries are likely to be important to recovery of endangered salmon and steelhead stocks is derived from salmon life cycle models and application of sensitivity analyses to identify critical life stages, and thus associated habitats, important to recovery. Kareiva et al. (2001) showed that improvement of survival of juvenile salmon during the estuarine and early ocean stage would significantly improve salmon population growth rates. Although they could not differentiate the contribution of the estuarine phase from the early ocean phase in their analysis, clearly the estuarine stage could be critical to recovery. Similarly, Greene et al. (in press)
demonstrated that variability in conditions in the nearshore zone of Puget Sound, which
serves as an extension of the Skagit Bay estuary, accounted for significant variability in
adult returns for Skagit Bay chinook salmon. Incorporating density dependence, as
indicated above, improved the resolution of the model output. Overall, the summary of
information from a variety of perspectives documents that salmon use and benefit from
the estuary and the habitats it contains. Identifying alterations that affect access or
quality of the habitats relative to the historical record can be used to assess the potential
of habitats within the estuary to represent a potential limiting factor of importance to
recovery of endangered salmon populations.

Considering the importance of estuarine habitat to juvenile salmon, what evidence
from the historical and more current record can be used to identify a change in the
availability of habitat in general, or more specific, loss of salmon type habitat in the
system. Thomas (1983) and Sherwood et al. (1990) have calculated losses of emergent
marsh and forested wetland habitats in the Columbia River estuary. They characterize the
change as substantial and likely a significant factor reducing the estuary’s opportunity
and capacity to support juvenile salmon. Approximately 121.6 km² of tidal marshes
(77% decline) and swamps (62% decline) that existed prior to 1870 have been lost
(Figure 21). Together with a 12% loss of deep-water habitat, these changes reduced the
estuary’s tidal prism from 12 to 20%. 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.

High elevation tidal marshes have been diked more than lower elevation marshes. New
tidal marsh formation has resulted primarily from vegetative colonization of
disposed dredge material. The location of tidal marsh habitat within the estuary has
changed as a result of modified flow regime, modified tidal action, and/or shipping
channel development and maintenance. Tidal swamp is the most impacted habitat type.
Almost all the tidal swamp habitat present in 1870 was converted to diked
floodplain/non-tidal habitat. Historically, few tidal swamps were present in the mainstem
areas of the Columbia River estuary, in contrast to what has occurred in the peripheral
bays. For example, there has been almost complete loss of all 1870 tidal swamp habitat
from Youngs Bay and Baker Bay effectively eliminating brackish tidal water from the
estuary. In the areas furthest upstream, losses of tidal swamps have been extensive,
although a substantial amount of tidal swamp acreage is still present, particularly in the
Cathlamet Bay region.

Johnson et al. (2003) in developing a restoration plan for the Columbia River
estuary refined the losses within particular zones of the estuary using information from
Thomas (1983), Graves et al. (1995), USACE (1996), and Garono et al. (2002). To
facilitate this comparison, the Columbia River estuary and lower mainstem was
delineated into eight distinct areas based on physical characteristics. A brief description
of each estuary area, a summary of habitat changes, and a qualitative description of
changes in select habitat characteristics were extracted from Johnson et al. (2003) and the
subbasin review developed by the Lower Columbia Fish Recovery Board as follows:
**Entrance**—dominated by subtidal habitat; highest salinity in estuary; historically a high-energy area of natural fluvial land forms, a complex of channels, shallow water, and sand bars; supports the Columbia Plume; abrupt changes have resulted from dredging and jetty construction that limit the ocean-fed supply of sediment; impacts have manifested in increased deep water habitat (18.9%) and a loss of medium-depth (41.1%) and tidal flat (43.6%) habitat types.

**Mixing Zone**—characterized by a network of mid-channel shoals and flats; highest variation in salinity based on tide cycle and river flow; relatively little change in acreage of the five major habitat types.

**Youngs Bay**—characterized by a broad flood plain and historically abundant in tidal marsh and swamp habitat; diking and flood control structures used to convert land to pasture resulted in 86.4% loss of tidal marsh and 95.7% loss of tidal swamp habitat. This subarea was responsible for the majority of lost tidal marsh habitat throughout the estuary.

**Baker Bay**—historically a high energy area from ocean currents and wave action; migration of mid-channel islands toward the interior of Baker Bay have sheltered the area; some tidal marsh habitat recently started to develop because of decreased wave action; potentially the most altered estuary area overall (-75.0% deep water, -71.3% medium depth, +74.9% tidal flats, -55.5% tidal marsh, and -100% tidal swamp habitat).

**Grays Bay**—pile dikes adjacent to the main Columbia River navigation channel have decreased circulation in the bay and caused flooding problems in the valley bottoms; accretion in the bay has led to development of tidal marsh habitat, increasing acreage 145.2% compared to historic conditions; dike construction for pasture conversion has isolated the main channel from its historic floodplain and decreased tidal swamp habitat 88.4% compared to historic acreage.

**Cathlamet Bay**—characterized by some of the most intact and productive tidal marsh and swamp habitat in estuary; large portion of area protected by the Lewis and Clark Refuge; other portions are heavily impacted by diking (Brownsmead area and Swenson Island) causing a 48.9% decline in tidal swamp; medium and deep water habitats have decreased (30.4% and 12.5%, respectively) as a result of dredge material disposal; fringe of dredge disposal areas has developed into tidal marsh habitat, resulting in 6.8% increase over historic acreage.

**Upper Estuary**—characterized by deep channels and steep shorelines on both sides of river; dominated more by tidal swamp habitat and less tidal marsh habitat; typically dominated by freshwater, except during low river flow or large flood tides; extensive diking and clearing has resulted in substantial loss of tidal marsh (64.3%) and tidal swamp (79.9%) habitat compared to historic acreage.

**Tidal Freshwater**—distinct in geology, vegetation, and climate; influenced by major tributaries; contains elongate islands that divide the river and form oxbow lakes,
sloughs, and side channels; historically dominated by a combination of tidal plant communities, ash riparian forests, and marshy lowlands; historic data for entire area is limited to other areas so historic comparisons are not as robust; from rm 46-102, increased upland habitat in the middle reach and substantial loss of non-tidal water/wetland, tidal flats, and tidal marsh habitat types, with no comparison category for tidal swamp habitat; from rm 105-146, increased non-tidal water/wetland and upland habitat and substantial loss of tidal flats and tidal marsh habitat types, with no comparison category for tidal swamp habitat.

Thomas (1983), as reported in the Estuary Subbasin Plan (2003), also investigated five categories of non-estuarine habitat (i.e. developed floodplain, natural and filled uplands, non-tidal swamps, non-tidal marshes, and non-tidal water) to identify the fate of floodplain areas that were removed from the estuarine system. Developed floodplain habitat was defined as all diked floodplain converted to agriculture, residential, or other land use. Natural and filled uplands included areas where measurable acreages have been filled, primarily through disposal of dredge material. Non-tidal swamps were areas of the diked floodplain that were never cleared or were cleared and converted back to swamp. Non-tidal marshes included areas of the diked floodplain that support emergent wetland vegetation; these were typically abandoned pastures dominated by rush and sedge. Non-tidal water consisted of former tidal sloughs that were separated from the river by dikes and tidegates. The largest increase, by far, 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).

As above, absolute changes in habitat opportunity alone should not be used to directly infer changes in the capacity of the estuary to support salmon. For instance, despite considerable loss of emergent and forested wetlands in the estuary and associated declines in macrodetrital production, the total area of estuarine shallows and flats actually increased 7% between 1870 and 1980. This was independently substantiated by Sherwood et al. (1990), who estimated $68.4 \times 10^6$ m$^3$ net sediment gain within the estuary between 1868 and 1958. Areas of sediment increase include peripheral bays such as Cathlamet Bay and Grays Bay, which had shoaling rates of 0.61 cm yr$^{-1}$ and 0.63 cm yr$^{-1}$ and net volumetric increases of $76.2 \times 10^6$ m$^3$ and $19.1 \times 10^6$ m$^3$, respectively.

Loss of estuarine wetlands not only reduced the total amount of shallow rearing habitat available to young salmon but also altered the magnitude and character of habitat capacity. The resulting decline in wetland primary production eliminated approximately 15,800 mt carbon year$^{-1}$ (84%) of macrodetritus that historically supported estuarine food webs. This macrodetritus originated from the vascular and macrophytic plants and microscopic algae historically produced within the estuary’s wetlands. 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). Nevertheless, the shifts in the sources and types of detritus available may have altered estuarine food webs, including those leading to salmon. For example, the epibenthic-pelagic food web supported by microdetrital sources favors production of calanoid
copepods and other pelagic organisms that typically are not consumed by juvenile salmon (Bottom and Jones 1990, Sherwood, et al. 1990).

As a result of loss of habitat, altering the spatial distribution of the food web may also be an important determinant of habitat capacity in the estuary. 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).

We have no objective means to quantify the ecological effects of the habitat shift from emergent and forested wetlands to shallows and flats. For example, no historic data are available for salmonid diet composition or stomach fullness within tidal wetlands to compare with other estuarine habitats. Although juvenile salmon may not directly benefit from the microdetrital food web, there is some evidence that they have higher stomach fullness in the mid estuary compared with other estuarine regions (Bottom and Jones 1990). One possible mechanism that has yet to be verified for the increased feeding rates is that enhanced detrital concentrations within the ETM may also stimulate secondary production in adjacent mid-estuary shallows and flats. However, we do know that prey production and salmon stomach fullness values are relatively high in protected flats compared with many estuarine habitats. Jones et al. (1990) found that the standing crop of benthic infauna in protected flats of the estuarine mixing region (approximately RM-7 to RM-21) was more than an order of magnitude higher (2.058 g m\(^{-2}\) AFDW (ash free dry weight) than benthic fauna standing crop in any of the other channel or unprotected flat habitats (0.098-0.136 g m\(^{-2}\) AFDW) within the same estuarine region.

One additional recent example, exemplifies the loss of habitat in the tidal freshwater region of the estuary, where we most lack the empirical evidence of change and contribution to expression of spatial structure and salmon life history diversity. 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 (Figure 22). It is obvious from the analysis that removing dikes alone would restore considerable amounts of shallow water estuary habitats. Further, diking entirely removes habitat from the estuarine system, while other anthropogenic factors change estuary habitats from one type to another (Thomas 1983). The degree to which estuary habitat types have been 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).

Mainstem estuary habitat in the Columbia River have for the most part been reduced to a single channel where floodplains have been reduced in size and off-channel habitat has been lost or disconnected from the main channel. Dikes prevent over-bank flow and affect the connectivity of the river and floodplain (Tetra Tech 1996); thus, the diked floodplain is higher than the historic floodplain and inundation of floodplain habitats only occurs during times of extremely high river discharge (Kukulka and Jay 2003). There is a critical level (i.e. the elevation of the diked floodplain) where water level must reach before substantial floodplain habitat are inundated (Kukulka and Jay 2003). Above this critical water level, large amounts of shallow water floodplain habitats
become available with small increases in water level up to an optimum threshold (Kukulka and Jay 2003). Under a modern bathymetry and flow regime scenario, the critical river discharge level in which significant shallow water habitats become available through floodplain inundation is relatively high and the frequency of occurrence of this river discharge is rare; thus, floodplain inundation is uncommon and availability of shallow water habitats is limited (Kukulka and Jay 2003). As is the case in the estuary (Bottom et al. 2001), loss of these vital mainstem floodplain habitats has likely reduced the productive capacity of the lower Columbia River for juvenile salmonids, particularly those juveniles employing life history strategies associated with these peripheral, shallow water habitats.

Substantial evidence indicates that the productive capacity of the Columbia River estuary has declined over the last century. The results generally illustrate that the habitat opportunity and capacity of the Columbia River estuary may have declined through reductions in the estuarine tidal prism, surface area, and the amount of peripheral wetland habitat. Moreover, changes in the detrital sources that support estuarine food webs have affected competitive and likely subsequent predatory interactions in the estuary with uncertain, but potentially significant, consequences for salmon survival.

Many of the changes in the biological production processes of the estuary described above can be attributed directly to physical causes. For example, the apparent shift from macrodetrital to microdetrital food chains in the estuary stems from the diking and filling of intertidal wetlands and the creation of deep reservoirs behind mainstem dams. While changes in the quality and quantity of prey resources could well be a proximal factor affecting the productive capacity of the estuary, the ultimate cause is the physical removal of those habitats that supported both macrodetrital production and the diversity of estuarine life histories among salmon.

This is not to say that all biological effects are physically driven or somehow irrelevant to salmon survival and abundance. Nonetheless, we find that many of the significant biological changes we now observe in the Columbia River estuary are best explained by physical modifications that have altered the habitat landscape to the benefit and detriment of different species and assemblages. These findings have important implications for developing restoration strategies that address the ultimate causes rather than the proximal symptoms of salmon decline. In summary, we conclude habitat change/loss has a likely medium to high impact on recovery potential for salmon populations, high for salmon life history strategies that utilize shallow water associated habitats, and medium to low for salmon life history strategies that utilize deeper habitats.

Toxics

In addition to physically altering or destroying salmon habitat, humans have degraded estuarine habitats through the release of toxic contaminants. With the exception of some metals and natural products, concentrations of toxic contaminants in the Columbia Estuary were historically low. However, beginning in the early 1800s,
activities such as agriculture, logging, mining, industrial discharges, and stormwater runoff began to degrade water quality in the Columbia Estuary. Currently, the Lower Columbia from Bonneville to the estuary mouth is the most urbanized section of the river, encompassing major the industrial centers of Portland and Vancouver. The Lower Columbia Estuary receives contaminants from over 102 point sources (Fuhrer et al. 1996), as well as non-point sources such as surface and stormwater runoff from urban and agricultural areas. The largest sources of effluent are the Portland and Vancouver sewage treatment plants and associated combined sewer overflows in the Willamette River and the Columbia River Slough (LCREP 1999). Contaminants may also be transported to the Lower Columbia from areas of known sediment contamination above the Bonneville Dam such as the Yakima River (Rinella et al. 2000; Fuhrer et al. 1996), Lake Roosevelt (Bortleson et al. 1994) and other tributaries (Fuhrer 1989; Roy F. Weston Inc. 1998).

A number of potentially toxic water-soluble contaminants have been detected in the Lower Columbia River Basin. The USGS NASQAN program has reported a wide range of current-use pesticides in the water column at its Lower Columbia sampling sites at Warrendale at RM 141 near the Bonneville Dam; the Willamette-Columbia River at confluence near Portland at RM 101.5; and the Beaver Army Terminal at RM 53.8 (Fuhrer et al. 1996; Hooper et al. 1997). These include simazine, atrazine, chlorpyrifos, metolachlor, diazinon, carbaryl, and others. Water concentrations and frequency of detection were highest at the Willamette/Columbia confluence, with detections in 80-100% of samples at concentrations up to 300 ng/L, but these compounds were frequently reported at the Beaver Army terminal near the estuary as well. Various trace metals have also been monitored as part of this program, revealing high concentrations of iron and manganese, especially near the Willamette/Columbia confluence, and high levels of arsenic in the Lower Columbia (Fuhrer et al. 1996). These compounds come partly from natural sources, but also historic anthropogenic activities, such as the use of lead arsenate as an insecticide for apples. Concentrations of other trace metals were similar to background concentrations in other North American streams (Fuhrer et al. 1996).

Contaminants that have been documented in Lower Columbia 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 polycyclic aromatic hydrocarbons (PAHs) and other semi-volatile compounds (Fuhrer and Rinella 1983; Fuhrer 1986; Harrison et al. 1995; Tetra Tech Inc 1996; US Army Corps of Engineers 1998; Roy F. Weston, Inc 1998). Many of these compounds, particularly lipophilic compounds such as PAHs and organochlorine compounds (OCs), are rarely detected as dissolved material in the water, but tend to bind to organic carbon or particulate materials, and commonly occur in association with fine-grained materials in the streambed or in suspension (Horowitz 1991; Tetratech Inc. 1993). Some of these, including PAHs, have been detected at levels that exceed State of Federal sediment quality guidelines or are considered harmful to humans and aquatic life (Tetra Tech Inc. 1996) in sediments from the lower Willamette River in the Portland area (concentrations up to 900 mg/kg wet wt (ODEQ 1994a; Harrison et al. 1995; Roy F. Weston, Inc 1998).
Suspended material may also be an important source of contaminant exposure in the Lower Columbia (LCREP 1999). This material is predominantly fine-grained, and contains many of the toxic compounds that have been detected in streambed sediments. McCarthy and Gale (2001) analyzed water samples collected with semi-permeable membrane devices from nine main-stem and six tributary sites throughout the Columbia River Basin (Washington and Oregon) and found dioxins, dibenzofurans, PCBs, organochlorine pesticides, and PAHs throughout the basin, with highest concentrations of many analytes in the Portland-Vancouver area. Metals including arsenic, lead, chromium, copper, iron, manganese, mercury, and zinc have also been detected in suspended sediments in the Lower Columbia (Fuhrer et al. 1996). Mobilization and transport of suspended sediments during extreme streamflow events can make them available to salmon and other aquatic organisms. For example, during the flood of February 1996, several legacy organic pesticides, including dieldrin and DDE, that are typically associated with the sediment phase were mobilized in the Lower Columbia and Willamette Rivers and detected in the water column at some sites for the first time (Kelly 1997). During this event, the estimated whole water concentration of p,p'-DDE exceeded the chronic ambient water-quality criterion for the protection of aquatic organisms by at least five-fold. Suspended particulates and associated contaminants may also occur in areas of high turbidity, such as the estuarine turbidity maximum, which may be an important feeding area for salmon (Bottom and Jones 1990). However, currently, the relative contributions of contaminants in water column vs. those in bed sediment to body burdens in resident biota are poorly understood (LCREP 1999; SEI 2001).

Fine-grained sediments are most likely to be deposited in areas with slower water velocities, including backwater areas in side channels and along the river’s margins, so these are areas where elevated concentrations of toxic contaminants are considered most likely (Tetra-Tech, Inc. 1994). In the main navigation channel of the Columbia, current velocities are generally high, so there is little deposition of fine-grained sediments. They typically make up less than 1% of the bulk bed material in the navigation channel (USACE 1998; McCabe et al. 1997), and contaminant concentrations in these coarse, sandy sediments are usually lower (USACE 1998; 1999). The backwater areas where fine-grained sediments tend to deposit also provide critical habitat for endangered salmon. As noted earlier in this document, smaller, unmarked chinook salmon that are likely to be naturally produced wild salmon are most commonly found in side channel, peripheral tidal marsh and forested marsh habitats, whereas larger hatchery developed chinook salmon dominate the more deeper-oriented mainstem habitat linked channel habitats. This suggests that wild salmon may especially at risk for exposure to and uptake of contaminants in the estuary.

Many of the contaminants found in streambed and suspended sediments in the Lower Columbia are accumulated by resident biota. A number of studies have identified trace metals, dioxins and furans, chlorinated compounds such as PCBs and DDTs, and PAH metabolites in non-salmonid fish from the Lower Columbia (Tetra Tech Inc 1993d,
1996; Brown et al. 2000; Foster et al. 2001a,b), in some cases at concentrations have exceeding health guidelines (LCREP 1999). Washington Although data on contaminant concentrations in listed salmon from the Lower Columbia are more limited, available data indicate that chemical contaminants are present in prey and tissues of juvenile salmon from the Columbia Estuary (Figures 23-27).

In an initial study\(^1\) conducted by the Northwest Fisheries Science Center, contaminant concentrations were measured in juvenile salmon collected from a number of Pacific Northwest estuaries between 1994 and 1998; Figure 23-24). As expected, contaminant concentrations were elevated in fish from urban estuaries and waterways, such as the Duwamish Estuary and in the Hylebos Waterway in Puget Sound. However, elevated concentrations of PCBs and DDTs were also found in whole bodies and stomach contents of juvenile chinook salmon from the Lower Columbia Estuary, at a site near the mouth of the Columbia just east of West Sand Island. Whole body concentrations of DDTs and PCBs in these fish were 44 ng/g wet wt and 53 ng/g wet wt, respectively, among the highest levels measured at estuarine sites in Washington and Oregon. Additional analyses from salmon collections in 1999 and 2000 showed that concentrations of PCBs and DDTs were consistently elevated in chinook salmon collected from near West Sand Island, with concentrations as high or higher than those measured in 1998.

The observation of salmon with elevated contaminant body burdens in the estuary, stimulated interest in additional sampling to better characterize exposure in outmigrant juvenile salmonids, and in 2000, NOAA Fisheries and the Army Corps of Engineers initiated a pilot study to measure contaminant concentrations in juvenile fall chinook from the Lower Columbia. In 2000-2002, tissue and stomach contents for chemical analysis were collected from juvenile chinook salmon from near the confluence of the Columbia and Willamette Rivers, near Longview, and at several sites within the Lower Columbia Estuary (White Island, West Sand Island, between East and West Sand Island, Chinook Point, East Trestle Bay, West Trestle Bay, Lower Desdemona Sands, and Middle Desdemona Sands). Composite samples of 10-15 whole bodies of fish from West Sand Island, Longview, and the Columbia/Willamette confluence were analyzed by GC/MS (Sloan et al. 1993) for PCBs, DDTs, and selected organochlorine pesticides and contaminants, including hexachlorobenzene, lindane, aldrin, dieldrin, endosulfans, mirex, heptachlor, and chlordanes. Individual fish (4-5 per site) from West Sand Island, East and West Trestle Bay, Upper, Middle and Lower Desdemona Sands, White Island, and Chinook Point, were analyzed by HPLC/PDA (Krahn et al. 1994) for PCBs and DDTs.

The primary contaminants found in whole body composite samples of Chinook salmon from the lower Columbia were PCBs and DDTs, which were found in measurable concentrations at all sampling sites (Figure 23-24). Average concentrations of PCBs in whole body composites ranged from 23 to 41 ng/g wet wt), while average DDT concentrations ranged from 32 to 39 ng/g wet wt, similar to levels observed earlier in fish from West Sand Island. Concentrations of PCBs and DDTs in salmon bodies were fairly

\(^1\) Data are from preliminary studies conducted by Dr. Mary Arkoosh, Hatfield Marine Science Center, Newport, Oregon, with supporting data from Puget Sound urban estuaries from Collier et al. 1998a, Stehr et al. 2000; Stein et al. 1995.
uniform at all sites sampled. Chlordanes, lindane, hexachlorobenzene, dieldrin, and mirex were also detected in fish from the confluence and Longview/Kalama, but not in those from West Sand Island. Aldrin, endosulfans, heptachlor, and hexachlorohexane were below detection limits at all sites. Analyses of individual fish from other sites within the estuary showed even higher body burdens of PCBs and DDTs in some individuals than those found in the composite samples. PCB levels were especially high near West Sand Island and West Trestle Bay (site means 70-90 ng/g wet wt). DDTs concentrations were also quite high (site means from 40-115 ng/g wet wt), especially in fish from West Sand Island, West Trestle Bay, and Lower Desdemona Sands. In individual fish, DDT levels were as high as 270 ng/g wet wt and PCB levels were as high as 340 ng/g wet wt.

Measurable concentrations of DDTs and PCBs, were also detected in stomach contents of juvenile fall Chinook salmon from West Sand Island, White Island, Longview, and the Columbia-Willamette Confluence (at other sites, stomach contents contaminants were not measured), with highest concentrations in fish from the Willamette/Columbia confluence (Figure 25). Concentrations of PCBs in stomach contents of fish from the confluence were comparable to those from in fish from the Hylebos Waterway in Puget Sound (Stehr et al. 2000), while DDT concentrations were higher than at any other sampled sites in the Pacific Northwest. Several additional chlorinated pesticides, including lindane, hexachlorobenzene, dieldrin, and certain DDT isomers (o,p-DDD, o,p-DDT, p,p-DDT) were detected only in stomach contents of salmon from the confluence and Longview sites, while chlordanes, mirex, and o,p-DDE were detected only at the confluence. Other measured compounds (hexahlorohexanes, aldrin, endosulfans, and heptachlor) were below detection limits at all sites.

The fish used for these analyses were from mixed hatchery and wild stocks (approximately 20-30% hatchery fish, based on fin-clip and coded wire tag data; A. Kagley, NWFSC, pers. comm.). Consequently, it is likely that DDTs and PCBs in hatchery food made some contribution to contaminant body burdens in hatchery fish, since these substances, especially PCBs, have been detected in hatchery feed and in juvenile Chinook salmon collected from Pacific Northwest hatcheries (M. Arkoosh, and G. Ylitalo, NWFSC, unpublished data; Figures 23-25). However, the findings of elevated levels of DDTs and PCBs in stomach contents of fish from sites within the Estuary indicates that fish are also being exposed in their environment through their natural prey. Additionally, DDT/PCB ratios were several times higher in bodies of salmon from Columbia River sites (average 1-4) than in hatchery fish (average 0.7), suggesting uptake of DDTs from the environment.

Less information is available on exposure to PAHs in juvenile salmon from the Lower Columbia. Data collected by NOAA Fisheries between 1998 and 2002 showed that concentrations of PAHs in stomach contents and PAH metabolites in bile were low to moderate in juvenile fall chinook from West Sand Island in comparison to levels found in fish from other estuaries along the Washington and Oregon Coast (Figure 26). In contrast, PAHs in stomach contents of juvenile chinook from the Willamette/Columbia and Longview were relatively high, higher than or comparable to concentrations in
juvenile salmon from the Hylebos Waterway in Puget Sound (Figure 26). No data are available on metabolites if PAHs in bile of fish from other sites in the Lower Columbia.

Other data on contaminant concentrations in salmon prey and other benthic invertebrates are limited. Little has been published for contaminants in aquatic macroinvertebrates, although some work has been done on clams, crayfish, and *Corophium*, a benthic amphipod that is important in the diets of several fish species, including salmonids (McCabe et al. 1986, 1993; Muir and Emmett 1988; Willis CF 1997; LCREP 1999). A small-scale reconnaissance study conducted by NOAA Fisheries Newport laboratory in the mid-1990s measured PAH concentrations in *Corophium salmonis* from several sites in the Lower Columbia, including Longview Bridge and the Multnomah Channel in the Columbia River, and sites at the Willamette River mouth, North Portland Harbor, and Hayden Island in the Willamette River. At the three sites where sediment PAH concentrations were highest (Hayden Island, Longview Bridge, and Multnomah Channel), no amphipods were present in sediments, suggesting *C. salmonis* has specific habitat requirements, including sediment particle size preferences, and, possibly, concentrations of contaminants, that may prevent them from occupying these sites. PAH concentrations in tissues of *Corophium* tissues were up to over 50 ng/g wet wt BaP equivalents or about 150 ng/g wet wt HAHs.

In general, these studies show that PCB and PAH concentrations in salmon or their prey from the Lower Columbia are comparable to those reported in juvenile salmon from other moderately to heavily urbanized sites, while DDT levels are high relative to levels in other Pacific Northwest estuaries. Although concentrations of contaminants were higher in stomach contents of collected from near Columbia/Willamette confluence, body burdens of bioaccumulative OCs were similar throughout estuary. Sources and pathways of exposure are unclear, and could include contaminated bed sediments, contaminated prey, and contaminants in suspended material, as well as hatchery feed for those fish that are of hatchery origin.

Studies suggest that, at least for some contaminants, exposure levels in juvenile salmon from the Lower Columbia are approaching concentrations that could affect their health and survival. In field studies in Puget Sound, at sites contaminated with PAHs, PCBs, and other OCs similar to those present in the Lower Columbia, juvenile salmon showed demonstrable effects, including immunosuppression, reduced disease resistance, and reduced growth rates, due to contaminant exposure during their estuarine residence (Arkoosh et al. 1991, 1994, 1998; Varanasi et al. 1993; Casillas et al. 1995a,b, 1998a). For example, juvenile chinook salmon from the Duwamish Waterway were not able to produce the normally enhanced secondary immune response observed in non-exposed control fish from the Nisqually estuary and fish from the releasing hatcheries (Arkoosh et al. 1991; Figure 27). Salmon exposed in the laboratory to PCBs and PAHs were also

---

Data are from preliminary studies conducted by Dr. Bruce McCain, Hatfield Marine Science Center, Newport, Oregon. Concentrations of PAHs in amphipods were measured using a semi-quantitative method that employs high performance liquid chromatography (HPLC) and a photo-diode array (PDA) detector (Krahn et al. 1993).
immunosuppressed (Arkoosh et al. 1994). Additionally, in disease challenge studies with *Vibrio anguillarum*, a marine bacterial pathogen that infects juvenile chinook salmon from estuaries along the Washington and Oregon coast (M. Arkoosh, in review), juvenile chinook salmon from the Duwamish Waterway were more susceptible to disease and exhibited higher cumulative mortality than fish from the releasing hatcheries on a non-contaminated estuary (Arkoosh et al., 1998; Figure 28). Similar effects were found in juvenile chinook salmon injected in the laboratory with extracts of sediments from the Hylebos Waterway in Commencement Bay (Arkoosh et al. 2000).

In related studies designed to assess the effects of contaminants on growth (Casillas et al. 1995a,b), juvenile fall chinook salmon collected from the Duwamish Waterway and held in the laboratory for up to 90 days did not grow as well as similarly-treated fish from the corresponding hatchery on the Green River. In contrast, juvenile salmon from the non-urban estuaries showed no difference in growth compared to fish from the corresponding hatcheries. Furthermore, concentrations of insulin-like growth factor, a plasma hormone involved in the regulation of growth, were lower in fish from the urban estuary than in fish from the corresponding hatchery or the non-urban estuaries and hatcheries. In a separate laboratory study (Casillas et al. 1998a), growth was reduced in juvenile chinook salmon exposed to PCBs and to extracts of sediments from the Hylebos Waterway, another urban estuary. Because growth of salmon during the first year of life appears to be critical to recruitment success (Holtby et al. 1990; Pearcy 1992; Unwin 1997), contaminant-related alterations in juvenile growth rates could increase the risk of salmon mortality.

In other studies in Puget Sound, juvenile salmon from urban estuaries, e.g. the Duwamish and Hylebos Waterways, had significantly higher levels of DNA damage (i.e., PAH-DNA adducts in liver) than salmon from relatively uncontaminated sites, as well as induction of cytochrome P4501A (CYP1A), the enzyme that metabolizes selected contaminants including PAHs, dioxins and furans, and dioxin-like PCBs (Stein et al. 1995; McCain et al. 1990; Varanasi et al. 1993; Collier et al. 1998a,b; Stehr et al. 2000). These biochemical alterations are not necessarily indicative of adverse health effects in themselves, but are associated with disease conditions including reproductive and developmental abnormalities and liver disease (ref).

Effects on growth and immune function are occurring at relatively low contaminant concentrations. At the two contaminated sites where most of the work described above was done, the Duwamish and Hylebos Waterways, average sediment PCB concentrations range from 400-500 ng/g dry wt, and average total sediment PAH concentrations are around 10,000 ng/g dry wt (Collier et al. 1998a; Krahn et al. 1998). Total body PCB concentrations in fish collected from these sites were in the 250-350 ng/g dry wt range, comparable to some fish sampled from sites within the Estuary. Contaminant concentrations in the laboratory-exposed salmon from the immune function and growth studies were comparable to those observed in field-collected salmon from the Duwamish and Hylebos Waterways (Casillas et al. 1998a,b).
Various health effects have been documented in non-salmonid fish and other aquatic biota from the Lower Columbia have also been observed, including fish with external abnormalities or skeletal deformities (Markle 1995; Tetra Tech Inc 1995b), alteration in endocrine response function (Goodbred et al. 1997; Foster et al. 2001a,b), and pollution-associated liver lesions (Myers et al. 1994). Although the exposure patterns and life histories of listed salmon may differ from those of these fish species, these data raise additional concern about the potential for exposure and health impacts on salmon.

To better characterize PCB concentrations associated with adverse effects in juvenile salmonids, Meador et al. (2002) developed critical body residue and sediment concentrations of PCBs associated adverse effects on salmon, based on a range of sublethal effects observed in salmonids in peer-reviewed studies conducted by NMFS and other researchers. He estimated a threshold concentration of 2400 ng/g lipid for protection against 95% of effects ranging from enzyme induction to mortality. For 2% lipid, which is the average concentrations for fish collected in the Columbia, the threshold is 48 ng/g wet wt. Mean PCB body burdens in juvenile salmon analyzed by the NWFSC were at or above these thresholds at several sites in the Lower Columbia (Figure 23). Of individual fish analyzed from sites within the estuary, 30-40% were above the effects threshold.

Effect thresholds for DDTs, another major contaminant of concern in the Lower Columbia, are less clear. Most reported effects of DDTs on salmonids are associated with whole body tissue concentrations in the 2000-3000 n/g wet wt range or greater, with some effects on early life stages (e.g., eggs, embryos, and fry) at tissue concentrations in the 500 ng/g wet wt range (Johnson and Pecor 1969; Poels et al. 1980; Burdick et al. 1964; Peterson 1973, Buhler et al. 1969; Allison et al. 1962; Hose et al. 1989), levels above those typically found in Lower Columbia Estuary salmon. However, a number of newer studies suggest that certain DDT isomers (e.g., o,p-DDT) may have endocrine-disrupting or immunotoxic effects (Khan and Thomas 1998; Milston et al. 2003; Zaroogian et al. 2001), although data on tissue residues associated with these effects are limited. Concentrations of o,p-DDT in tissues and environment are generally low, but because it has estrogenic activity it could work in conjunction with other estrogenic contaminants (e.g., plasticizers, pharmaceuticals, and surfactants) to alter reproductive processes or other physiological functions. Recently, Nendza et al. (1997) estimated a no-observable effect concentration for DDTs of 22-50 ng/g wet wt for impacts related to bioaccumulation and bioconcentration, based on studies with a number of fish from marine estuaries (Nendza et al. 1997). Many juvenile salmon sampled from the Lower Columbia have DDT body burdens at or above this level. Furthermore, invertebrate species, including some salmon prey, are generally quite susceptible to effects of DDT. Results of laboratory and field investigations, as well as equilibrium partitioning calculations, suggest that thresholds for chronic effects generally occur at total DDT concentrations in sediments of about 2 ng/g dry wt, or approximately 10 ng/g dry wt (Long et al. 1995; Pavlou et al. 1987). These DDT concentrations are not uncommon in Lower Columbia sediments.
In addition to bioaccumulative contaminants, waterborne contaminants such as dissolved metals and current use pesticides may pose a threat to listed salmon. Scholz et al. (2000) reported that the organophosphate pesticide, diazinon, disrupted olfactory function in Chinook salmon at concentrations of 1-10 ug/L, so that fish failed to show normal anti-predator responses or homing behavior (Figure 29). Moore and Waring (1996) and Waring and Moore (1997) found that exposure to diazanon and carbofuran in a similar range could desynchronize the reproductive physiology of prespawning Atlantic salmon (Salmo salar) by inhibiting the male’s ability to detect sex pheromones. Concentrations of diazanon in the 1-10 ug/L range have been reported in NASQAN sampling in the Lower Columbia, and other pesticides with related modes of action that would be likely to have the same effects (e.g., chlorpyriphos, malathion, aldicarb, carbaryl, carbofuran) are detected even more frequently and at higher concentrations.

Similarly, Baldwin et al. (2003) found that short-term exposure to copper at concentrations in the 3-6 ug/L range affected olfactory function in coho salmon so they could no longer respond normally to test odorants. This could impair the ability of juveniles to find prey and avoid predators, or interfere with homing and reproductive behavior in adults. Dissolved copper concentrations at the Lower Columbia 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). Other contaminants in the water column, including endocrine-disrupting substances such as synthetic hormones, are only beginning to be characterized in the Lower Columbia, but could also have substantial impacts on salmon.

More research is clearly needed to document exposure and associated effects of chemical contaminants on endangered Columbia Estuary salmon, but the available data show clearly that environmental concentrations and tissue burdens of several classes of contaminants are within the range where they could potentially affect survival and productivity of listed stocks.

In summary, we conclude that toxic contaminants have a likely medium to high impact on recovery potential for salmon populations, high for salmon life history strategies that utilize shallow water associated habitats, medium for salmon life history strategies that utilize more deeper associated habitats occupied by salmon.

Caspian Tern Predation of Juvenile Salmon

In the preceding analyses, we considered diversity of life history strategies in relation to diversity of habitats as the means to assess detrimental impacts of limiting factors. Although loss of habitat may affect recovery by reducing diversity and spatial structure of salmon populations, the direct effect on mortality and hence abundance and productivity can only be conjectured, not measured. However, recent developments in the Columbia River estuary has identified a direct mortality factor that lends itself to an assessment on population growth rate, another key VSP attribute important to sustainability of salmon population. This involved the recent movement of the
piscivorous bird predator, Caspian Tern (*Sterna caspia*), into the Columbia River estuary. Increasing populations of piscivorous birds (primarily Caspian terns) nesting on islands in the Columbia River estuary annually consume large numbers of migrating juvenile salmonids (Roby et al. 1998) and thus constitute one of the factors that may currently limit salmonid stock recovery (Roby et al. 1998; Independent Multidisciplinary Science Team 1998; Johnson et al. 1999). Reducing Caspian tern predation in the estuary, is one potential mechanism to reduce mortality, thereby increasing population growth rates of Endangered Species Act (ESA) listed salmonid in the Columbia River Basin. Here, we consider mortality of juvenile salmon as a result of predation by Caspian Terns as a limiting factor.

Caspian terns are highly migratory and are cosmopolitan in distribution (Harrison 1983; Harrison 1984). Nesting has been reported on the west coast of North America from Baja, California to the Bering Sea, in the interior from the Gulf Coast of Texas to Lake Athabasca, Saskatchewan, and on the east coast from the Florida panhandle to Labrador. Outside of North America, nesting has been reported in Australia, New Zealand, South Africa, Asia, and Europe.

The numbers of Caspian terns in western North America more than doubled between 1980 and 1999 (Cuthbert and Wires 1999). One reason for the increase is that human-created habitat provides high quality nest sites and is associated with population increases in many parts of North America (Cuthbert and Wires 1999).

In the early 1990s, a substantial increase in the size of a newly established Caspian tern nesting colonies on man-made islands in the Columbia River estuary was noted by NOAA Fisheries staff. Several estuary islands on which piscivorous birds nest were created from or augmented by materials dredged to maintain the Columbia River Federal Navigation Channel. Before 1984, there were no recorded observations of terns nesting in the Columbia River estuary, when approximately 1000 pairs apparently moved from Willapa Bay to nest on newly deposited dredge material on East Sand Island. In 1986, those birds moved to Rice Island. The Caspian tern colonies in the estuary have since expanded to 9,000-10,000 pairs, the largest ever reported. In 1999, the colony was encouraged to relocate to East Sand Island. In 2001, the majority of the West Coast population nested on just four acres on East Sand Island; in 2002, the terns nested on six acres.

Caspian terns arrive in the Columbia River estuary in April and begin nesting at the end of the month (Roby *et al.* 1998). To avoid mammal and avian predators, terns construct their nests on islands (Harrison 1984) and show a preference for barren sand. They are piscivorous in nature (Harrison 1984), requiring about 220 grams (roughly one-third of their body weight) of fish per day during the nesting season. 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) (Figure 23).

Salmon and steelhead constitute a major portion of tern diets, particularly when the birds nested on Rice Island. Diet analyses indicated that juvenile salmonids constituted 77.1% of prey items in 1997 and 72.7% of prey items of Caspian terns nesting
on Rice Island (Collis et al. 2002). During the May peak of smolt out-migration of steelhead, yearling chinook salmon, and coho salmon through the estuary, when Caspian terns are in their incubation period, the diet of Caspian terns was consistently over 90% juvenile salmonids (Collis et al. 2002). This concentration on salmon as a food source translates into substantial juvenile mortality during the outmigration period.

Efforts to relocate the terns to East Sand Island from Rice Island that began in 1999, have succeeded in reducing consumption of smolts without affecting tern productivity. East Sand Island is a site lower in the estuary with abundant alternate prey sources. Over the last few years, consumption of salmonids in the estuary has been lower than previous levels, while consumption of alternative prey species has increased. Relocating the colony to East Sand Island, which is lower in the estuary and closer to periodically abundant Pacific herring [Clupeidae] and anchovies [Engraulidae] has contributed to the reduction. Observed diets, which consisted of almost exclusively salmonids at Rice Island (77% in 1999 and 90% in 2000), shifted to 46%, 47% and 33% salmonids at East Sand Island in 1999, 2000 and 2001 respectively (Collis et al. 2001a, Roby et al. 2003). These data represent substantial declines in juvenile salmonid mortalities from Caspian tern predation. In 2000, salmonid consumption was estimated at 7.3 million smolts, which is 4.4 million less than in 1999--the last time a substantial number of terns nested on Rice Island (Collis et al. 2001a, USFWS 2001). In 2001, salmonid consumption was estimated at 5.9 million smolts, which is 5.9 million less than in 1999 (Collis et al. 2001a). These observational data were substantiated by PIT tag detections on the two islands in 1999 and 2002. Significantly fewer PIT tags (approximately 2 to 4 times fewer tags per pair of terns) were detected per nest on East Sand Island in 1999 and 2000 than were detected on Rice Island in 1999 and 2000.

In a recent evaluation of the impact of Caspian tern predation on juvenile salmon (Good et al. 2003), two approaches were recognized as providing the types of predation rate estimates needed to incorporate into a salmon life cycle models that are used to assess the effects of various factors on risk of extinction in the Columbia River basin (Kareiva et al. 2001). One approach to evaluating the extent of Caspian tern predation and resultant salmonid mortality uses bioenergetics modeling. Since 1997, biologists with the Bonneville Power Administration-funded research project ("Avian Predation on Juvenile Salmonids in the Lower Columbia River," - a joint project of Oregon State University, the U. S. Geological Survey, the Columbia River Inter-Tribal Fish Commission, and Real Time Research Consultants) have used observed salmonid consumption at tern colonies in a bioenergetics model (Roby et al. 1998) to estimate the consumption of salmonids in the Columbia River estuary. Smolt consumption, estimated from bioenergetics modeling, from 1999 to 2002 ranged from a low of 5.9 to a high of 11.7 million smolts eaten.

Another approach uses 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. 2001b, Ryan et al. 2001). Since 1987, researchers in the Columbia River basin have placed over five million PIT tags in juvenile salmonids for a variety of studies (Ryan et al. 2001). Identifying PIT tags on bird colonies can provide a minimum
estimate of proportion of the stocks that were consumed by terns in these colonies. In recent years, approximately one million juvenile salmonids have been PIT-tagged annually (Collis et al. 2001b), the vast majority of which are steelhead and chinook from the Snake River basin. Using PIT tag detection equipment, over 115,000 PIT tags were detected on Rice Island in 1998 and 1999 (Ryan et al. 2001). Collis et al. (2001b) indicate that the majority of these PIT tags detected were from steelhead and chinook, coho and sockeye salmon. Of the PIT tags placed in steelhead smolts in 1997 that were detected at Bonneville dam, 2.8% of wild smolts and 5.4% of hatchery-raised smolts were subsequently detected on the Rice Island tern colony (Collis et al. 2001b). For steelhead that were PIT-tagged in 1998 and detected at Bonneville Dam, 11.7% of wild smolts and 13.4% of hatchery-raised smolts were subsequently detected on the Rice Island tern colony (Collis et al. 2001b). For yearling chinook salmon PIT-tagged in 1998 and detected at Bonneville Dam, 0.5% of wild smolts and 1.6% of hatchery-raised smolts were subsequently detected on the Rice Island tern colony (Collis et al. 2001b).

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 of salmon recovery, efforts focused on determining if a unique predation rate could be identified. The effort focused on the Caspian tern colonies on East Sand Island in the lower estuary of the Columbia River because the colony currently represents the majority of the West Coast Caspian tern population. The focus period was 1999-2002 because this represents the time period (after relocation from Rice Island) during which this colony has dominated Caspian tern predation activity in the Columbia River estuary. Bioenergetics modeling, which has been used to estimate the effect of Caspian tern predators on juvenile salmonids on Rice Island (Roby et al. 2003), was used to calculate predation rates (estimated # of salmon consumed/estimated # of salmon available in the estuary) using updated and refined estimates of the number of outmigrating salmon that run the river or are transported to below Bonneville Dam. Similarly, PIT tag detections, which have also been used to estimate the effect of Caspian tern predators on juvenile salmonid outmigrants (Ryan et al. 2003), were used to generate estimates of predation rates on salmon. Although the relationship between tern abundance and predation rate is not known with certainty, the estimates (using either bioenergetics modeling or PIT tag data) showed 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 (Figure 31). Moreover, PIT tag detection allows possible ESU-specific predation rate estimates to be derived. Support for a linear relationship between estimates of predation rate and the number of terns nesting on East Sand Island comes from per capita consumption rates (# of smolts consumed/adult tern), which have been relatively constant throughout the range of colony sizes on East Sand Island from 1999-2003. 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 (Figure 32).
NOAA Fisheries has developed a life cycle model under the auspices of the Cumulative Risk Initiative (CRI) at the Northwest Fisheries Science Center to assess salmonid population trends and the impact of an anthropogenic activity on those trends. This model has application when mortality rates can be constructed and attributed to a particular source. The value of life cycle models derive from providing an objective outcome for comparing the influence of various factors influencing population growth rates, rather than attempting to estimate real gains from any management action. Assessing the impact of predation by Caspian terns on juvenile salmonids during a particular life history phase was amenable to such evaluation.

Using the CRI model, Good et al. (2003) estimated the impact of Caspian tern predation on the population growth rate ($\lambda$) 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 $\lambda$, 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 $\lambda$ 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.

The PIT tag detection approach enables the calculation of ESU-specific estimates of predation rate (and hence proportion increase in $\lambda$). Good et al (2003) used the CRI model to estimate impact of Caspian tern predation on the population growth rate ($\lambda$) of steelhead and spring Chinook salmon ESUs using predation rates estimated from PIT tag detections for which reasonable estimates of the number of smolts available to be consumed could be generated. Predation rates for 20,000 Caspian terns on four of the five ESA-listed steelhead and spring Chinook salmon ESUs were estimated using linear regression. The maximum proportional increase in $\lambda$ corresponding to complete elimination of mortality due to tern predation ranged from 1.9% to 4.9% for steelhead ESUs.

Several factors must be considered when interpreting the results of these calculations. Perhaps the most important factor is that this type of calculation assumes that there is no compensatory mortality later in the life cycle, and that any reduction in tern predation is fully realized. In their assessment of predation impact by Rice Island terns on salmonids in 1997-1998, Roby et al (2003) hypothesized that tern predation was 50% additive. Given these limitations and uncertainties, the estimates of percent change
in population growth rates should be viewed as maximum potential improvements. Realized improvements in population growth would likely be lower from any management action that reduces Caspian tern predation impacts on salmonid ESUs. These results may not be as easy to achieve as they are to calculate. 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 affects 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 a result of the dominant migratory periods employed by salmonids with a stream type life history. Salmon from this life history type move in great numbers coincident 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, characteristic of fall Chinook in the Columbia River basin, the impact is less than for the stream type salmonids (Roby et al. 2003). Good et al. (2003) concluded that gains in $\lambda$ 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 $\lambda$ by managing terns were considered to benefit other salmon ESUs in the basin, albeit to a much lesser degree. In summary, consistent with the desire to rank the contribution of Caspian tern predation on juvenile salmon use of the Columbia River estuary, we conclude that Caspian tern predation has a likely high to medium impact on recovery potential for salmon life history strategies that exhibit a stream type life history pattern and medium to low for salmon that exhibit an ocean type life history strategy.
LIMITING FACTORS ANALYSIS SYNTHESIS

The overall purpose of this document is to evaluate and rank candidate limiting factors in the estuary with respect to the potential of each factor to affect population status or suppress population specific recovery. Such an analysis is intended to help facilitate recovery of endangered salmon stocks in the Columbia River basin. Identifying and then fixing factors that are having significant affects on viability is expected to improve the recovery potential of populations. In addition, valuing the influence of such factors in policy and management decisions should, at the very least, improve the status of these populations. The focus of this analysis was on effects of estuarine factors on population viability, not on the relative importance of the estuary relative to other factors operating at other life stages. This broader analysis is outside the scope of this review and will be considered in the Integration Phase of the overall life cycle analysis.

Our general approach was to assess how limiting factors in the estuary would affect viability of salmon populations. Because population specific data on effects of limiting factors was unavailable, the endpoint of this approach was an evaluation of how each limiting factor addressed in this review could change the distribution and performance of life history strategies limiting factors was unavailable, our approach was to evaluate how each limiting factor addressed in this review could viability, especially the distribution and performance of life history strategies expressed by the two major life history types present in the Columbia River basin: stream type and ocean type salmonids (Tables 1-2). The overall effect of each estuarine factor in affecting population viability or of viability of different life history strategies was summarily ranked as having a high, medium, or low effect.

The first step in this analysis was to assign each ESU to a dominant life history type. The different life history strategies employed by the populations and the likely mixes of life history strategies for each life history type representing each ESU under historic conditions were then defined (Table 3). This relied, in large part, upon Rich’s (1919) analyses of timing of different life history strategies in the estuary. He demonstrated that all life history strategies were evident for long periods of time throughout the year, which we term the historic norm. We recognize the limitations of the data set (Rich collected information for only 2 years), as characterized by Burke (2001) and the extrapolations we have made. However, the data set does represent the most complete historical assemblage available to characterize life history patterns in the Columbia River estuary. The likely mix of life history strategies employed by each life history type and the characteristic reduction in life history expression by each life history type evident today was defined for current conditions (Table 3) from Bottom et al. (2001). Finally, the role of four estuarine related factors in affecting change (i.e., the role of water flow, availability of salmon specific habitats, predation by Caspian terns, and toxics effects on the quality of habitats) was examined for several different habitat types within the estuary and plume.
In this section, integration of ESU’s, life history type, life history strategy, limiting factor impacts, habitat attributes, and population viability are linked and summarized. The intent of this analysis is to compare and contrast the importance of each limiting factor on population viability for each ESU as defined by their life history type. This is done by considering the importance or relative ability of each limiting factor to affect the existence or at the very least suppress expression of a life history strategy (i.e., represent a potential causal factor for difference in life history strategy contribution to the viability of an ESU listed in Tables 5-6) within different habitat types in the estuarine and plume environments. By linking the impact of a limiting factor to expression of life history strategies, we draw inferences concerning the potential changes in overall population viability caused by a limiting factor. Factors are ranked as having a high, medium, or low ability to affect change in population viability. For flow, habitat, and toxics, the analysis considers the ability of a limiting factor to affect the availability and the quality of habitats used by salmonids; in the case of tern predation, direct affects on population growth rate are considered.

A weakness of our analysis is the resolving power to accurately discriminate impacts of limiting factors between estuarine zones or habitat types within each zone. As discussed in Section III, there are at least four major zones in the estuary (see section II) and many habitat types within each zone (Table 4); habitat use, timing, and distribution of juvenile salmon likely varies between and within each zone. Knowledge of how different juvenile life history strategies specifically use these habitats and zones is largely absent. Moreover, our present knowledge base is not robust enough to acknowledge differential effects of limiting factors on either habitat within a zone or between zones or to be extensively discriminatory. Our ability to draw inferences regarding estuarine limiting factors on habitat type is primarily based upon how depth and velocity is affected by a limiting factor regardless of where these types of habitats in the estuary are located. To address these limitations, we collapsed the estuary from Bonneville to the mouth into one zone and the plume as a second major zone. Within the estuary, 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 while the plume was considered as one habitat unit. Our inability to discriminate limiting factor other than at a coarse scale does not imply that development of recovery actions or future research should not consider smaller spatial scales. Other factors (such as landscape connectivity within and between habitat zones) will need to be considered in the future when considering how to distribute recovery efforts.

Limiting factors were subsequently ranked as having a high, medium, or low ability to limit the viability of particular life history strategies. Recall that strategies salmonids utilize are linked to viability of the life history type/ESU overall completing the link from limiting factor impact to ESUs. To guide our determination of how limiting factors could affect viability of life history types and subsequently salmonid populations (Tables 5-6), we simultaneously considered the effects of a limiting factor on a specific habitat type (i.e., did it cause a significant change), direct impacts to juvenile salmon, and impacts on population viability. We assumed if there was a significant affect on quality of the habitat or a significant reduction in abundance of fish rearing in the habitat, that
this change affected all VSP parameters. If fish were rare in a habitat and there was little likelihood of a change in habitat quality, we focused our consideration primarily on potential impacts on spatial structure and diversity.

To more formally guide our evaluation, we used the following hypotheses or assumptions about the effects of specific limiting factors. These hypotheses embody the rules used in the limiting factor analyses.

1. Tern predation differentially affects the larger yearling strategies, especially steelhead, more than smaller fish such as fingerling chinook (Ryan et al. 2003). Tern predation is assumed to be distributed in both the plume and estuary zones.

2. The main effect of flow reductions is to affect amount of habitat available to fish; the main effects of habitat changes is on distribution, quantity and quality of habitat, whereas toxics affects habitat quality.

3. Any reduction in quality or quantity of shallow water habitat affects smaller juvenile salmonids employing strategies such as fry and fingerlings more than subyearlings and yearlings.

4. Subyearling salmonid habitat use is primarily of medium and deep channel habitat.

5. Fry and early fingerling life history strategies do not move into the plume, but more likely utilize the surf zone if they exit the estuary proper.

6. A reduction in the amount of shallow water habitat reduces the capacity of the estuary to rear fish, thereby reducing spatial structure and diversity. Changes in flow in the estuary primarily affect all VSP parameters. Tern predation affects diversity and productivity/abundance parameters and less spatial structure of habitat.

7. Reductions in flow above Bonneville delay arrival timing of all migrants in the estuary and the time it takes fish to migrate through the estuary.

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 impact occurs. Toxics are primarily assumed to be a problem in the riverine tidal zone.

9. Cumulative impacts within any habitat were not considered.

Within the estuary (Table 5), we conclude that the most important limiting factors are flow and habitat changes and that their major affect is on shallow water habitats and life history strategies that depend upon these habitats. This conclusion is based upon the following. First, flow and habitat has a significant affect on ocean type populations because these populations are dominated by small size classes (fry and fingerlings), make extensive use of shallow water habitats, and have the longest residence time in the estuary (i.e., they are most dependent upon shallow water habitats) (e.g., Bottom et al.
A major function of these shallow water habitats for these small size classes is to support their feeding and growth; high growth rates experienced here can help population members avoid some of the predation that these small fish experience (Simenstad et al. 1982). Yearling and subyearling fish are generally not in the habitats where they are most vulnerable to the effects of these two limiting factors. Because the estuary may function for these size classes primarily as a physiological transition and migratory corridor, loss of these habitats is less critical.

Secondly, the affect of both flow and habitat changes has primarily been on both quality and quantity of shallow water habitats. As summarized by Thomas (1983) and Johnson et al. (2003), the losses of shallow water habitats due to the combined effects of flow and habitat changes are dramatic while losses of deeper water habitats appear to be minimal. We also predict that these changes will also increase as a result of climate change effects that appear likely to occur in the future. Third, while toxic contamination affects ocean type populations, we predict that few population members are affected because effects are local and that they are primarily those fish using shallow water habitats (i.e., primarily fry and fingerlings).

Fourth, tern predation is primarily focused on larger members of both stream and ocean type populations which are less abundant than smaller life history strategies in shallow water areas. As a result, steelhead and salmon populations with significant numbers of subyearlings/yearlings appear to be especially at risk to this limiting factor. Most populations that produce yearlings are stream type populations which are largely associated with deeper water habitats so in general we expect that these populations are most at risk to tern predation. The proportion of tern predation occurring in each habitat is unknown but we assumed that because it is distributed over both environments (plume and estuary), this reduced the magnitude of this risk factor from what would be expected if predation was primarily restricted to only one habitat. The effect of tern predation will primarily be on the diversity of ocean type populations and have less of an effect on other VSP measures (Table 7). For stream type populations, tern predation will affect all VSP measures.

Finally, both the flow and habitat limiting factors affect all VSP parameters for ocean type populations. The loss of shallow water habitat and changes in its distribution and quality caused by flow and habitat changes will reduce the capacity of estuarine habitats to support ocean type populations; this will reduce abundance and productivity of these populations. Further, loss and degradation of shallow water habitat will also diminish the spatial structure and number of life history pathways available to the fish. This has the potential to make these populations more vulnerable to effects of extreme events such as severe droughts or strong El Nino events. Because relatively few stream type fish use shallow water habitats, we predict that flow and habitat will have less of an effect on capacity of these habitats to rear and support stream type populations but a more significant affect on diversity and spatial structure of these populations (Table 7). 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. This will increase ability of these populations to persist in the face of environmental extremes such as droughts or strong El Ninos.

From the perspective of ocean type populations in the estuary, we therefore conclude that changes in the quantity and quality of shallow water habitats most impacts viability of these populations. Although our approach here focused on evaluating effects of single limiting factors, any recovery actions directed at shallow water habitats should consider the cumulative effects of all factors affecting ocean type populations in these habitats. This may necessitate a more habitat based view of recovery actions rather than one simply focused on single limiting factors. Thus, some recovery actions directed at habitat restoration in shallow water areas may need to address toxic contamination in combination with actions directed at flow and habitat.

Within the Columbia River plume, we conclude that tern predation and flow alterations are the most important limiting factors (Table 6). Data that we reviewed suggest that terns appear to prey primarily on larger life stages. As a result, steelhead and other populations with significant numbers of yearlings appear to be especially at risk to this limiting factor. We ranked this factor as medium, however because we assume that not all tern predation occurs in the plume; some predation occurs in the estuary. The proportion of tern predation occurring in each habitat is unknown but we assume that because it is distributed over both environments (plume and estuary), that this would reduce the magnitude of this risk factor from what would be expected if predation was primarily restricted to only one habitat.

We conclude that flow alterations are a significant limiting factor in the plume. Primarily as a result of flow but also know doubt also a result of physical changes to the estuary (e.g., dredging and diking), the shape, behavior, size, and composition of the plume has been changed. For example, because water that would have moved downstream and been part of the plume is stored, affecting spring runoff, the size of the plume in spring is smaller than it was historically. Altering dynamics of the plume affects the roles identified to date, potentially affecting juvenile salmon beneficial use of this habitat. Benefits are likely to including enhanced growth, reduced predation rates, diet, and increased residence time in coastal areas. Because it is known that larger fingerlings, subyearlings, and yearling life stages associated with both stream and ocean type populations use the plume, all these life stages are at risk to flow changes in the plume. Since stream type populations are dominated by these larger life stages, we predict that stream type populations are most at risk to effects of flow changes in the plume. Moreover, all elements of population viability are potentially affected by flow changes to the plume. For example, a reduction in size of the plume could reduce the capacity of this habitat to be productive while loss of any of these life history strategies will affect diversity (Table 4).

A major uncertainty in evaluating limiting factor effects in the plume environment is use of the plume by small life history strategies such as fry. While sampling methods clearly find larger juveniles in the plume, fry and smaller fingerlings (< 80mm) have not been captured. This may be because small fish do not use the plume or that the sampling
methods and when samples in the plume have been collected simply missed these small fish. We observed small juvenile salmon at the entrance to the plume (at the mouth of the Columbia River estuary) and speculate they, at the very least, are tidally flushed into the plume region. Because of the affinity of the smaller juvenile stages to associate with shoreline habitat, juveniles utilizing fry and fingerling strategies may be more aligned to the surf zone, rather than the region encompassed by the plume proper that is currently being sampled. This is an uncertainty that needs to be addressed in the future.

This analysis has only considered the effects of four limiting factors within the plume and estuary on viability of salmon populations. Other factors, identified earlier and not considered here may also have a role in recovery of anadromous species in the basin. The evaluation of these four factors was based upon the relative effects of each limiting factor within these two environments and the availability of empirical information to include in the evaluation. Decisions on the overall role of the estuary within the context of all recovery actions will occur as part of additional analyses that are being conducted. Such comparative analyses will be challenging due to variability in information and approaches used to assess limiting factors in all life stages. In general, we suggest a strategy that focuses on crafting and evaluating alternative scenarios involving all life stages of the animals can be a useful approach. These evaluations will likely require the use of a number of measurements including VSP parameters, landscape principles, costs, risk, and uncertainty. Even within any one habitat such as the estuary or tributary, a variety of factors (such as landscape connectivity within and between habitat zones) will need to be considered in deciding how to distribute recovery efforts directed at any suite of limiting factor.

The questions that should be asked in any comprehensive analyses should not revolve around if the estuary has a role. We believe the available empirical knowledge base now being acquired demonstrates that the estuary and plume have a role in salmon recovery. We recommend that questions focus more on what conditions should the estuary and other non-estuarine habitats that are part of the landscape salmon be restored to rather than should the Columbia River estuary be included in recovery considerations. Some level of functioning conditions are needed in all habitats, including the estuary, to restore viability of Columbia Basin salmon and steelhead populations. Here, we have provided guidance on what factors need to be addressed in the estuary environment. Our history of continuously trying to increase salmon numbers by focusing on only one life stage or another of the animal suggests that more comprehensive approaches that address needs of the fish at all life stages are needed.
ACKNOWLEDGEMENTS

We thank Jennifer Burke for her insights and discussions on many aspects of salmon use of the Columbia River Estuary. Critical reviews of this document were provided by (to be added)
REFERENCES


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.


Burke 2001


Dawley 1989


Emmett et al. 2003


Foster, EP; Fitzpatrick, MS; Feist, GW; Schreck, CB; Yates, J; Spitsbergen, JM; Heidel, JR. 2001b. Plasma Androgen Correlation, EROD Induction, Reduced Condition Factor, and the Occurrence of Organochlorine Pollutants in Reproductively Immature White Sturgeon (Acipenser transmontanus) from the Columbia River, USA. Arch. Environ. Contam. Toxicol. 41:182-191.


Garono et al. 1995

Garono et al. 2002


Greene and Beechie in press


Healey, M.C., and W.R. Heard. 1984. Inter- and intra-population variation in the fecundity of chinook salmon (Oncorhynchus tshawytscha) and its relevance to life history theory. Canadian Journal of Fisheries and Aquatic Science 41:476-483.


Jay 2001


LCREP 1999

Levin


Murphy et al. 1988


Simenstad et al. 1978


TABLES
Table 1. A summary of the characteristics of stream-type and ocean-type salmonids as compiled from various literature sources such as Myers et al. (1998) and Healey (1991).

<table>
<thead>
<tr>
<th>Stream Type Fish</th>
<th>Ocean Type Fish</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Species</strong></td>
<td><strong>Species</strong></td>
</tr>
<tr>
<td>Yearling coho</td>
<td>Subyearling chinook</td>
</tr>
<tr>
<td>Yearling chinook</td>
<td>Pink</td>
</tr>
<tr>
<td>Steelhead</td>
<td>Chum</td>
</tr>
<tr>
<td>Yearling Sockeye</td>
<td>Coho Fry</td>
</tr>
<tr>
<td><strong>Attributes</strong></td>
<td><strong>Attributes</strong></td>
</tr>
<tr>
<td>Long period of fw rearing (&gt;1yr)</td>
<td>Short period of fw rearing</td>
</tr>
<tr>
<td>Shorter ocean residence</td>
<td>Longer ocean residence</td>
</tr>
<tr>
<td>Uses ocean north of ocean type</td>
<td>Uses ocean south of stream type</td>
</tr>
<tr>
<td>Short period of estuarine rearing</td>
<td>Can use marshes extensively</td>
</tr>
<tr>
<td>Reduced use of marshes</td>
<td>Small size at time of migration</td>
</tr>
<tr>
<td>Large size at time of migration</td>
<td></td>
</tr>
</tbody>
</table>
Table 2. Description of life history types and selected attributes associated with Columbia River anadromous salmonid populations based upon **historic** use of the system. Various sources of information, such as Bottom et al. (2001) and J. Burke (NOAA, Fisheries, personal communication), were used to develop this table. Sizes and times should be considered estimates.

<table>
<thead>
<tr>
<th>Life History Strategy</th>
<th>Attributes</th>
</tr>
</thead>
</table>
| Early fry             | Time of estuarine entry: March- April  
|                       | Size at estuarine entry: <500mm  
|                       | Estuarine residence time: 0-40d  
|                       | Freshwater rearing: 0-60d  |
| Late fry              | Time of estuarine entry: May-June, present thru Sept.  
|                       | Size at estuarine entry: <60mm  
|                       | Estuarine residence time: < 50d  
|                       | Freshwater rearing: 20-60d  |
| Early fingerling      | Time of estuarine entry: April-May  
|                       | Size at estuarine entry: 60-100mm  
|                       | Estuarine residence time: < 50d  
|                       | Freshwater rearing: 60-126d  |
| Late fingerling       | Time of estuarine entry: June-Oct, present thru winter  
|                       | Size at estuarine entry: 60-130d  
|                       | Estuarine residence time: 0-80d  
|                       | Freshwater rearing: 50-180d  |
| Subyearling (smolt)   | Time of estuarine entry: April-Oct  
|                       | Size at estuarine entry: 40-130d  
|                       | Estuarine residence time: little  
|                       | Freshwater rearing: 20-180d  |
| Yearling              | Time of estuarine entry: Feb-May  
|                       | Size at estuarine entry: >80mm  
|                       | Estuarine residence time: little  
|                       | Freshwater rearing: extended  |
Table 3. Linkage between salmonid ESU, life history type (ocean or stream type), and dominant life history strategies of juvenile salmon in the Columbia River. General contribution to the outmigrant population represented as Abundant (>50%), Medium (10-50%), Rare (1-9%), or Absent (<1%) are listed for each ESU under historic (early 1900s) and current conditions (historic/current).

<table>
<thead>
<tr>
<th>ESU</th>
<th>Life History Type</th>
<th>Life History Strategy</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Early Fry</td>
<td>Late Fry</td>
</tr>
<tr>
<td>Lower Columbia River</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Chum Salmon</td>
<td>Ocean</td>
<td>High</td>
</tr>
<tr>
<td>Snake River Sockeye</td>
<td>Stream</td>
<td>Absent</td>
</tr>
<tr>
<td>Salmon</td>
<td></td>
<td>Absent</td>
</tr>
<tr>
<td>Upper Columbia River</td>
<td>Stream</td>
<td>Absent</td>
</tr>
<tr>
<td>Steelhead</td>
<td></td>
<td>Absent</td>
</tr>
<tr>
<td>Snake River Steelhead</td>
<td>Stream</td>
<td>Absent</td>
</tr>
<tr>
<td>Middle Columbia River</td>
<td>Stream</td>
<td>Absent</td>
</tr>
<tr>
<td>Steelhead</td>
<td></td>
<td>Absent</td>
</tr>
<tr>
<td>Upper Willamette River</td>
<td>Stream</td>
<td>Absent</td>
</tr>
<tr>
<td>Steelhead</td>
<td></td>
<td>Absent</td>
</tr>
</tbody>
</table>

Preliminary draft, 19 December 2003
Table 3. Linkage between salmonid ESU, life history type (ocean or stream type), and dominant life history strategies of juvenile salmon in the Columbia River. General contribution to the outmigrant population represented as Abundant (>50%), Medium (10-50%), Rare (1-9%), or Absent (<1%) are listed for each ESU under historic (early 1900s) and current conditions (historic/current).

<table>
<thead>
<tr>
<th>ESU</th>
<th>Life History Type</th>
<th>Early Fry</th>
<th>Late Fry</th>
<th>Early Fingerling</th>
<th>Late Fingerling</th>
<th>Subyearling</th>
<th>Yearling</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lower Columbia River Chum Salmon</td>
<td>Ocean</td>
<td>High</td>
<td>High</td>
<td>Absent</td>
<td>Absent</td>
<td>Absent</td>
<td>Absent</td>
</tr>
<tr>
<td>Snake River Sockeye Salmon</td>
<td>Stream</td>
<td>Absent</td>
<td>Absent</td>
<td>Absent</td>
<td>Absent</td>
<td>Rare</td>
<td>Abundant</td>
</tr>
<tr>
<td>Upper Columbia River Steelhead</td>
<td>Stream</td>
<td>Absent</td>
<td>Absent</td>
<td>Absent</td>
<td>Absent</td>
<td>Rare</td>
<td>Abundant</td>
</tr>
<tr>
<td>Snake River Steelhead</td>
<td>Stream</td>
<td>Absent</td>
<td>Absent</td>
<td>Absent</td>
<td>Absent</td>
<td>Rare</td>
<td>Abundant</td>
</tr>
<tr>
<td>Middle Columbia River Steelhead</td>
<td>Stream</td>
<td>Absent</td>
<td>Absent</td>
<td>Rare</td>
<td>Rare</td>
<td>Medium</td>
<td>Abundant</td>
</tr>
<tr>
<td>Upper Willamette River Steelhead</td>
<td>Stream</td>
<td>Absent</td>
<td>Absent</td>
<td>Absent</td>
<td>Absent</td>
<td>Rare</td>
<td>Abundant</td>
</tr>
</tbody>
</table>
Table 3. Continued.

<table>
<thead>
<tr>
<th>ESU</th>
<th>Life History Type</th>
<th>Early Fry</th>
<th>Late Fry</th>
<th>Early Fingerling</th>
<th>Late Fingerling</th>
<th>Subyearling</th>
<th>Yearling</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lower Columbia River Steelhead</td>
<td>Stream</td>
<td>Absent</td>
<td>Absent</td>
<td>Rare</td>
<td>Rare</td>
<td>Medium</td>
<td>Abundant</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Absent</td>
<td>Absent</td>
<td>Absent</td>
<td>Absent</td>
<td>Rare</td>
<td>Abundant</td>
</tr>
<tr>
<td>Upper Columbia River Spring</td>
<td>Stream</td>
<td>Absent</td>
<td>Absent</td>
<td>Rare</td>
<td>Rare</td>
<td>Rare</td>
<td>Abundant</td>
</tr>
<tr>
<td>Chinook Salmon</td>
<td></td>
<td>Absent</td>
<td>Absent</td>
<td>Absent</td>
<td>Absent</td>
<td>Rare</td>
<td>Abundant</td>
</tr>
<tr>
<td>Snake River Spring/Spring/Summer</td>
<td>Stream</td>
<td>Absent</td>
<td>Absent</td>
<td>Rare</td>
<td>Rare</td>
<td>Rare</td>
<td>Abundant</td>
</tr>
<tr>
<td>Chinook Salmon</td>
<td></td>
<td>Absent</td>
<td>Absent</td>
<td>Absent</td>
<td>Absent</td>
<td>Rare</td>
<td>Abundant</td>
</tr>
<tr>
<td>Snake River Fall Chinook Salmon</td>
<td>Ocean</td>
<td>Absent</td>
<td>Absent</td>
<td>Medium</td>
<td>Medium</td>
<td>Medium</td>
<td>Abundant</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Absent</td>
<td>Absent</td>
<td>Medium</td>
<td>Medium</td>
<td>Medium</td>
<td>Abundant</td>
</tr>
<tr>
<td>Upper Willamette Chinook Salmon</td>
<td>Ocean</td>
<td>Rare</td>
<td>Rare</td>
<td>Medium</td>
<td>Medium</td>
<td>Rare</td>
<td>Abundant</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Absent</td>
<td>Absent</td>
<td>Rare</td>
<td>Rare</td>
<td>Medium</td>
<td>Abundant</td>
</tr>
<tr>
<td>Lower Columbia River Fall Chinook</td>
<td>Ocean</td>
<td>Medium</td>
<td>Medium</td>
<td>Medium</td>
<td>Medium</td>
<td>Medium</td>
<td>Rare</td>
</tr>
<tr>
<td>Salmon</td>
<td></td>
<td>Medium</td>
<td>Medium</td>
<td>Medium</td>
<td>Medium</td>
<td>Medium</td>
<td>Rare</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Rare</td>
<td>Rare</td>
<td>Rare</td>
<td>Rare</td>
<td>Abundant</td>
<td>Rare</td>
</tr>
</tbody>
</table>
Table 4. Major types of estuarine habitats and some of their important attributes in the Columbia River below RM 46 (partially after Thomas 1983 and Johnson et al. 2003).

<table>
<thead>
<tr>
<th>Major habitat types</th>
<th>Important Attributes</th>
</tr>
</thead>
<tbody>
<tr>
<td>Tidal swamps</td>
<td>Vegetation is mostly shrub and woody species. Higher elevations.</td>
</tr>
<tr>
<td>Tidal marshes</td>
<td>Dominant vegetation varies. Includes emergent marshes. Tidal channels often present. Depths generally range from MLLW to above MHHW.</td>
</tr>
<tr>
<td>Tidal flats</td>
<td>Range in depth between MLLW and 6 ft below MLLW. Usually not vegetated.</td>
</tr>
<tr>
<td>Medium deep</td>
<td>Depth range: 3-18 ft.</td>
</tr>
<tr>
<td>Deep</td>
<td>&gt; 18 feet in depth.</td>
</tr>
</tbody>
</table>
Table 5. Linkages between limiting factors associated with the estuary, the expression of life history strategies, life history type (ocean type and stream-type) and ESU (only examples are provided—see Table 3). Only two, general habitat types in the estuary were considered—shallow, low velocity and deep, channel habitats. Factors were ranked as having a high, medium, or low ability to limit the viability of particular life history strategies and viability of the life history type/ESU overall.

<table>
<thead>
<tr>
<th>ESU</th>
<th>Life history type</th>
<th>Life history strategy</th>
<th>Shallow, low velocity</th>
<th>Deep, channel</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Flow</td>
<td>Habitat</td>
<td>Terns</td>
</tr>
<tr>
<td>SR Fall,</td>
<td>Early fry</td>
<td>High</td>
<td>High</td>
<td>Low</td>
</tr>
<tr>
<td>LCRC,</td>
<td>Late fry</td>
<td>High</td>
<td>High</td>
<td>Low</td>
</tr>
<tr>
<td></td>
<td>Early fingerling</td>
<td>High</td>
<td>High</td>
<td>Low</td>
</tr>
<tr>
<td></td>
<td>Late fingerling</td>
<td>High</td>
<td>High</td>
<td>Low</td>
</tr>
<tr>
<td></td>
<td>Subyearling</td>
<td>Low</td>
<td>Low</td>
<td>Medium</td>
</tr>
<tr>
<td></td>
<td>Yearling</td>
<td>Low</td>
<td>Low</td>
<td>Medium</td>
</tr>
<tr>
<td></td>
<td>Overall</td>
<td>High</td>
<td>High</td>
<td>Low</td>
</tr>
<tr>
<td>LCRS,</td>
<td>Early fry</td>
<td>Medium</td>
<td>Medium</td>
<td>Low</td>
</tr>
<tr>
<td>UCRSC</td>
<td>Late fry</td>
<td>Medium</td>
<td>Medium</td>
<td>Low</td>
</tr>
<tr>
<td></td>
<td>Early fingerling</td>
<td>Medium</td>
<td>Medium</td>
<td>Low</td>
</tr>
<tr>
<td></td>
<td>Late fingerling</td>
<td>Medium</td>
<td>Medium</td>
<td>Low</td>
</tr>
<tr>
<td></td>
<td>Subyearling</td>
<td>Low</td>
<td>Low</td>
<td>Medium</td>
</tr>
<tr>
<td></td>
<td>Yearling</td>
<td>Low</td>
<td>Low</td>
<td>Medium</td>
</tr>
<tr>
<td></td>
<td>Overall</td>
<td>Medium</td>
<td>Medium</td>
<td>Low</td>
</tr>
</tbody>
</table>
Table 6. Linkages between limiting factors associated with the plume, the expression of life history strategies, life history type (ocean type and stream-type) and ESU. The plume was only considered as one habitat zone. Factors were ranked as having a high, medium, or low ability to affect distribution of life history strategies.

<table>
<thead>
<tr>
<th>ESU</th>
<th>Life History Type</th>
<th>Life History Strategy</th>
<th>Plume</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Flow</td>
<td>Habitat</td>
</tr>
<tr>
<td>SR Fall,</td>
<td>Ocean-Type</td>
<td></td>
<td></td>
</tr>
<tr>
<td>LCRC, WC</td>
<td>Early fry</td>
<td>Low</td>
<td>Low</td>
</tr>
<tr>
<td></td>
<td>Late fry</td>
<td>Low</td>
<td>Low</td>
</tr>
<tr>
<td></td>
<td>Early fingerling</td>
<td>Medium</td>
<td>Low</td>
</tr>
<tr>
<td></td>
<td>Late fingerling</td>
<td>Medium</td>
<td>Low</td>
</tr>
<tr>
<td></td>
<td>Subyearling</td>
<td>Medium</td>
<td>Low</td>
</tr>
<tr>
<td></td>
<td>Yearling</td>
<td>Medium</td>
<td>Low</td>
</tr>
<tr>
<td></td>
<td>Overall</td>
<td>Medium</td>
<td>Low</td>
</tr>
<tr>
<td>LCRS, UCRSC</td>
<td>Stream-Type</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Early fry</td>
<td>Low</td>
<td>Low</td>
</tr>
<tr>
<td></td>
<td>Late fry</td>
<td>Low</td>
<td>Low</td>
</tr>
<tr>
<td></td>
<td>Early fingerling</td>
<td>Medium</td>
<td>Low</td>
</tr>
<tr>
<td></td>
<td>Late fingerling</td>
<td>Medium</td>
<td>Low</td>
</tr>
<tr>
<td></td>
<td>Subyearling</td>
<td>Medium</td>
<td>Low</td>
</tr>
<tr>
<td></td>
<td>Yearling</td>
<td>Medium</td>
<td>Low</td>
</tr>
<tr>
<td></td>
<td>Overall</td>
<td>Medium</td>
<td>Low</td>
</tr>
</tbody>
</table>
Table 7. Overall relative rating of each limiting factor for estuary and plume based upon their potential effect on each VSP parameter.

<table>
<thead>
<tr>
<th>Life History Type</th>
<th>VSP Parameter</th>
<th>Estuary</th>
<th>Plume</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ocean Type</td>
<td>Flow</td>
<td>Habitat</td>
<td>Terns</td>
</tr>
<tr>
<td>Abundance/Prod.</td>
<td>High</td>
<td>High</td>
<td>High</td>
</tr>
<tr>
<td>Diversity</td>
<td>High</td>
<td>High</td>
<td>Low</td>
</tr>
<tr>
<td>Spatial Structure</td>
<td>High</td>
<td>High</td>
<td>Low</td>
</tr>
<tr>
<td>Overall</td>
<td>High</td>
<td>High</td>
<td>Medium</td>
</tr>
</tbody>
</table>

| Stream Type       | Abundance/Prod. | Low | Low | High | Low | Low | High | Low |
| Spatial Structure | Medium         | Medium | Low | Low  | Medium | Low  | Medium | Low  |
| Overall           | Medium         | Medium | Medium | Low  | Medium | Low  | Medium | Low  |
FIGURES
Figure 1. The Columbia River estuary extends from the upper extent of tidal influence at Bonneville Dam (RKm 240) through the oligohaline zone of the river mouth into the coastal zone of the plume in the Pacific Ocean. Inset shows the estuarine study region (to RKm 75) for the Columbia River Estuary Data Development Program (Simenstad et al. 1990a).
Figure 2. Historical and contemporary early life history types for one brood-year of chinook salmon in the Columbia River estuary. Historical timing and relative abundance based on historical sampling throughout the lower estuary (Rich 1920). Contemporary timing and relative abundance derived from Dawley et al. (1985) sampling at Jones Beach. (From Bottom et al. 2001).
Figure 3. Comparison of the monthly averaged Columbia River interior sub-basin virgin, adjusted, and observed river-flow estimates 1970-1999. Flow regulation and irrigation depletion have greatly decreased spring and summer flows (May to August), while increasing flows from September to March. (From Bottom et al. 2001).
Figure 4. Changes in the annual flow cycle of Columbia River flow at Beaver, 1878-1903 vs. 1970-1999. (From Bottom et al. 2001).
Figure 5. Peak freshet day vs. year suggests that the freshet is now about two weeks earlier than in the 19th century in the Columbia Basin. (From Bottom et al. 2001).
Figure 6. The incidence of flows above 18,000 m$^3$s$^{-1}$ (the pre-1900 estimated bankfull flow level) and above 24,000 m$^3$s$^{-1}$ (the present bankfull flow level). The present bankfull flow level has only been exceeded in four years since 1948. (From Bottom et al. 2001).
Figure 7. Monthly average flows at Beaver (1878-1999), present and historical bankfull
flow levels, and warm and cold-PDO cycles. Historically, there was a major
difference between the warm and cold phases of the PDO cycle in disturbance
frequency. This has been largely eliminated by flow regulation and diking;
overbank flow is now a rare event. (From Bottom et al. 2001).
Figure 8. Relationship between average river flow registered at Bonneville Dam for 10 days prior to sampling in the Columbia River plume in May and the maximum distance offshore juvenile chinook and coho salmon were captured in surface trawls along a transect extending east along latitude 46.15, just south of the mouth of the Columbia River.
Figure 9. Biomass of *Cancer magister* megalopae captured in May 2001 and 2002 in the ocean, front and plume habitats using a neuston net. Box plots demarcate the 10th, 25th, 50th, 75th and 90th percentiles. An ANOVA (blocked) was used to identify significant differences. Biomass of this species was 17 and 71 times higher in the front habitat compared to the average of the ocean and plume habitats.
Figure 10. Abundance of A) yearling coho B) yearling chinook C) juvenile chum salmon and D) juvenile steelhead captured in the ocean, front and plume habitats using a Nordic rope trawl. Box plots demarcate the 10th, 25th, 50th, 75th and 90th percentiles of densities. From Robertis et al. (submitted) Frontal regions of the Columbia River plume: II Distribution and feeding ecology of juvenile salmon. Can. J. Fish. Aquat. Sci.
Figure 11. Habitat opportunity in Cathlamet Bay as a function of river discharge. Habitat opportunity is defined as a site with water velocity less than 30 cm/sec. Habitat opportunity shown in hours normalized to a 30-day month is based on this velocity criterion. There are marked differences between habitat opportunity in the modern and pre-development systems. (From Bottom et al. 2001).
Figure 12. The change in availability of shallow water habitat in the tidally influenced region between RM 50 and RM 90 on the Columbia River under unmodified and modified flow conditions only. The top panel represents condition under virgin flow with no dikes, where extensive inundation of the floodplain occurs for long durations. The bottom panel represents conditions under modern flow conditions with no dikes, where river staged lowered and much less inundation of floodplain for shorter duration occurred. (From Kukulka and Jay 2003).
Figure 13. Catch per unit effort for juvenile chinook salmon for 2002 and 2003 at several sites in the mouth of the Columbia River estuary (circle), in the mixing zone (square), or in the tidal freshwater zone (Curtis Roegner, NOAA Fisheries, pers. comm.)
Figure 14. Mean fork length of juvenile chinook salmon for 2002 and 2003 at several sites in the mouth of the Columbia River estuary (circle), in the mixing zone (square), or in the tidal freshwater zone (Curtis Roegner, NOAA Fisheries, pers. comm.)
Figure 15. Box plots of size of juvenile salmon captured in peripheral habitats (trapnets) and near the main channels (beach seine) of the Columbia River estuary. Marked salmon (adipose clipped) represent hatchery fish whereas unmarked fish (adipose fin present) represent an unknown mixture of naturally produced and hatchery released juvenile salmon.
Figure 16. Proportion of Chinook salmon ESU’s originating from various parts of the basin identified in samples taken from peripheral habitats of the Columbia River estuary during 2002 between April to August. LCR – Lower Columbia River, UCR – Upper Columbia River, UWR – Upper Willamette River, SR – Snake River, MCR – Middle Columbia River, su/fa – summer/fall run, sp – spring run, sp/su – spring/summer run. From Paul Moran, NOAA Fisheries, pers. comm..)
Figure 17. Stock composition of subyearling chinook salmon in Columbia River plume study area June 1998 – 2001.
Figure 18. Temporal presence of naturally and hatchery produced juvenile chinook salmon in Skagit Bay, WA, an enclosed oligohaline region associated with the Skagit River watershed.
Figure 19. Movement of juvenile Chinook salmon in the Skagit River estuary system. Juveniles more prevalent in near shore habitat (blue), then move to deeper, shore oriented habitats later in the year, and more offshore habitats (yellow) towards the end of the year. The proportion of wild (unmarked salmon) ranged from 98%, 82%, and 73% in the peripheral, deeper, shore oriented, and channel habitats, respectively.
Figure 20. The relationship between freshwater wild chinook smolt population size and density of juvenile wild Skagit Chinook in Skagit River delta habitat, 1992-2002. The number of chinook per unit area within the delta levels-off as the total number of outmigrants increases, indicating density dependent use of the delta. E. Beamer. Skagit Bay Cooperative Research Department. Pers. Comm.
Habitat Changes in the Columbia River estuary

Total area loss = 24%

Data Source: Thomas, T.W. 1983. Changes in Columbia River estuary habitat types over CREDD

Figure 21. Change in acreage of various habitat types used by salmon in the Columbia River estuary from 1870 to 1980.
Figure 22. Daily Shallow-Water habitat (SWH) Area from 1974 to 1998 for virgin (a) and observed (b) river flows without dikes, and for virgin (c) and observed (d) flows with dikes, from Kukulka and Jay (2003).
Figure 23. Mean concentrations (± SE) of total PCBs (ng/g wet wt) in whole bodies of juvenile fall Chinook salmon sampled from Pacific Northwest estuaries. Sites from the Lower Columbia Estuary are indicated in white.
Figure 24. Mean concentrations (± SE) of DDTs (ng/g wet wt) in whole bodies of juvenile fall Chinook salmon sampled from Pacific Northwest estuaries. Sites from the Lower Columbia Estuary are indicated in white.
Figure 25. Mean concentrations (± SE) of PCBs and DDTs (ng/g wet wt) in stomach contents of juvenile fall Chinook salmon sampled from Pacific Northwest estuaries. Sites from the Lower Columbia are indicated in white.
Figure 26. Mean concentrations (± SE) of high molecular weight polycyclic aromatic hydrocarbons (HAHS) (ng/g wet wt) in stomach contents and metabolites of HAHs in bile of juvenile fall Chinook salmon sampled from Pacific Northwest estuaries. Sites from the Lower Columbia are indicated in white.
Figure 27. The leukocyte primary (■) and secondary (□) in vitro plaque forming cell response per culture (PFC/culture) against an antigen. The mean (±SD) PFC response was analyzed in chinook salmon from the Green-Duwamish System and the Nisqually System. The asterisk (*) indicates the secondary PFC/culture that is significantly higher (p≤0.05) than that observed in the primary response. From Arkoosh et al. 1991.
Figure 28. Cumulative mortality of juvenile chinook salmon collected from the Duwamish Waterway (urban) and the associated hatchery hatchery-estuary system and the Nisqually hatchery estuary system (non-urban) following disease challenge with the marine bacterium *Vibrio anguillarum*. Adapted from Arkoosh et al. 1991.
Figure 29. Foraging activity in the presence of olfactory signals of a potential predator in juvenile Chinook salmon exposed to diazinon. Control fish (solid bar) responded to the conspecific skin extracts by reducing their foraging activity and freezing. The magnitude of the antipredator response was reduced in diazinon exposed fish (2 h at 0.1, 1.0, and 10.0 mg·L⁻¹), and they were more active and fed more often than controls. The effect of diazinon was significant at the 1.0 and 10.0 mg·L⁻¹ exposures ($p = 0.05$, Fisher’s test). From Scholz et a. 2000.
Figure 30. Dose–response curves and threshold determinations for sublethal copper neurotoxicity. Data were obtained from six treatment groups (control and five copper exposures; n = 6 fish per group). The evoked electro-olfactogram (EOG) amplitudes for all copper exposures were normalized to the mean response of the controls and expressed as a group (mean ± 1 standard error). The dashed line indicates a benchmark criterion of 0.75, or a 25% reduction in evoked response (relative to controls). Three vertical lines in the upper right show the lower limits of the 95% confidence interval for the control response to the three different odorants. Filled circles indicate the benchmark concentrations for the different olfactory pathways (L-serine, taurocholic acid [TCA], and the amino acid mixture). Note that the benchmark values are nominal concentrations, or a change (increase) from the copper present at approximately 3 mg/L in the source water for the Northwest Fisheries Science Center hatchery (Seattle, WA, USA). From Baldwin et al. 2003.
Figure 31. Estimated predation rates on all steelhead in the Columbia River estuary by Caspian Terns (1999-2002) using bioenergetics modeling (black symbols) and recovery of PIT tags (blue symbols). Error bars on bioenergetics estimates represent 95% confidence limits.
Figure 32. Per Capita Smolt Consumption by Columbia River Estuary Caspian Terns on all Salmonids 1997-2003 (with 95% CI). From D. Lyons and D. Roby. Oregon Fish Cooperative Unit, Oregon State University. (Pers. Comm.)