WASHINGTON

CONTAMINATION ECOLOGY OF SELECTED FISH AND WILDLIFE OF THE LOWER COLUMBIA RIVER

A Report to the Bi-State Water Quality Program

By the Columbia Basin Fish and Wildlife Authority

April 23, 1996

OREGON

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COLUMBIA BASIN FISH & WILDLIFE AUTHORITY

April 24, 1996

Messenger Delivery

Mr. Don Yon
Oregon Department of Environmental Quality
Water Quality Division
811 SW 6th Avenue
Portland, Oregon 97204

Re: Oregon Department of Environmental Quality Contract # 096-94 and
Washington Department of Ecology Contract # C9400129
Final Product Delivery

Dear Don:

Enclosed is a revised Contaminant Ecology report dated April 8, 1996. The final Combined Fish and Wildlife Report was hand delivered to your office on April 4, 1996. Both of these reports are the result of subcontracted research efforts. CBFWA has revised the format and writing but not the content.

Sincerely,

Janice M. Eckman

Janice M. Eckman
Acting Executive Director

Enclosure

cc: Brian Offord - Washington Department of Ecology
Susan Glasser - CBFWF Fiscal Contract Administrator
Clayton Hawkes - CBFWA Project Coordinator

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WP V5.2 FILE ON DISK ALSO ENCLOSED.

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COORDINATING AND PROMOTING EFFECTIVE PROTECTION AND RESTORATION OF FISH, WILDLIFE AND THEIR HABITAT IN THE COLUMBIA RIVER BASIN

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ACKNOWLEDGEMENTS

This report builds on the report Lower Columbia River Basin Bi-State Water Quality Program Fish and Wildlife Literature Review (July 29, 1994), which was completed by the Columbia Basin Fish and Wildlife Authority's subcontractor - Nora and Steven Berwick of WILDSystems. WILDSystems completed a first draft of this report on October 14, 1994.

EXECUTIVE SUMMARY

Scope of the Review

The Lower Columbia River Bi-State Water Quality Program is assessing the impacts of contaminants to the biota of the lower Columbia River and estuary (LCR) with a variety of surveys. This literature review concentrates on the effects of changes in the LCR ecosystems and the impacts of these changes on the influence of contaminants on fish and wildlife. Contaminant effects can be measured in terms of contaminant bioaccumulation, persistence, and health/pathologies. The contaminant picture in fish and wildlife is also significant as an indirect indicator of potential human health issues, and as direct agents of contaminant poisoning to humans who consume the fish and wildlife.

Four target species were selected as representative indicator animals by the Bi-State Water Quality Program, Fish and Wildlife Committee: largescale sucker (*Catostomus macrocheilus*), bald eagle (*Halieetus leucocephalus*), mink (*Mustela vison*), and river otter (*Lutea canadensis*). Four other representative species were deemed particularly useful as trophic representatives. These additional species might clarify understanding of contaminant bioaccumulation through the food chain:

1. phytoplankton (freshwater diatom) *Asterionella formosa*;

2. estuarine zooplankton *Eurytemora affinis*;

3. benthic/epibenthic amphipod, *Corophium salmonis*; and

4. chinook salmon smolt, *Onchorynchus tshawytcha*.

Reasons for Concern

We attempted to document bioaccumulation of contaminants. Heavy metals such as mercury are deposited in relatively low concentrations to the river from such sources as paper mills, pesticides, metal production, sewage, and other sources. Mercury is then concentrated between 10,000 and 80,000 times in fish, then further concentrated in their predators, the mink and otter. These furbearers can die from 1-2 ppm (parts per million) in their diet. High concentrations of mercury have been documented in the Owyhee River and Brownlee Reservoir and other parts of the Columbia system.

DDT was banned in 1972, but it continues to reside in sediment and becomes mobile when the river dredging resurfaces and redistributes the sediment. A breakdown product of DDT, DDE is very long lasting. Water is saturated with DDT at only 1.2 parts per billion. With bioaccumulation up the food chain, other organisms obtain much greater doses: 0.08 ppm in water plants, 0.42 ppm in clams, 1.28 ppm in flatfish, 13.8 ppm in osprey eggs, and 22.8 ppm in merganser tissue. The potential impacts for Columbia River salmon were demonstrated long ago. In 1968, 700,000 young coho salmon died in Lake Michigan as they absorbed the last oil from their DDT-contaminated egg yolk sacs.

Sources

Sources of TCDD (dioxin) are municipal waste and combustion, fuel burning, pulp and paper mill bleaching processes, chlorine treated sewage, and forest fires after previous pesticide applications. PCBs are byproducts of the production of coolants, plastic, and insulation. Potential PCBs sources in the estuary are 5 chemical industry facilities, 6 pulp and paper mills, 4 lumber/plywood mills, 3 metal production facilities, 2 electric power plants, and 7 Superfund contaminant sites. DDT and its relatives from agricultural and other pesticides now resides in soils and sediments and is mobilized by dredging as described above. Other sources include copper from industrial plating and sewage, lead from fuel exhausts, pesticides, paints, mining, and mercury from the sources described above and from natural (geology and soil) sources.

Environmental Changes

Habitat changes in the LCR began in the mid 19th century when irrigation withdrawals served an area which increased from 2000 km^2 in 1900, to 32,000 km^2 in 1980. Impacts from dredging, diking, grazing and forestry soon followed. Since 1930, 14 massive mainstem hydropower dams were constructed, and over 100 tributary dams, further altering flows, sediment load and deposit, salinity, and seasonal fluctuation.

Effects

Dioxins have been found at concentrations exceeding guidelines in the LCR and are widely dispersed and persistent in the system. They are carcinogenic at low concentrations (0.013 ppm). TCDD levels in the Columbia River are so high that EPA has implemented a Total Maximum Daily Load control level.

DDT and its relatives are widely found in the flora and fauna of the Columbia Basin and are very persistent with a half life for DDT of about 20 years (longer for DDE). PCBs are related to and correlated with levels of DDT/DDE. Over 2.1 ppm DDE and 2.4 ppm PCB are found in the LCR bald eagle tissue where eggshell thinning is considered significant. Breeding success is about 25 percent less than the rest of the state, which had the second highest DDT/DDE levels of a sample of 29 states.

Metals are ionically bound to clays, peats, and other environmental features suspended in water and average 10 times the concentration of the ambient river water. The toxicity of such metals increases with temperature and respiratory activity, which is also related to temperature. River temperatures have increased with dams slowing velocity, irrigation withdrawals, removal of streamside vegetation by riparian grazing and willow conversion to pasture, sedimentation and the other changes. Salmonids are more sensitive to heavy metal toxicity than other kinds of fish.

Contaminants in One Basin

Twenty-two of the 31 subbasins in the Columbia River are above Bonneville Dam (RM 146). Although other subbasins may well contribute a similar mix and amount of contaminants, the Yakima River Subbasin has been recently documented. Pesticides regularly exceeded standards for chronic toxicity and this is reflected in measurements of the biota. Northern squawfish and largescale sucker exceeded maximum recommended concentrations of DDT/DDE; 1100 ug/kg for the sucker and 3000 ug/kg for the squawfish). Mercury concentrations of 320 ug/L in sucker and 810 in mountain whitefish are about 10,000 times recommended levels.

PHYTOPLANKTON (freshwater diatom), *Asterionella formosa*

As a unicellular primary producer, this diatom does not have a diet, life history, or demography in the sense employed for the other species reviewed. *A. formosa* densities do vary in time and space. They are found in the water column and require the basic mineral nutrients, light, a degree of water quality, and appropriate ionic ratios. Light is more likely limiting than nutrients in an estuarine environment. Upriver phytoplankton production constitutes the primary source of phytoplankton biomass (about 75 percent). *Asterionella* is rapidly flushed through the system except in inlets and bays where residence is much longer and contamination enhanced. Most die and sink upon encountering the salt wedge, and are removed by flushing or are sequestered in sediment. Zooplankton grazing accounts for about 1 percent of the removal.

Phytoplankton take up organochlorines through adsorption or absorption, eliminating them more slowly than they accumulate. Because of their low solubility, this initial step is significant in the contamination of a system, increasing residue burden with exposure over time. Some phytoplankton metabolize DDT to DDE and DDD. This can be a very effective process. For example, about 50 percent of zinc appeared to be taken up by *A. formosa* related to position in the water column, the pH, and the biochemistry of it's uptake. At lower pH, free metal ions (aluminum, lead and mercury) increased ion concentration, and uptake activity, ion transport and enzyme activity covaries with pH and phytoplankton concentration. These variables need to be studied as they constitute the proximate biological fixing of contaminants in the LCR.

ZOOPLANKTON, *Eurytemora affinis*

E. affinis is an estuarine zooplankton that inhabits the epibenthic mixing zone. *E. affinis* and *Scottolana canadensis* are very abundant in this tightly defined region, even during spring flushing, summer low flow, and strong tides. *E. affinis* abundance (100,000/m^2 or more) is apparently influenced by salinity, tides, and circulation and may also be proportional to plant food. It is not known whether high density throughout much of the year is accomplished by numerical replacement, seeking refuge, or vertical movement in the water column. Entrained in tidal currents and assisted by an endogenous swimming rhythm based upon sensing salinity, *E. affinis* maintains position in the estuary by staying high in the water column when flow is upriver, and sinking when the tide changes. *E. affinis* grazes organic detritus and phytoplankton, removing up to 4.6 $mgCm^{-3}$/day, or 1.2 percent of the total phytoplankton carbon available.

Although *E. affinis* is abundant and important in diets of invertebrates and fish in the Columbia River estuary, little has been done on the effects of contaminants on them. It is generally known that heavy metals, particularly silver, copper, and mercury, can be toxic to such copepods through feeding on contaminated phytoplankton. Sublethal effects on behavior, feeding, egg production, morphology, growth, and development are implied. The anti-fouling agent tributyltin (TBT) may constitute an issue in the LCR as well. In Chesapeake Bay it exists at three times the minimum concentration at which 50 percent *Eurytemora* mortality occurred within 72 hours.

Movement and distribution patterns of *A. formosa* and *E. affinis* may relate to contaminant patterns in the LCR. The entrainment of rising and falling *Eurytemora* in the mixing zone suggests that any contaminants they house will also cycle instead of flush or disappear in sediment. With highest predation by fish and invertebrate predators in the epibenthos as well as on animals in the water column around this area of mixing, a selection for contaminant effects on certain taxa is possible.

BENTHIC/EPIBENTHIC AMPHIPOD, *Corophium salmonis*

C. salmonis is a tube-dwelling 5 mm-long amphipod of the top 10 cm of sediments, particularly the fine sediments of shallow bays and shoals. They appear to prefer salinity below 10 parts per thousand, living in fresher water of the upper estuary tidal flats. They are detrital deposit feeders filtering water through their tubes.

Juveniles are produced in late summer broods of 10-15, after which females disappear. There are two peaks of juvenile production - fall and spring. Adults die in winter when densities are at a peak. Males seem to sustain higher predation by juvenile salmon, starry flounder, and other fish, possibly because they are more active outside of their burrows. Populations may migrate within estuarine areas in response to seasonal salinity changes.

As seen elsewhere, this amphipod is a dominant benthic macroinvertebrate and an important prey item for many fish including juvenile salmon, waterfowl, and other invertebrates. Densities in Gray's Harbor ranged from 216 to 49,675 per m^2, increasing in early summer. Production due to *C. salmonis* was nearly 11 g/m^2, a level nearly 100 times that of typical temperate grassland production, and over 90 percent of the total productivity of the site.

The metabolism (pathways, products, transfer) of contaminants by Corophium is not known. *Corophium* persists in heavily contaminated environments. Mercury, cadmium, and zinc are the most toxic to *C. salmonis*. DDT and PCB (Aroclor) were highly toxic on two *Corophium* species that have been studied. Some may metabolize organochlorines to DDE/DDD. High dioxin and furan concentrations in *Corophium* near a pulp mill was mirrored in its predators (e.g. western grebes, bald eagles), and underlines the potential significance of such pathways. Changing environmental conditions cause *Corophium* males to leave their tubes for the water column, rendering more vulnerable to such predation.

JUVENILE CHINOOK SALMON, *Oncorhynchus tshawytscha*

The use of the LCR estuary varies by salmon species and stocks. Chinook salmon exhibit semelparous anadromy and are organized into distinct stocks. Life history variations in age at migration to the sea, estuarine residence, and behavior, spreads the risk of mortality in the environment. "Stream types" (spring/summer chinook) spend a year or more as juvenile smolt in fresh water streams. After several years in the ocean, adults return to natal streams in the spring or summer. The "ocean type" (fall chinook) migrates to sea as fry and fingerlings within three months of hatching, often rearing in the estuary to smolt size, and returning several years later as adults to spawn in the fall. Fall chinook fry and fingerling tend to move downstream along shorelines, sometimes residing several weeks in an area of the river or more time in portions of the estuary. Because of their migration behavior and residence time in the estuary, fall chinook may be more vulnerable to contaminants in the LCR.

The distribution and residence (e.g. from a week to six months) in the estuary seems to depend on optimal habitat selection, food resources, predators, and salinity gradients as the fish change osmoregulation. Early exposure to full sea water results in slowed growth rates. The transition must be gradual and seems to be controlled by the local movements of the salmon within the estuary. Under optimal salinity (which is also dependent on water temperature), they experience the greatest growth rates of their life histories in the estuary (3.5 to 10 percent of body weight per day). Juvenile survivorship is between 30 and 60 percent as they move through stages to the estuary where survivorship may be less than 15 percent. The agents of this high mortality are not known. Chinook fry seem to have a vertical circadian movement in the estuary, inhabiting the top 3 m at night. The determinants of these movements and any consistency with specific popuiation segments is not known. Such considerations are important in assessing vulnerability to locally discharged contaminants.

Compared to other juvenile salmon, chinook salmon prefer slower (but moving), shallower water with small particle substrate. Aquatic insects dominate the diet of juvenile chinook in the Columbia River. In the Columbia River reservoirs, animals which reflect the fine particle size sediments (*Daphnia*), small terrestrial insects, and other small animals such as zooplankton are the major food items. In the Bonneville pool, the diet of juvenile salmon is 90 percent *Corophium*.

Little documentation exists on estuarine habitat utilization. Common prey are algae, small fish, plankton, and fish larvae, the most common being benthic/epibenthic insects and copepods from a detritus-based food chain in marsh channels and mudflats. Zooplankton, and *Corophium* of intertidal areas, dominate. The primary predators in the estuary are cutthroat trout, chinook and chum salmon, cod, auklets, cormorants, mergansers, terns, gulls, and harbor seals. However, juvenile salmon do not constitute much of the gut content of these predators. Predation on juvenile salmon may be moderated by the habitat complexity of the estuarine habitat. Several studies address effects of heavy metals, but not the potential critical sublethal effects.

Juvenile chinook are an important prey resource for many LCR predators such as squawfish, walleye, bald eagles, bass, cormorants, mergansers, grebes, and other birds and could be an important bioaccumulator of heavy metals and organochlorines, including PCBs and dioxins. Bald eagles consume a high proportion of young salmon in the spring when young eagles are starting to fledge. The role salmon play in contaminant concentration and transfer has not been studied. In a polluted area of Puget Sound, juvenile salmon concentrate aromatic hydrocarbons 650 times, and PCBs four-fold.

We recommend that studies on these four representative species include:

- measurement of contaminant concentrations in phyto/zooplankton;
- adsorption patterns of organochlorines and dioxins under varying physical environments;
- toxic effects of contaminants on *A. formosa* / *E. affinus*;
- the community structure of phyto/zooplankton at various river sites;
- cycling, flushing, and sediment patterns of different contaminants in the LCR;
- retention of contaminants in *E. affinus* in the estuarine mixing zone;
- contaminant uptake, abundance and production of *C. salmonis* in polluted and unpolluted waters;
- determinants of tube evacuation in *C. affinus*;
- effects (lethal and sublethal) of contaminants on juvenile chinook;
- factors (physical, morphological, physiological) influencing contaminant uptake in juvenile chinook (e.g. water temperature, residency and mobility);
- food web energetics and efficiency;
- stock distribution and relative vulnerability of chinook;
- relationship of chinook contamination, rates of predation by chinook consumers, and contaminant profiles in these predators.

LARGESCALE SUCKER, *Catostomus macrocheilus*

Efforts to assess the effects of contaminants in the LCR have led to bottom feeders whose food is coincident with sediment-bound contaminants, and which are significant items in the food chain. Largescale suckers sieve through the bottom sediments for food and constitute the prey of numerous birds and mammalian predators. They are distributed throughout the Columbia River Basin and populations are large (12,000-15,000/km along the Hanford reach).

Largescale suckers are mass spawners in graveled areas of rivers and lakes, producing between 625 and 1574 eggs. They are mobile and live over 20 years, achieving a weight of 3.2 kg, and over 1/2 m in length. Fry and yearlings tend to inhabit shallow backwaters, with the adults in larger water, where they can constitute the dominant fish species. Benthic plankton, diatoms and algae, salmon eggs, mollusks, and aquatic insect larvae are ingested with some bottom ooze.

Studies of the National Pesticide Monitoring Program (1960's-1980's) showed metal concentrations in fish of 0.05 to 0.73 ug/l (cadmium), 0.05 to 0.1 ug/l (lead), and 0.01 to 0.09 ug/l (mercury). In the Columbia River estuary, PCBs are present in fish including largescale sucker (0.1 to 1.0 ppm) at levels higher than national means, and DDE is present (0.1 to 0.7 ppm) as well. The dioxin concentration in largescale suckers averages 0.24 to 0.3 ppt (parts per trillion).

Studies specific to the largescale sucker are rare, with most information incidental to commercial species. Largescale suckers are important foods of walleye, squawfish, channel catfish, sturgeon, blue heron, and bald eagles, However, other fish species only prey on juvenile suckers, and therefore may not obtain significant amounts of contaminants. Fifteen years of data collected after 1971 indicate that heavy metals have declined in the LCR except for mercury. Chlorinated hydrocarbon concentrations have also declined. However, a controlled study of largescale sucker ecology and contamination (related to source and dispersal) may now be warranted.

BALD EAGLE, *Halieetus leucocephalus*

Bald eagle declines between 1950 and 1975 were linked to ingestion of DDT. PCBs are related to the DDT complex of chlorinated hydrocarbons and similar effects via reduced calcium mobilization and consequent egg shell thinning. Mercury and lead are also found in Oregon eagles. The effects of dioxin on bald eagles are poorly known, although reproductive success is diminished in other piscivorous birds. A level of 3.6 ug/g DDE will cause declines in raptor productivity. Oregon has one of the largest bald eagle populations in the United States, but it is among the least productive. Factors directly contributing to the decline include human disturbance and habitat loss, particularly through nesting failure. Contaminants such as DDT/DDE and PCBs, as well as heavy metals such as lead and mercury have been associated with bald eagle declines in the LCR area, partially through accumulation in the food chain.

Although bald eagles mature at 5 years and can live over 30 years, the mortality of immature birds is nearly 90 percent. Monitoring of the major eagle concentrations in Oregon (Klamath, Cascades, and LCR) showed nesting underway in March, hatching in May, and fledging in August. In addition to resident bald eagles, bald eagles of Alaska and Canada migrate here in winter. A large percentage of the adults are non-breeders. The percent and survivorship of young as related to food supplies and the timing of the breeding cycle tends to relate to salmon runs. Subadults are less successful hunters than adult birds and suffer food robbery by adults at communal feeding sites while engaging in intraspecific fighting to a greater degree than adults. For these reasons, subadults as a cohort are much more affected by resource shortages, doubling the mortality experienced during abundance.

Bald eagles require medium size prey, roosting trees, and freedom from disturbance. Breeding eagles have a home range of 660 ha with 0.5 km of shoreline. Human disturbance increases foraging time and nesting birds are particularly sensitive. In Puget Sound, 52 percent of the nests were occupied, most were near water, and in areas characterized by stand heterogeneity. Although three-quarters of the nests were within some form of disturbance within 1/2 mile, they became much less productive if the disturbance was within 1/4 mile. Nests above the canopy were significantly more successful. In the LCR, 57 percent of the residents and most of the migrants used flats and tidal marshes. The primary contributors to breeding site selection were prey availability and minimization of energy loss through wind, radiation, precipitation, and temperature. Conifer stands provide the best microclimates, and tidal flats were particularly important for subadult scavenging. Feeding peaks at dawn and generally occurs over shallow water at low tide.

Prey choice may reflect contamination pathways. During breeding small fish are preferred prey with hunting providing 57 percent of the food, the rest obtained by scavenging and piracy. LCR eagles consume more fish (71 percent) than other areas studied, with the largescale sucker being the most important item (17 percent), salmon the fifth most important (8.6 percent), and all waterfowl constituting 11 percent of the diet. A shift from fish to waterfowl occurs as salmon spawning concludes by early December, and hunting produces more dead birds until about February. In the Klamath River basin, waterfowl, voles, and hare constitute the bulk of an eagle's diet. Klamath rodents and lagomorphs contained lead and DDT/DDE is found in 80 percent of the waterfowl. Elevated levels of PCBs found in the bald eagle probably came from fish while DDE is magnified through ingestion of fish-eating waterfowl.

The LCR eagles may be severely affected by DDE, PCBs, and dioxin. In tests from 1980 to 1987, eagles had 2.13 ppm DDE, 2.4 ppm PCBs, 0.43 ppm lead, and 3.07 ppm mercury. Oregon eagles had significant egg shell thinning (1980-1984 samples) and showed the second highest concentrations of DDE of the 15 states sampled. Eggshells were 10 percent thinner than pre-DDT averages. It appears that DDT residues accumulate and concentrate throughout an eagle's life as it continues to consume contaminated prey. Current studies indicate that bald eagles are the most contaminated of all the piscivorous birds with DDE levels around 1.0 ug/g, PCBs between 5 and 10 ug/g.

In the LCR breeding success is 55 and 39 percent for the estuary (1980-1987), with an average of 61 percent for Oregon and 63 percent for Washington (1989-1993). This is poor compared to Chesapeake Bay, where the survival rates of bald eagles are over 90 percent and breeding territories increased 250 percent over the 9 years following the 1972 ban on DDT. The Chesapeake Bay population was growing at about 13 percent per year.

The depressed productivity of the LCR bald eagle population relative to others in North America is cause for concern and requires investigation of the interaction of contaminant effects with habitat, prey, and competitor changes. Habitat changes affect the pathways and accumulation of contaminants by altering prey availability and foraging behavior. River dynamics as influenced by human activity such as shipping and dams, have changed food web relationships and contaminant accumulation The loss of riverside old growth and the remainder vulnerable to development makes the maintenance of the requisite riverside old growth problematic. Food source changes may be related to the loss of perch and nest site changes. In the past century, estuarine area has decreased by a quarter and nearly a fifth of the medium to deep water areas and three-fourths tidal wetlands reduced due to diking. Wetlands are key areas of detrital production - the base of the estuarine food chain.

The current dietary prominence of bottom-feeding suckers for bald eagles may also reflect habitat and population declines of salmonids. A high dietary intake of diving and fish-eating waterfowl and marine birds may also serve to increase the consumption of contaminants because of additional links in the food chain. These variables have yet to be studied as an interactive system or correlated with specific demographic responses in a eagle population.

We recommend the following bald eagle studies:

- trophic web studies in selected sampling zones;
- determinants of bald eagle prey selection;
- effects of various contaminants on eagles and their prey under various controlled conditions of stress, metabolism, and environment;
- a system simulation of rates and variables in the LCR bald eagle ecosystem.

MINK, *Mustela vison*

As has been found for the bald eagle, mink of the LCR have elevated levels of DDE and PCB residues and mink seem to be particularly sensitive to contaminants. Concern for the health of this furbearer relates both to their value as an indicator of system health and as an economic asset. The percentage of the statewide mink harvest contributed by the 2 LCR counties dropped by over 1/3 in the quarter century after 1950.

Mink are nocturnal and solitary until a wide search for females begins in March and April. Gestation ranges between 38 and 85 days depending on the delayed implantation of the arrested blastula with early implantation yielding larger litters providing a population control mechanism responsive to environmental conditions.

Mink inhabit a variety of wetland types from the arctic tree line through all of the U.S. except Arizona. Densities are highest in swamp associated with woody cover and abundant fish. Adjustments in habitat use respond to prey abundance. They are more terrestrial in fall and spring (particularly males), with a larger component of terrestrial prey. Home ranges are 1.5-3.0 km long and assume the shape of the water body. They avoid open areas such as broad rivers and homogeneous shorelines, which are often the result of channelization, dredging, reservoirs, and home construction along the shore. In the LCR, mink sign is found in tidal marshes and riparian willows with rip-rap regularly used above Bonneville Dam. They use cavities in tree roots or rock piles as dens along shorelines not heavily grazed.

Common mink diet includes small mammals such as muskrat, fish, birds, herptiles, crustaceans, and insects. Along the Columbia River, crayfish, sculpin, and carp were most common in the spring diet, sculpin in summer, birds, suckers, and bass in the fall, and birds, suckers, and crappie/sunfish in winter. The consumption of mammals and birds were inversely proportional.

Toxicity experiments on captive farm mink have not examined interactive effects of different contaminants. Mink appear relatively tolerant to DDT/DDE (up to 771 ppm in fat when fed 100 ppm). They are very sensitive to dioxin. In one study one-half of the animals died within 28 days of consuming 4.2 ug/kg TCDD in food. Levels of TCDD in the LCR exceeds 5 ppt - the level considered as a threshold before reproductive impairment by the Canadian Wildlife Service. PCBs, DDT/DDE, and TCDD were identified as persistent in the LCR environment. PCBs are particularly evident in mink from the lower reaches of the Columbia River. Mink are also sensitive to PCBs, which accumulates and is proportional to total intake, eventually impairing reproduction. About 2/3 of the mink fed 10 ppm died (females being the most sensitive). All died at 3.6 ppm if fed long enough (105 days) to simulate chronic consumption. The PCBs affect reduced growth, anorexia, liver and kidney degeneration. Animals with PCB levels around 1.0 ppm suffered losses of 11 of 12 litters, with the single litter experiencing total mortality of kits within one day.

LCR mink contaminant levels are higher than in other areas of Oregon. Subadult mink had generally twice the PCB concentrations of adults (5.9 ug/g), which also have the highest DDE concentrations. Concentrations tend to correlate with distance from the source of PCBs. Lead concentrations of over 10 ug/g cause toxicity. Lead was also related to the distance from source in the river and reached levels of 34 ug/g.

Mink do not suffer heavy predation, or parasite and disease losses. However, contaminant poisoning has been implicated in mink losses, particularly from PCBs and mercury poisoning by contaminated fish and birds. Juveniles exceed adults in the population by 2-3 times. Sex ratios are nearly equal, although demographic data for the LCR population were not found. It appears that mink in the LCR have dramatically declined in the past 20 years, likely in response to PCB pollution.

Habitat loss and contamination seem to have resulted in a dramatic reduction (to possible extirpation) of LCR mink. Because of their local distribution and residence, contaminant levels and losses reflect local river-borne contaminant sources. Harvests appear to reflect declines in populations and does not appear to be a significant mortality factor. Damming, dredging, and filling of riverain habitat has undoubtedly reduced available habitat over the past 100 years. For example, it is estimated that 20 to 50 percent of the LCR populations of furbearers, including mink, lived in tidal marshes, and over 3/4 of this area has disappeared. There are 33 major sources of TCDD, PCBs, and heavy metals in the LCR as well as many non-point contaminant sources. The trophic concentration of contaminants is both placentally transferred to mammalian young, as well as in milk.

NORTHERN RIVER OTTER, *Lutea canadensis*

River otter are rare in most of their historic range which once extended throughout North America. However, they are considered stable or increasing in Oregon and Washington. LCR river otter consume a variety of fish and crayfish and as a top consumer are considered an indicator of the health of this river system. This furbearer has economic significance as well with a pelt value of $3,000,000 in 1976. A 1978-9 study of otter tissue in the LCR contained dioxin concentrations higher than reported by the Canadian Wildlife Service as causing reproductive impairment in mink.

River otter mature and mate in the spring at 2 years of age, and live until 16 although males may not be successful mates until 5 or 6 years of age. Implantation is delayed until February with a gestation of 2 months yielding a litter of 2 or 3. The young begin to venture from the den at about 2 months, and are weaned at 5 months remaining with the mother until just before the next litter.

Males move 10 km/night with a maximum dispersing distance of 42 km in a yearling male. Family groups move much less. Food has the greatest influence on movements and otter will, for example, remain at a spawning bed for 40 days. Peak activity is at midnight and dawn and during winter months. About 40 percent of observations are of solitary animals.

Females occupy an area about 7 km in diameter, and males about twice that. Home ranges overlap and can be up to 70 km long. Several activity centers are found in each home range where they spend at least 10 percent of the time. Dens are in or near the water's edge and employ natural or man-made structures. Beaver dams and logjams were used 32 and 18 percent of the time, respectively, and dense riparian vegetation characterized most of the rest.

Otter are most abundant along coastal streams and estuaries, are scarce near human activity, and almost entirely aquatic. Stream habitats are preferred to lakes and reservoirs. Primary food items are fish and crustaceans with lesser portions of mammals (muskrat) and carrion (particularly waterfowl). Along the LCR carp, crayfish, suckers, and sunfish are dominant prey with some salmon, birds, mollusks and mammals, fish constituting 80 percent. Mud flats provide slow moving fish and invertebrates although escape cover is necessary precluding use of some reservoir areas. Lakes are more acceptable in winter when ice provides cover. In the Columbia River, activity sites were found in willow, spruce, sedge, and cattail - marsh vegetation types. Critical habitat along the LCR are sloughs and tidal creeks with willow, dogwood and sitka spruce. These areas contain abundant crayfish, carp, and sculpin, with the low tides concentrating these prey. Most prey are found in logs and near undercut banks. Shallow sloughs and backwaters are probed for invertebrates and slow fish.

Recruitment just about balances mortality in Oregon otter with contaminant levels relatively low (0.1 to 1.7 ppm), although very few samples are from the LCR. The only study in this area indicated that all individuals were contaminated with DDE and PCBs with the highest levels of PCBs from the livers of 2 males (23 and 17 ppm) - much higher than in experimental mink that died of contaminant poisoning. In New York river otters, fish contaminant concentrations were correlated with river otter tissue contaminants, demonstrating a consistent relationship for PCBs and DDE, but not mercury. Similar cause-effect relationships with contaminants (particularly mercury) have been tied to local extirpation of otter in Sweden.

River otter are particularly vulnerable to over-harvest and destruction of habitat because of their restricted areas of movement. The LCR river otter population may be very low with about 0.5 to 1.5 otter per stream mile. Wetlands are used by more than 50 percent of the population and much of this habitat has been lost (77 percent of tidal wetlands in the estuary). Harvest trends may reflect markets, which are related to the number of trappers. An accurate population estimate and demographic data and more detailed studies of contamination in the LCR are needed before conclusions can be drawn about the effects of contaminants at this time. Research on river otter should address:

- current population densities and dynamics;
- sensitivity of different population segments to different contaminants;
- the seasonal contaminant burden;
- the extent of specific contaminant bioaccumulation;
- correlation of specific contaminants with specific habitats.

INTRODUCTION

The Columbia River follows a 1,250 mile course through a watershed of 259,000 square miles encompassing large parts of British Columbia, Washington, Oregon, Idaho and small portions of Alberta, Montana, Wyoming, Utah, and Nevada (NPPC 1986). It is the fourth largest river in the United States with an average annual stream flow of 141 million acre feet. The Basin contains 150 tributaries, including the 1,000 mile long Snake River. Dropping over 2,400 feet in height, the Columbia River has been greatly developed for power generation. The Columbia River may be the most dammed river basin in the world with 28 federal dam projects and many nonfederal dam projects. Of the 31 subbasins identified in the Columbia River Basin Fish and Wildlife Program, 22 are above Bonneville Dam (NPPC 1986).

The Lower Columbia River (LCR) Bi-State Water Quality Program was formed in 1990 and a four year plan was developed to assess the water quality. Inventory and reconnaissance surveys have been conducted to document the chemical contaminants in the LCR. The scope of this literature survey for the Bi-State Program is to understand the impacts of ecosystem alterations and contaminants on the fish and wildlife in the LCR and its estuary. We compiled and summarized available literature on the designated key species (bald eagle, mink, river otter and largescale sucker) and selected representatives as indicators of risk focusing on impacts of chemical contaminants and habitat changes in the LCR.

To achieve a technical analysis of the contaminant impacts in LCR, this report provides some additional information on the quantities and types of contaminants that are flowing from the upper Columbia River. Literature was collected and reviewed for contaminant impacts in subbasins such the Snake River, Yakima River and the upper Columbia River in British Columbia, Canada. Many of these same LCR contaminant sources exist in the upper Columbia River subbasins.

The selection of the fish target species has a significant influence on any evaluation. The fish tissue concentration of chemical contaminants varies considerably. Many of the game species have higher concentrations than the target species, such as the largescale sucker. These other fish species may exhibit more serious impacts on wildlife species, as well as humans through sport and tribal fishing. The evaluation of all this information must rely on careful consideration of bioconcentration factors, persistence of the chemical in the river environment; mutagenic effects, reproductive effects and other toxicological effects. The total evaluation of all the contaminants and the interacting factors is beyond the scope of this initial phase.

The current approach to water quality compliance focuses on the water analysis and the comparison to the Environmental Protection Agency (EPA) water quality standards. This tends to require vast quantities of data and studies to eventually move state and federal agencies to enforce compliance with point and, especially, the non-point sources. The combination approach of water analysis and fish tissue residue is used to document the noncompliance of water pollutants. These water pollutants contain organic and inorganic chemicals that will bioaccumulate in aquatic insects, clams, fish, and wildlife to high levels that affect populations. Humans health can also be affected when theses species are consumed. For example, mercury that is deposited in rivers may be at low levels of concentration in the water, yet it will bioaccumulate to 10,000 to 81,670 times in freshwater fish (U.S. EPA. 1992a). In fish-eating mammals, such as otter and mink, mercury concentration of 1-2 ppm in the diet can cause death. In humans, the consumption of mercury-contaminated fish produces symptoms such as numbness of the extremities, tremors, spasms, personality changes, difficulty in walking, deafness, blindness, delayed development of nerve cells in fetuses and death. Abnormally high mercury levels has been documented in the fish within the Owyhee River and Brownlee Reservoir (ODEQ 1992).

LITERATURE SEARCH METHODOLOGY

The methods used in this study involved collecting, summarizing and analyzing data from previous studies designed to identify the impacts from exposure to chemical contaminants, changes in the extent and quality of critical habitat, the life history and diet of the largescale sucker, bald eagle, mink, and river otter.

Key word searches were conducted for literature addressing "Asterionella and estuaries", "Asterionella and contaminants/contamination", "Asterionella and pesticides", "Asterionella and toxins", "Asterionella and pollutants", "Asterionella and Columbia River", "phytoplankton and Columbia River", "phytoplankton and estuaries", and "phytoplankton and contaminants/contamination". Similar searches were conducted with the genus names and taxon of *E. affinis* and *C. salmonis*. Juvenile chinook salmon were searched for under their common name. Papers selected for annotation dealt with not only the species under question, contaminants, and the LCR (or comparative habitats), but those that were more contemporary and those that represented a breadth of issues pertaining to the complexity of their population and community dynamics.

For largescale sucker the following key words were used: largescale sucker, *Catostomus macrocheilus*, contaminants, toxicology, catostomids, pesticides, metals, pesticide pollution, pollution effects, agro-chemicals, environmental impact, sediment toxicology, and organochlorine. Author searches were also conducted to search literature produced by scientists noted for their work involving largescale suckers.

In addition to key word searches and through cross-checks with the bibliographies of articles and documents obtained, prominent authors on bald eagles were author searched. Key word searches were for documents addressing "bald eagles and estuaries", "bald eagles and contaminants/ contamination", "bald eagles and pesticides", "bald eagles and toxins", "bald eagles and pollutants", and "bald eagles and Columbia River". Since thousands of articles and papers have been written that pertain to bald eagle life history (see Stalmaster 1987 for extensive bibliography), the key word search focused on identified life history research that was related to contaminants. Papers selected for annotation addressed not only bald eagles, contaminants, and the Columbia River estuary (or comparative habitats), but also those that were more contemporary and those that represented a breadth of issues pertaining to the complexity of bald eagle population and community dynamics. In other words, the annotated articles sample the issues rather than exhaust a few limited issues.

For mink and otter, current data bases were searched using combinations of key words; "Columbia River", "contaminant and/or toxicity and/or pollutant", "mink and/or river otter". Articles related to general life history, habitat use, food habits, population status and effects and/or concentrations of environmental contaminants in river otter were chosen for review and annotation. Studies conducted in Washington or Oregon, especially in relation to the Columbia River, were important criteria for summarization. Where there were no studies specific to the Columbia River, studies conducted in other regions of North America were selected as a basis for comparison.

Literature searches were conducted at the Portland State University, Oregon State University, and University of Washington, Evergreen State College, and Grays Harbor Community College libraries. The Washington State Library contains all government documents pertaining to research in Washington. Agency sources included: Oregon Department of Environmental Quality; Oregon Department of Fish and Wildlife; U.S Department of Fish and Wildlife - Wildlife Division, Portland; National Marine Fisheries Service, Seattle; National Biological Survey, Corvallis; Hatfield Marine Science Center, Newport; CREST, Astoria; and the Washington Department of Fish and Wildlife.

Databases searched included Biosis™, UWIN, and Zoological Records housed at University of Washington and Environmental References at Portland State University. Contacts were also made with personnel at Tetra Tech, NMFS and CREST.

SOURCES OF CHEMICAL CONTAMINANTS IN THE LOWER COLUMBIA RIVER AND ESTUARY

Industries along the Columbia River have contributed to water quality degradation. Between Bonneville Dam and Longview, Washington, the Columbia River receives discharge from three pulp mills in Washington. The types of industrial waste discharging into the Columbia River include pulp and paper, plywood, wood products in general, chemicals, aluminum reduction, food processing, gravel washing, milk processing and sand and gravel (Tables 1 and 2).

A major main source of contaminants is by the use of pesticides and herbicides on agricultural crops. The soil acts as a reservoir for pesticides (Edwards 1973). Rain and snow (fallout) cause the transfer of contaminants from crops to the river, either thorough runoff, subsurface leaching, or groundwater transmission. Contaminants are taken up by invertebrates that break the contaminants down and the bi-products are retained longer, especially in heavier soils with a high organic matter content. Seelye et al. (1982) demonstrated the potential for uptake of DDE and PCBs by fish as a result of dredging.

Table 1. LCR contaminant sources.

Location	RM	Potential Sources	References
Reed Island	124	reference station	Johnson & Norton 1988
Gresham	120	urban runoff	Tetra Tech 1992
James River II	120	paper mill	Tetra Tech 1992
Troutdale	120	Reynolds metals, urban runoff	Tetra Tech 1992
Camas Slough	118	pulp mill, urban runoff	Johnson & Norton 1988
Portland	105-106	urban runoff, paper mill	Tetra Tech 1992
Vancouver	105-102	paper mill, aluminum smelter, chemical storage, STP, urban runoff	Johnson & Norton 1988
Near Sauvie Island	103	Aluminum Company of America	Tetra Tech 1992
Near Sauvie Island	98	Salmon Creek STP	Tetra Tech 1992
St. Helens	86	urban runoff	Tetra Tech 1992
West of St. Helens	83	Chevron Chemical Company	Tetra Tech 1992
Near Kalama River	75	Trojan Nuclear Power Plant	Tetra Tech 1992
Kalama	75	chemical manufacture	Johnson & Norton 1988
Longview	67-56	pulp mills, aluminum smelter, former Hg-cell chloralkali plant, log yards, light industry, STP, urban runoff	Johnson & Norton 1988
	41	James River II, Wauna Mill	Tetra Tech 1992
Astoria	17	urban runoff	Tetra Tech 1992
Warrenton	9	urban runoff	Tetra Tech 1992
Ilwaco	3	boatyards	Johnson & Norton 1988

Table 2. Sources and routes of entry of contaminants in the LCR.

Chemical	Routes of Entry / Sources	References
TCDD	combustion facilities	Tucker et al. 1983
	municipal waste facilities	Tucker et al. 1983
	as an impurity in a number of industrial chemicals, as a component of smoke emissions from municipal, hospital, and management waste incinerators heating plants	Westing 1984
PCB	isomers used in aerosols (large transformers), carbonless paper, paints, electronic industry, coolants	Peterle 1991
	used in industry as heat transfer agents (heating to cooling), plasticizers and insulating materials	Leithe 19??
DDT	runoff and percolation from agricultural lands	Strobbe 1971
	drift from aerial and land applications	Strobbe 1971
DDE	DDT metabolite	Peterle 1991; Johnson and Finley 1980
Cu	industrial	Peterle 1991
	sewage effluent, plating industry	Peterle 1991
Hg	Industrial use: fur processing, coal, electrical, dental treatments, paper mills, paints, fossil fuels, seed dressings, cement manufacturing, mining, petroleum, pesticides, milling and metal alloy production, chlorine and caustic soda production, fungicidal paints, preservatives, batteries, switches, thermometers, barometers, medicinal uses. Agricultural: insecticides and fungicides, seed dressings	Peterle 1991
Pb	mining, smelting processing	Train 1979
	lead bearing paints, gasoline additives, automotive exhaust	Peterle 1991
	pesticides	Peterle 1991

Dioxins are discharged to the Columbia River from both point and nonpoint sources. Sources include leaded and diesel fuels, bleached wood pulp, wood preservatives, sewage chlorination, petroleum refinement, hazardous hospital and municipal waste incinerators and forest fires. Hydroelectric dams have also been identified as a significant source of PCBs. Sources of PCBs in the environment include coolants, electrical transformers and plastics. Even though PCBs and most of the chlorinated pesticides have either been banned or severely limited in use during the past 15 years, they persist and are still found in aquatic plants and animals. Sewage and industrial wastes contribute ammonia, phenol, cyanide, copper, zinc, cadmium, nickel, chromium and mercury. Pulp and paper mill effluent also contribute contaminates from debarking, kraft pulping, caustic extraction and mechanical pulping processes.

CHANGES ASSOCIATED WITH THE ALTERATIONS OF THE COLUMBIA RIVER AND ESTUARINE ENVIRONMENTS

Historical Alterations

Dams, dredging and filling for navigational improvements, rapid growth in human population, over fishing, and natural climatic variation are a few of the factors which have impacted the Columbia River.

Human Intervention

Human alteration of Columbia River flows began with the instigation of irrigation in the middle 1800's (Table 3). Dramatic increases in the amount of land under irrigation have influenced water quality in the LCR. Irrigated land has increased from 2,000 km^2 in 1900 to 31,600 km^2 in 1980 (Weitkamp 1993; Sherwood et al. 1990). The total annual water withdrawal in 1985 was 13,300 million m^3 (Weitkamp 1993). Other human activities which have altered LCR water quality included diking, dredging, disposal of dredge spoils, filling, jetty construction, construction of upriver dams and watershed activities such as forest harvest. Greater than 29,700 ha (80%) of swampland on the Columbia River estuary has been impacted through diking. Dredging and the construction of dikes began in 1873 and continues today (Thomas 1983; Simenstad et al. 1984). By 1976, between five and 10 million m^3 of material was dredged annually (Sherwood et al. 1990; Simenstad et al. 1984).

Hydroelectric Projects

Fourteen hydropower dams were constructed on the mainstem of the Columbia River between 1933 and 1983 (Weitkamp 1993). Dam projects on the mainstem of the Columbia, listed in order of proximity to the ocean are: Bonneville, The Dalles, John Day, McNary, Priest Rapids, Wanapum, Rock Island, Rocky Reach, Wells, Chief Joseph and Grand Coulee. In addition, more than 100 dams are located on the tributaries. Dams may affect water quality through altering flow timing and rates and may also be sources of PCB contamination.

MAJOR CHEMICAL CONTAMINANT GROUPS

2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD)

Studies on dioxins and their effect on flora and fauna include Henny et al. (1981), Beak Consultants (1989), Keenan et al. (1990), Tetra Tech (1990), Farrington (1991), Green (1991), Parsons et at. (1991), Tetra Tech (1992), Northwest Power Planning Council (1994), Sanderson et al. (1994) and U.S. Fish and Wildlife Service (unpublished). Dioxins have been found in wildlife and fish at concentrations that exceed guidelines for the protection of human health in the LCR. Dioxins are considered highly toxic and carcinogenic at low

Table 3. Time-line of activities that have affected water quality on the LCR.

Year	Location	Activity	References
8,000 BC	Columbia River	NW Indians occupy the region relying on salmon as a major food source	NWPPC 1992
1827	Fort Vancouver, WA	Hudson's Bay Company initiates Pacific Northwest logging industry	NWPPC 1992
1840's	Near Walla Walla, WA	First farm irrigation systems installed adjacent to mission near Walla Walla	NWPPC 1992
1840's	Columbia River	Logging and lumber export begins	Weitkamp 1993
1850's	Columbia River Basin	Mining and logging and livestock production increases	NWPPC 1992
1861	Columbia River	Idaho gold rush increases ship traffic	NWPPC 1992
1868	Columbia River Estuary	First dikes built	Simenstad et al. 1984
1873	Columbia River	First dredging of the Columbia River	Simenstad et al. 1984
1885	Mouth of Columbia River	Construction of South jetty begins	Simenstad et al. 1984
1888	Columbia River Basin	Mining, logging and livestock production beginning to have noticeable effect on soil and water quality	NWPPC 1992
1902	Columbia River Basin	Reclamation Act authorized federal aid to settle land and develop farms	NWPPC 1992
1910	Columbia River Basin	Irrigation acreage in the basin increases four fold	NWPPC 1992
1933	Near Wenatchee	Rock Island Dam is first dam completed across the Columbia	NWPPC 1992
1938	Bonneville Dam	Bonneville Dam begins operation	NWPPC 1992
1963	Hanford, WA	First nuclear power plant on Columbia begins operation	NWPPC 1992
1976	Portland, OR to mouth of Columbia River	122 m deep channel dredged annually	Simenstad et al. 1984; Sherwood et al. 1990

concentrations (0.013 ppq) (EPA 1986 and Peterle 1991). The 2,3,7,8'-TCDD dioxin compound is considered the most potent form of dioxin. It is lethal to a number of laboratory mammals at dosages of less than 100 μg/kg (Westing 1984). Dioxin is created by the chemical interaction of chlorinated compounds with organic matter. Table 4 summarizes the characteristics of the major contaminate groups.

Polychlorinated biphenyls (PCBs)

Studies on PCBs include Henderson, et al. (1971), Kimbrough (1974), Poland et al. (1979), Henny et al. (1981), Schmitt, et al. (1981), Kocibaca and Schwetz (1982), Poland and Knutson (1982), Schmitt, et al. (1983), Henny et al. (1984), Safe (1984), Schmitt et al. (1985), Brunstršm and Reutergardh (1986), Nikolaidis et al. (1988), Buchman (1989), Schmitt et al. (1990), Farrington (1991), Nosek et al. (1992), Tetra Tech (1992), Anthony, et al. (1993) and Sanderson et al. (1994). Jensen et al. (1969) found concentrations as high as 218 ppm in livers of white shark - *Carcharodon carcharias* (Zitko 1972). This reflects the fact the PCBs tend to accumulate in tissues with high lipid levels (Vernberg et al. 1977). One of the most important properties of PCBs is their tendency to bioaccumulate in fish to levels higher than ambient water as a result of high lipid solubility and the slow rate with which fish metabolize and eliminate PCBs (Khan et al. 1979). PCBs accumulate in fish via water exposure and their diet (Khan et al. 1979). The principal mechanism in fish, lobster, crab and skate appears to be hydroxylation. These organisms metabolize the simplest PCB component, biphenyl, to the 4-hydroxy derivative and to a lesser extent 2-hydroxy (Khan et al. 1979). PCBs have been found to be widely dispersed throughout the LCR, show a tendency for persistence, and are highly toxic to many animal species.

Dichloro-diphenyl-trichloroethane (DDT)

DDT studies include Henderson et al. (1969), Henderson et al. (1971), Schmitt et al. (1981), Schmitt et al. (1983), Hudson et al. (1984), Johnson and Finley (1984), Schmitt et al. (1985), Buchman (1989), Schmitt et al. (1990) and Farrington (1991). DDT has been found to be pervasive in flora and fauna of the Columbia River in numerous studies. DDT also rapidly accumulates in invertebrates to several thousand times the exposure level in concentrations as low as 80 ng/L (Johnson and Finley 1984) and proves to be extremely deleterious to many species.

Dichloro-diphenyl-dichloroethylene (DDE)

Studies on DDE include Henderson et al. (1969), Henderson et al. (1971), Schmitt et al. (1981), Henny et al. (1984), Schmitt et al. (1985), Schmitt et al. (1990), Tetra Tech (1992) and Anthony et al. (1993). DDE is one of the primary metabolites of DDT in invertebrates and produces biological effects similar to those of the parent compound. DDE has been found in Columbia River fish and wildlife at potentially hazardous concentrations. It, like DDT, accumulates in organisms and food chain systems (Peterle 1991).

Metals

Metal contaminants such as mercury (Hg), lead (Pb) and copper (Cu) affect a variety of aquatic, terrestrial and avian animals. Studies on metals include Fuhrer (1986), Buchman (1989), Henderson et al. (1972), Walsh et al. (1977), U.S. Army Corps of Engineers (1979), May and McKinney (1981), Eisler (1987 and 1988), Fuhrer and Horowitz (1989), Schmitt et al. (1990), United States Environmental Protection Agency (USEPA) (1991) and Tetra Tech (1992).

CONTAMINANT SOURCES AND TRANSPORT

Contaminate Transport and Dispersion

Contaminants can be introduced and accumulated in bodies of water by several different pathways. These pathways include sediment uptake and release, chemical degradation, chemical and photochemical formation, direct absorption, fallout (rain or snow), spray transfer, and volatilization (Table 5).

Adverse impacts to fish and wildlife resulting from biomagnification of organochlorine pesticides and PCBs have not been well documented. Primary mechanisms of PCB dispersion are believed to include movement by animals and through sediment transport.

Concentrations of pesticides in water are controlled by solubility, adsorption - desorption, partitioning, hydrodynamics and other factors (Haque and Freed 1975). The solubility of most of the environmentally important pesticides and other chemicals in pure water is very low. The remainder of the pesticides are adsorbed onto particles or are partitioned into suspended organic solids or liquids (Haque and Freed 1975).

DDT refuses to stay where it is put. It filters and flows into streams where it still retains its toxic properties via wind, soil and rain. Transport of p,p'-DDT can follow several routes which lead to chemical degradation. One leads to p,p'-DDE which is probably a metabolic dead-end. Another path leads to a hydroxylated derivative, dicofol. Via dehydrochlorination, p,p'-DDD is the main excretory product from mammals that ingest DDT. Finally, reduction and oxidation occurs resulting in p,p'-DDA and similar polar metabolites (Lockwood 1976). Planktonic organisms have also been found to convert p,p'-DDT to p,p'-DDE (Kiel and Priester 1969, Bowes 1972; Rice and Sikka 1973).

Because DDT is fat soluble, it is stored in living tissue, accumulated by animals in natural food chains and increasingly concentrated toward the top of such ecosystems (Strobbe 1971). Certain physical properties of DDT are important in determining its behavior in the biosphere. DDT residues tend to accumulate in lipids of plants and animals and are very persistent in nature, estimates of their half-life range upward to 20 years (Peterle 1991). DDT and dieldrin are long term contaminants of the total environment and small traces can be found in almost all compartments of the ecosystem (Edwards 1973). DDT also has a

Table 4. Characteristics of major chemical contaminant groups identified.

Chemical	Characteristics	References
TCDD	highly toxic	USEPA 1986
PCB	high lipid solubility	Khan et al. 1979
DDT	relatively non-volatile	Edwards 1973
	low vapor pressure (1.5 x 10^7 mm @ 20°C)	Peterle 1991
	high chemical stability	Peterle 1991
	relatively insensitive to light	Peterle 1991
	low water solubility (0.2 to 1.2 ppb)	Peterle 1991; Woodwell et al.
	high solubility in fats	Woodwell et al. 1973; Strobbe
	very persistent in nature	Woodwell et al. 1973
	dieldrin found to increase the rate of DDT accumulation	Salabaser 1982
	levels in fish found to vary with temperature	Peters and Weber 1977
DDE	most common metabolite of DDT	Peterle 1991; Johnson & Finley 1980
	conversion of DDT to DDE is pH dependent	Peterle 1991
Cu	toxicity is increased by reduction in water hardness, temperature and dissolved oxygen	
	toxicity is decreased in the presence of chelating agents such as EDTA and NTA, humic acids, amino acids suspended solids	
Hg	persistent in the environment	Peterle 1991
Pb	Pb salts mostly low in solubility	

vapor pressure high enough to assure direct losses from plants and soil into the atmosphere (Jukes et al. 1973). DDT residues enter the biota through direct absorption and through food webs (Jukes et al. 1973). DDT residues may reenter complex food webs from organic sediments by the direct consumption of detritus (Jukes et al. 1973).

Metals cycling processes can be categorized in several ways. Metal cations appear to form organically bound complexes with peat which readily bonds with them (Wieder and Lang (1986). Clay particles capture metals at their ion exchange sites (Boto and Patrick 1979; Pickering 1979). Pickering (1979) explained how a high proportion of heavy metal loading in an aquatic system can be associated with the clay fraction. Copper ions act much like other divalent cations on clay, with their own behavior depending largely on the coppers strongly held copper water ligand (McBride and Mortland 1974). Protonated ligands (e.g., silicate or phosphate) can act as bridges between the clay surface and the metal ion, thus enhancing uptake (Farrah and Pickering 1976). Colloidal particulate always contained the highest heavy metal content, typically more than 10 times that of the dissolved material (Perhac 1972).

Table 5. Pathways for chemical introduction into the aquatic environment.

Chemical	Biological Dispersion	Physical Dispersion	References
TCDD	plants uptake from soil	photodegradation	Tucker et al. 1983
	dermally adsorbed		Tucker et al. 1983
	intestinal absorption		Tucker et al. 1983
	accumulates in adipose tissues, lab animals		Tucker et al. 1983
PCB	accumulates in lipids	accumulates in sediment	Hutzinger 1978
	bioaccumulates in fish to levels higher than ambient water		Khan et al. 1979
	fish accumulate through diet		Khan et al. 1979
	fish metabolize through hydroxylation		Khan et al. 1979
	main organ involved in metabolism is liver		Khan et al. 1979
		leaching, volitilization, movement in the air	Peterle 1991
DDT	stored in invertebrate tissues		Edwards 1973
	stored in vertebrate tissues		Edwards 1973
	accumulates in lipids in plants and animals		Woodwell 1973
	passes from mother to unborn through placenta		Strobbe 1971
		direct absorption	Woodwell 1973
DDE	accumulates in organisms		Peterle 1991
Cu		readily complexes by inorganic and organic substances and is adsorbed on to particulate matter	
Hg	Transported in food chain	volatilizes in air	Peterle 1991
Pb		Precipitation, lead dust fallout, erosion and leaching of soil, municipal and industrial waste discharges runoff and fallout deposits from streets and other surfaces. Aerial distribution from automotive exhaust.	

Since metals are frequently bound in sediments, the transport of sediments may be important in the metal cycling process. Physical sedimentation helps confine metals to substrates through a physical incorporation. Once incorporated, plants may take them up or anaerobic processes may change the state of the metal to either a more soluble product or one that can be released into the atmosphere. If particulate organic matter becomes settled into an anaerobic environment, fermentation processes and anaerobic respiration become operative (Bott 1976). Sediment transport processes are directly related to the hydrological processes occurring in these systems.

BIOLOGICAL UPTAKE AND EFFECTS OF CONTAMINANT GROUPS

The fate of organic pesticides in the aquatic environment is governed by various physical, chemical and biological forces of adsorption on solid surfaces, biologic and chemical degradation, hydrolysis, uptake, retention and release by biological systems and precipitation reactions.

An extensive study on bald eagles (*Haliaeetus leucocephalus*) by Anthony et al (1993) found that eggshell fragments showed considerable amounts of thinning association with high concentrations of DDE and PCBs. Other results indicated an accumulation of DDE and PCB concentrations in the blood with age. Steidl et al. (1991) reported eggshell thinning and reduced reproductive success in relation to DDE and PCB in osprey (*Pandion haliaetus*) eggs and fish in Delaware Bay. Fish eating birds may be a greater source of organochlorine compounds in eagle diets than fish (Anthony et al. 1993; Frenzel 1984; Kozie and Anderson 1991). DDE causes thinning of eggshells experimentally in raptors and other birds (Bitman et al. 1969; Porter and Wiemeyer 1969; Wiemeyer and Porter 1970, Longcore and Samson 1973; Lincer 1975).

DDT has a tendency to concentrate in the fatty tissues of animals, possibly because DDT is more soluble in fat than in water (Morton 1976). Aquatic plants, such as algae, have also been observed to absorb DDT from the water rather than chemically or biologically utilize it (Vance and Drummond 1969). Since animals feed on each other and on plants, the buildup of DDT concentrations in animals is often referred to as the food chain concentration effect (Morton 1976). The effects of DDT toxicity include ataxia, wing-drop, jerkiness in gait, continuous whole-body tremors, falling and convulsions (Hudson, Tucker and Haegele 1984). The p,p'-isomer appears to be more toxic than the o,p-isomer. DDT rapidly accumulates in invertebrates to several thousand times the exposure level in concentrations as low as 80 ng/L (Johnson and Finley 1980).

In fish, the uptake of many organic pollutants comes from the ingestion of food (Table 6). Therefore, such contaminants are frequently taken up in direct proportion to the dietary intake, which depends on the quantity of food consumed and also on the concentrations of the contaminants in the food (Spigarelli et al. 1983; Hilton et al. 1983). Three processes could contribute to organochlorine accumulation: ingestion of contaminated food, direct absorption from water through the gills, and absorption through the skin (Kerr and Vass 1973). Temperature effects on PCB accumulation are due to the temperature-controlled food consumption, growth and lipid content of the fish (Spigarelli et al. 1993). PCBs produce numerous effects which include weight loss, edema, hepatotoxicity, immunotoxicity, decreased reproductive success, teratogenicity, promotion of cancer and enzyme induction (Sanderson et al. 1994). Fish kills from PCBs occur at various concentration levels. Some fish are killed by exposures as low as 0.1 ppb while other species can survive concentrations over 1,000 ppb (Peterle 1991).

Dispersion patterns of metals contaminants have been well documented in many studies. Heavy metal toxicity increases with higher water temperatures because of elevated respiratory activity. Also the metal solution itself causes increased respiratory activity. The absorption and release of metals can also depend upon temperature. Apart from the effects of water temperature on the toxicity of heavy metals as a result of physiologic changes in the organism, these two parameters can, due to chemical processes in water and sediment, decisively influence heavy metal availability (Lloyd 1965).

Mercury is considered a nonessential element for fish and is found to be a highly toxic element for living organisms. Fish exposed to mercury in polluted rivers have higher mercury levels in the summer than winter.(Amend et al. 1969). In a typical biological food chain for mercury, the decay of organic material in the aquatic environment, possibly enriched by the disposal of sewage and industrial effluent, together with detritus, provides a rich source of nutrients in both the bottom sediments and the overlying water body. Microorganisms and microflora incorporate and accumulate metal species into their living cells from these supply sources. Subsequently, small fish become enriched with the accumulated substances. Predatory fish then eat the small fish and birds and mammals eat the fish. Even at low concentrations, mercury and its compounds present potential hazards due to enrichment in the food chain (Hartung 1972). Microorganisms are also capable of transforming inorganic mercury to the more toxic monomethyl- and dimethyl-mercury (Jensen and Jernel 1969) .

Metal pollution in a stream can kill many of the organisms present, resulting in low species diversity (Brock 1969). Every organism has a certain ability to cope with non-essential metals or excessively available essential metals (Schat and Bookum 1992). Organisms may be affected in various ways such as by a reduced growth rate or inability to complete a particular stage in their life history. Salmonidae are usually more sensitive to heavy metals that the Cyprinidae and other fish (Herbert 1965). Some heavy metals occur naturally in low levels and are required by plants and animals. Copper is one of the micronutrients required for plant growth (Pickering, 1979; Ouzounidou et al. 1991) and is a constituent of the protein component of several enzymes in plants (Ouzounidou et al. 1991). It is also nutritionally essential for certain fish (Miller, et al. 1992). The availability of the micronutrients, such as copper, is significantly controlled by the colloidal components of the soil (e.g. humic acids) (Pickering, 1979; Salomons and Fišrstner, 1984). Excessive quantities of metals are toxic to plants (Brown and Wells 1990; Ouzounidou et al. 1991). The easiest way to detect responses to toxic metals is in the inhibition of root growth (Eleftheriou and Karataglis 1989).

Metals uptake in an aquatic system is a continuum between absorption and the release of sorbed copper (Pickering 1979). This continuum consist of metals being mobilized through the system by being bound or by sediment uptake. Metals can then be mobilized from the sediment and taken up by the biota. Plants use the sediment as nutrition. In the next step, the plants can be eaten by fish or other organisms and thus the metal is once again placed back into the sediments and the environment. Excessive intake of copper results in its accumulating in the liver (Försner and Wittman 1981). Metals such as lead and mercury in very low concentrations can cause chronic poisoning which results in a wide variety of neurological and other problems. Mercury concentrations that exceeded dietary levels interfere with successful reproduction in nesting mallards (Heinz 1979).

Table 6. Contaminate effects on biota.

Chemical	Effect on Animals	References
TCDD	death of herbivores due to uptake of soils fish, long latent period of lethality highly toxic, carcinogenic in reproduction, teratogenic, mutagenic immunotoxic, histopathologic effects lab experiments on fish have shown exposure results in retardation, increased liver size, induced MFO activity field observation reveal responses such as decreased gonad size, increased liver size, induction of hepatic MFO, reduced circulating levels of sex steroids (concentrations as high as 124 pg g^-1)	Tucker et al. 1983 Peterle 1991; Servos et al. 1994 Peterle 1991 Servos et al. 1994 Servos et al. 1994
PCB	avian reproductive failure, transfers across the placenta and milk of animals	Peterle 1991
DDT	Coho and Chinook salmon developed severe deformity, degeneration of fins and opercules and malformation of the skull affect tissues of central and peripheral nervous system cause bursting on lateral line nerve fibers	Edwards 1973 Peters and Weber 1977
DDE	potential impact on eggshell thinning, alters reproductive success in birds	Peterle 1991
Cu	precipitation of copper salts of mucus on the gills causing suffocation and direct damage of the gill	
Hg	neurotoxic to all organisms, mutagenic	Peterle 1991
Pb	reduces dehydratase levels in birds which causes malfunction of hemoglobin physiological changes in oxygen in birds, and effects growth rates in birds	Peterle 1991

CONTAMINANT GROUP CONCENTRATIONS

The processes described above influence the accumulation of contaminants in the biota of the LCR. Table 7 presents water quality criteria for the selected contaminant groups. Table 8 presents the concentrations of the target contaminant groups addressed in this report in selected LCR species.

Table 7. Water quality criteria for chronic toxicity of major contaminates selected.

Chemical	Domestic Water Supply Limit	Aquatic Life	Freshwater	References
TCDD	-	100 ug/kg	-	Westing 1984
PCB	5 ppm	-	0.014 ug/L	Cahn et al. 1977 Tetra Tech 1992
DDT	-	0.001 ug/L	0.001 ug/L	Train 1979
DDE	-	-	0.001 ug/L	Tetra Tech 1992
Cu	1.0 mg/L	0.1 times LC50	-	Train 1979
Hg	2.0 ug/L	0.05 ug/L	0.10 ug/L	Train 1979
Pb	50 ug/L	0.01 times the 96-hour LC50	-	Train 1979

One can browse through an extensive body of literature in the fields of pharmacology and toxicology to identify factors most important in understanding and predicting the dynamics of absorption and retention of pesticides and metals by organisms. Some of the most important factors include lipid solubility, molecular size and degree of ionization (Edwards 1983).

UPPER COLUMBIA RIVER

In the upper Columbia River, the current high priority emphasis is on salmon recovery in the Snake River subbasins (NPPC 1992a; Snake River Recovery Team 1994). Dealing with chemical contaminants are an important factor in realizing the goals of salmon and steelhead enhancement (NPPC 1992b). Throughout the Columbia River Basin, salmon and steelhead habitat has been degraded below the optimum carrying capacity for the spawning and rearing of healthy populations. On public and private land, agricultural practices, timber harvesting, road building, cattle grazing, and mining have increased sedimentation in streams, causing increases in water temperatures, and contributing to increased mortalities of juvenile fish (Meehan 1991). As described for alterations to the LCR, irrigation withdrawal from rivers and streams has decreased flows that reduces the rearing habitat of juvenile fish. Pulp and paper mills contribute dioxin and furans (Johnson et al. 1993). Radioactive chemicals have been released from the Hanford nuclear facilities into the Columbia River (Robertson and Fix 1977). Abatement of these sources is important for achieving salmon recovery.

Table 8: Levels of Contaminants in Biota in the Columbia River

From Henderson et al. 1969
Data from fall 1967 and 1968
Location: Bonneville Dam, OR

Species	Organochlorine Insecticides (PPM)									
	DDE	TDE	DDT	DDTand met	Dieldrin	Aldrin	Endrin	Lindane	Heptachlor	Heptachlor Epoxid
Largescale Sucker	0.29	0.27	0.15	0.61	0.05	0.01	-	0.01	0.02	0.01
Black Bullhead	0.12	0.13	0.05	0.30	0.01	-	0.01	-	-	-
Rainbow Trout	0.09	0.01		0.10	-	-	-	-	-	-
Northern Squawfish	0.69	0.58	0.14	1.42	-	-	-	-	-	-

From Henderson et al. 1971
Data from Fall 1969
Location: Bonneville Dam, OR

Species	Organochlorine Insecticides (PPM)						
	DDE	TDE	DDT	DDTand met	Dieldrin	BHC	Est. PCBs
Largescale Sucker	0.36	0.17	0.11	0.64	0.01	0.01	1.04
Chiselmouth	0.70	0.41	0.09	1.20	0.01	0.02	0.98
Northern Squawfish	1.87	0.45	0.10	2.42	0.01	0.01	1.19

From Walsh et al. 1977
Data from 1971 - 1973
Location: Bonneville Dam, OR

Species	Metal Residues (mg/kg)			
	Cadmium	Lead	Mercury	Arsenic
Largescale Sucker	0.05	0.15	0.02	0.20
Northern Squawfish	0.05	0.00	0.80	0.05
Carp	0.08	0.12	0.05	0.14

Table 8: Levels of Contaminants in Biota in the Columbia River

From Schmitt et al. 1981
Data from 1970 - 1974
Location: Bonneville Dam, OR

Species	Organochlorine Insecticides (PPM)						
	p,p' DDE	p,p' DDT	Dieldrin	Aldrin	Heptachlor	p,p' DDD	PCBs
Largescale Sucker	0.73	0.05	0.01	0.00	0.00	0.23	0.25
Northern Squawfish	0.56	0.00	0.01	0.00	0.00	0.12	0.30

From May et al. 1981
Data from 1976 - 1977
Location: Bonneville Dam, OR

Species	Metal Residues (mg/kg)				
	Cadmium	Lead	Mercury	Arsenic	Selenium
Largescale Sucker	0.73	0.05	0.01	0.00	0.00
Northern Squawfish	0.56	0.00	0.01	0.00	0.00

From Schmitt et al. 1985
Data from 1980 - 1981
Location: Cascade Locks, OR

Species	Organochlorine Insecticides (PPM)						
	p,p'DDE	p,p' DDT	Dieldrin	Endrin	Heptachlor	p,p' DDD	PCBs (Aroclors)
Largescale Sucker	0.54	0.05	0.01	0.00	0.00	0.21	0.60
Northern Squawfish	0.64	0.00	0.01	0.00	0.00	0.14	0.25

Table 8: Levels of Contaminants in Biota in the Columbia River

From Parsons et al. 1991
Data from August and November of 1989
Location: river mile 2 to river mile 335

Species	TCDD levels
Coho salmon	0.09
Fall chinook salmon	0.17
Summer steelhead trout	0.07
White sturgeon	0.73
Largescale sucker	0.28
Carp	1.08

From Anthony et al. 1993
Data from 1991
Location: Columbia River Estuary

Species	Organochlorine Insecticides (PPM)									
	DDE	TDE	DDT	DDTand met	Dieldrin	Aldrin	Endrin	Lindane	Heptachlor	Heptachlor Epoxid
Largescale Sucker	0.29	0.27	0.15	0.61	0.05	0.01	-	0.01	0.02	0.01
Black Bullhead	0.12	0.13	0.05	0.30	0.01	-	0.01	-	-	-
Rainbow Trout	0.09	0.01		0.10	-	-	-	-	-	-
Northern Squawfish	0.69	0.58	0.14	1.42	-	-	-	-	-	-

From these sources, chemical contaminants flow into streams from point and non-point sources. These sources contribute water pollutants such as organic pollution (sewage enrichment, nitrogen and phosphorus), dioxin and furans, heavy metals (arsenic, mercury, copper, cadmium, chromium, lead), pesticides (herbicides, insecticides and fungicides), wood preservatives and hazardous materials (PCBs, trichlorethylene, benzenes, and phenols). Bioaccumulation occurs throughout the aquatic community (Miller et al. 1992) affecting various species of fish and wildlife, and also causing potential human risk. Cumulative impacts of these water pollutants affect aquatic life and may also bioaccumulate in resident fish where consumption may affect human health (Warren and Doudoroff 1971; Hart and Fuller 1974).

BRITISH COLUMBIA, CANADA

In the upper Columbia River Basin, British Columbia, Canada, dioxin and furans enter the river from Canadian pulp and paper mills and flow into Lake Roosevelt (Environment Canada 1989; E.V.S. 1990). The Washington Department of Ecology has documented the chemical contaminants in fish tissue in Lake Roosevelt (1991a, 1991b, 1991c, 1993). The concern is the high sport fishing in Lake Roosevelt and subsequent human consumption of fish.

Although this example only illustrates a portion of the chemical contaminants existing in the Columbia River flowing into the U.S., the need exists to inventory chemical contaminants across the international border. A comprehensive ecosystem approach to understanding the chemical contaminants impacting fish, wildlife and humans must include the entire Columbia River Basin throughout Oregon, Washington, Idaho, British Columbia and Montana.

SNAKE RIVER SUBBASIN

The Snake River Basin drains 29,740 square miles and includes Idaho, northeastern Oregon and southeastern Washington. The Snake River in Washington state begins at the Washington/Oregon border and flows northwesterly along the border of Washington and Idaho to RM 139.1. At this point the Snake River is completely within Washington State. It flows southwesterly to its confluence with the Columbia River at RM 324.2. The Washington state portion of the Snake River Subbasin encompasses about 4,351 square miles. The drainage includes all parts of Asotin, Garfield, Columbia, Walla, Franklin and Whitman counties.

The upper portion of the Snake River forms the state line between Idaho and Washington and Oregon for its 108 miles from Lewiston, Idaho (RM 139.1) to Hells Canyon Dam (RM 247), the upper limit of accessible river to anadromous fish. In this stretch of river, the Snake drains 29,740 square miles. Major tributaries include the Salmon, Clearwater, Grande Ronde, Imnaha, and Tucannon rivers.

Sources of chemical contaminants in the Snake River subbasin are agriculture, mining, food processing, municipal sewage and other industrial discharges. For example, historic mining in Idaho continues to discharge various metals to natural waters. In Panther Creek, a tributary to the Salmon River, a historic mining operation in the 1940's caused the decimation of the salmon and steelhead run in the watershed. Salmon restoration is hindered because high quantities of acid mine water containing copper, cobalt, cadmium and various other metals are discharge into Panther Creek (Everson and Reiser 1985). A similar problem exists in Bear Valley Creek, tributary of the Middle Fork Salmon River. A major mining reclamation project was implemented to abate the discharge of metals and low level radioactive rare earth metals that impact the population of wild spring chinook salmon and steelhead trout (Everson and Konopacky 1985).

Many other sources of chemical contaminants exist in the Snake River subbasin that contribute to the cumulative impact of chemical contaminants in the mainstem Columbia River. A complete literature review and contaminant inventory is beyond the scope of this project.

YAKIMA RIVER SUBBASIN

The Yakima River subbasin contributes a substantial load of chemical contaminants to the Columbia River. Water quality in the basin is significantly affected by intensive agriculture, irrigation, grazing, timber harvesting and urbanization. The U.S. Geological Survey (USGS) selected the Yakima River as one of the basins to be included in the National Water Quality Assessment Program. Consequently, there have been several reports available on the in the water, sediment, and fish.

The USGS (1992) have summarized chemical contaminant data for surface water, sediment, benthic invertebrates and fish tissue in the Yakima River basin from 1976 through 1984. Pesticide data for water exceeded Washington State standards for chronic toxicity. All of the following compounds exceeded standards: aldrin/dieldrin (0.0019 ug/l), DDT+DDD+DDE (0.001 ug/l), endrin (0.0023 ug/l), parathion (0.013 ug/l) and PCB (0.014 ug/l).

The fish tissue data in USGS Report 91-453 have the most relevance to the focus of this project. The trace elements arsenic (As), lead (Pb) and zinc (Zn) in whole-fish tissue were elevated more than one and one-half times when compared with national baseline data. Elevated concentrations occurred in the largescale sucker and the black crappie. Mercury (Hg) concentrations (780 u/kg) were high in edible fish tissue. From 1968-1985 water years, PCB and the pesticides dieldrin, DDT, DDE, and DDD were the predominant organochlorine compounds found in fish throughout the Yakima basin. Salmonids generally had smaller concentrations than resident species. For 1985, concentrations for DDT+DDE+DDD in resident whole fish ranged from 1,100, in largescale sucker, and to 3,000 ug/kg, in northern squawfish. These values exceed the maximum recommended concentration (1,000 ug/kg) by the National Academy of Science for the protection of fish-eating birds (USGS 1992).

The USGS (1994) published the major- and minor-element data for sediment, water, and aquatic biota, for the period of 1987 through 1991. For arsenic, the largescale sucker had median concentration of 0.30 ug/g, however, the other species had higher bioaccumulation. The Asiatic clam had 4.1 ug/g and the waterweed (*Elodea sp.*) had 1.6 ug/g of arsenic. Mercury concentrations in fish tissue were 0.32 ug/g in largescale sucker, 0.38 ug/g in carp, 0.81 ug/g in mountain whitefish, and 0.26 ug/g in rainbow trout. These data suggest that the largescale sucker, used as a target species in the Bi-State Program may not be the best selection for representing bioaccumulation and potential hazards to human health. The selection of the fish species may be more relevant if it corresponded to those species consumed more frequently by the Yakama Tribal members who fish in the Yakima River. For comparison of results, the copper analysis provides an example:

Species	Minimum	Median	Maximum (in ug/g)
Bridgelip sucker	7.7	14.0	19.0
Carp	28.0	55.0	100.0
Largescale sucker	23.0	26.0	32.0
Mountain whitefish	5.6	6.4	11.0
Rainbow trout	18.0	91.0	480.0
Asiatic clam	25.0	28.0	34.0
Caddisfly	9.2	13.0	21.0
Stonefly	27.0	32.0	38.0
Curlyleaf pondweed	9.2	11.0	22.0
Waterweed	13.0	19.0	65.0

Carp, rainbow trout, Asiatic clam, stonefly and the waterweed data were higher values than the largescale sucker at the maximum concentration. For understanding the bioaccumulation of contaminants in the aquatic community, the selection of the target species should include both bottom feeding and predator fish species, freshwater clams consumed by mink and otter and the aquatic insects that represent the lower trophic level of food for fish. The single use of the largescale sucker would be inadequate for understanding the interactions of chemical contaminants in the aquatic ecosystem related to fish and wildlife species.

EFFECTS OF CONTAMINANTS ON ALTERNATIVE TROPHIC REPRESENTATIVES IN THE LOWER COLUMBIA RIVER

Asterionella formosa
Eurytemora affinis
Corophium salmonis
Oncorhynchus tshawytscha

INTRODUCTION

Through the Bi-State Program, the health of the lower Columbia River (LCR) and its estuary is being assessed through the status of certain indicator species, including largescale sucker, bald eagle, mink and river otter. The primary source of contaminants for the four target wildlife species is the prey they consume that is sequestered, concentrated, and transported by various mechanisms. These species appear to integrate and characterize many trophic and ecosystem interactions, they are particularly sensitive to changes in the ecosystem, and they can be used evaluate the effect of changes upon the ecosystem (Clark 1974). The criteria for selection of alternative trophic level species was based primarily upon their relative importance as producers in the LCR, their position in the food web of the lower river and its estuary according to the CREDDP 1984 model, as well as recommendations by the members of the Bi-State Water Quality Program Fish and Wildlife Work Group.

Additional species were chosen for study because of their importance in other trophic levels, such as phytoplankton and zooplankton. One representative producer and three representative consumers were selected as being useful in determining the passage of contaminants to the higher consumers in the LCR (Table 1). We selected these additional species:

-- Phytoplankton - *Asterionella formosa*, a diatom which favors low phosphorus levels and vacates an environment with added phosphorus

-- Water Column Zooplankton - *Eurytemora affinis*, a prominent calanoid copepod

-- Benthic Deposit Feeder - *Corophium salmonis*, a common benthic deposit feeder throughout the Columbia River, a major contributor to production on demersal slope areas and channel bottoms, and an important food organism for higher level consumers such as juvenile salmon

-- Juvenile Chinook Salmon - *Oncorhynchus tshawytscha*, a prey item for bald eagles and occasionally mink and river otter. The adult forms are ecologically and economically important in the Pacific Northwest.

We examined literature for descriptions of habitat, life history, population dynamics, population trends, diet, and their response to contaminants. We tried to analyze and synthesize these data and to identify weaknesses in the data base.

Table 1. Biomass and production of functional groups of organisms (condensed from "The Dynamics of the Columbia River Estuarine Ecosystem, Volume II", 1984).

Taxon	Biomass (gCm^2)	Production ($gCm^{-2}\ yr^{-1}$)
PRODUCERS		
Phytoplankton	3.622	46.907
Benthic Algae	12.421	25.774
Vascular Plants	380.200	392.600
CONSUMERS		
Zooplankton		
(Suspension Feeders)	1.458	14.585
(Predators)	0.010	0.026
Larval Fishes	0.260	0.260
Epibenthic Invertebrates	0.010	0.089
Benthic Infauna		
(Deposit Feeders)	0.616	1.349
(Suspension Feeders)	0.021	0.085
(Predators)	0.165	0.165
Fish	0.083	0.053
Avifauna	0.012	0.009
Marine Mammals		
(Predators)	0.040	0.007
Terrestrial/Aquatic Mammals	0.146	0.102

Inventory and reconnaissance surveys have been conducted on the LCR as part of the Bi-State Water Quality Program documentation of the presence of chemical contaminants in the LCR and its estuary. However, the extent, the mechanisms, and the ramifications of contamination by organochlorines, heavy metals, and other toxins are not yet fully known. Most studies on contamination in the LCR have used higher trophic level species as bioindicators. The lower trophic levels may also incur direct mortality or indirect effects on population health and viability due to contamination. The pathways of contamination and accumulation in the Columbia River estuary include such organisms as phytoplankton, zooplankton, microcrustaceans, and juvenile fish. These organisms interact with physical parameters (such as salinity, pH, and trace metal concentration) in such complexity and detail that the interactions themselves, not just the rate of their changes, appear instantaneous.

Estuarine dynamics and physical processes tend to dominate the life cycles of the smaller organisms in the plankton and benthos. Factors affecting the transport and fixing of contaminants in the estuarine ecosystem often are the same variables directly impacting population trends and life histories of estuarine micro-organisms and their predators. Flow rates, salinity regimes, and other estuarine dynamics can be key parameters in both micro-organism species composition and in contaminant accumulation. In addition, elucidating the effects of ecosystem alteration and contamination in micro-organisms can help clarify the causality that is alluded to in so many studies but rarely quantified or examined at the level of micro-habitat interactions. For example, the turbidity maximum, also known as the null zone or salt wedge, defines a small area within the estuary which has unique dynamics and properties which affect the macro-habitat of the estuarine ecosystem. Unlike the intertidal zone or the forest edge, it has not yet been studied exclusively in terms of its species composition and dynamics. By considering several trophic level organisms the community and system dynamics that are affected by contamination or alteration can be viewed more holistically rather than merely alluding to such dynamics.

FINDINGS

Asterionella formosa

Habitat

A. formosa is a freshwater diatom which can only tolerate very low salinity. It is found in many freshwater environments throughout the world, including the Columbia River. As a diatom in need of nourishment from photosynthesis, it can exist only where it gets enough light, that is, above the compensation point. In an estuary or highly turbid environment, the light extinction coefficient may occur at depths closer to the surface than in other aqueous environments (McClusky 1989). This means that it could potentially exists in its living form throughout the river water column, but is more often limited to higher depths due to light-limitation. It is found in high abundances in the mid and upper Columbia River estuary, yet it encounters high mortality on reaching the zone of salinity intrusion (Frey et al. 1984; Lara-Lara et al. 1990a). The dead, sinking diatoms are still an important component of the ecosystem, albeit a slightly less vital one.

Biophysical Variables and Relative Abundance

As a diatom, *A. formosa* is a single-celled algae whose basic necessities for life are nutrients (phosphorus, nitrogen, iron, zinc, vitamins, and other trace metals), adequate light and water quality, and appropriate ionic concentrations and ratios. In estuaries, phytoplankton are subjected to a high degree of environmental variability to which they must adapt. For phytoplankton in the LCR and estuary, light rather than nutrients is probably the limiting factor on productivity and survival (Frey et al. 1984; Lara-Lara et al. 1990a).

Columbia River estuary phytoplankton are composed primarily of freshwater diatoms, of which *A. formosa* is one of the most abundant species (Frey et al. 1984). Much of the phytoplankton biomass in the Columbia River estuary comes from phytoplankton production upriver. Although in situ production of phytoplankton occurs within the estuary, it appears to be at much lower rates than upriver production. Seventy-five percent of the total organic carbon in the estuary was found to be detrital carbon. Of the 25 percent that is live carbon, 75 percent is imported from the main river stem (Frey et al. 1984). Once the phytoplankton carried by the river flow hit the zone of salinity intrusion, or the salt wedge, they encounter high mortalities and sink to the bottom, so that phytoplankton removal from the estuary and lower river is primarily due to two factors: flushing from high flow rates and salinity induced mortality and subsequent sinking (Lara-Lara et al. 1990b). Grazing by zooplankton appears to account for only 1 percent of the total phytoplankton biomass removal in the estuary (Frey et al 1984).

Contaminants

A. formosa is the only primary producer included as a bioindicator in this review. The interactions of this diatom with pesticides, pollutants, and other contaminants are distinct from those of the other species, for whom contamination usually implies some threat to their reproduction and survival. For *A. formosa*, contamination can either help or hinder its productivity and its total biomass in that certain pollutants may cause blooms, others may be toxic, and others may change the diatom community composition. It, then, can be useful as an indicator less through its population 'status' at a single instant in time but more through changes in its population status over time (Williams 1964). This is related to a high intrinsic rate of growth relative to the larger species under consideration, to its algal existence which makes its diet one of light and nutrients rather than one of other biota, to its ability to absorb or bind to pollutants and toxins rather than incidentally consuming them through the food chain, and to the fact that death of this diatom does not imply removal from the ecosystem. A dead diatom can still be an important diet item for primary consumers. Instead, removal is related to a host of factors such as sedimentation and flow rates. In addition, *A. formosa* is useful to consider not just as a possible bioindicator but an integral variable in the causal pathways leading to contamination in the higher trophic levels.

Planktonic algae are often the mechanism of pollutant incorporation into the trophic structure. Microflora can incorporate and accumulate metals, such as mercury, and other toxins, such as PCBs or DDT, into their cells from the aquatic environment. Other mechanisms of pollutant introduction to the food web are sedimentation of pollutants, incorporation of pollutants by microorganisms, or uptake of contaminants by fish from water through their gills (Lebovitz and Everson 1994).

In trace concentrations, both Zn and Cu are biologically essential, although their uptake patterns by *Asterionella* appear to be very different. In lakes, unlike oceans, there is little evidence that concentrations of dissolved metals are positively correlated with nutrient concentrations. In a productive soft-water lake the mass of zinc associated with the standing crop was about half of that in solution at the time of the *Asterionella* bloom. The mass of copper associated with the standing crop was less than 5 percent. Zinc appeared to be taken up by the algae, while copper does not. The differences in zinc concentrations within the lake were the result algal cycling and reduced vertical mixing in the water column (Reynolds and Hamilton-Taylor 1992).

Riseng et al. (1991) examined the relationship between pH, aluminum concentration, and chelator manipulations on the growth of two species of *Asterionella*. They provide a brief overview of the effects of pH and toxic ions on phytoplankton. pH affects trace metal speciation and can affect trace metal toxicity. Increased levels of mineral acids in precipitation resulting in dissolution of elements from sediments have increased levels of toxic Al ions in acidified lakes. Aluminum appears to have toxic effects on phytoplankton in concentrations as low as 135 ug/L by interfering with intracellular functions such as ion transport of metabolic enzyme activity. Biological responses within organisms are generally because of free metal ion activity, and bioavailability of metals can be a function of chelation and water column pH. Metals can either have reduced effects on organisms at lower pH due to competition with hydrogen ions for active sites (Type I metals), or they can have increased effects on organisms at lower pH due to increased concentration of the free metal ion species in solution and at biologically active sites (Type II metals). This second option appears to be that of lead, mercury, and aluminum, so that these metals could significantly affect organisms such as *A. formosa*, that grow between pH 4 and 7. In addition, if aluminum, lead, and mercury behave similarly under similar physical conditions, aluminum results may be able to be extrapolated for lead and mercury. This experiment also included the effects of interactions with EDTA, which is the experimental complexing ligand which controls trace metal ion activity. Toxic metal may interact with EDTA in experiments to affect growth rates.

The researchers found that measured reactive aluminum covaried with pH and EDTA concentrations in patterns distinct from those found with free aluminum ions. Free aluminum ion concentration was pH dependent, so that as pH decreased from 7 to 5, ion concentration increased 1 to 4 orders of magnitude. pH had a species specific effect on growth rates. *A. formosa* exhibited zero growth at pH 5 and grew best at pH 6 and 7. At pH 7, *A. formosa* showed growth increase after addition of 200 ug/L of aluminum at both 50 and 5.0 mM EDTA. At pH 6 and 5.0, *A. formosa* grew with aluminum concentration at 200 ug/L but exhibited zero growth when concentration rose to 800 ug/L, although growth rates increased at the same aluminum concentration at 50 mM EDTA. These results demonstrated that pH controls trace metal speciation and chelation with EDTA, which may affect phytoplankton growth. Aluminum may act like a Type I metal instead of the Type II it is currently classified as, since it appears to be affected by decreasing pH through competition with hydrogen ions at the cell surface which may decrease toxicity. If this is the case, it may not serve as an adequate analog for mercury and lead, assuming they are currently classified properly. In addition, the myriad interactions which appear to influence the toxicity of a metal do not address uptake. In fact, it may be safe to assume that at levels which are toxic, no uptake would be occurring. The basic point to be taken from such a study is that the bio-physiological parameters affecting metal uptake and toxicity are many, are complex, and tend to influence and interact with each other.

Eurytemora affinis

Habitat

E. affinis is one of the few truly estuarine zooplankton. It is endemic to the estuarine mixing zone (Bottom and Jones 1984). In the mixing zone salinity ranges between that of freshwater from incoming river flow to that closer to the marine waters brought in by tidal changes and mixing. The only other zooplankton endemic to estuaries found in the Columbia River estuary has been *Scottolana canadensis* (Bottom and Jones 1984). Both *E. affinis* and *S. canadensis* are the most abundant zooplankton in the mixing zone of the Columbia River estuary (Bottom and Jones 1984). *E. affinis* can maintain itself in the mixing zone even during the extreme flow changes from spring to late summer, when high spring freshet flows dwindle rapidly to low late summer flows.

The ability of *E. affinis* to maintain itself in the mixing zone throughout the year at relatively high abundances has consistently perplexed researchers in the literature. This is because of the substantial tidal influence, high river flow rates, and, therefore, high flushing rates leading to net seaward flow which characterize the mixing zone. Position maintenance hypotheses for *E. affinis* have included an extremely high reproductive rate which would replace the numbers of estuarine mixing zone resident zooplankton, refuge in bays and other inlets, vertical migration dependent on estuarine upwelling, or passive transport in water moving upstream (Bottom and Jones 1984; Hough and Naylor 1991).

E. affinis is also more abundant at depth than at the surface (Haertel 1969; Cordell et al. 1992). It is the most abundant epibenthic species in the Columbia River estuary, with especially large numbers dominating the species community in late spring (Simenstad et al. 1984). Williams (1983) found that *E. affinis* was always present in the channels of the lower and central Columbia River estuary but was found in higher concentrations in times of higher river flow in the central estuary and in autumn low flow throughout the estuary. Williams hypothesized that *E.affinis* distributions in the estuary are probably most influenced by food supplies. Simenstad et al. (1984) reported that *E. affinis* appears to move up the estuary during periods of low flow (e.g. late summer-fall). Cordell et al. (1992) found *E. affinis* to affiliated with low salinity water masses and/or high turbidity water masses.

Life History

Synthesizing habitat observations such as these and conducting their own observational research on the distribution of *E. affinis* in a highly tidal, mixed estuary in North Wales, Hough and Naylor published two studies which propose to answer the age-old estuarine biology conundrum of how those zooplankton stay there. The answer appears to be an endogenous circatidal swimming rhythm in *E. affinis*. The zooplankton, then, take advantage of tidal currents, like surfers selectively choosing which wave they would like to ride, to entrain themselves in the estuary.

These researchers (Hough and Naylor 1991) found that *E. affinis* was consistently more abundant at downriver sampling sites during flood tide and at upriver sampling sites during ebb tide. In the mid-estuary, abundances tended to be greater on the flood tide and were higher overall than at either the more upriver or downriver sites. Vertical distributions of *E. affinis* likewise fluctuated, depending on time, season, and time of day. The greatest number of individuals were found on flood tides, clustered in the middle vertical zone. According to the researchers, these results indicate a behavioral strategy based on selective tidal transport to retain position within the estuary. The maximum net abundance of copepods occurred during different tides with difference in estuarine positioning, at flood tide most abundant at the seaward end of the estuary and at ebb tide at the landward end. *E. affinis* was consistently found higher in the water column when flow was upriver and on tides increasing in amplitude and appeared to sink when the tide changed. The physical dynamics of the estuary are not sufficient explanation for this. Instead, the copepod must time its swimming activity based on the tides to ensure its presence in the water column. Cues for determining their position relative to their 'preferred' zone appear to be salinity cues corresponding to estuarine zonation and flux. Peak survivorship of *Eurytemora* is between low salinity and 20 ppt.

In their second study Hough and Naylor explained the behavioral mechanism behind the swimming activity of *E. affinis* as an endogenous circatidal swimming rhythm (Hough and Naylor 1992). This would be different from more frequently observed circadian based vertical migrations. Tidal, as opposed to circadian, vertical migrations might be advantageous to organisms living in estuaries where being carried out to sea by river flow could be thwarted by tidal transport back into the estuary, especially for organisms such as *E. affinis* who live part time epibenthically and then migrate up into the faster moving water column on a tidal basis. Only one other organism has thus far been shown to have a circatidal swimming rhythm.

The researchers observed that the activity of *E. affinis* collected from a site mid-way in a tidally mixed estuary and near the limit of tidal influence peaked around 1.5 hours before expected high tide, dropped before high tide, and was maintained at low levels during ebb tide. Later in the same spring-neap tidal cycle, activity peaked after expected high tide and decreased over ebb tide to a minimum during expected flood tide. Animals collected near the limit of tidal influence during early spring tides showed peak activity just after expected high tide and a similar pattern to those collected in mid-estuary during late spring.

The researchers therefore concluded that this was evidence in *E. affinis* of an endogenous circatidal swimming rhythm of upward swimming behavior and that this rhythm as observed suggests the underlying behavioral mechanisms to the tidal abundances observed in the 1991 study. The activity observed from animals collected in mid-estuary in early spring tides and near the estuary mouth occurred before expected high tide and would lead to their rising in the water column on the flood tide and then being carried upstream. The decline in activity before high tide would lead to sinking to the bottom over the ebbing tide which would prevent resuspension and transport downstream of the animals. The rhythms observed appeared to be modulated by estuarine position as well as tidal range. The environmental cues for this rhythm are not yet known. As it appears that *E. affinis* is distributed relative to preferred salinity zone, salinity might provide the cue. Cordell et al. (1992) found similar distributional and activity patterns for *E. affinis* in the Columbia River estuary and interpreted their data as supporting that of Hough and Naylor (1991). This behavioral mechanism for position maintenance might, however, be specific to *E. affinis* and not *S. canadensis*, which instead appears to maintain itself in the estuary through passive concentration by the physical trapping and resuspension processes of the estuarine turbidity maximum, which is also known as the estuarine mixing zone (Cordell et al. 1992).

Little research has been done on the life history of *E. affinis* in Columbia River estuary, although several extensive studies have catalogued the distribution and abundance of *E. affinis* in the estuary, including seasonal and salinity variations. Since *E. affinis* diets are primarily composed of phytoplankton and organic detritus and since *E. affinis* is a small, fast-reproducing organism, it is possible to infer their reproductive maxima relative to their overall abundances and to their seasonal correlations with phytoplankton abundances. Therefore, if *E. affinis* abundances can be directly related to *E. affinis* production and if food availability also conditions *E. affinis* production, the following sections on population dynamics and diet should be read as indicators of the seasonality of *E. affinis* production and growth.

Diet

E. affinis is a grazer which consumes phytoplankton and other water born organic particles. Frey et al. (1984) determined grazing rates on phytoplankton by zooplankton in the Columbia River estuary. Maximum grazing removal occurred in later spring and summer, averaging 4.6 $mgCm^{-3}$ per day. This was equivalent to 1.2 percent of the total phytoplankton carbon available. The annual phytoplankton removal due to zooplankton grazing was estimated to be 669 $mgCm^{-3}$ per year. However, the in situ production of Columbia River estuary is low relative to both other estuaries and the amount of consumers in the estuary. The researchers stated that the higher trophic levels, such as zooplankton, are able to remain productive because phytoplankton biomass is supported by import from the river more than in-estuary production. A more recent study by Lara-Lara et al. (1990b) confirmed this hypothesis, demonstrating that 75 percent of estuarine phytoplankton came from upriver production.

A substantial amount of phytoplankton are killed when they reach the higher salinity in the intruding salt wedge. The sinking organic detritus that once was living diatoms can also be important food for epibenthic zooplankton, such as *E. affinis*. In conjunction with this observation, abundances of *E. affinis* have been shown to shift following the seasonal shift in the null zone, or salt wedge intrusion. Thus, the observations by Simenstad et al. (1984) and Williams (1983) over the principal factors influencing zooplankton densities and population structures in the estuary might not be contradictory but compatible. Food resource availability might be directly related to seasonal and spatial variations in salinity intrusion and null zone processes.

Contaminants

No research has been conducted on the effects of contaminants on *E. affinis* in the Columbia River or its estuary. Little to no research had been done on this genus and contaminants effects in general. However, many assumptions have been made about food chain bioaccumulation leading to the contaminant levels observed in Columbia River estuary and LCR predators. The absence of research, especially researching addressing the community dynamics of zooplankton, invertebrates and fish assemblages, on this potential link in the chain is then notable. *E. affinis* is abundant in the estuary and is important in the diets of many juvenile fish and larger invertebrates (Simenstad et al. 1984; Critic et al. 1976).

Dawson (1979) summarized the effects of pollution on estuarine copepods such as *Eurytemora*. Heavy metals from human activities can be toxic to zooplankton. Feeding on phytoplankton which have adsorbed heavy metals can be a source of contamination for copepod zooplankton. The negative effects of heavy metal contamination depend on the type and concentration of metal and the species affected, although the author highlights mercury, silver, and copper as most toxic, followed by cadmium, zinc, lead, chromium, nickel, and cobalt, in order of decreasing toxicity. The degree of toxicity of the metal to zooplankton will be affected by the form of the metal in the water, the presence of other metals acting synergistically, environmental conditions, and life history stage of the organism. Effects can be lethal or sublethal. The implications of sublethal effects in copepods can include morphological change, inhibitory effects on growth and development, and behavioral change.

The author stated that two metals amply studied in marine environments have been mercury and copper. Mercury has been identified as moving through the food chain from nearshore phytoplankton to offshore consumers, although there appears to be little difference in copepod sensitivity to mercury nearshore or offshore. This is not the case for copper. It has been shown to affect local populations of copepods differently. Feeding rate and egg production appear to be the factors most sensitive to sublethal quantities of heavy metals. Copper and mercury can act synergistically to multiply the effects of heavy metal concentrations in copepods. Mercury has not been shown to amplify as it moves up trophic levels.

The effects of PCBs on estuarine copepods have not been studied (Dawson 1979). PCBs do not appear to be concentrated from pelagic zooplankton to fish up the trophic structure. DDT has been shown to be concentrated through trophic levels from plankton to birds, although other evidence suggests DDT concentration depends on species-specific trophic interactions.

Chesapeake Bay fish and invertebrates studied the Acute toxicity of tributyltin (TBT) to , including *E. affinis* (Bushong et al. 1988). TBT is often used as a biocide in anti-fouling paints for commercial and recreational watercraft and is effective because it is so toxic. Salinity throughout the experiment ranged from 9.8 to 12.1 percent and pH ranged from 8.15 to 8.31. TBT concentrations were between 0.2 and 32.0 ug/l in four experiments. Control survival for *E. affinis* was around 80 percent or greater. *E. affinis* was one of the two most sensitive organisms tested. Its 72-h LC_{50} (least concentration at which 50 percent mortality occurs) value was 0.6 ug/l. In the Chesapeake Bay, TBT concentrations have been reported to reach values almost three times higher than this. The researcher state that similar levels of TBT have been found in other others with high boating activity. The LCR and estuary is one such area, although it was not mentioned in this paper.

Population Dynamics

High densities of zooplankton in the estuary, especially during late spring in the lower estuary, early summer in the estuarine mixed zone, and late summer in the freshwater zone. Densities of *E. affinis* were responsible for the spatial and temporal abundances of zooplankton found, since *E. affinis* was the most abundant taxon in the estuary, with densities consistently above 100,000 per meter squared. The center of population abundance shifted up and down the estuary as freshwater inputs from river flows varied. *E. affinis* was predominantly found in the estuarine mixing zone but the center of abundance moved from the marine and mixing zones in the spring to the mixing and freshwater zones in the summer (Bottom and Jones (1984).

Simenstad et al. (1984) surveyed similar abundances and distributions of *E. affinis* in the epibenthos of the estuary. It was the most abundant single species there, although numbers fluctuated with the seasons. *Eurytemora* dominated the shallows in late spring, as well as the deeper waters during spring freshet. These researchers hypothesized that *E. affinis* abundances and population structure relative to other zooplankton in the estuary appeared to be principally determined seasonally and spatially by circulation, salinity intrusion, and null zone processes. This would be in contrast to the motile macroinvertebrates, one of their predators, whose abundances and assemblages would be determined by prey resources. However, Williams (1983) hypothesized that *E. affinis* distributions may be more influenced by food resources, whereas an infaunal species such as *C. salmonis* may be distributed more according to sediment types.

Corophium salmonis

Habitat

C. salmonis is a tube-dwelling amphipod that lives in fine sediment substrate in which it embeds itself. It appears to prefer firm sediments composed of silt or muddy sand (Albright 1982) and is typically found only within the top 10 cm. of sediment (Holton et al. 1984). Studies of *C. salmonis* habitat preferences in the Columbia River estuary have found higher densities of the organisms in fine sediments located in shallow bays and shoaling areas of the central and upper estuary (Williams 1983). *Corophium salmonis* distributions are probably highly dependent on sediment type but also appear to be limited by the salinity of the surrounding water and interstitial salinity. The organisms appear to prefer salinity less than 10 ppt (Williams 1983; Holton et al. 1983). Reish and Barnard (1979) stated that the ability of estuarine species of *Corophium* to tolerate salinity changes is related to the amount of fresh sediment supplied. Embryos and larvae are more sensitive to salinity changes.

C. salmonis individuals in the Columbia River were found to be associated with estuarine conditions but able to live in fresh water, being more frequent in the central and upper estuary in riverine areas than near the estuary mouth or the bays near the river mouth (Williams 1983). One explanation for its frequency in riverine areas is the tendency of samples to collect young males in the water column who have left their tubes in search of better habitats and/or mates (Williams 1983; Holton et al. 1984; Albright 1982). Simenstad et al. (1984) found *Corophium* species to be most common in the upper estuary tidal flats and demersal flats.

Life History

Holton et al. (1984) conducted an extensive study on the life history of *C. salmonis* in the Columbia River estuary. They sampled sites at Grays Bay and Desdemona Sands between August 1980 and June 1981. A summary of their results follows. Desdemona Sands had higher salinity than Grays Bay, ranging from 10.5 ppt in late December of 1980 to 3 ppt in late February 1981. Juveniles were produced between August and November at Grays Bay. After November, the disappearance of breeding females left the population dominated by males and juveniles. Over the spring the juveniles released in early fall matured to compose the adult population, while those released later in fall matured to compose the population of immature adults. In May 1981, there was a second peak of juveniles, with the result that the population had two distinct cohorts falling into juveniles and adults. The data from both Grays Bay and Desdemona Sands suggested that the *C. salmonis* population produced two generations a year, one in spring, the other in fall. However, at Desdemona Sands the pattern of animal residence differed. Juveniles disappeared in the fall, and adults and older immature adults reappeared in the spring when reproduction occurred again. The two generational cycle, then, is based on the fall production of juveniles, the death of the reproducing adults in winter, and the maturation of the fall juveniles into reproducing adults which produce the spring new juveniles. Temperatures stimulating reproduction appeared to be above 7°C.

Densities at Grays Bay increased throughout the fall and winter, peaking at 31,754/m^2 in February of 1981. Densities in March and April suggested a disappearance of the reproducing population. At Desdemona Sands, the population disappeared in September 1980, reappearing in April 1981 and peaking in August at 96,096/m^2, probably due to recruitment of juveniles from fall and spring generations. Mean brood sizes were 14.17 at Grays Bay and 16.10 at Desdemona Sands. Albright (1982) found mean brood sizes in Grays Harbor, W to be 11.4 but noted that this number was probably too low due to sampling techniques. Although Desdemona Sands had significantly larger broods, Grays Bay supported significantly longer mature females, 5.03 mm and 4.90 mm, respectively. This may be because of an interaction of high summer temperatures and reduced nutrient transfer to the gonads causing more rapid maturation and less rampant reproduction.

Males were smaller than females on average. Sex ratios hovered around 1.0, except for four sampling dates at Desdemona Sands when males outnumbered females. Usually the situation is reversed, with females outnumbering males. Albright (1982) found male-female ratios in Grays Harbor to be lower for individuals greater than 4.0 mm in length and this was more pronounced in late spring and summer than in early spring. Males seem to get eaten more by juvenile chinook and other fish predators, either because they are selectively preyed upon or because they are more active outside the burrow and thus better food targets. Albright (1982) hypothesized that the sex ratio difference for the larger size class was because of predation on males who left their tubes to look for mates. Williams (1983) noted that the higher overall densities of *Corophium* sampled in the water column than in more protected areas was probably because of the tendency of males to leave their tubes and enter the water column.

The Holton et al. (1984) study proposed another explanation for the presence of *C. salmonis* in the water column and relate it to the observed density fluctuations. The researchers state that the density fluctuations observed at the two sites are not explainable solely by juvenile recruitment but must also be due to other factors such as winter adult immigration, and spring adult emigration, die-off, or predation by salmon and starry flounder at Grays Bay. Predation by juvenile chinook and starry flounder primarily occurs between June and September (Higley and Holton 1975). Predation, then, does not seem to be a likely explanation for the disappearance of the Desdemona Sands population during fall and winter. Predator impact may, however, be a viable explanation for the Grays Bay male:female ratios which declined during summer there. The researchers cite vertical migration to take advantage of tidal currents which would redistribute the organisms to more favorable habitats as a better explanation for the Desdemona Sands population change in the fall. Habitat suitability for *C. salmonis* would be determined by temperature, salinity, and sediment characteristics. The most noticeable difference between the two sites that would account for the differing density distributions was salinity. Salinity in the fall at Desdemona Sands rose to above 10 ppt, coinciding with the disappearance of *C. salmonis* at this site and the immigration of adults to Grays Bay. This suggests that *C. salmonis* populations in the Columbia River estuary adapt to changing environmental conditions through migration into tidal currents.

Diet

C. salmonis are deposit feeders who scavenge and consume detritus. They can sometimes be considered filter feeders, using the current generated from water filtered through their tubes (Holton et al. 1984). They feed through ingesting diatoms, sediments, and other windfalls they receive from water filtered through their tubes (Reish and Barnard 1979). More research has been done on the organisms that eat *C. salmonis* than on its diet or foraging ecology.

Contaminants

Contaminants effects on estuarine amphipods, such as *C. salmonis*, have been little studied. Reish and Barnard (1979) provided some general observations on the effects of pollutants on estuarine amphipods. Amphipods are absent from severely polluted harbors and may be more sensitive to heavy metals although little data existed at the time of their review to support this hypothesis. In general, amphipods have been found to be more sensitive to the effects of environmental change than other substrate dependent invertebrates, such as polychaetes and mussels, suggesting that their absence may possibly be an indication of degradation.

Although amphipods may be highly sensitive to the toxicity of contaminants, they also may be metabolizers of some contaminants present in non-toxic levels. Invertebrates and fish are known to be responsible for the metabolism of *p,p'* DDT to *p,p'* DDE and *p,p'* DDD, although other organisms such as algae or copepod zooplankton have also been found to be capable of organochlorine degradation (Addison 1976). The metabolism of organochlorines by invertebrates appears to be species specific, even within the same order or genus. In larger crustaceans, such as lobsters, *p,p'* DDT was metabolized to *p,p'* DDD and *p,p'* DDE in approximately equal proportions (Addison 1976). Smaller invertebrates such as planarian worms have been found to perform similar functions (Addison 1976). Although specific studies demonstrating the DDT metabolism of Corophium were not found, they may be able to degrade low concentrations of DDT as well.

Reish (1993) studied the amphipods *C. insidiosum* and *Elasmopus bampo* to determine the effects of metals and organic compounds on their survival and on bioaccumulation in these species. In addition, the author summarized the toxicological research on amphipods since their last review in 1979. The author identified no studies on *C. salmonis*, although several had been conducted using other members of the *Corophium* genus. *C. insidiosum* was the only *Corophium* species upon which studies of organic toxins had been performed. The effects of heavy metals on amphipods had been most studied. In the research portion of this study, the researcher examined the effects of chlorides of cadmium, copper, lead, mercury, and zinc, as well as DDT and Arochlor 1254 (PCB), on the two amphipod species.

The 96-h LC_{50}s (least concentration at which 50 percent mortality is incurred) were determined for the toxicants on each species (see Table 2). Mercury and copper were the most toxic metals to both species, and cadmium was also highly toxic. Chromium was the least toxic metal to *C. insidiosum*. DDT and PCB were highly toxic for both species, and *C. insidiosum* was more sensitive to their toxic effects. Heavy metals, especially copper, mercury, and zinc, were found in higher concentrations in *C. insidiosum* than in *E. bampo*. Uptake of DDT and PCB occurred in *E. bampo* but not *C. insidiosum*.

These marine gammaridean amphipods seemed to occupy an intermediate position with respect to the 96-h LC_{50} data relative to others of their genus. *C. insidiosum* was more sensitive to the toxic effects of the organic compounds, so that little uptake would occur because the organisms were instead quickly dead. However, *C. insidiosum* seemed to uptake the metals more efficiently than *E. bampo*. The researcher observed that the heightened sensitivity of *C. insidiosum* to metals but total lack of sensitivity to the organic compounds in terms of uptake, with the reverse trend exhibited by *E. bampo*, may be illustrative of different uptake capabilities in the two species. This would imply that uptake and accumulation effects need to be evaluated separately from toxic effects and are species specific.

Table 2. Effect of toxicants on marine gammaridean amphipods (mean 96h LC_{50} in mg l^{-1} and 95 percent confidence intervals) (Reish 1993).

Toxicant	*Corophium insidiosum*	*Elasmopus bampo*
Arsenic	1.1 (0.8-1.6)	2.75 (1.8-4.3)
Cadmium	0.68 (0.3-1.8)	0.9 (0.6-1.3)
Chromium	11.0 (9.0-13.4)	3.4 (1.9-6.0)
Copper	0.6 (0.3-1.4)	0.25 (0.2-0.4)
Lead	>5.0	>10.0
Mercury	0.02 (0.09-0.06)	0.02 (0.02-0.04)
Zinc	1.9 (0.7-5.6)	12.5 (2.6-60.3)
DDT	0.00007-0.0004	0.002 (0.002-0.003)
PCB	0.009 (0.004-0.02)	0.04 (0.02-0.07
Altosid		>100.0
BTI	--	12.8
W-S Diesel	0.9 (0.5-1.5)	2.6 (1.7-3.0)

Vermeer et al. (1993) analyzed eight estuarine bird species and their prey in the Somass River estuary, BC, Canada for residues of the dioxin, 2,3,7,8-substituted polychlorinated dibenzodioxin (PCDD), and of the furan, polychlorinated dibenzofuran (PCDF). The estuary is adjacent to a pulp and paper mill on Vancouver Island. The bird species analyzed included western grebes, a prey species of Columbia River bald eagles (Watson et al. 1991), as well as mergansers. Prey species analyzed were Pacific staghorn sculpins, spine sticklebacks, and *Corophium* species and were collected from near the pulp mill. Sediment samples from near the pulp mill were also analyzed for toxin residues.

Elevated concentrations of PCDD and PCDF congeners were found in grebes and ducks. Elevated dioxin concentrations were also found in surface sediment samples from the estuary (Table 3). Western grebes and common mergansers had the highest dioxin concentrations. The common merganser prey collected consisted primarily of juvenile salmon. *Corophium* was eaten by all of the birds analyzed and exhibited high dioxin concentrations. The birds, then, may have become contaminated from consuming *Corophium or from feeding on other fish which eat Corophium*. However, the researchers noted that the furan to dioxin congener ratio in the birds analyzed was higher than in the *Corophium* sample, suggesting another dietary or non-dietary source of furan congeners than those sampled.

Table 3. PCDD and PCDF residues in fish and *Corophium* amphipods from the Somass River estuary, Vancouver Island, April 1991 (Vermeer et al. 1993).

Dioxins and furans[a]	Residues in ng kg^{-1}[b]			
	Staghorn sculpins (6)[c]	Three spine sticklebacks (6)	*Corophium* amphipods (20)	Sediment
2,3,7,8-T4CDD	ND	ND	ND	10
1,2,3,7,8-P5CDD	0.4	0.6	2.9	161
1,2,3,6,7,8-H6CDD	ND	ND	29	1677
1,2,3,7,8,9-H6CDD	ND	ND	13	327
1,2,3,4,6,7,8-H7CDD	ND	ND	38	5831
O8CDD	ND	ND	180	20,441
2,3,7,8-T4CDF	1.8	2.8	5.1	123
2,3,4,7,8/1,3,4,8,9-P5CDF	ND	ND	ND	65
1,2,3,4,6,7,8-H7CDF	ND	ND	15	5069
O8CDF	ND	ND	27	5634

ND = Not detected.
[a]2,3,7,8-Congeners not listed were less than detection limits.
[b]Wet wt for organisms, dry wt for sediments.
[c]Number of samples pooled for analysis.

The ratios and concentrations of dioxins and furans were similar in the *Corophium* and the sediment samples. This is not surprising, since *Corophium* make their tubes from the sediment. There was a greater concentration of low molecular weight congeners in *Corophium*, indicating selective uptake of low molecular weight congeners. The researchers stated that it was unexpected to find that *Corophium* formed such a high proportion of the birds' diets, yet in these degraded habitat conditions tube-dwelling amphipods were some of the sole survivors. Gammarid amphipods have been observed to be abundant in this estuary despite overall low species diversity. These observations may be useful or analogous to the Columbia River estuary which itself receives the downstream output of pulp mills.

Population Dynamics

Albright (1982) studied the population dynamics of *C. salmonis* in Grays Harbor, Washington. *C. salmonis* is one of the dominant benthic macroinvertebrates in the muddy sand of inner Grays Harbor and is important prey item for many fish, including juvenile salmon, and sometimes for waterfowl and other invertebrates. Densities at two stations ranged from 216 to 49,675 individuals per meter squared. Densities were highest 1.8 m above MLLW. In general, population densities decreased in early spring, increased dramatically with the appearance of juveniles in mid-May, peaked in July, and leveled off in August and September. The author found high production values for *C. salmonis* relative to other marine macroinvertebrates with similar life spans. Turnover rates, as the ratio of production to mean biomass, were also high. For example, at one station 1.8 m above MLLW total production was 10.72 grams per meter squared and the mean biomass there was 1.14 grams per meter squared over the period 1 April to 30 September in 1980. Production levels were lower at stations only .6 m above MLLW. The author states that these results indicate that the amount of organic material made available to predators by *Corophium* is higher than estimates of biomass alone would allude to and that therefore *Corophium* species are important secondary producers and are key prey species.

Holton et al. (1984) found the total annual production of *C. salmonis* at Desdemona Sands in the Columbia River estuary to be 13,150 mg AFDW/m^2 and at Grays Bay in the estuary to be 8,228 mg AFDW/m^2. The ratio of production to mean biomass (P:B), or turnover rate, was 12.29 at Desdemona Sands and 5.49 at Grays Bay. This high ratio for Desdemona Sands is because of low biomass values during most of the year and high spring and summer production. P:B values at both sites were higher than those for other invertebrates at those sites. The total annual macrofaunal production values were relatively low for estuarine fine sediment habitat communities. The huge majority of the production at both sites was contributed by *C. salmonis* (> 90 percent). *C. salmonis* likewise dominated the community assemblages of both sites numerically. The researchers state that the small macrofaunal communities sampled were relatively simple and that this is probably because of the high degree of environmental instability to which they are subjected, through rapid salinity changes typical of the middle and upper estuary. These zones may be more susceptible to disturbance. It is interesting to compare this observation with those made in the Saumass River estuary in Canada in terms of possible contaminant implications (see discussion of Vermeer et al. 1993 in Contaminants section).

Williams (1983) found *C. salmonis* to be most abundant in spring and autumn in the mid-upper estuary. Simenstad et al. (1984) hypothesized that motile macroinvertebrate assemblages are structured by the distribution of their prey resources, salinity intrusion, and predator assemblages. *C. salmonis* can be seen as partially motile, having the ability to swim in the water column and doing so during an important fraction of its life history (Holton et al. 1984; Williams 1983).

Oncorhynchus tshawytscha (Juvenile)

Life History

Within their anadromous and semelparous life history strategy, chinook salmon (*O. tshawytscha*) populations display a broad array of life forms and behavioral tactics which include variation in age at seaward migration, variation in length of freshwater, estuarine, and oceanic residence, variation in ocean distribution and ocean migratory patterns, and variation in age and season of spawning migration. Healey (1991) attributes a large part of this variation in *O. tshawytscha* populations to the fact that the species occurs in two behavioral forms: 'stream-type' and 'ocean-type'.

Gilbert (1913) designated stream-type chinook as those that spend one or more years as fry or parr in fresh water before migrating to sea, perform extensive offshore oceanic migrations, and return to their natal river in the spring or summer, several months prior to spawning. Males of this form have been reported to occasionally mature precociously without going to sea. Ocean-type chinook are typical of populations that migrate to the sea during their first year of life, normally within three months after emergence from the spawning gravel, and then spend most of their oceanic life in coastal waters, returning to their natal river in the fall, a few days or weeks before spawning (Figure 1).

According to Healey (1991), this variation represents adaptation to uncertainties in juvenile survival and productivity within diverse freshwater and estuarine nursery habitats. By evolving a variety of juvenile and adult behavior patterns, the risk of mortality is spread across years and habitats (Real 1980; Stearns 1976). Since the focus of this report is to summarize existing information on chinook smolt as an alternative trophic level species to the four designated indicator species (bald eagle, mink, river otter, and largescale sucker) with regard to chemical contamination in the LCR, the following discussion encompasses only the juvenile life history stages of *O. tshawytscha* and emphasizes smolt where data are available. The text below on life history and habitat is primarily adapted from Healey (1991).

I. Freshwater Residence and Downstream Migration

The downstream movement of stream- and ocean-type chinook fry is probably a dispersal mechanism that distributes fry among suitable rearing habitats. In the case of stream-type populations, fry migration serves principally to distribute the chinook among suitable freshwater nursery habitat, whereas for populations that spawn close to tidewater (ocean-type), downstream dispersal carries the fry to estuarine nursery areas. Healey (1982, 1980), Levy and Northcote (1982), and Northcote (1976) document the importance of estuaries as nursery rearing habitat for recently emerged chinook fry.

In late spring after the initial downstream dispersal of fry after emergence in the spawning beds, there appears to be a second dispersal that carries some populations to the ocean or simply redistributes the population within the river system, presumably to more suitable summer rearing habitats. For populations that remain a year in fresh water, there is a third late fall redistribution to suitable over-wintering habitat, usually from the tributaries to the river main stem. Finally, in the spring of the second year there is a migration of yearling smolts to the sea (Healey 1991).

Overall, during late spring and fall redistributions in fresh water, chinook populations tend to shift into deeper water and move toward the ocean. These habitat changes are consistent with the shorter term habitat changes recorded by Chapman and Bjornn (1969) and Lister and Walker (1966), in which juvenile chinook moved into deeper, faster water as they grew in size. Healey (1991) hypothesizes that these redistributions may punctuate developmental stages as well as achieve more efficient utilization of the freshwater nursery habitat, and that the tendency for redistribution to carry the fish downward toward the ocean may be coincidental. The author also states that such movement pattern may also be adaptive by shortening the length of spring migration for yearling smolts, particularly for headwater spawning populations in large rivers, like the Columbia.

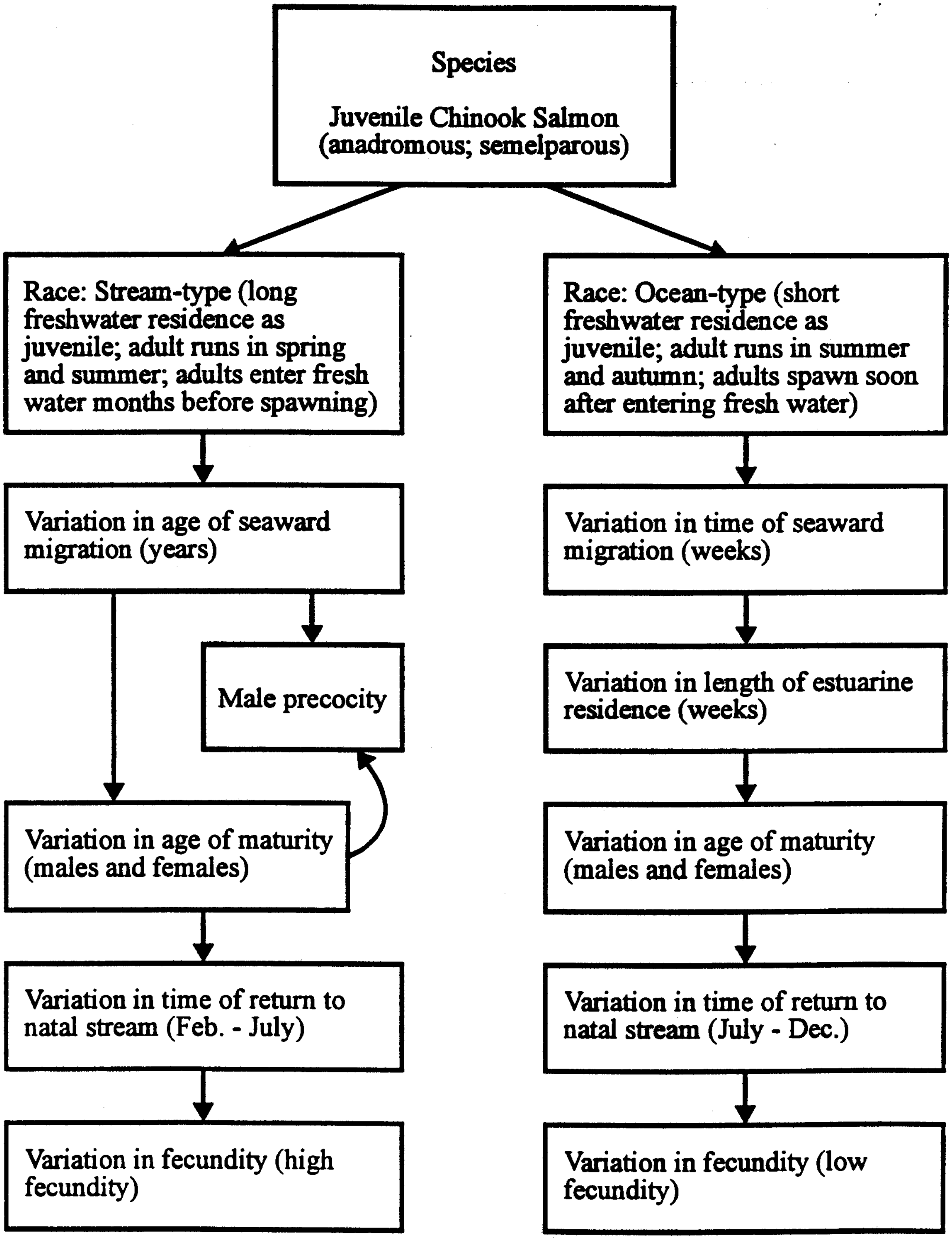

Figure 1. Life history structure of chinook salmon showing the division of the species into two races (ocean- and stream-type) and the range of tactical variation within each race (Healy 1991).

I.a. Fry Migrants

Healey (1991) reports that factors controlling the emergence of chinook fry from the spawning beds are not well studied. Upon emergence, fry swim, or are displaced, downstream. Downstream movement of fry occurs mainly at night, although small numbers may move during the day. Once migrating downstream, chinook fry may continue to the river estuary, or may stop migrating and take up residence in the stream for a time period ranging from a few weeks to a year or more. River discharge and competition with siblings for food and habitat may both play a role in stimulating downstream movement of chinook fry (Healey 1980; Kjelson et al. 1981; Lister and Walker 1966; Major and Mighell 1969; Reimers 1968). Other factors, such as interactions other species may also influence movement of chinook (Stein et al. 1972; Taylor and Larkin 1986; Taylor 1988), with habitat segregation occurring as a mechanism for reducing competition rather than a result of competition (Bjornn 1969; Everest and Chapman 1972).

I.b. Fingerling Migrants

Although factors stimulating downstream movement of subyearling chinook are not known, downstream migration of fingerlings follows fry migration (Healey 1991). Fingerling migrants are distinguished from fry by lack of a yolk sac and by their larger size. Fry migrants usually range from 30-45 mm in fork length while fingerling migrants normally range from 50-120 mm in fork length and have actively been feeding for some time (Healey et al. 1977; Lister et al. 1971; Mains and Smith 1964).

Although downstream movement of chinook fingerlings is generally complete by the end of June in most rivers sampled (Healey and Jordan 1982; Lister and Walker 1966; Lister et al. 1971; Reimers and Loeffel 1967), data exist which indicated downstream migration at other times of the year as well (Northcote 1976; Bjornn 1971). Reimers and Loeffel (1967) observed juvenile chinook in some Columbia River tributaries as late as October. They attributed the extended residence in the Columbia River tributaries to slow growth, and proposed that size was an important variable in determining when fingerlings migrate downstream. However, as Healey (1991) notes, downstream migrant fingerlings vary in size, both within and between rivers (Table 4). He therefore suggests that some factor other than size also plays a role in the downstream migration of stream-type chinook fingerlings. Furthermore, since subyearling chinook have been documented to move out of tributaries and into a river main stem, or to relocate downstream with the approach of winter (Bell 1958; Chapman and Bjornn 1969; Park 1969), and since suitable summer habitat may not be suitable winter habitat, the disappearance of chinook from some Columbia River tributaries in October or November observed by Reimers and Loeffel (1967) may indicate a relocation to an instream wintering area rather than a seaward migration.

The majority of fingerlings migrate downstream at night although such downstream movement is also observed throughout the day (Lister et al. 1971; Mains and Smith 1964). Mains and Smith (1964) also reported fingerling preference of shoreline during their

Table 4. Fork length (mm) of age 0.0 and 1.0 riverine smolts in various rivers and years (Healey 1991).

River	Year	Fork length		Source
		Mean	Range	
Age 0.0				
Sixes (OR)	1969	62.0	40-91	Reimers (1971)
Nitinat (BC)	1980	52.7	44-67.5	Healey (unpubl. data)
Cowichan (BC)	1966	77.3	63-98	Lister et al. (1971)
	1967	72.1	60-91	"
	1978	63.8	57-75	Healey (unpubl. data)
Nanaimo (BC)	1979	68.8	52-84	Healey & Jordan (1982)
	1980	63.5	49-76	"
Big Qualicum (BC)	1972	66.5		Paine et al. (1975)
Age 1.0				
Yakima (WA)	1959	125.5	105-170	Major & Mighell (1969)
	1960	124.6	105-170	"
	1961	127.0	105-170	"
	1962	134.0	90-170	"
	1963	132.6	90-160	"
Snake (OR)	1954	101.0	55-147	Mains & Smiths (1964)
	1955	101.0	55-147	"
	1957	68.4	45-105	Bell (1958)
	1958	67.9	50-95	"
Upper Columbia (OR)	1955	84.2	55-140	Mains & Smith (1964)
Taku (BC, AK)	1961	73.3	45-110	Meehan & Siniff (1962)
Crooked Cr. (AK)	1961	93.5	90-140	Waite (1979)

migration studies in the Columbia and Snake rivers. However, in the lower Fraser River and in the Nanaimo River, most fingerlings migrated in the fastest moving water in the center of the rivers studied (Healey and Jordan 1982). According to the 1978 studies of Cramer and Lichatowich in the Rogue River, the rate of downstream migration of chinook fingerlings appears to be both time- and size-dependent as well as related to river discharge and the location of chinook in the river.

Direct estimates of chinook fingerling growth in fresh water exist only from studies of Kjelson et al. (1982) for the Sacramento-San Joaquin estuary, California. Inferences about growth in other river systems can be made from seasonal changes in the size of resident chinook or from the size of downstream migrants. Healey (1991), however, cautions against such extrapolations because the length of freshwater residence is not precisely known for either the downstream migrant fish or for those captured during river residence.

I.c. Yearling Smolt Migrants

Stream-type chinook do not migrate to the ocean during their first year of life, but instead migrate during the spring following their emergence from the spawning gravel, and, in northern rivers, sometimes not for an additional year (Healey 1983). Although stream-type chinook generally return from the sea to their natal river in spring, exceptions have been reported to occur in some Oregon rivers, such as the Rogue, where spring-run adults produce subyearling smolts (Cramer and Lichatowich 1978; Nicholas and Hankin 1988). Chinook overwintering in large rivers tend to move out of the tributary streams and into the river mainstem, where they occupy deep pools or crevices between boulders and rubble during the winter. A fall redistribution of juvenile chinook from 'preferred' summer habitat to 'preferred' winter habitat has been observed in several river systems (Bjornn 1971; Carl and Healey 1984; Chapman and Bjornn 1969; Don Chapman Consultants 1989; Reimers and Loeffel 1967). Generally, yearling smolts migrate seaward in the early spring, sometimes preceding the main migration of fry and fingerlings and sometimes mixed with them (Bell 1958; Major and Mighell 1969; Meehan and Siniff 1962; Waite 1979). Raymond (1968) reported the rate of downstream migration of yearling smolts in the Columbia and Snake rivers to be positively correlated with discharge and the travel rates through free-flowing and impounded sections of these rivers to be similar. However, the rapid migration of chinook smolt through impoundments on the Columbia River indicates that yearling smolt undertake a directed migration that is independent of river flows (Healey 1991). Yearling smolt vary greatly in size. The existence of two distinct size groups of fish in the same river (Table 4) suggests that there may be important differences in microhabitat affecting chinook growth in rivers.

II. Estuarine Residence

Chinook salmon, because of their many juvenile life history patterns, have the most varied pattern of estuarine utilization of all *Oncorhynchus* species. Chinook fry appear to remain in estuarine nursery areas until they attain about 70 mm fork length, after which they disperse to nearby marine areas (Healey 1991). Estuarine residence times by juvenile salmonids appears to vary greatly with respect to spatial, temporal, and biological factors -within species, between species, estuaries, and locations within estuaries (Lebovitz 1992). Variation also occurs in relation to salmonid migration strategies and physical conditions, particularly size (Reimers 1973; Healey 1980; Healey 1982; Levy and Northcote 1982; McCabe et al. 1986; Nicholas and Hankin 1988; Fisher and Pearcy 1989). Juvenile chinook peak residence is May-July (R=Jan-Nov). Residence time can vary from one week to six months reflecting the different migration strategies and migration distances the species employ.

Our current understanding of the processes which influence juvenile salmonid residency in estuarine areas is incomplete, but it may be correlated to the three primary benefits which estuaries are believed to provide for juveniles: productive foraging habitat, predator avoidance, and adaptation to increasing salinity during juvenile migration to the sea (Reimers 1973, Healey 1980, Levy and Northcote 1982, Simenstad et al. 1982, Nicholas and Hankin 1988, Fisher and Pearcy 1989, Shreffler et al. 1990). It is likely that the above three functions are closely interrelated and may not be easily distinguished from one another. It also seems that any combination of these functional conditions existing simultaneously is essential for salmonids during some portion of their juvenile life. It is possible that without the existence of estuarine habitat, specific salmonid stock/race biological requirements may not be met.

II. a. Fry Migrants

Many of the ocean-type juvenile chinook that migrate downstream immediately after emerging from the spawning gravel reside in the river estuary and rear there to smolt size. Recently emerged chinook fry have been reported to rear in the Sacramento and Columbia River estuaries (Kjelson et al. 1982, 1981; Rich 1920), in the Skagit River estuary (Congleton et al. 1981), in the Fraser River estuary (Dunford 1975; Goodman 1975; Levings 1982; Levy and Northcote 1982, 1981; Gordon and Levings 1984), the Nanaimo River estuary, the Campbell River estuary and other estuaries on the east coast of Vancouver Island (Healey 1982, 1980; Levings et al. 1986), and the Nitinat and Somass River estuaries on the west coast of Vancouver Island (Birtwell 1978; Healey 1982).

Because of differing salinity regimes, estuary rearing is qualitatively different from river channel rearing upstream. In some estuaries the salinity of the rearing habitat is low (e.g., Sacramento River: Kjelson et al. 1982; Fraser River: Levy and Northcote 1982, 1981;) or is unknown. However, observations from the Cowichan, Nanaimo, Courtenay, Campbell and Nitinat River estuaries indicate that chinook fry may rear where salinity is commonly 15-20 ppm or more (Healey 1982, 1980; Levings et al. 1986). Although chinook fry may be unable to survive immediate transfer to 30 ppm salinity, they are able to survive transfer to 20 ppm or less, and osmoregulatory capability develops quickly in fry exposed to intermediate salinity (Clarke and Shelbourn 1985; Wagner et al. 1969; Weisbart 1968). Rich (1920) reported chinook fry in the Columbia River estuary as early as December, and, during March and April, fry were abundant in the estuary. Rich (1920) did not measure salinity at the capture sites but did note that the smallest fry appeared to avoid brackish water and were consistently associated with freshwater inflows to the estuary.

Studies on several Vancouver Island estuaries and on the Fraser River estuary (Healey 1982, 1980; Levings 1982; Levy and Northcote 1982, 1981) indicate that these estuaries are important nursery areas for chinook fry and that the distribution of chinook changed seasonally and tidally during a twice daily pattern of dispersal from low tide refuges in tidal channels to the edges of the marshes at the highest points reached by the tide. As the year progressed, the main concentration of fry redistributed themselves seaward through the delta area of these Vancouver Island estuaries. The researchers attributed this latter behavior pattern to larger fish apparently preferring deeper water and the fact that larger fish are able to osmoregulate in higher salinity. However, this seasonal migration of fish may be attributed to increasing temperatures in shallow tidal channels, particularly at low tide since Healey (1980) found that fry moved away from sampling stations where temperatures exceeded 20°-21°C, although Levy and Northcote 1981) did not observe this seasonal movement in the Fraser River estuary.

Kjelson et al. (1982), in their studies of the Sacramento-San Joaquin River delta, observed that in freshwater rearing areas of this delta, chinook fry distribution changed diurnally and with fish size. During the day, chinook fry concentrated in the upper 3 meters of the water column and became randomly distributed in the water column at night. Furthermore, small fry concentrated near shore in shallow water during the day, migrated offshore at night. Larger fish were concentrated further offshore than smaller fry.

The proportion of downstream migrant fry that rear in estuaries is not well known. According to Healey (1991), only 30 percent or less of the estimated downstream migrants could be accounted for in the estuaries of the Nanaimo and Nitinat River systems on Vancouver Island. Although the fate of the remaining 70 percent is unknown, Healey (1991) concludes that is unlikely that the fry reared someplace else, despite their apparent ability to survive and grow in habitats with high salinity. Studies by Healey (1982, 1980) on other potential nursery areas in the vicinity of the Nanaimo River in 1976 and 1977 failed to reveal any significant numbers of chinook fry outside of the estuary in April and early May. There may be high mortality of downstream migrant fry shortly after completion of their downstream migration, but the agents of this mortality are unknown.

It has been estimated for the Columbia River estuary that survival of fry, presmolt, and smolt is between 60 percent and 100 percent until the smolt reach the estuary, where survival is estimated to be 13 percent (RASP 1992). The agent(s) or process of this apparent mortality are not known. To date, there has been no systematic study conducted to identify the limiting factors for salmonid survival in the estuarine habitat (Emmett 1993, pers. comm.). The lack of knowledge concerning the utilization patterns of salmon in the Columbia and other less modified estuaries, the effects that estuarine alterations have had on salmon populations, estuarine salmonid habitat requirements, and the reasons for juvenile mortalities in the estuary, represent a significant constraint on the exploration of alternative management and restoration opportunities.

The residence times of subyearling chinook in estuarine habitats have been estimated by mark and recapture studies in the Nanaimo, Nitinat, Fraser, Skagit, and Sacramento-San Joaquin estuaries (Healey 1982, 1980; Congleton et al. 1981; Levy and Northcote 1982; Kjelson et al. 1982). The residence times calculated showed variability between sites and the factors responsible for the different resident times are speculative. Numbers ranged from a maximum of about 60 days in the Nanaimo to an average of 19 days for the Nitinat Lake to about 8 days for the Fraser River marsh to 3 days for one channel in the Skagit River marsh to an average maximum residence time of 58 days in the Sacramento-San Joaquin River estuary.

In the Sixes River estuary in Oregon, Reimers (1973) reported that subyearling fall chinook remained in the estuary for up to about three months. Based on mark-and-recapture experiments with hatchery fish Dawley et al. (1986), found that subyearling chinook salmon slowed to about 70 percent of their riverine migration rate upon entering the Columbia River estuary, but generally passed through the estuary within 6 days after arrival. The only juvenile salmon to remain in the estuary for several months were subyearling chinook salmon originating in streams below Jones Beach (RKm 75). Dawley et al. (1986) reported rapid growth for fish that remained in the estuary for an extended period of time. McCabe et al. (1986), however, found it difficult to ascertain length of residence time for subyearling chinook in the Columbia River estuary because of the various groups of wild and hatchery fish that enter the estuary in spring and summer.

II. b. Fingerling and Yearling Migrants

Based on studies in the Nanaimo and Nitinat River estuaries in British Columbia, Healey (1991) notes that estuarine habitat vacated by fry in late May and June is taken over by fingerling smolts. Studies in Oregon coastal estuaries (Myers 1980; Myers and Horton 1982; Reimers 1971; Reimers et al. 1978), revealed few fry migrants, and it appears that the first chinook to enter these estuaries are fingerling smolts. The period of greatest abundance of fingerling smolt appears to be June to August in Oregon estuaries, and the smolt appear to reside in these estuaries well into October.

Muir and Emmet (1988) noted the outmigration of salmonid smolts to the Columbia River estuary as usually beginning in late April, peaking in May, and declining for all salmonid species except subyearling chinook salmon which continue to migrate into the fall (Park 1969), with smolt outmigration generally occurring during periods of increasing river flow and rising water temperature (Mains and Smith 1964).

Habitat Requirements

Specific habitat requirements of the varying stages of stream- and ocean-type juvenile *O. tshawytscha* chinook do not appear to be well studied. Based on the current literature review, this is particularly true for juvenile chinook in the LCR and its estuary. The information provided below on habitat utilization is based on available data from other sites, and where data do exist, from the Columbia River and its estuary.

I. Habitat Utilization in Fresh Water

The process by which chinook take up residence in a stream is not well studied. Chapman and Bjornn (1969) and Everest and Chapman (1972) in studying habitat segregation between stream-type chinook and steelhead in the Snake River reported juvenile chinook to be most abundant where substrate particle size was small, velocity was low, and depth was shallow. Juvenile chinook were found in small numbers in virtually every habitat investigated, and fish size was positively correlated with water velocity and depth for both species. However, species differed in size because in differences in emergence timing and fry size between species.

In sampling various habitats in the lower Taku River for chinook, coho, and riverine sockeye, Murphy et al. (1989) found chinook mainly in riverine habitat and seldom in beaver ponds or off-channel sloughs. Velocity and turbidity were the main factors associated with chinook distributions with chinook being rare in still water or where the velocity was greater than 30 cm/sec. The movement of fish offshore and into faster water represents a shift from predominantly sandy substrate to predominantly bolder and rubble substrate. According to Chapman and Bjornn (1969), chinook prefer finer substrates than steelhead of comparable size, but both species exhibited a strong preference for the 'rubble' habitat. Healey (1991) notes, however, that any interpretation of substrate preferences is confounded by velocity preference and needs further study. Don Chapman Consultants (1989) and Edmundson et al. (1968) reported changing day-night distributions of juvenile chinook in streams -- chinook moved from their nocturnal bottom residence in inshore quiet waters or pools over sandy substrates to previous day-time rifle and glide areas.

II. Habitat Utilization in Estuaries

A number of investigators (Healey 1982, 1980; Kjelson et al. 1982; Levy and Northcote 1982; Myers and Horton 1982; Reimers 1973) have reported on the importance of estuaries as rearing areas for subyearling chinook salmon. Little has been documented about the specific use of the Columbia River estuary by subyearling chinook salmon (McCabe et al. 1986). Marsh and wetland areas are believed to provide juvenile salmonids with three primary functions: productive juvenile feeding areas, shelter from predators, and acclimation to salt water (Lebovitz 1992).

II.a. Productive juvenile feeding areas

In most cases, juvenile salmonids experience the greatest rates of growth in their life histories while in the estuaries (Shreffler et al. 1992). Both juvenile chinook and chum have been observed to add 3.5 to 5.8 percent (and up to 10 percent) to their body weight per day while in the estuary. Food requirements for juvenile salmonids in the Nitinat estuary has been estimated at 3 times their daily growth rates when growth was less than 4 percent per day and slightly lower when growth increased above 4 percent (Healey 1982). Estuarine feeding behavior is not well understood, but is generally opportunistic, although selective feeding does occur depending upon habitat conditions. Common food items include algae, insects, plankton, small fish, and fish larvae. The prey items of juvenile chinook and chum are most commonly benthic or epibenthic insects, crustaceans and copepods (Healey 1982; Simenstad et al. 1982). The estuarine salmonid food chain is a detritally-based one, with the primary source of detritus apparently dependent upon the decay of marsh and wetland vegetation (Wolfe et al. 1983). However, the specific sources and amounts of particulate and dissolved organic carbon upon which these food chains are based not known and need study. Marsh channels and mudflats are believed to provide a great number of prey items for juvenile salmonids, both directly, and indirectly as a sheetflow of carbon source and phytoplankton (Emmett 1993, pers. comm.).

II.b. Shelter from predators

In general, salmonid smolt are reported to be particularly vulnerable to predators, and estuarine marshes provide a rich feeding and nursery environment for rapid growth to a threshold size which confers some protection from predators (Parker 1971). Primary predators are reported to be adult cutthroat (*O. clarki*), chinook (*O. tshawytscha*) and chum (*O. keta*), Pacific coast cod (*Microgadus*), rhinoceros auklets *Cerorhinca monocerata*), cormorants (*Phalacrocorax* spp.), mergansers (*Mergus* spp.), harbor seals (*Phoca vitulina*), and orcas (*Orcinus orca*). Since gut analyses show relatively infrequent occurrence of salmonids in the predators, the estuarine refuge of small channels, debris, banks, turbidity, and euryhaline changes appear to be effective habitat factors in reducing predation (Healey 1982; McMahon and Holtby 1992; Simenstad et al. 1982). In the Columbia River estuary, other important predators of juvenile salmon appear to be Caspian terns (*Sterna caspia*) and gulls (*Larus* spp.) (Emmett 1993, pers. comm.).

II.c. Acclimation to salt water

The primary physiological process involved in the osmotic transition from fresh to sea water by anadromous salmonids is mediated by gill Na-K ATPase. Chinook show a preference for salinity below 2 ppt until they reach 70 mm in length when they begin to move into more saline waters (Healey 1980). Premature exposure to full strength sea water (30 ppt) results in slowed growth rates. More energy must be expended on osmotic homeostasis in higher salinity when juveniles are smaller, reducing the amount of energy available for growth. Water temperature is also important in the ability of juveniles to regulate sodium concentrations. Optimal freshwater rearing temperatures for various salmonids range from 10 to 17.5 °C, while optimal salt water temperatures "at transfer" range from 13 to 14.5 °C (Clarke and Shelbourn 1985). The osmotic transition must be gradual and, depending upon fish races or stocks, appears to be controlled by migration of the fish into the marsh channels which vary in salinity as freshets flow in and the tides change. At present no studies on this crucial element of salmonid life history have yet been conducted for the Columbia River estuary.

Many of the ocean-type juvenile chinook that migrate downstream immediately after emerging from the spawning gravel reside in river estuaries and rear there to smolt size. Levy and Northcote (1981) investigated the relationship between occurrence and abundance of chinook fry in various marsh habitats of the Fraser River estuary according to the physical characteristics of the habitat. Juvenile chinook abundance was significantly correlated with twelve of the twenty-two habitat characteristics studied. A multiple regression analysis revealed that only two characteristics explained chinook catch variation; area of low tide refugia and elevation of tidal channel banks. The results suggested that fry chinook prefer tidal channels with low banks and many subtidal refugia. Chinook and associated fish species also tended to be associated with larger tidal channels.

For the Columbia River estuary, McCabe et al. (1986) found an uneven distribution of subyearling chinook throughout the estuary with more than 95 percent of the juveniles captured in the intertidal and pelagic areas. Catches in channel bottoms, tributaries, sloughs, and coves were small. The subyearlings were most abundant from May through September in the 1980-1981 field studies conducted.

Many subyearling chinook salmon enter the Columbia River estuary at a larger size than those entering other estuaries, such as the Nanaimo River estuary in British Columbia (Healey 1980) and the Sacramento-San Joaquin estuary in California (Kjelson et al. 1982). According to McCabe et al. (1986), the mean length of subyearlings in the Columbia River estuary increased from March to October. However, the authors concluded that the size variation could not be attributed solely to growth in the estuary. Subyearlings continually enter the estuary throughout the year and hatchery releases of different sized fish could be responsible for some of the length changes observed. McCabe et al. (1986) suggested that the decline, or at least lack of increase, in the mean fork length of subyearlings in June was probably due to a large hatchery release(s) of smaller juvenile chinook salmon.

According to McCabe et al. (1986), length differences between subyearlings collected in intertidal and pelagic habitats indicated that larger subyearlings preferred pelagic areas to the shallow intertidal habitats of the estuary. Myers (1980) observed that the mean length of juvenile chinook salmon collected in channel areas was greater than that of those collected in intertidal areas. Dawley et al. (1982) observed similar size distributions at Jones Beach (Rkm 75), Columbia River.

Diet

The feeding ecology of Columbia River juvenile chinook salmon has been studied in several riverine habitats, where aquatic insects dominated the diet (Chapman and Quistorff 1983; Becker 1970, 1973; Dauble et al. 1980, and in semiestuarine habitats, where zooplankton was the major dietary component (Critic et al. 1976; McCabe et al. 1983; Kirn et al. 1986). Rondorf et al. 1990 found the diet of subyearling chinook to be predominately midges, by numbers, and adult caddisflies (Trichoptera) by weight, in riverine nursery habitats, but mostly *Daphnia* and terrestrial insects in reservoir habitats. The low number of benthic invertebrates found in the diet of subyearling chinook suggested that the fish most likely fed at the surface and in the water column.

Riverine and Reservoir Habitats

According to Rondorf et al. (1990), the creation of reservoirs and associated habitats has altered the invertebrate prey available to subyearling chinook salmon migrating through the forebay, mid-reservoir, and riverine reaches of the Lake Wallula system of the Columbia. Vannote et al. (1980) characterized rivers as a continuum of physical structures and biotic responses, and considered alterations to be a shift in the overall continuum responses. The upstream development of low velocity habitats in reservoirs results in the accumulation of fine particulate organic matter (50 um to 1 mm in diameter) rather than coarse particulate matter such as that found in higher velocity habitats. This coarse particulate matter supports large aquatic insects, such as caddisflies, which are important components of the diet of subyearling chinook salmon in riverine habitats. The shift in diet to smaller invertebrates, such as zooplankton, which utilize the small-sized particles in low velocity habitats of embayments, is accompanied by increased availability of smaller prey items.

Rondorf et al. (1990) proposed that the shift in diet to smaller, less-preferred food items such as *Daphnia* spp. in the embayment might be the result of their high density and relatively high rate of capture in the water column. Subyearling appeared to have a low preference for cyclopoid zooplankters and selected mostly large *Daphnia* spp. from the available prey population. Although juvenile chinook salmon preferred large insects and selected prey items according to size, the rank availability of food items in the environment was the only factor clearly associated with diet composition in reservoir habitats. The authors concluded that this ability to shift diet enables subyearling chinook salmon to use reservoir habitats for nursery areas as a substitute for natural areas in the riverine reach, and that the switch to alternative prey, such as *Daphnia* spp. and terrestrial insects, reflected the availability of these items in reservoir habitats. However, the use of *Daphnia* spp. may entail a higher foraging cost per energy unit gained because of the small size of prey.

Muir and Emmett (1988) studied the food habits of migrating salmonid smolts passing the lower most dam at Bonneville in the Columbia River from April through August 1984. Outmigrating spring subyearling chinook salmon fed intensively on *Corophium* spp. (90 percent Index of Relative Importance -- IRI), but they also utilized dipteran adults (2 percent IRI), hymenopterans (3 percent IRI), and homopterans (4 percent). During May, *C. salmonis* was the chief prey item, while in June, *C. spinicorne*, homopterans and hymenopterans were also eaten. Dipterans (Chironomidae) were frequently consumed in the spring, but in small quantities. For yearling chinook salmon, Muir and Emmett (1988) found *Corophium* spp. to be the most important prey item (97 percent IRI). For most of May, *C. salmonis* represented a large proportion of the diet, but in late May and June, *C. spinicorne* increased in importance. Hymenopterins were also important in June. It is not known what effect juvenile salmonid feeding had on the reservoir's food resources, especially on *Corophium*.

Becker (1973) reported that subyearling chinook salmon in the Hanford reach of the Columbia River showed a preference for suspended, moving organisms. Why then is *Corophium*, a tube-dwelling benthic invertebrate, so extensively utilized by salmonids passing through the Bonneville reservoir? *C. salmonis* reportedly undergo vertical migrations, both daily and seasonally, with migrational peaks occurring in the evening hours and during the spring months (Davis 1978; Wilson 1983). These migrations, coupled with higher river flows during the spring, may keep these amphipods suspended in the water column for long periods of time, making them susceptible to predation by salmonids. In the Duwamish estuary (Meyer et al. 1981), the Columbia River estuary (McCabe et al. 1983), and the Bonneville reservoir (Muir and Emmett 1988), the importance of *Corophium* declined in summer as river flows declined. According to Meyer et al. (1981), the proportion of

Corophium in the diet of salmonids was highest in the evening and lower during the day when the proportion of insects in the diet increased. Therefore, *Corophium* availability appears to peak in spring during the major chinook outmigration and coincides with the peak feeding times (evening) of juvenile salmonids (Johnson and Johnson 1981; Rondorf et al. 1985).

Estuarine Habitat

In the Columbia River estuary (McCabe et al. 1983); the Sacramento-San Joaquin Delta, California (Saski 1966); the Sixes river, Oregon (Reimers et al. 1978); Grays Harbor, Washington (Herrmann 1971); Duwamish estuary, Washington (Meyer et al. 1981); and other Pacific Northwest and Canadian estuaries not cited, *Corophium* spp. were the major food item for juvenile salmonids. Insects were of secondary importance (especially chironomids), increasing in importance in the upper estuarine areas. In the Columbia River estuary, *Daphnia* spp. were important prey for subyearling chinook salmon in summer (McCabe et al. 1983). The food habits of chinook in the Columbia River estuary are essentially identical to those described by Muir and Emmett (1988) for migrating smolts passing Bonneville Dam in the Columbia River, both in types of foods consumed and their seasonal occurrence, while being considerably different from those reported from free-flowing riverine areas.

Studies by McCabe et al. (1986) on the Columbia River estuary as a foraging area for subyearling chinook salmon revealed that although the subyearlings fed in pelagic areas, intertidal areas were used more frequently. The authors proposed that the higher Index of Feeding (IF) values for subyearling chinook in intertidal habitats was related to prey distribution and abundance. Large populations of *Corophium salmonis* occur in a number of intertidal areas of the Columbia river estuary (Durkin and Emmett 1980). Commonly eaten insect species (e.g., chironomids), are also found in the tidal flats and/or marsh areas. According to McCabe et al. (1986), the IF differences between intertidal and pelagic subyearling chinook salmon from through September were related to the importance of *Daphnia* spp. as a prey item in pelagic habitats. However, since *Daphnia* spp. are relatively small zooplankters, many may have to be consumed to equal the biomass of one larger invertebrate.

McCabe et al. (1986) also postulated that the differences of IF values observed for subyearling chinook feeding in intertidal and pelagic habitats could be caused by fish behavior. Subyearling chinook in pelagic habitats could be actively migrating seaward, spending little time pursuing prey. The authors observed subyearlings in the intertidal zones to be generally shorter in mean length than those in the pelagic zones, and suggested that the intertidal habitat yearlings were not migrating as rapidly and were spending more time searching for food than their pelagic cohorts. McCabe et al. (1986) attributed the switch from *C. salmonis* to other prey species in the estuary as the season progressed to changes in *C. salmonis* behavior and/or abundance, or increases in the relative abundances of other prey. *Daphnia* spp. populations, for example, peak in July-August (Haertel and Osterberg

1967), and the dietary importance of *C. salmonis* in the McCabe et al. (1986) coincided with the spring freshet. Information appears to be lacking on the effect of the spring freshet on *C. salmonis* populations.

Contaminants

Juvenile chinook do not appear to be acutely affected by heavy metals or organochlorines. However, they may suffer from more chronic effects from the presence of these substances in their environment and prey and may be therefore better bioaccumulators. Toxicity and accumulating ability may be opposed capacities since a dead fish may be eaten less, depending on predator consumption patterns. For instance, bald eagles consume the carcasses of adult salmon but not those of juvenile salmon, so that toxicity for juvenile salmon may remove it from the food web of potential predators.

McCain et al. (1990) observed the effects of PCBs and aromated hydrocarbons (AHs) on juvenile chinook and their prey in the Duwamish Waterway in Puget Sound. This area contains high levels of PCBs in its surface sediments, amphipods, and fish such as English sole. Juvenile chinook spend time there for two months between April and June, feeding on epibenthic crustaceans and gammarid amphipods and consuming higher proportions of gammarid amphipods than larger chinook. The mean concentrations of AHs and PCBs in food collected from the stomachs of the juveniles were 650 times and 4 times higher respectively than in a control site, indicating bioaccumulation of these substances.

Toxicity tests conducted for metals have shown cadmium and copper to be relatively toxic to juvenile chinook (Hamilton and Buhl 1990; Finlayson and Verrue 1982; Chapman 1978). Mercury is also extremely toxic (Hamilton and Buhl 1990). Toxic effects vary with life history stage of juvenile chinook and with water salinity. Younger chinook fry in fresh water tend to be more sensitive to the effects of heavy metals than older fry in brackish water. Sensitivity to cadmium and zinc has been found to be twice as great for young fry in fresh water as opposed to older fry in brackish water. However, sensitivity to mercury has been found to be six times greater in older fry in brackish water than in young fish in fresh water (Hamilton and Buhl 1990). Chapman (1978) tested the toxicity of metals to various stages of juvenile salmonids. Newly hatched chinook alevins were more tolerant of zinc and cadmium than their older juveniles. Juvenile chinook salmon were also found to be more metal tolerant than steelhead juveniles. In general, tolerance to metals increased with age, despite the initial high tolerance of newly hatched salmonids. Acute lethal levels of metals for juvenile salmonids tended to be much higher than minimum water safety levels (Chapman 1978). The higher tolerance of chinook to the toxic effects of metals may indicate a greater capacity for accumulation.

Finlayson and Verrue (1982) tested for the acute effects of copper, zinc, and cadmium on juvenile chinook salmon. These metals showed additive effects of copper and zinc at high copper to zinc ratios but not at lower ones. Other ratios showed antagonistic, as opposed to additive, effects of the copper-zinc-cadmium mixtures. These researchers provided estimates of safe metal concentrations for these metals. The mean metal concentrations at the 96-hour LC_{50} for Cu:Zn:Cd ratios of 1:3:0.02 and 1:12:0.08 were (in ug/liter) 37 and 18 for Cu, 121 and 218 for Zn, and 1.1 and 1.8 for Cd. The safe copper concentrations (0.1 times 96-hour LC_{50}) for these two mixtures would be 4 and 2 ug/liter. Dilution of the original mixtures to the safe copper concentrations would yield safe Cd concentrations below its individual concentrations, indicating selective removal of copper wastes (e.g. from acid-mine wastes) can decrease overall waste toxicity.

Saiki et al. (1992) studied the effects of elevated dissolved salt concentrations in tile drainwater (WWD) in juvenile chinook. They found reduced survival and growth in juvenile chinook exposed for 28 days to WWD. High trace elements of chromium and selenium were present in the test water. While selenium may be a source of chronic toxicity, the researchers believed the main effects on growth and survival to be caused by higher concentrations of total dissolved salt. Selenium has been shown to bioaccumulate in aquatic food chains and to be a health hazard to birds (Schuler et al. 1984).

Juvenile chinook diets in the LCR are mainly composed of aquatic insects and zooplankton. They also eat invertebrates such as *Corophium salmonis* (Albright 1982). Juvenile chinook appear to feed on organisms found in the water column, so that benthic invertebrates such as *Corophium* become vulnerable to juvenile chinook predation only when they leave their tubes and enter the water column. Likewise, epibenthic zooplankton such as *Eurytemora affinis* become available for consumption during the periods when they rise in the water column, although juvenile chinook feeding strategies appear to be based on preference, not necessarily relative abundances. Critic et al. (1976) found insects to be the most common prey item in the spring and fall and zooplankton to be most commonly consumed in August and September in the LCR. However, during May, June, and July, zooplankton composed only 23 percent of the food in the chinook stomachs. *Daphnia* dominated the composition of zooplankton found in juvenile chinook stomachs. *Daphnia* was also one of the most abundant zooplankton collected and peaked at water temperatures of 21°C in August. While *Bosmina* was also highly abundant, the juvenile chinook fed selectively on *Daphnia,* probably because of its larger size.

Elevated levels of contaminants in insect prey have been found in the Yakima River subbasin (USGS 1994). Contaminant accumulation in zooplankton in the Columbia River has not been studied. Tribulytin has been shown to be toxic to *Eurytemora affinis* in the Chesapeake Bay (Bushong et al. 1988) and mercury, silver, and copper are known to be highly toxic to copepods (Dawson 1979). The effects of organochlorines on estuarine zooplankton are not well known. *Corophium* spp. are known to accumulate dioxins and metals such as copper, mercury, and zinc (Reish 1993; Vermeer et al. 1993).

Population Trends

Salmon runs in the Columbia River Basin have declined from an estimated 10 to 16 million pre-development on the river to current natural run averages of 1 million (Figure 2). This is

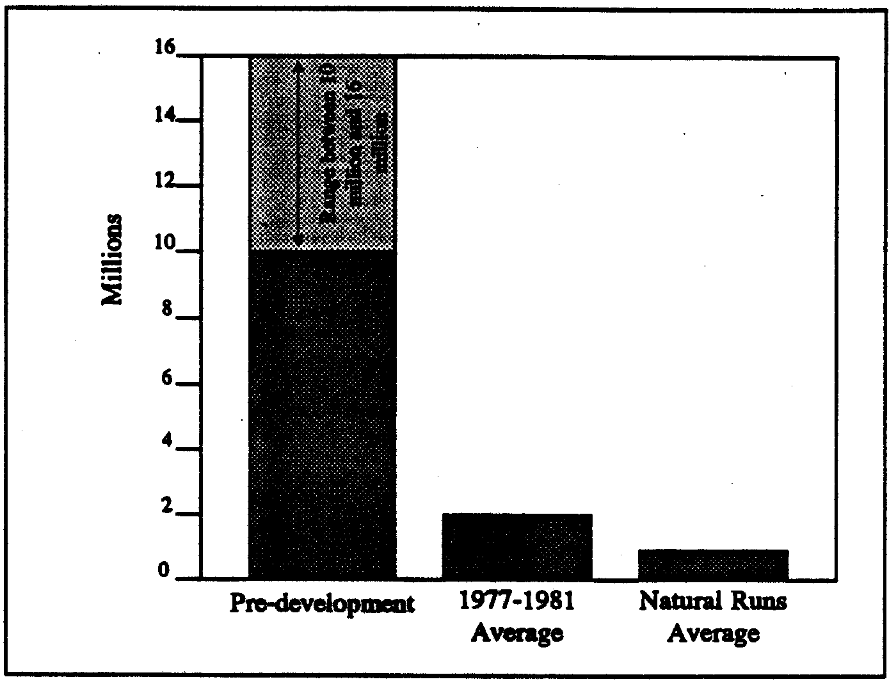

Figure 2.
Columbia River Basin Salmon Runs--An Historical Perspective

despite efforts to augment wild stocks with hatchery stocks. These efforts, however, have been plagued by the failure of hatchery stocks to adequately mimic the genetic diversity of natural salmon stocks. Spring chinook salmon runs in the Snake River have fluctuated over the past thirty years between 50,000 and 5,000 adult fish. Fall chinook runs in the Snake River have steadily declined from nearly 20,000 in 1967 to averages of barely 1,000 since 1977 (NPPC 1992).

These figures are for adults returning up the river to spawn. The number of juveniles migrating to sea is determined by the numbers of returning adults that spawn, and the number of adults that reach their natal stream and spawn is determined by ocean predation, harvesting, and mortality caused by the dam system in the Columbia River Basin. Numbers of juvenile chinook migrating to the Pacific through the LCR are not well known, especially since wild and hatchery stocks may be considered as separate populations or summed together. It is known, however, that the numbers have declined over the years. In addition, hatchery stocks which augment the wild population numbers have lower fitness than the wild stocks (Cramer 1992).

DISCUSSION AND ANALYSIS

Asterionella formosa

Discussion of Existing Data and Current Data Gaps

Phytoplankton are known to uptake organochlorines either through absorbing them, that is binding with them at the cell surface, or absorbing them, where absorption normally follows adsorption (Kerr and Vass 1973). However, the mechanisms and dynamics influencing uptake have been poorly elaborated in the literature and not discussed at all within the dynamics of the LCR. In general, phytoplankton sensitivity and uptake of organochlorines varies with environmental parameters such as light intensity, temperature, interspecific competition, and geographic location (Ehrlich et al. 1977). Phytoplankton elimination of organochlorines tends to occur less quickly than accumulation, usually over a period of days. This implies that with continued exposure, the residue burden of an organism will increase over time (Addison 1976). Since the solubility of most pesticides and other chemicals in pure water is low, the adsorption of water borne contaminants onto phytoplankton is an important method of their introduction into the food web. Absorption or adsorption of DDT by algae and other aquatic plants tends to occur without biological or chemical utilization of it by the absorbing organism (Vance and Drummon 1969). The current concern for organisms in the LCR revolves around DDE not DDT, suggesting that metabolism must have occurred.

Certain phytoplankton have also been identified as metabolizers of p,p'-DDT into p,p'-DDE and p,p' DDD (Patil et al. 1972; Bowes 1972; Rice and Sitka 1973). Metabolism of PCBs varies, but often leads to mono- or di-hydroxy chlorobiphenyls (Hutzinger et al. 1974). However, little information exists regarding PCB metabolism in phytoplankton or in estuaries. In addition, the banning of DDT for agricultural use in 1972 implies that current levels in biota today would probably be from accumulation of DDE from the soil rather than from DDT or DDE plankton bound in the water column. However, evidence from the Yakima River subbasin suggests that DDT and DDE are still entering the water from terrestrial soil runoff. This evidence comes in the form of high levels of these contaminants in aquatic insects, suggesting they acquired these contaminants from their prey, i.e. phytoplankton (USGS 1994). Limited studies have been conducted on the effects on these contaminants on phytoplankton. In microhabitat and laboratory studies, mimicking the conditions of the phytoplankton's ecosystem are important because phytoplankton responses to contaminants are highly dependent on pH, salinity, trace metal concentrations, and other physical factors (Riseng et al. 1991).

Physical factors, such as seasonal changes in river flow rates or salinity regimes, in the LCR probably affect contaminant cycling, deposition, or efflux through the river but no studies exist to quantify these process. Such processes are related to the cycling, deposition, and efflux of phytoplankton in the river to the extent that a contaminant becomes bound in the plankton. It would be useful to know the proportion of a contaminant load that falls to the bottom immediately, becomes flushed out, becomes bound to plankton and detritus or does other things. This would be possible to measure for single instances of contaminant release such as the 1986 release of a large PCB load into the river from Bonneville Dam.

While some phytoplankton may incur toxic effects from contaminants, the most important and relevant effects of contaminants on phytoplankton would be changes in plankton community composition and productivity (Williams 1964; Ehrlich et al. 1977). To some extent chemicals such as dioxins and heavy metals may become bound in the plankton. Therefore, dynamics of phytoplankton and zooplankton throughout the LCR might parallel a certain proportion of contaminants and their dynamics. In addition, quantifying relative phytoplankton abundance and productivity in regions of high and low sediment contamination and quantifying the health of their dependent biota would be another method of ascertaining contaminant impacts on phytoplankton and dependent communities.

Studies have been conducted on the abundance, distribution, and productivity of *A. formosa* in the LCR. Amspoker and McIntire (1986) determined that the Columbia River estuary is more influenced by river flow rates than other estuaries, based on evidence from the relative abundances of freshwater versus marine diatom assemblages. The researchers hypothesized that river flow and estuarine circulation were more important factors in the determination of diatom flora than photoperiod, nutrients, or temperature. *A. formosa* was an abundant freshwater diatom found throughout most of the estuary, suggesting high flushing rates. In addition, they described a lengthened residence time of freshwater in tributaries and bays. Both of these factors affect both contaminant and phytoplankton dynamics. Contaminants deposit from their source differentially based on different flow rates. For instance, during periods of low flow localized contamination may be more prevalent, but during storm periods contaminants may be flushed quickly to sea (Stewart 1994, pers. comm.).

Phytoplankton sinking and flushing patterns may affect contaminant distribution, if the contaminants are bound to the plankton. In addition, the dynamics of phytoplankton and other suspended particles may be relevant to contamination patterns as indicators of the flow and distribution of contaminants bound to organic detritus in general. Although *A. formosa* contributes to much of the primary production in the Columbia River estuary, overall annual water column primary production is low due to light limitations from high flows and turbidity (Frey et al. 1984). However, the overall productivity of the estuary is high due to phytoplankton imports from upriver (Lara-Lara et al. 1990a; Frey et al. 1984). This implies contaminants bound to upriver phytoplankton become flushed to the estuary rather than deposited with sinking plankton upriver and that upriver contaminants may be able to insinuate themselves into mid-estuary dynamics by virtue of high flows and flushing rates of phytoplankton. Upriver freshwater phytoplankton encounter high mortalities when they hit the salt wedge. The cells sink and become food for depth abundant *E. affinis*, but particle removal due to grazing is low in the Columbia River estuary compared to removal from export to ocean or sinking (Lara-Lara et al. 1990b).

Contamination does not appear to be a factor in phytoplankton deaths in the LCR. Instead, pertinent questions are where do the diatoms absorb contaminants and at what stage do those diatoms then get eaten, after death and sinking or at surface, and, if the upriver diatoms are eaten would they be more contaminated either due to longer residence in the estuary or because of closer proximity to original source of contamination. Death and sinking of *A. formosa* on encountering the salt wedge means that these diatoms sink to become important food for epibenthic and benthic invertebrates, such as *E. affinis* and *C. salmonis*. *C. salmonis* is important prey for juvenile salmonids and waterfowl, which are in turn eaten by bald eagles. That much of the freshwater diatoms sinking upon reaching the salt wedge originate from upriver implies that upriver contamination could be affecting the food webs in the mid-estuary.

While *A. formosa* mortalities in the LCR do not appear to be related to contaminants, the diatom is affected at lethal levels by some metals, such as aluminum (Riseng et al. 1991). However, the toxic effects are highly dependent on other bio-physical conditions such as pH and trace metal concentrations. Other contaminants may be toxic as well, but their effects on *A. formosa*, especially mitigated by other important variables, have not been well studied.

Recommendations

--measurements of existing contaminant concentrations in Columbia River phytoplankton

--laboratory studies of adsorption patterns of organochlorines and dioxins in *A. formosa* and other abundant Columbia River phytoplankton, including mitigating variables such as pH, salinity, alkalinity, total suspended particle load, trace metal concentrations

--laboratory studies of toxic effects of organochlorines and dioxins on *A. formosa* and other abundant Columbia River phytoplankton, including mitigating variables

--field studies of relative abundances and community structure of phytoplankton at various sites in the river, including geographic and extent of contamination variability in the sites

--laboratory and field studies addressing the cycling, flushing, and sedimentation patterns of various contaminants through the Columbia River, including mitigating biophysical variables

--microhabitat study addressing contamination accumulation through the food web

Eurytemora affinis

Discussion of Existing Data and Data Gaps

As discussed in the findings, no research has been conducted on the effects of contaminants on *E. affinis*, or any other zooplankton, in the Columbia River and little to no research has been conducted on this genus and contaminant effects in general. Basic knowledge is needed about the potential for and pathways of bioaccumulation in *Eurytemora*. It and *Scottolana canadensis* are the only two zooplankton endemic to estuaries in any abundance that have been identified in the Columbia River estuary (Bottom and Jones 1984). Since estuaries are highly variable and highly difficult to study, knowledge about contaminant effects on estuarine zooplankton appears to be lacking.

Like *A. formosa*, patterns of *Eurytemora* movements and distributions in the estuary may a useful gauge for estimating contaminant patterns through the LCR. *E. affinis* is an interesting choice for an alternative trophic species because new data have recently demonstrated that its methods of entrainment in the estuarine mixing zone are based on endogenous swimming rhythms. This behavioral mechanism takes advantage of tidal currents, so that *Eurytemora* organisms selectively rise and fall in the water column in order to maintain themselves in the estuarine zone (Hough and Naylor 1992; Hough and Naylor 1991). This evidence suggests that any contaminants bound in these zooplankton or the phytoplankton they eat may be able to rise and fall in the water column seasonally and tidally.

While a large proportion of the contaminant load may not be affected by such patterns, it is still an interesting area to consider in terms of possible variable affecting contaminant distribution and food web consumption patterns. Consumption of *E. affinis* by fish and invertebrate predators is assumed to occur in the epibenthos where greatest concentrations of *E. affinis* generally occur. However, during times of migration in the water column, *E. affinis* becomes vulnerable to predation by less demersal organisms. In addition, any study of the microhabitat interactions in the estuarine mixing zone which included *E. affinis* would be able to map any regional constancy in contaminants by virtue of their presence in *E. affinis*. That is, because *E. affinis* consistently maintains itself in the estuarine mixing zone, studies sampling contaminant loads in *E. affinis* may be able to provide estimates of the relative contamination of this region of the river.

E. affinis consumes epibenthic algae and detritus, probably including dead *A. formosa* cells. This zooplankton is in turn eaten by epibenthic organisms, including *C. salmonis* which filter feeds on epibenthic detritus and plankton. *Corophium* is consumed by largescale suckers, juvenile chinook salmon, and waterbirds in large quantities. All of these predators are eaten by bald eagles. *E. affinis* may also be ingested by the benthic feeding largescale sucker. While it is not consumed by juvenile salmon in any significant quantities, its sheer abundance in the LCR makes it an important marker of contaminant patterns and effects in the ecosystem.

Little research has been conducted on pathways of accumulation of substances such as DDE or dioxins which may involve *E. affinis*. In pelagic zooplankton, PCBs do not appear to be concentrated from zooplankton to fish but more research is needed on estuarine zooplankton's responses to contaminants (Dawson 1979). In addition to laboratory research on contaminant effects and potential accumulation on *E. affinis*, useful knowledge may be gained by examining in detail the amounts of contaminants (if any) associated with *E. affinis* at various points along its cycle of entrainment in the estuary. Potential sublethal effects of contaminants on the important behavioral mechanism resulting in the endogenous swimming rhythm are not known.

Tributyltin has been demonstrated to have toxic effects on *E. affinis* (Bushong et al. 1988). Tributyltin is an often used biocide in antifouling paints for commercial and recreational watercraft. Both commercial and recreational boats make ample use of the LCR. Heavy metals may also have toxic effects on zooplankton, although the negative effects of heavy metal contamination depend on the type and concentration of metal and the species affected. Mercury, silver, and copper are generally acknowledged as very toxic to zooplankton (Dawson 1979). Copper is also a frequently used antifouling paint for boats.

Recommendations

--laboratory research of sublethal and lethal effects of contaminants on *E. affinis*, including mitigating physical parameters

--laboratory research on uptake or metabolic capability of *E. affinis* with regards to contaminants

--investigation into potential for *E. affinis* to represent contaminant levels (most likely at adjusted values) in estuarine mixing zone

--intensive microhabitat study of estuarine mixing zone and its flow, contaminant, detritus, and consumption dynamics

Corophium salmonis

Existing Data and Data Gaps

C. salmonis is one of the most important prey items of fish and waterfowl in the LCR food web, including largescale suckers and juvenile salmonids (Bottom et al. 1984; Higley et al. 1976). *Corophium* spp. have been noted to persist in contaminated sediments with low species diversity of infaunal communities (Vermeer et al. 1993). The LCR itself supports infaunal communities with low species diversity and productivity where *Corophium* accounts for most of the biomass (Holton et al. 1984). These two pieces of evidence are not proof in of themselves that *Corophium* in the LCR survive when few others do, but they do point to interesting areas of inquiry that have not yet been researched. If *Corophium* does well in contaminated environments and it comprises much of the prey base in the LCR, one may question whether current fish diets are accurate reflections of prior diets in a more pristine Columbia River, whether *Corophium* would dominate to the same extent in cleaner rivers, and the effects on contaminant accumulation of a shift (if such a shift occurred) to a diet based on the infaunal detritus eater, *Corophium*.

Invertebrates, including larger crustaceans, are known to metabolize contaminants such as DDT (Addison 1976). *Corophium* metabolism of contaminants is not known. However, gammarid amphipods such as *Corophium* have also been identified as being highly sensitive to the toxic effects of certain contaminants. Reish (1993) studied the effects of metals and organic compounds on survival and bioaccumulation in *C. insidium*. *C. insidium* was highly sensitive to the toxic effects of DDT and PCB 1254 Arochlor and did not uptake either chemical. However, *C. insidium* was able to uptake heavy metals, although mercury and copper were also highly toxic. These results imply that *Corophium* may only accumulate PCBs and DDT in low levels; in higher concentrations, it dies and therefore does not pass on contaminants. However, it may accumulate metals in higher levels before incurring toxic effects.

Vermeer et al. (1993) identified high dioxin and furan concentrations in *Corophium* spp. in a pulp mill polluted estuary. Proximal sediments and birds eating *Corophium*, such as western grebes, also contained elevated dioxin concentrations. Western grebes are consumed in small quantities by bald eagles in the LCR (Watson et al. 1991). *Corophium* appeared to selectively uptake low molecular weight dioxin and furan congeners. The tracing of these particular congeners through the food web may be a fruitful method of identifying certain contamination patterns.

C. salmonis individuals have an interesting behavior pattern of leaving their tubes and entering the water column, thus incurring higher mortalities due to predation. This behavior, however, benefits juvenile salmonids and other water column feeders. *Corophium* tube flight has been ascribed to male desire, one researcher stating that the males leave their tubes in search of mates (Albright 1982). However, an intensive study of infaunal community dynamics in the Columbia River estuary led other researchers to conclude that *Corophium* vacate their tubes when their habitat becomes unsuitable (Holton et al. 1984). Habitat suitability for *C. salmonis* would be determined by temperature, salinity, and sediment characteristics. The most noticeable difference between the preferred site and the abandoned site was salinity. At salinity greater than 10.5 ppt, the *C. salmonis* population disappeared from one sample site in the LCR, while concomitantly at another sample site with lower salinity the population increased (Holton et al. 1984). This suggests that *C. salmonis* populations in the Columbia River estuary adapt to changing environmental conditions through migration into tidal currents. Further experimental research could further elucidate the factors and mechanisms behind these migrations. The presence if *C. salmonis* in the water column is crucial for the fish that prey on them. Habitat change leading to migration would be an important variable determining food web interactions and subsequent contamination patterns.

Studies have been conducted on the effects of dredging on *C. salmonis* populations in Grays Harbor, Columbia River, Washington (Albright 1982; Albright and Armstrong 1982). However, the definitions of the potential effects of dredging have been limited to habitat change and increased turbidity and have neglected possible contaminant effects and interactions. Dredging brings contaminants such as DDE up to the soil's surface, releasing otherwise buried chemicals. Other potential side effects of dredging include the obligate increase in shipping and ship-borne contaminants, redistribution of contaminated soils, habitat change affecting food web dynamics, and potential impacts on other infaunal populations more sensitive to turbidity changes. These impacts could have results such as increased predator reliance on *C. salmonis* in the absence of other prey species or changes in the relative uptake and food web incorporation of contaminants such as low-molecular weight dioxins and furans.

Recommendations

--investigation into relative abundances and production values of *C. salmonis* in polluted and unpolluted areas

--laboratory studies on contaminant metabolic and uptake capacities of *C. salmonis*

--further research on mechanisms leading to *C. salmonis* vacating of tubes and water column abundances

Oncorhynchus tshawytscha (Juvenile)

Existing Data and Data Gaps

Juvenile chinook salmon (*O. tshawytscha*) are important food resources for upper trophic level predators on the LCR such as squawfish, walleye, bass, bald eagles, cormorants, and other piscivorous birds. Diets of juvenile chinook are primarily composed of *Daphnia* spp, insects, and benthic invertebrates such as *C. salmonis* (Dawley et al. 1985).

Few studies have discussed the effects of contaminants on juvenile chinook salmon and those that have generally addressed the effects of metals, not organochlorines. In addition, these studies quantified toxic effects of metals such as copper, cadmium, and zinc, but did not address the issue of sublethal contamination effects. Such effects are more pertinent in terms of bioaccumulation than toxic effects.

Habitat alteration such as that incurred by dams may impact contamination pathways by reducing the biomass of juvenile chinook available to predators and changing the diet composition of predators. Such changes may affect which contaminants are present in the food web and which concentrations of contaminants are accumulated. In addition, wild stocks have decreased while management attempts to supplement them with hatchery stocks have resulted in differential behavior and migration patterns in juvenile chinook (Cramer 1992; Dawley et al. 1985). These differences may affect the relative diets of wild and hatchery stocks which in turn affect bioaccumulation pathways.

Saiki et al. (1992) showed that higher than normal dissolved salt concentrations can impact juvenile chinook growth and survival. Changes in dissolved salt concentration can be related to habitat changes through issues such as agricultural runoff of salts. In addition, reduced juvenile growth may increase migration time (Healey 1991) and thus increase chances of predation. Increased predation of juvenile chinook relative to other prey items could change the concentrations or types of contamination in higher trophic level species, such as bald eagles. However, in the absence of data on bioaccumulation of various contaminants in juvenile chinook it is difficult to estimate the degrees or vectors of such changes.

Bald eagles consume juvenile chinook as a high proportion of their diet during smolt outmigration, when bald eagle young are beginning to fledge (Watson et al. 1991). Juvenile chinook, then, are potential sources of contamination for both adult eagles and newly hatched young. Nestling bald eagles in the Columbia River estuary showed detectable levels of PCBs and DDE in their blood (Anthony et al. 1993). Nestlings probably accumulated these residues not from their mother's blood but from their prey (Garrett et al. 1988).

In addition, bald eagles consume birds which eat juvenile chinook, such as cormorants. Such interactions merit further study, both in the lab to quantify chemical accumulation in fish and bird lipids and in the field. Although juvenile chinook are only one prey species among many consumed by bald eagles, their relative presence or absence in bald eagle diets can impact overall eagle contaminant levels. For instance, if largescale suckers contain higher residues of pesticides than juvenile chinook, a habitat alteration or other change affecting the relative consumption of suckers and salmonids will impact contaminant levels in the eagles.

Recommendations

--studies on sublethal and lethal effects of contaminants of juvenile chinook salmon

--studies on potential for uptake and accumulation of contaminants in juvenile chinook salmon and various factors affecting these processes, such as diet, gill vs. prey entry of contaminant, relative residency in estuary or upriver

--studies on overall food web productivity: effects of contaminants on, results observed in upper trophic levels, effects of prey availability changes on

--studies addressing potential contaminant effects in higher trophic level species resulting from changes in juvenile chinook abundances and availability

EFFECTS OF CONTAMINANTS ON LARGESCALE SUCKER (*Catostomus macrocheilus*) ON THE LOWER COLUMBIA RIVER

INTRODUCTION

Largescale suckers (*Catostomus macrocheilus*) are distributed widely and are generally abundant throughout the Columbia River system (Gray and Dauble 1977; La Bolle et al. 1985). Largescale suckers are important prey for a number of vertebrate predators, including fish and birds. Suckers, like other native fish populations, are affected by fishing and industrial activities on North American rivers. Only in recent years however, has the need to manage non-game fisheries such as the largescale sucker been recognized.

Largescale suckers are a component of the bottom feeding community. Numerous contaminants in natural systems are found to bind to sediment or end up settling on the bottom within the sediments. While sieving through the sediment for food, largescale suckers can take up sediment-borne contaminants. Therefore, largescale suckers can be an indicator of sediment contamination.

Levels of chemical contaminants such as dichloro-diphenyl-trichloroethane (DDT), dichloro-diphenyl-dichloroethylene (DDE), polychlorinated biphenyls (PCBs), polychlorinated dibenzo-*p*-dioxins (PCDDs) and various metals such as mercury (Hg), lead (Pb), and copper (Cu) have been shown to have a deleterious effect on the natural benthic community of a river. Such contaminants bring about a physical change in the substratum and many of the benthic types indigenous to the area are eliminated or severely altered from addition of chemicals to their environment. PCDDs show extreme toxicity and also show a potential to accumulate in fish (Servos et al. 1994). Fish exposed to PCDDs experience growth retardation, increased liver size, and induced MFO activity (Servos et al. 1994).

The purpose of this portion of the study is to characterize what is known about the effects of chemical contaminants on the largescale sucker community inhabiting the lower Columbia River. As part of the Fish and Wildlife Study of the Lower Columbia River Bi-State Water Quality Program, specific objectives are to evaluate the availability and completeness of information found in the existing literature regarding largescale sucker population dynamics, population trends, and diets. The scope of this report includes a qualitative and quantitative analysis and synthesis of the data found. This study also seeks to identify gaps and weaknesses in the current data base.

FINDINGS

Life History

Largescale suckers are an ecologically important non-commercial species of fish in the Northwest present in all types of water environments from estuarine and riverine to lacustrine (Dauble 1986). They are mass spawners with several males attending a single female. Spawning occurs in a variety of physical situations, in the inlet and outlet streams of lakes and graveled lakeshores (McCart and Aspinwall 1970). The onset of spawning is probably temperature-regulated (Nelson 1968). Spawning largescale suckers are sexually dimorphic (Dauble 1986). Largescale suckers were found to spawn in areas which were characterized by rapid flow over gravel where freshets were common (Nelson 1968). Number of eggs per female typically range from 625 to 1,574 Dauble (1986).

The maximum age of a largescale sucker found in the Lower Granite Reservoir, Snake River, Washington was 22 years (Scoppettone 1988). Most of the largescale sucker population is generally mobile (Dauble 1986). Growth rates are greatest during the first four years of life, with increases from 37 to 73 mm per year until maturity. Largescale suckers typically achieve a length of 61 centimeters and a weight of 3,200 grams (McPhail and Lindsey 1970). Peak spawning occurs in late May and June at water temperatures from 12 to 15°C.

Habitat Requirements

Fry reside in shallow pools and backwater areas, usually over mud and cobble substrate. Yearlings were found to be most abundant in backwater areas of lakes and rivers at depths of less than one meter, however, numerous yearlings can also be found over cobble bottom in protected areas of a main river. Adult largescale suckers are found in riverine, estuarine and lacustrine areas (Dauble 1986). Larval suckers were found to be pelagic and common in nearshore areas of low current velocity. findings in finding that sucker larvae are positively phototaxic and numerous in nearshore areas of a reservoir below Hanford (Hjort et al. 1981; Dauble 1986). First year suckers were found in shallow pools (10 to 60 cm depths) and backwater areas, usually over mud and cobble substrates. Adults appear to be most dominant resident fish species in the main river drift (Dauble 1986).

Diet

Largescale sucker diet consists almost entirely of benthic organisms and of organisms associated with bottom vegetation. Immature fish mainly consume plankton and aquatic insect larvae mixed with small quantities of bottom ooze. Planktonic forms includes *Cladocera*, *Copepoda*, *Hydracarina* and *Ostracoda*. Insect larvae include mainly Chironomidae, *Trichoptera* and *Ephemerida*. Bottom ooze is found in larger quantities in the digestive tracts of mature suckers (Carl 1936). The species is characterized as opportunistic, omnivorous fish. The quantities of bottom ooze in most of the specimens examined shows that the mature sucker feeds directly off the bottom. Sand is also present in most guts of the largescale suckers. The character of the food materials indicates that it does not select its food. Adults are also observed grazing cobble substrate (Dauble 1986).

Tables 1 and 2 illustrate that the food of the largescale sucker consists almost entirely of bottom forms and of organisms associated with bottom vegetation. Diatomaceae was found to be the most common item found in the diet of the largescale sucker. Aquatic insects were found at high levels in lake dwelling fish while diatoms were found at high levels in river fish (Carl 1936). Miscellaneous materials found by Carl (1936) include insect eggs, fish eggs (*Oncorhynchus nerka*), wood fibers, seed capsules, conifer needles, organic detritus, sand, and gravel. Generally, accumulate organic pollutants when they ingest food.

Table 1. List of animals and plants found in the digestive tracts of largescale suckers collected from various rivers and lakes in British Columbia, Canada (Carl 1936).

PLANTS	
Filamentous algae	*Nostoc, Spirogyra, Mougeotia, Ulothrix, Chara*
Desmidiaceae	*Closterium*
Diatomaceae	*Synedra, Navicula, Fragillaria, Surirella, Tabellaria, Melosira*
Bryophyta	*Selaginella, Sphagnum*
ANIMALS	
Protozoa	*Difflugia*
Annelida	*Lumbriculus*
Bryozoa	*Plumatella polymorpha*
Ostracoda	Two undefined species
Copepoda	*Epischura nevadensis, Diaptomus ashlandi, Cyclops bicuspidatus, Cyclops viridis, Canthocamptus minutus*
Cladocera	*Daphnia longispina, Acroperus harpae, Scapholeberis mucronata, Pleuroxus denticulatus, Pleuroxus* sp. *Bosmina longispina, Eurycercus lamellatus, Chydorus sphaericus, Alona costata, Alona affinis, Leptodora kindtii, Leydigia quadrangularis*
Amphipoda	*Gammarus limnaeus, Hyalella azteci*
Hydracarina	Two undefined species
Insecta	Plecoptera nymphs, Ephemerida nymphs, Odonata (including *Ischnura* sp. nymphs), Hemiptera (*Corixa* sp.), Trichoptera (*Setodes* sp., *Hydropsyche* sp., and other larvae), Simuliidae larvae, Culicidae (*Chaoborus* sp. larvae), Chironomidae larvae, Tipulidae larvae
Mollusca	*Planorbis* sp., *Gyraulus vermicularis, Valvata lewisi, Cochlipa* sp., *Musculium truncatum*

Table 2. Percentage occurrence in 87 stomachs of largescale suckers Girard collected from various rivers and lakes in British Columbia, Canada, (Carl 1936).

Hirudinea	0
Ostracoda	15
Copepoda	15
Cladocera	16
Amphipoda	14
Hydracarina	10
Aquatic insects	78
Terrestrial insects	4
Mollusca	25
Plants: diatoms, algae	38
Detritus	72

Therefore, contaminants are taken up in direct proportion to the dietary intake and the concentrations of the contaminants in the food (Spigarelli et al. 1983; Hilton et al. 1983).

Contaminants

Contaminant studies were conducted by the National Pesticides Monitoring Program from the late 1960's through the early 1980's. Sucker contaminants detected include DDE, DDT, TDE, DDT and it's metabolites, Dieldrin, Aldrin, Lindane, Heptachlor, Heptachlor Epoxid, BHC, PCBs, Aroclors, TCDDs, p,p' DDE, p,p'DDT, p,p'DDD, cadmium, lead, mercury, arsenic, and selenium. Table 4 presents these data showing levels of the various contaminants found in largescale suckers. Figure 1 plots table 3 data to show the levels of toxicants detected in largescale suckers from the lower Columbia River from 1967 to 1986.

Table 4 presents data showing levels of the various metals found in largescale suckers from the National Pesticides Monitoring Program and U.S. Fish and Wildlife research. Figure 2 plots table five data to show the levels of metals detected in largescale suckers from the lower Columbia River from 1971 to 1986. There seems to be no general trend in the concentrations of any of the metals with the exception of arsenic which was detected in the 1971-1973 monitoring but not in the 1976-1977 monitoring.

Table 3. Concentrations of target contaminant groups addressed in this report in ppm.

	1967-1968 Henderson et al. 1969 Bonneville Dam, OR	1969 Henderson et al. 1971 Bonneville Dam, OR	1970-1974 Schmitt et al. 1981 Bonneville Dam, OR	1980-1981 Schmitt et al. 1985 Cascade Locks, OR	1986 Anthony et al. 1993 Columbia River Estuary
DDE	0.29	0.36	-	-	-
p,p' DDE	-	-	0.73	0.54	0.07
TDE	0.27	0.17	-	-	-
DDT	0.15	0.11	-	-	-
DDT & met.	0.61	0.64	-	-	-
p,p' DDT	-	-	0.05	0.05	0.02
p,p' DDD	-	-	-	0.21	0.08
Dieldrin	0.05	0.01	0.01	0.01	-
Est. PCB	-	1.04	0.25	0.06	0.85

Water quality standards for the above contaminants are as follows: DDE - 0.001 ug/L, DDT - 0.001 ug/L, PCBs - 5 ppm.
Note: - indicates not reported

Population Trends

Reimers et al. (1967) examined the distribution of fishes in the lower Columbia River. The researchers commented that large populations of *C. macrocheilus* were spread throughout the lower Columbia River down into brackish water. It was also reported that *C. macrocheilus* generally ascend to the tributaries only on spawning migrations. Peak population estimates were made at 14,961 and 12,343 per km for June and July 1981 in the Hanford reach of the Columbia River (Dauble 1986). Tag return data from Dauble (1986) indicated the majority of the largescale sucker population was mobile.

DISCUSSION AND ANALYSIS

Introduction

Information on largescale suckers in the lower Columbia River is limited to a few studies. The majority of the information found on largescale suckers came from incidental reports contained in research on salmonids or other more economically important species.

Because largescale suckers are opportunistic omnivorous species, relationships between important prey items and species health are difficult to assess. The relationship between largescale suckers and contaminant effects on predators such as eagles, squawfish

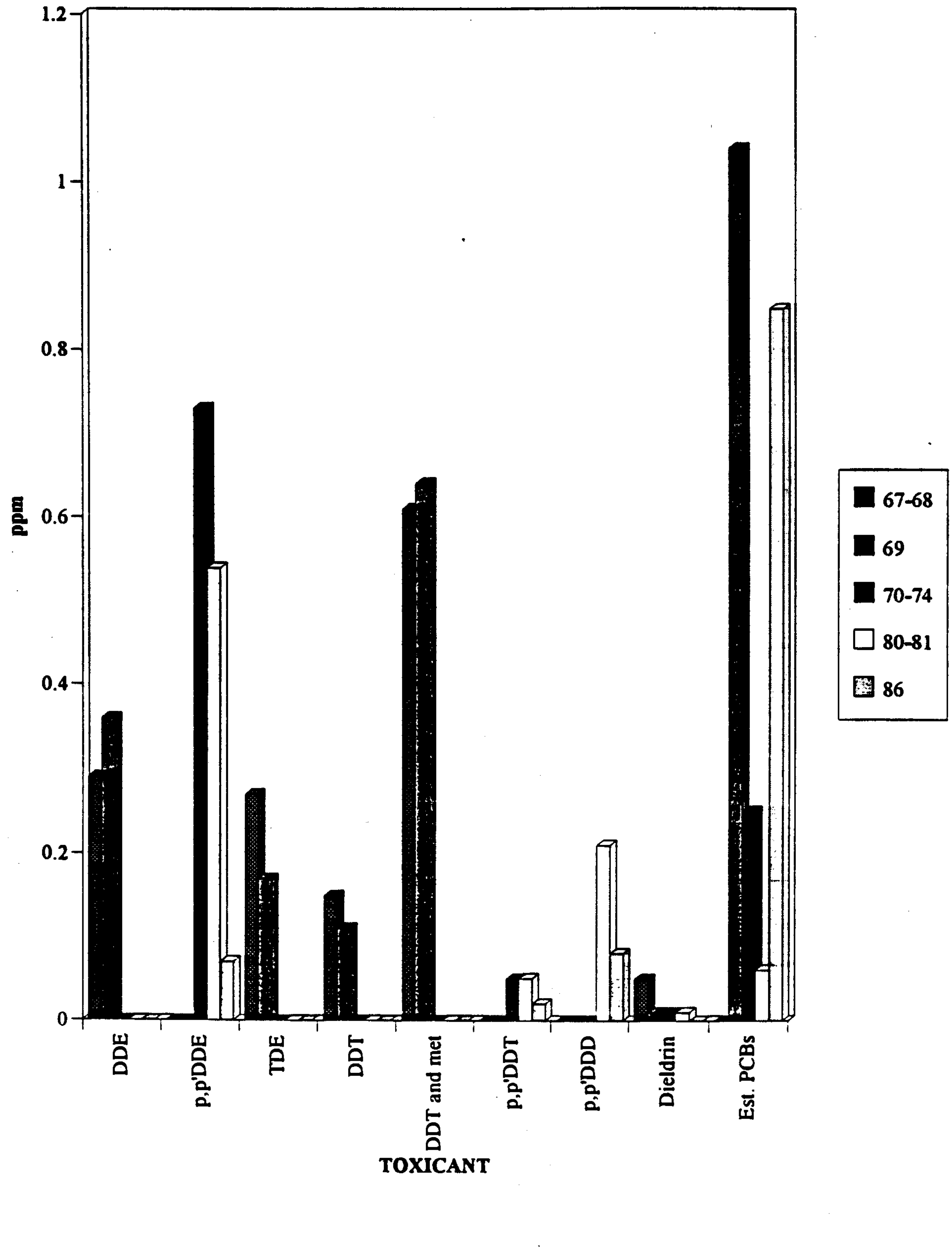

Figure 1. Levels of toxicants detected in largescale suckers from the lower Columbia River from 1967 to 1986

Table 4. Concentrations of target metal groups addressed in this report in mg/kg.

	1971-1973 Walsh et al. 1977 Bonneville Dam, OR	1976-1977 May et al. 1981 Bonneville Dam, OR	1986 Anthony et al. 1993 Columbia River Estuary
Cadmium	0.05	0.73	0.05
Lead	0.10	0.05	0.01
Mercury	0.02	0.01	0.09
Arsenic	0.20	0	-

Water quality standards for the above contaminants are as follows: Lead - 0.50 ug/l, Mercury - 2.0 ug/l.
Note: - indicates not reported

(*Ptychocheilus oregonensis*), channel catfish (*Ictalurus punctatus*) and white sturgeon (*Acipenser transmontanus*), walleye (*Stizostedian vitreum*) and great blue heron (*Ardea herodis*) is also limited at best.

Discussion of Existing Data and Data Gaps

Gaps in the information base on largescale suckers and species specific effects of contaminants are numerous. No direct studies on largescale suckers have been conducted. Most of the data comes from research on other species of suckers. The majority of the largescale sucker data found examines their hybridization with other species of suckers.

Impact of Habitat Alteration on Prey Choice

Loss of habitat and habitat degradation may not directly impact largescale suckers due to their opportunistic, omnivorous feeding habits. Water and sediment contamination by toxic substances does however appear likely to have an impact on largescale sucker populations. Further research is needed to explain the potential impacts of pollutants on species health.

Impact of Habitat Alteration on Food Web Dynamics and Contaminant Presence

Dauble (1986) found juvenile largescale suckers to be important prey for a number of vertebrate predators, including fishes and birds. Juveniles were found in the guts of northern squawfish, channel catfish and white sturgeon. Sucker fry was found to make up over 50 percent of the diet of the walleye in Lake F. D. Roosevelt on the upper Columbia River. Suckers were also found to be the most common resident fish prey of the bald eagle in the Hanford reach of the Columbia River (Fitzner and Hanson 1979) and also contribute to the diet of the great blue heron. However, resident fish, such as largescale suckers, may only

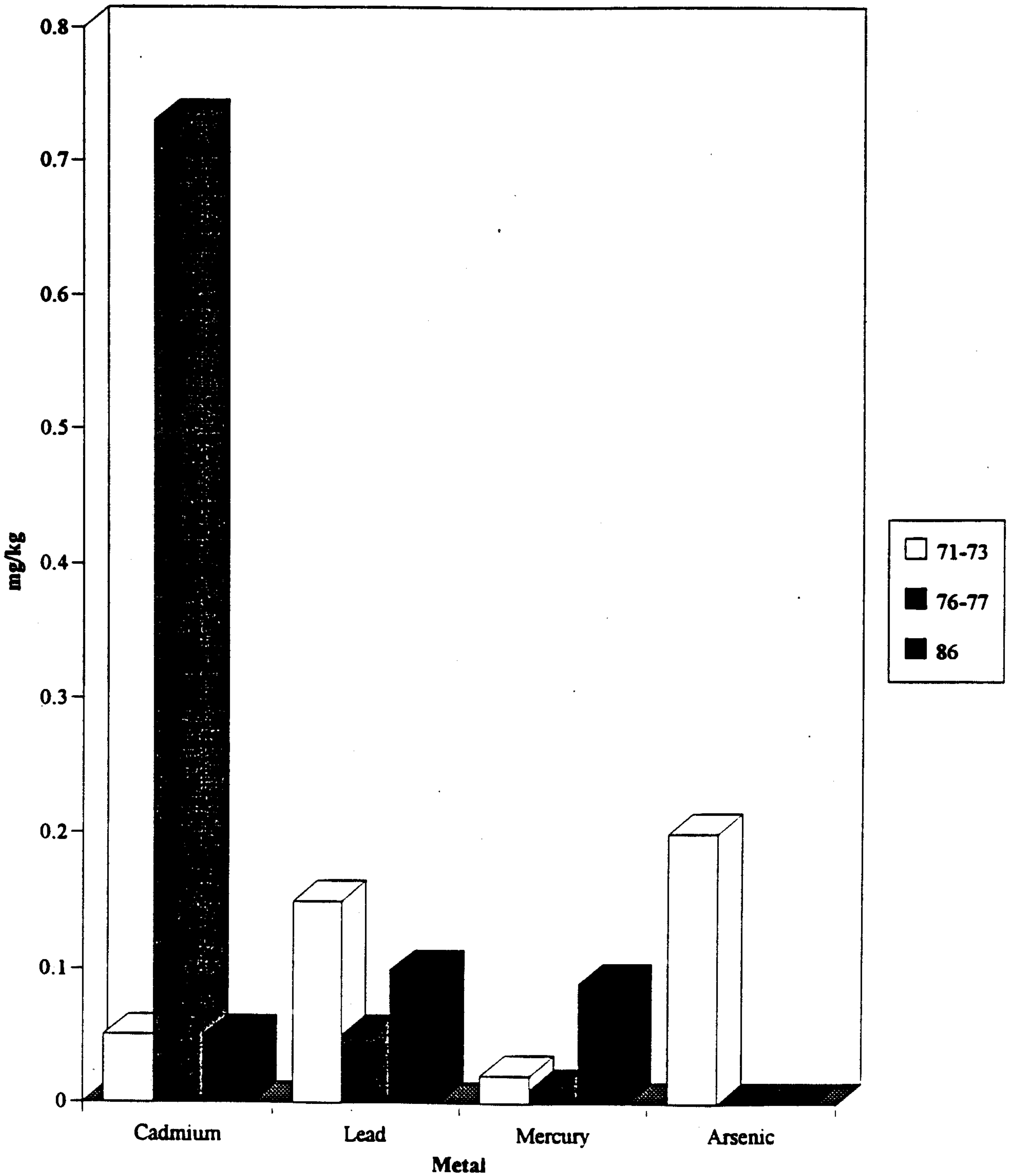

Figure 2. Levels of metals detected in largescale suckers from the lower Columbia River from 1971-1986

comprise a small portion of Columbia River eagle and heron total diets. Algal periphyton is the major food ingested by all sizes of largescale suckers and consisted primarily of diatoms. Chironomidae larvae are found to be the most important insect in the diet of fry.

Although not an obviously critical link in the lower Columbia River's food chain, the largescale sucker is nonetheless an important component of the trophic system. Largescale suckers are not the primary predator on any component of the lower Columbia River's food chain nor are they the primary prey item of any of the river's top consumers. If lost from the river's food chain, direct impacts on any individual species would be difficult to predict. Largescale suckers are however very abundant in the river. Given their numbers, a systemic effect from their loss is foreseeable. If such a large component of one level of the river's trophic system were to be lost, energy transfer would undoubtedly be impaired.

Interactions of Habitat Alteration and Contamination Resulting from Human Economic Activities

The data in table 4 and figure 1 indicates a trend in contaminate levels from 1967-1986 detected in the lower Columbia River. The general trend shows a decrease in all the toxicants with the exception of PCBs which show a substantial rise in the detected level from 1980-1981 to 1986. A common source of PCB isomers is from their use in aerosols, large transformers, carbonless paper, paints, the electronic industry and as coolants (Peterle 1991).

The data was obtained from samples collected from Bonneville Dam, Oregon, and the Columbia River estuary. Bonneville Dam is downstream from Columbia Aluminum in Goldendale, Washington which could effect some of the levels of contaminants detected in the studies. Point sources for the Columbia River estuary are numerous. They include 5 chemical industry facilities, 6 pulp and paper mills, 4 lumber and plywood mills, 3 metal production facilities, 2 power generating plants, and 7 Superfund contaminant sites.

Parsons et al. (1991) conducted a study to determine the impact that effluent from paper mills located along the Columbia River in Oregon and Washington had on fish. The study focused particularly on 2,3,7,8 TCDD. Largescale suckers impacted by the mills average 0.30 parts per thousand (ppt) while suckers collected upstream of the mills averaged 0.24 ppt. The data indicate that mill effluent did not elevate fish TCDD residues above background.

When addressing the levels of metals detected in largescale suckers in the monitoring program (table 5, figure 2), marked increases are not consistent through the years. Cadmium shows a substantial increase in 1976-1977 whereas lead shows its peak level in the 1971-1973 and another peak in the 1986 monitoring program and mercury showed an increase from the 1976-1977 to 1986 monitoring years. Mercury is found in a wide range of materials and comes from many industrial activities such as in industrial fur processing, coal use, electrical materials use and manufacturing, dental treatments, paper mills, paints, fossil fuels, seed dressings, cement manufacturing, mining, petroleum, pesticides, milling and metal alloy production, chlorine and caustic soda production, fungicidal paints, preservatives, batteries, switches, thermometers, barometers, medicinal uses, and in agricultural uses as insecticides and fungicides (Peterle 1991).

The extensive diking and dredging of lower Columbia River wetlands for agricultural use may have removed large areas of spawning habitat from the system. Nelson (1968) reports that in the upper Columbia River banks were modified by farming practices adjacent to spawning areas. This modification could have effected largescale spawning populations though no studies have been conducted.

Dredging and filling for navigational improvements may be a factor in releasing contaminants from sediment particles such as metals and DDT. Dredging began in 1873 and continues today (Simenstad et al. 1984). By 1976, between 5 and 10 million m^3 of material was dredged annually (Sherwood et al. 1990 and Simenstad et al. 1984). A study on yellow perch in the Great Lakes demonstrated that several common environmental contaminants can be accumulated by fish directly form resuspended sediments. Khan et al. (1979) report the fish accumulated PCBs directly from the water after the PCBs were released from the sediments. The possibility also exists that contaminants associated with sediment particles that collected on the gills of the fish were taken up directly by the gill tissue.

Ongoing and Proposed Largescale Sucker Studies Within the Lower Columbia River Basin

-DOE: WA -- Fish tissue contamination monitoring in the lower Columbia River.

EFFECTS OF CONTAMINANTS ON THE BALD EAGLE (*Halieetus leucocephalus*) ALONG THE LOWER COLUMBIA RIVER

INTRODUCTION

The lower Columbia River (LCR) supports a large population of resident bald eagles (*Halieetus leucocephalus*). The special valence accorded to the bald eagle as national bird and its location at the top of the food chain, making it susceptible to environmental contaminants, give it status as an indicator of ecosystem health. Bald eagles are predators who eat fish, waterfowl, and occasionally small mammals, so that these raptors consume the output of aquatic systems.

Bald eagle prey availability can be significantly altered by the presence of human activity and pollutants. Many physical parameters determine the species assemblages of smaller organisms in the estuary. Disturbance of the food chain, habitat alteration, shooting, electrocution, and pesticides all affect bald eagle populations. Many chemicals and toxins may be implicated in bald eagle population declines, including dieldrin, endrin (both banned for pesticide use in 1984), mercury, lead, and PCBs. Habitat loss and human disturbance are also significantly correlated with nesting failures, although their relationship to population declines is less certain (Green 1985).

National declines in bald eagle populations from the 1930s into the 1970s were attributed to habitat loss and contaminants in the environment, particularly dichloro-diphenyl-trichloroethene (DDT) and its metabolite dichloro-diphenyl-dichloroethylene (DDE). DDE is stored in the fatty tissue of fish and waterfowl and accumulates in bald eagles as they feed. DDE can be ingested in lethal quantities but most impacts bird populations through sublethal effects on reproduction, causing eggshell breakage or added embryos. Since the ban of DDT in 1972, DDE residues bald eagle eggshell have dropped and reproductive rates have increased in most regions of the United States (Weimeyer 1993; Green 1985).

The Oregon resident bald eagle population is one of the nation's largest, yet it is still listed as threatened in the state. Despite relatively high numbers of bald eagles in Oregon, bald eagle productivity is low when compared to other recovering populations throughout the United States. Most bald eagle populations are recovering ones, moving back towards density-dependent dynamics after over half a century of having their populations regulated by human-induced mortality and reduced fecundity (Buehler et al. 1991; Green 1985).

The relatively low productivity of LCR bald eagles is cause for concern over continued impacts of pesticides and other contaminants on bald eagle reproductive success. DDE may continue to impact eagle populations (Anthony et al. 1993). In addition, PCBs, heavy metals, and new pesticides used in lieu of banned ones, such as highly toxic organophosphorus pesticides, may be impacting LCR estuary bald eagle populations (Anthony et al. 1993; Henny and Anthony 1989).

The purpose of this paper is to report on existing literature related to bald eagles, the Columbia River estuary, and contaminant effects. As part of the Fish and Wildlife Study of the Lower Columbia River Bi-State Water Quality Program, specific objectives are to evaluate the existing literature on the basis of bald eagle population responses to the presence of contaminants, bald eagle habitat, life history, population dynamics, population trends, and diets; to analyze and synthesize these data on the basis of study methodology, qualitative information provided, and quantitative information provided; and to identify weaknesses in the data base.

FINDINGS

Life History

Bald eagles have been known to live up to the age of 36 but more often die before becoming adults, with immature eagle mortalities estimated at 90 per cent (Green 1985). Bald eagles can be aged through plumage changes, attaining at adulthood the characteristic white headfeathers that give them their name.

Bald eagles prefer long-term monogamy for mating. Breeding periods begin with courtship and nesting activity in the late fall and end when the young become independent five to seven months later. The timing of breeding varies with latitude, beginning as early as late September in Texas or as late as March in Alaska (Green 1985; Stalmaster 1987). Breeding chronology appears to be timed in order to coincide with maximum food supplies for the young. Females usually lay a clutch of between one and three eggs. Clutch size in the west is typically two eggs (Anthony 1983). Both sexes participate in incubating the eggs. Incubation lasts 34-36 days, young fledge between 70 and 98 days after hatching, and fledglings remain dependent for 60 to 80 more days as they learn to hunt.

Isaacs et al. (1983) documented the breeding chronology of Oregon bald eagles. Pre-nesting activities began in February, nesting attempts were underway by mid-March, incubation occurred between March and late May, hatching lasted from early April to late May, and young stayed in the nests from early April to mid-August. Young kept near the nest from August to as late as October. Despite the apparent synchrony of breeding activities at the nest sites monitored, there was site-specific variation in the timing of breeding activities.

Bald eagles reach sexual maturity around five years of age and remain able to reproduce for 20 to 30 years after that (Johnsgard 1990). A substantial proportion of the adults may be non-breeders in some populations, even given enough nesting habitat. This phenomenon may be associated with intraspecific competition and variable annual food supplies (Johnsgard 1990). Average rearing success can improve with food availability. The researcher was interested in testing whether habitat limits breeding success or whether food abundance limits reproduction. If the latter was the case, food surpluses during the breeding period should increase breeding density, clutch size, and offspring survival. Along the Chilkat River in southeast Alaska, more nests are active when major food patches exist than in the years when the food patches are absent. Proximity to the spring patches of food also influences the laying date of the eagles. Pairs within 3 km of thawed salmon carcasses laid eggs earlier than more distant nesting pairs, and nest success was higher at areas that were experimentally supplemented with food, although clutch size was not significantly larger. With an earlier laying date juveniles may have more time to acquire skills for enduring winter weather and variable food supplies (Hansen 1987).

Younger subadult eagles may also be less successful than adults at acquiring food in winter. At gravel bars feeding stations supplied with salmon carcasses, eagles often kleptoparasitized other eagles. Subadults had their food stolen from them by adults more often than they could successfully steal from the adults. This was despite the observation that subadults also initiated intraspecific aggression more often than adults, so that their success rate was low relative to the adults. Aggressiveness in food acquisition could be a form of social dominance. Adult eagles were the only age class to procure sufficient food from the feeding station, suggesting that the younger birds were forced to seek out other, non-monitored feeding areas. These dynamics partially explain the high mortalities of younger birds, especially during food shortages (Stalmaster and Gessaman 1984).

Subadults have been found to be more abundant when prey is more abundant and available. Hansen and Bartelme (1981) found that the proportion of subadults in the wintering bald eagle population on the Skykomish River in Washington increased in the year when salmon runs were larger. They studied the bald eagle population during the winters of 1977-78 and 1979-80. There were larger salmon runs in 1979-80. Eagle numbers increased from 17 to 31 in 1979-80, and the proportion of subadults rose from 22.5 per cent to 57.7 percent of the population in 1979-80. Fitzner et al. (1981) found similar results among wintering eagles in the Hanford National Environmental Research Park along the upper Columbia River. The proportion of subadults there was likewise higher when prey was more abundant. Garret et al. (1988) documented that subadults also move northwards in spring later than the adult portion of the population and appear to be more dependent on carrion and crippled prey.

Watson et al. (1991) observed the foraging ecology and behavior of bald eagles along the Columbia River estuary. Eagles exhibited different behaviors for breeding and non-breeding seasons, the most marked being also the most obvious: nesting behaviors accounted for the majority of the difference. The eagles also engaged in more acts of intra-specific aggression during the winter. Eagles also exhibited seasonal differences in prey choice. During the breeding season smaller fish were the preferred prey more often than during the non-breeding season. Eagles hunted their prey most often (57 percent of the time), but also scavenged and pirated. There were no differences in foraging methods from season to season, but scavenging proved more successful (98 percent success rate) than attempts to capture live prey (66 percent) which was in turn more successful than piracy (46 percent). There was a negative correlation between total foraging time and overall foraging success ($r=-0.57$), implying that those eagles that were less successful hunters also (or therefore) spent more time hunting. Influences on foraging behavior included water depth, tide, and time of day. Eagle predation of fish peaked at dawn. Eagles used shallow water predominantly to capture fish and foraged more often during low tide than would be expected, based on expectancies derived from the amount of time eagles were observed during each tidal cycle.

Temporary human disturbance, such as recreational boating, may affect temporary bald eagle behaviors, such as foraging, more than the type of permanent disturbance, such as human housing and industry (McGarigal et al. 1991). Eagle foraging time increased within 400 meters of high-use foraging areas during the weekends, when human activity, especially boating, was greater. This suggests that human activity may alter eagle hunting efficiency. Eagles appeared to alter their habitat use patterns both temporally and spatially in response to a stationary boat experimentally placed near the center of their high-use foraging area. Eagles subjected to relatively high boating activity concentrated their foraging more in the early morning hours before sunrise than pairs subjected to relatively low boating activity. Spatially, the eagles avoided an area within 300-400 meters of the boat situated in the center of their previously much used foraging area. This avoidance also reduced foraging efficiency, resulting in more time spent actively hunting.

Habitat Requirements

Bald eagle range throughout North America. Eagles in the northern latitudes (Alaska and Canada) migrate to the lower forty-eight states in winter. The species decreases in size gradually from their northern to southern range and once were thought to be two species based on this geographical gradation.

Bald eagle habitat needs are based on three basic requirements: an adequate supply of moderate to large-sized fish, nearby nesting sites, and freedom from disturbance during the nesting period (Johnsgard 1990). Habitat choices is limited by nesting site requirements and by prey abundance. In Oregon (Klamath Basin, Cascade lakes, Oregon coast and LCR) from 1978-1982, the majority (85 percent) of bald eagle nests existed in large stands of tress, usually ponderosa pine, Douglas-fir, or sitka spruce, near bodies of water with substantial fish and waterfowl populations (Isaacs et al. 1983).

Bald eagle nest site parameters include proximity to open water, large trees with sufficiently high sturdy branches, and stand heterogeneity. Eighty-nine percent of the nests had some human disturbance within one mile and 74 percent of the disturbances were within half a mile. There was no direct association between disturbance and nest success or activity. However, close proximity of human activity to nesting sites was often associated with unproductive nests. These findings imply a certain tolerance of bald eagles to human presence in their habitat, although presence does not equal disturbance, which was defined as the presence of such permanent human activities as roads, buildings, docks, airstrips, towns, houses, clearings, logging, and farming. The point at which presence seemed to become disturbance was within 0.25 miles or closer to the nest. Within this range more unproductive nests were observed. In southeast Alaska, bald eagle nests in trees above the canopy were more active (70 percent v. 46 percent of canopy-level nests). In addition, nests in Sitka spruce were more successful (36 percent) than those in black cottonwood (12 percent). Timber type and degree of human activity significant factors. Habitat quality and food abundance influence breeding activity, although variations in the food supply might be a stronger determinant of year to year population variations, so that areas with ostensibly good bald eagle habitat might exhibit negative productivity trends (Hansen 1987).

Along the Columbia River estuary, the dominant habitat type for bald eagles is open-water, mudflat, and marsh habitats in the mid and upper estuary. Eagles occur throughout most of the estuary in all seasons but also always in relatively low numbers. Eagle densities were never more than one bird per kilometer and lower estuary open-waters supported no eagles (Hazel et al. 1984). This was probably because of inadequate perch sites, although it could also be ascribed to lack of nearby shallow water for foraging. Most eagle nests are situated in coniferous stands bordering the estuary and on river islands. Most frequent use by bald eagles in the LCR is characterized by extensive intertidal marshes, shallow water, and exposed mudflats at low tide, yielding stranded fish prey and ample perch habitat (Figure 2). In the area between Tongue Point and Tenasillahe Island, 57 percent of the resident bald eagles in the LCR were clustered, because of prey availability of prey appeared to be the most important factor in breeding site selection (Garrett et al. 1988).

Bald eagles in the Columbia River estuary appear to make ample use of available tidal flats. Tidal flats allow for scavenging and foraging of prey in shallow water, which may be particularly important for subadults not yet adept at efficient foraging strategies and hunting (Figure 1). Other important habitats for perching, roosting, and foraging are old growth and mature stands of conifers and bottom land hardwood (Garrett et al. 1993; Hansen 1987).

Winter roosts are distinguished from nests because they are primarily used at night and are often communal and tend to winter based on available food, so that some dry habitats can support wintering eagles. In winter, habitat choice can also be a behavioral strategy to minimize energy loss (Stalmaster and Gessaman 1984). On the Nooksack River in Washington, which is characterized by mild winters with heavy rain, bald eagles winter between October and March. Metabolic heat production of the bald eagles in response to the environmental parameters measured, including temperature, rainfall, wind speed, and long-wave radiation, was highest in January and at a gravel bar station, lowest in February and in coniferous habitat, and intermediate in December and in deciduous habitat. Higher rates of heat loss are incurred under clear skies, such as those over open habitat in January. The coniferous forest had the mildest winter microclimate regime of the three microhabitats, with higher night temperatures, lowest rainfall, and highest downward long-wave radiation. Mature, old-growth coniferous forests may be chosen for roosting rather than riparian deciduous stands because wind and radiative heat loss are reduced in this habitat.

Nesting site choice may be more influenced by behavioral strategies than by prey abundance or the nest site parameters that apply during breeding season (Green 1985). Breeding bald eagles, however, are territorial and require a substantial food base to support their relatively large size, therefore, they have large home ranges. Frenzel (1983) estimated an average home range of 660 hectares for eight breeding pairs in Oregon and an average distance between nesting territories of 3.2 km, with an average of 0.5 km of shoreline within each territory.

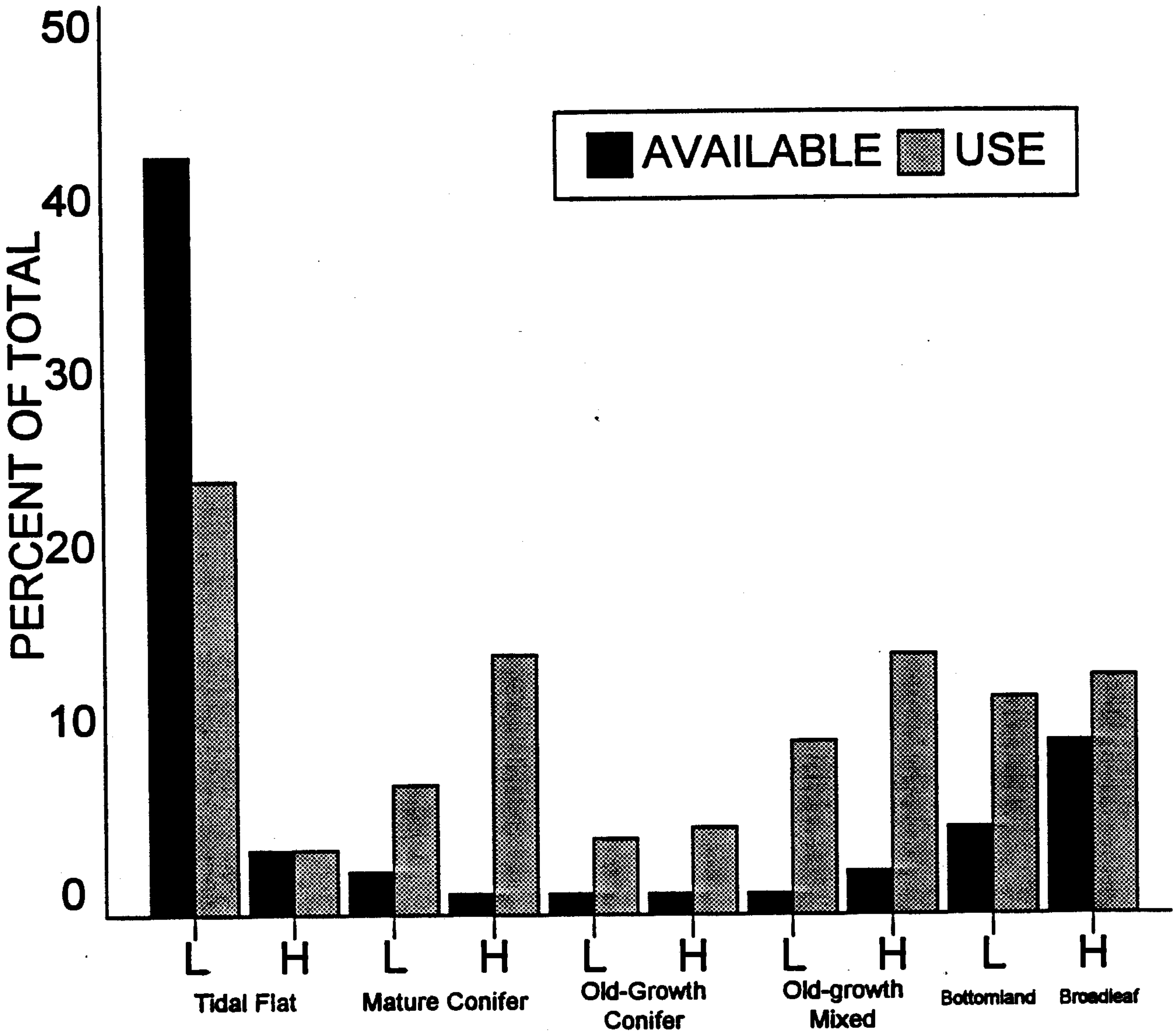

Figure 1. Percent habitat availability of and use by resident and non-resident bald eagles during low (L) and high (H) tide. Columbia River estuary, 1984-1986 (Garrett et al. 1993).

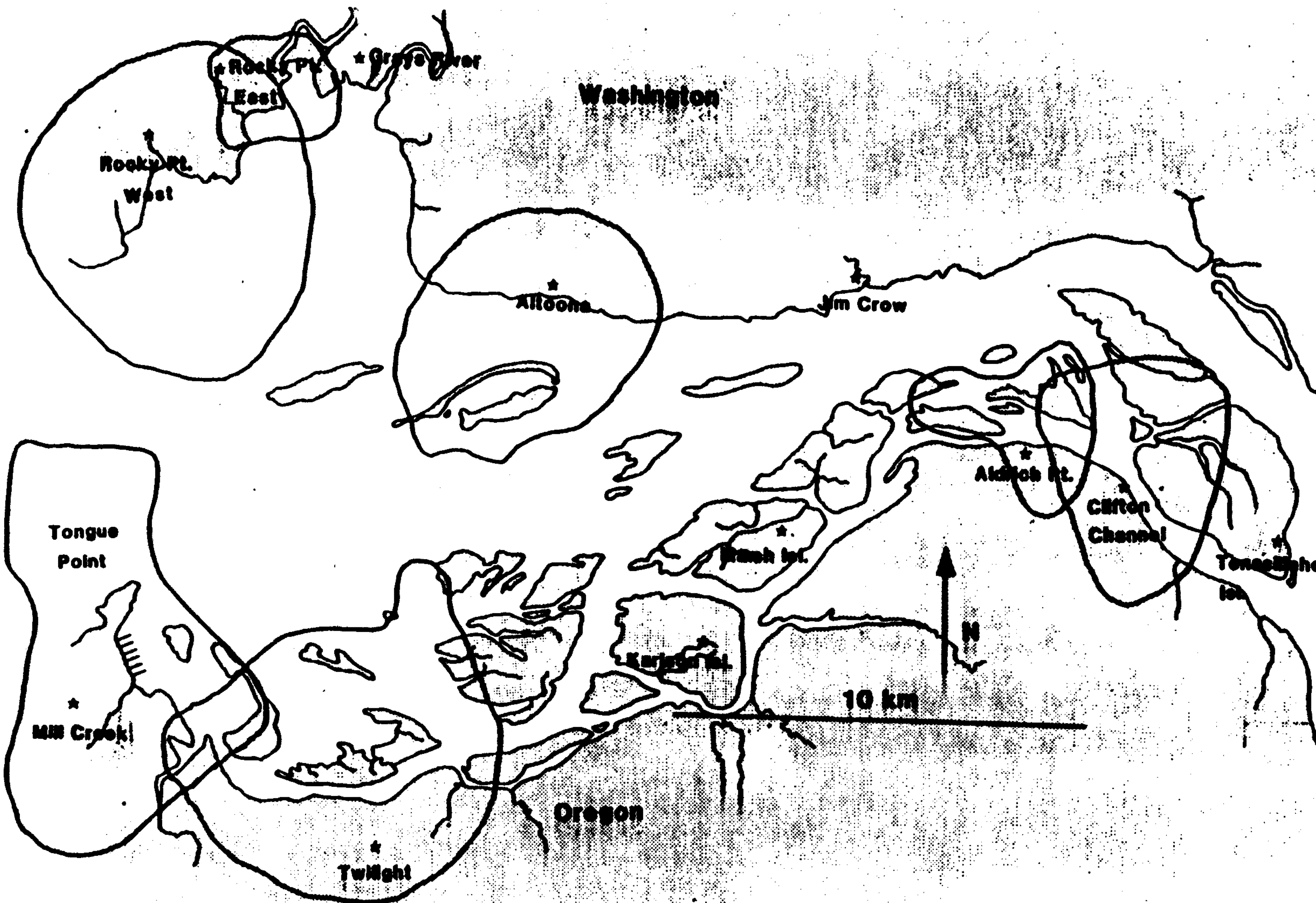

Figure 2. Locations of high use nest territories in the lower Columbia River, 1984-86.(Garrett et al. 1988).

Diet

Fish are considered the food of choice for bald eagles, although they also consume waterfowl and mammals when such prey is readily available or highly abundant. Eagles hunt and scavenge for both live and dead fish. Live fish are vulnerable only near the water surface, since eagles hunt by extending their talons inches into the water. Wintering bald eagles in the Klamath Basin forage on avian and mammalian prey and not on fish, eating primarily waterbirds and voles. Piscivorous birds are one trophic level up from fish and may contain higher levels of contaminants. In addition, different contaminants may be present in different prey. No organochlorines were found in the Klamath Basin voles or in black-tailed jackrabbits and only low concentrations of mercury were detected in 40 percent of the vole samples. Fifty percent of the jackrabbit samples contained lead, and all of the vole samples contained detectable levels of lead. Low concentrations of DDE were present in 80 percent of the waterfowl collected from bald eagle foraging areas (Frenzel and Anthony 1989). Anthony et al. (1993) found contaminants in the prey samples of the Columbia River estuary bald eagles. PCBs were present in the fish samples, including the large-scale sucker, at levels higher than the national mean and DDE was present although not in excessive concentrations. Elevated levels of PCBs in the bald eagles might then have been from consumption of fish prey, while elevated DDE levels might have come through one more level of magnification, that is, from other waterfowl which prey on fish.

Fitzner et al. (1981) suggested that bald eagle diets depended on food availability, not just prey abundance. They observed that bald eagle numbers in the winter months were correlated with abundances of salmon redds. They noted, however, that the statistical correlation belied the biological facts. In the mid-Columbia River, chinook spawn in early November and carcasses are available for consumption between late November and mid-December, so that chinook availability does not span the entire wintering period. Chinook availability for eagles is dependent on carcass availability then rather than spawning times. Waterfowl availability is dependent on the numbers of sick and injured birds not on the healthy bird count, so that waterfowl availability for eagles can be enhanced by the presence of human hunters. Waterfowl were found to be the principal bald eagle food in January and February in the Hanford National Environmental Research Park (Fitzner et al. 1981). Hansen and Bartelme (1981) found similar results for wintering bald eagles on the Skykomish River in Washington, where the eagles fed on chinook and coho salmon carcasses until mid-February, after which they were thought to have consumed non-anadromous fish and waterfowl.

Hansen (1987) found that the fortuitous thawing of salmon carcasses frozen in a river can benefit fledgling survival rates in the spring. Stalmaster and Gessaman (1984) observed that eagles preferred partially eaten or torn carcasses, probably because of the energy savings provided by the already opened flesh. Wintering bald eagles also benefitted more from carcasses that thawed in shallows or gravel bars, since decomposition rates of the carcasses were higher in water.

In the Columbia River estuary, fish is the most common prey (71 percent), with the most frequent species being resident catostomids and cyprinids, and anadromous clupeids and salmonids (Table 1). The proportion of fish consumed by eagles in the Columbia River estuary is higher than other areas studied, such as the Chesapeake Bay, southern Louisiana, and southeast Alaska (Garrett et al. 1988). Fish consumption is probably actually higher than the percentages observed from remains at prey nests since fish tend to degrade faster and since foraging observations showed a higher take of fish (Watson et al. 1991).

The high variability of LCR bald eagle diets may be related to prey availability rather than prey preferences either prey distribution or individual prey preferences (Garrett et al. 1988). Eagles preferred smaller fish in the breeding season than the non-breeding season, probably because of the increased capture of juvenile salmonids moving through the estuary in June. Other prey categories were waterfowl, seabirds, and small to medium-sized mammals. Tidal flats and shallow water increase fish availability, both live and dead. In the Columbia River estuary Grays Bay and the estuary islands are important foraging areas (Hazel et al. 1984). There was a 16 percent overall increase in avian consumption, at the expense of fish consumption, with the majority of that increase accounted for by waterfowl (Garrett et al. 1988). Dietary shifts in winter are common and probably reflects a seasonal change in prey availability rather than changes in habitat use since the resident bald eagles stay near the same nest site year-round.

This is wrong

Contaminants

In the LCR, bald eagles are the most contaminated of the piscivorous birds sampled, with high concentrations of ~~DDE, PCBs,~~ ~~and~~ TCDD (USFWS 1993). Bald eagle egg levels of DDE ranged between 0.96 and 1.3 ug/g. Levels of PCBs were between 5.1 and 10 ug/g and levels of TCDD were ~~between 60 and~~ 61 pg/g. This is compared to western gulls, for example, from the same sample area who had DDE levels ranging from 0.81-1.4 ug/g, PCBs levels from 1.4-2.2 ug/g, and TCDD levels from 4.0-7.0 pg/g. Sediment and prey samples from the same area, including the amphipod *Corophium* and largescale suckers (*Castostomus macrocheilus*), also showed measurable levels of dioxins. The concentrations of TCDD found in the fish samples exceeded the EPA fish consumption guideline for human health protection. The next most contaminated bird sampled was the double-crested cormorant. Like the bald eagle, cormorants are piscivorous residents of the Columbia River estuary. Another proposed study would research environmental contaminants in bald eagles nesting along the Columbia River and would relate levels of organochlorine pesticides, PCBs, mercury, dioxins, and furans in bald eagle eggs to biological parameters such as productivity (USFWS 1994c).

Many of the bald eagle population declines in the United States documented between 1950 and 1975 were associated with contamination by DDT and its metabolites (DDD and DDE). DDT and DDE are known to be linked with bald eagle eggshell thinning (Weimeyer 1993). Polychlorinated biphenyls (PCBs) may have similar effects but since they are often highly correlated with the presence of DDE, it is difficult to discern the effects of PCBs

Table 1. Prey remains from bald eagle nests in Columbia River estuary, Oregon and Washington, 1984-86 (Watson et al. 1991).

Classification	Individuals No.	Individuals %	% of Class	Nests where prey identified No.	Nests where prey identified %
Fish					
Catostomidae				15	83
largescale sucker (*Catostomus macrocheilus*)	32	17.3	24.2		
Clupeidae				13	72
American shad (*Alosa sapidissima*)	24	13.0	20.2		
Cyprinidae				12	67
common carp (*Cyprinus carpio*)	20	10.8	15.2		
peamouth (*Mylocheilus caurinus*)	18	9.7	13.6		
other cyprinids	8	4.3	6.7		
Salmonidae				12	67
salmon (*Oncorhynchus* spp.) and steelhead (*Salmo gairdneri*)	16	8.6	12.1		
Centrarchidae				5	27
black crappie (*Pomoxis nigromaculatus*)	2	1.0	1.5		
other centrarchids	3	1.6	2.3		
Percidae				2	11
yellow perch (*Perca flavescens*)	2	1.0	1.5		
Acipenseridae				3	17
sturgeon (*Acipenser* spp.)	3	1.6	2.3		
Gadidae				2	11
unidentified	3	1.6	2.3		
Embiotocidae				1	6
shiner perch (*Cymatogaster aggregata*)	1	0.5	0.8		
Subtotal	132	71.0			
Birds					
Anatidae				12	67
mallard (*Anas platyrhynchos*)	9	4.9	18.4		
green-winged teal (*A. crecca*)	4	2.2	8.2		
northern pintail (*A. acuta*)	2	1.0	4.1		
American wigeon (*A. americana*)	2	1.0	4.1		
Canada goose (imm.) (*Branta canadensis*)	3	1.6	6.1		
other anatids (*Aythya* spp.)	2	1.0	4.0		
Podicipedidae				6	33
western grebe (*Aechmophorus occidentalis*)	8	4.3	16.3		
Phalacrocoracidae				3	17
cormorant (*Phalacrocorax* spp.)	5	2.7	10.2		
Laridae				4	22
gull (*Larus* spp.)	5	2.7	10.2		
other	2	1.0	4.0		
Alcidae				1	6
common murre (*Uria aalge*)	2	1.0	4.1		
Other birds	5	2.7	10.1	3	17
Subtotal	49	26.1			
Mammals					
Leporidae				2	11
brush rabbit (*sylvilagus bachmani*)	2	1.0	50.0		
Other mammals	2	1.0	50.0	2	11
Subtotal	4	2.0			

independent of DDE induced shell thinning (Weimeyer 1993). The consequences of eggshell thinning can be reduced embryo survival or reduced productivity, where productivity is defined as number of successful offspring per nesting area. No North American raptor population with eggshell thinning greater than 18 percent has been able to maintain a stable population (Lincer 1975).

Both organochlorines and polychlorinated pesticides in extremely high doses can have lethal effects. The effects of lead ingested by bald eagles through waterfowl can also be lethal (Frenzel and Anthony 1989; Pattee et al. 1981). Both mercury and lead have been detected in sublethal concentrations in Oregon bald eagles, and although mercury is considered to have adverse effects on reproduction, these effects are not well known (Wiemeyer et al. 1989). The effects of dioxins, particularly 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD), on bald eagle reproductive success are not well known (USFWS 1994a). However, in other piscivorous birds TCDD has been shown to have negative impacts on reproductive success. TCDD concentrations in the Columbia River are high enough that the water quality control of a total maximum daily load (TMDL) had to be implemented by the EPA (USFWS 1994a).

Between 1975 and 1977, Kaiser et al. (1980) found that 166 of 168 bald eagle carcasses from 29 states contained measurable levels of PCBs and 165 of 168 had measurable levels of DDE. Dieldrin, another common pesticide, was present in a substantial portion as well. Despite the near omnipresence of DDE and PCBs, the frequency of detection of these contaminants has declined between 1972 and 1977. DDE detection has declined from 100 to 83 percent, PCBs from 92 to 84 percent, and dieldrin from 67 to 42 percent. The same study found evidence of lead shot poisoning in nine specimens.

Wiemeyer et al. (1993) examined environmental contaminants and related productivity indices in 105 bald eagle eggs from 15 states between 1980 and 1984 and evaluated trends in contaminant concentrations from 1973-1984. Oregon was among the states with statistically significant eggshell thinning, the other states being Wisconsin, Ohio, Maine, Maryland, and Virginia, although biologically significant eggshell thinning (defined as greater than 15 percent in a population) did not occur within any of the states. However, because the researchers selected eggs for study that had failed to hatch, they caution that these results should be viewed as possibly biased. DDE, PCB, and mercury residues were present in every breeding area sampled. Eggs from Oregon contained the second highest mean concentration of DDE, after eggs from Maine.

DDE and PCBs to be highly intercorrelated ($r = 0.765$). DDE was negatively correlated with mean five-year young production across the sample sites ($r = -0.545$), as were PCBs but less so ($r = -0.399$) (Wiemeyer et al. 1993). Both DDE and PCBs were also negatively correlated with adjusted shell thickness. The researchers identified 3.6 ug/g DDE as a threshold level of contamination above which raptor productivity will decline and 16 ug/g DDE as a level above which greater than 15 percent shell thinning occurs. Other researchers have suggested that eggshell thinning is a parallel symptom of DDE poisoning rather than causally related to declines in productivity (Nisbet 1989).

Noble and Elliot (1990) reviewed levels of contaminants in Canadian raptors, based on data collected by the Canadian Wildlife Service between 1966 and 1988. The researchers estimated minimum critical levels of organochlorine and mercury residues in raptors above which acute lethal toxicity may ensue. For bald eagles, minimum DDE levels were 250 mg/kg wet weight in the brain, 100 mg/kg in the liver, and 1.2-30 mg/kg in eggs. For PCBs, minimum levels were 500-3000 mg/kg in the brain, unknown in the liver, and > 50 mg/kg in eggs. For mercury, minimum brain levels were > 50 mg/kg, liver levels were 20-45 mg/kg, and egg levels were > 0.5 mg/kg. Few raptor tissue samples analyzed actually demonstrated evidence of acute toxicity from contaminants. The researchers observed that species which fed primarily on migratory birds were most contaminated and species whose diets were principally mammalian were least contaminated. Factors influencing this trend may be local pesticide use, food from aquatic sources, and wintering range.

In the LCR, DDE and PCBs may be severely affecting the reproductive success of the resident bald eagles, based on their review of existing literature on bald eagles and contaminants in the western United States (Henny and Anthony 1989). Anthony et al. (1993) examined the presence of environmental contaminants in bald eagles in the Columbia River estuary. The researchers collected 19 intact eggs, 12 eggshell fragments, 22 blood samples, and 2 carcasses of bald eagles and determined breeding success for 22 occupied territories between 1980 and 1987. DDE, PCBs, and 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) were high in fresh eggs, adult blood, and the carcasses. Nestling blood showed detectable levels of DDE and PCBs, while adult blood showed higher concentrations, indicating accumulation with age. In nestlings, mean DDE concentrations were 0.05 ppm wet mass, mean PCB concentrations were 0.04 ppm, mean lead concentrations were 0.23 ppm, and mean mercury concentrations were 0.47 ppm. In adults, mean DDE concentrations were 2.13 ppm, PCB concentrations were 2.40 ppm, mean lead concentrations were 0.43 ppm, and mean mercury concentrations were 3.07 ppm (Table 2).

These contaminants were also evident in the bald eagle prey samples, including those of the large-scale sucker. PCB residues in the fish samples were greater than the national mean (0.85 ppm in the large-scale sucker), yet DDE concentrations were not excessively high in the fish prey (Table 3). The researchers hypothesized that the elevated levels of PCBs in the eagles might be from fish prey, while elevated levels of DDE might be from consumption of waterfowl which prey on fish. Mean eggshell thickness was 10 per cent thinner than pre-DDT averages. The researchers described a negative correlation between breeding success and eggshell thickness ($r = -0.52$, $n = 14$, $P < 0.05$). Breeding success, defined as living offspring after 8-11 weeks, of LCR bald eagles was 39 percent during the study period, while statewide averages were 62 percent.

Garrett et al. (1988) believed the residues of DDE and PCBs in LCR bald eagles were not merely passed on from the mother to the egg, because DDT residues have previously been

Table 2. Concentrations of environmental contaminants (ppm wet mass) in blood from bald eagles, Columbia River estuary. 1984-86[a] (Anthony et al. 1993).

Contaminant	Nestlings (n = 15)	Subadults (n = 4)	Adults (n = 3)
pp' DDE	0.05 0.01-0.24	0.31 0.15-0.70	2.13 1.00-3.20
PCBs	0.04 ND[b]-0.13	0.53 0.14-1.50	2.40 1.40-3.50
Lead	0.23 0.03-0.70	0.17 ND[b]-0.27	0.43 0.05-1.00
Mercury	0.47 0.19-1.40	1.50 ND-3.10	3.07 1.30-4.10

[a]Values are mean and ranges.
[b]ND = None detected at minimum level of detection (organochlorines and PCBs--0.01; heavy metals--0.02).

Table 3. Concentrations of organochlorines, PCBs and heavy metals (ppm wet mass) in whole body samples of fish prey species, Columbia River estuary, 1986[a] (Anthony et al. 1993).

Contaminant	Large scale sucker[b]	Peamouth[b]	American shad[b]	Northern squawfish[b]
pp' DDE	0.07 ND[c]-0.12	0.41 0.34-0.52	ND = 0.14	0.20 ND-0.37
pp' DDD	0.08 0.05-0.14	0.15 0.08-0.21	0.08-0.11	0.21 0.10-0.40
pp' DDT	0.02 ND-0.03	d ND-ND	0.05-0.05	ND-0.08
Total PCBs	0.85 0.74-0.29	2.1 0.68-3.30	0.26-0.49	1.7 1.00-2.30
Cadmium	0.052 0.034-0.084	0.061 0.043-0.084	0.054[e]	0.17[e]
Lead	0.10 ND-0.17	ND-0.16	ND[e]	ND[e]
Mercury	0.094 0.042-0.17	0.12 0.061-0.16	0.039[e]	0.19[e]

[a]Values are arithmetic means and ranges.
[b]Each fish sample was a homogenized composite of 3-5 individuals of the same species that were collected from the same local area; n = 4, 3, 2, and 3 for sucker, peamouth, shad, and squawfish, respectively, unless otherwise noted.
[c]ND = none detected at minimum level of detection (organochlorines and PCBs--0.01; lead--0.08).
[d]Means were not calculated if <50% of the samples contained detectable concentrations of <3 samples were analyzed.
[e]n = 1.

shown to dilute with avian growth. This would indicate that accumulation with age and detection of organochlorine residues early in the bald eagles' life histories, suggesting bald eagle prey as the source of contamination. The source of DDE residues in the LCR bald eagle populations is of interest since DDT has been banned for years. It is coming from waterfowl that migrate to Central and South America or from persistent residues in Columbia River sediments and water. However, elevated levels of contaminants are also found in breeding LCR resident eagles, mink, river otter, and black-crowned night herons that eat mostly fish, suggesting a local source of contamination (Henny 1984;. Henny et al. 1981).

Studies of other piscivorous birds can elucidate the mechanisms and relative severity of bald eagle contamination. One study currently in progress is examining the concentrations and effects of environmental contaminants in Columbia River Basin great blue herons to determine if they might serve as good indicator species (USFWS and Oregon State University 1994). Another study in progress is designed to test for contaminants levels in double-crested cormorant egg embryos from the Lewis and Clark National Wildlife Refuge along the Columbia River estuary and to test for the effects of PCBs and dioxins on cormorant reproductive success (USFWS 1994b).

Henny et al. (1984) presented results for the impacts of DDE on black-crowned night-herons along the Columbia River. One egg was collected from each of 220 nests throughout the intermountain west during 1978-1980. DDE was present in all 220 eggs. Heptaclor epoxide was more common along the two Columbia River sites than in samples from other sites sampled. PCBs were present in 50 percent of the eggs sampled and were most common in Columbia River and Ruby Lake, NV eggs. The researchers found negative correlations between eggshell thickness and log-transformed residues of DDE ($r = -0.559$) and of PCBs ($r = -0.264$). Data pooled from all the colonies sampled demonstrated that the percentage of successful nests, clutch size, and number of young per successful nest decline as DDE residues increase. DDE residue levels in the two Columbia River sites sampled were just as high as the relatively more contaminated more southern colonies and did not decline over the study period, as other northern colonies sampled did. DDE and PCBs were found in the fish prey samples and regurgitated prey samples of Columbia River night herons, as opposed to their absence in Ruby Lake samples. This suggests that Columbia River herons ingest the contaminants locally, while the southern populations encounter contaminants during their migrations to Latin America in the winter.

Organophosphorus pesticides, which have become more common following the ban on organochlorine pesticides, may also adversely affect bald eagles (Henny et al. 1987). Eight bald eagle deaths in Iowa, California, and Idaho can be attributed to famphur and fenthion applied to the backs of livestock for insect control. These chemicals are less persistent in the environment but more toxic to birds. However, organophosphorus pesticides are probably not a threat to eagle populations in the LCR, since these pesticides have little or no capability to spread throughout the ecosystem. Mortalities due to organophosphorus poisoning in LCR bald eagles have yet to be documented (Schuler 1994 pers. comm.).

Buehler et al. (1991) conducted a study in the Chesapeake Bay to determine survival rates and population dynamics of bald eagles in order to determine the extent their supposed population recovery following the banning of DDT. Average adult bald eagle survival rates were found to be 91-98 percent, breeding territories had increased nearly 250 percent over a period of nine years, and nesting success rates ranged from 58 to 82 percent over those years. The average number of young per successful nest was 1.71 and the breeding population appeared to be growing at a rate of 12.6 per cent per year. These numbers and the modelled maximum intrinsic growth rate were higher than usual or expected, so that the researchers concluded that since the ban of DDT, population growth rates have been exponential during population recovery.

Population Trends

For bald eagles, productivity and breeding success are useful indices of the population health and dynamics at a given point in time. Reproductive success can be a useful measure of population stability (Garrett et al. 1988). Productivity is the number of young raised per nest and breeding success is the percent of offspring which live to 8-11 weeks in a population (Anthony et al. 1993). These measurements can account for fecundity, survivorship of fledglings and mortality of fledglings, which in turn feed into and reflect adult dynamics. Since the main effects of contaminants on bald eagle populations are sublethal impacts on reproduction, young production and survivorship can be a better indication of the population status relative to contaminants than adult survivorship.

The LCR resident bald eagle population is one of three main breeding populations in Oregon. Between 1978 and 1982, 147 bald eagle breeding sites that were monitored by Isaacs et al. (1983). The nests were clustered in three regions: Klamath Basin, Cascade lakes, and the Oregon coast and LCR. In the three regions, breeding chronology was nearly synchronous, although nesting at higher elevations tended to occur later than at lower ones. In 1975, north of the Columbia River and west of the Cascades, including Puget Sound basin, 218 nests were found, 114 of which were occupied (Grubb 1976).

As of 1992, Oregon supported 227 bald eagle sites, 205 of which were occupied. The LCR had a total of 37 bald eagle sites in Washington and Oregon (USFWS 1994a). Garrett et al. (1988) documented 32 nesting pairs of bald eagles in the Columbia River estuary. Hazel et al. (1984) measured bald eagle densities along the Columbia River estuary between 1980 and 1981. Bald eagles occurred throughout most of the estuary in all seasons in relatively low numbers. Densities ranged from 0.1 bird/km of transect to 0.7 bird/km of transect.

The influx of migrant bald eagles in winter augments the population numbers of residents. Oregon supports the sixth highest wintering bald eagle population in the nation (653 in 1980) (Opp 1981). The Klamath Basin, which spans Oregon and northern California, has the largest wintering population of bald eagles in the lower 48 states. The migrants probably originate from British Columbia and Saskatchewan (Opp 1981). In 1990, the State of Oregon midwinter bald eagle count found 701 bald eagles. This was 42 percent higher than in 1989 (USFWS 1994a). In 1994, 107 wintering eagles were counted in the Columbia River Recovery Zone (Isaacs 1994). The five-year averages of bald eagles observed in Midwinter Eagle Counts in Oregon had consistently risen over the period 1979-1994 (Isaacs 1994).

Year-round abundances of resident bald eagles have been correlated with prey abundances in the literature (Watson et al. 1991). Numbers of wintering bald eagles and subadult eagles seem to be especially influenced by prey abundance. Frenzel and Anthony (1989) attributed the high numbers of bald eagles wintering in the Klamath Basin to the relatively high abundances of waterfowl and small mammals there. Fitzner et al. (1981) found that numbers of wintering bald eagles in the Hanford National Environmental Research Park were correlated with prey abundances and that subadults accounted for a greater proportion of the population when prey was abundant. However, the researchers challenged the idea that prey abundances in and of themselves were meaningful for the eagle population. Instead, they suggested that prey availability, determined by the amount of dead or moribund waterfowl and salmon in winter rather than the amount of live animals, should be the variable considered. Hansen and Bartelme (1981) also found a positive correlation between winter eagle numbers and abundance of salmon, as well as a higher proportion of subadults when food was more abundant.

Annual variations in the food supply of bald eagles may cause variations in bald eagle fledging rates (Hansen 1987). Areas with adequate habitat may have negative productivity trends for bald eagles if the food supply has extreme fluctuations. Prey abundances can be influenced by natural fluctuations in populations or by human impact, including contamination. However, Hansen also suggested that the bald eagle populations may themselves merely be a natural oscillatory cycle driven by competition for food between breeders and non-breeders. This would imply that resuscitated populations in other habitats may result in oscillatory population trends that may be indicators of the health of the population. Buehler et al. (1991) also cautioned against assumptions of what a 'natural' and healthy bald eagle population dynamic should look like over time, since no reliable data exist for such populations. They suggested that the exponential population growth they observed in Chesapeake Bay bald eagles was a stage of bald eagle population recovery and that the population, given no more habitat alteration or contamination, would eventually hit density-dependent growth.

Bald eagle productivity is an important indicator of the population status. Productivity is measured as number of young raised per nest and is still abnormally low in some states, even those with relatively high numbers of eagles, including Washington and Oregon. Garrett et al. (1988) cautioned that the productivity guidelines set by Sprunt et al. (1973) for population stability may be too low. These guidelines were 50 percent breeding success and 0.70 productivity. The productivity numbers are also averages which conceal the often highly variable annual productivity of Columbia River bald eagles.

As of 1985, Oregon bald eagle productivity was 0.77 and Washington's was 0.78 (Green 1985). Table 4 provides a summary of productivity data for the Columbia River estuary from 1980-1987, giving separate data for Oregon and Washington. Table 5 adds to this data for the years 1989-1993 for the Columbia River Bald Eagle Recovery Zone. These numbers are lower than all but three of the states containing eagles and than the recommended minimum average productivity of 1.0 required for delisting of the bald eagle as a threatened species (Green 1985). The breeding success of bald eagles in the Columbia River Recovery Zone (Zone 10) was lower and more erratic than Oregon statewide averages for the years 1989-1993, with rates ranging between 33 and 79 percent and five-year averages not higher than 55 percent, while statewide averages for those years were between 50 and 67 percent (Isaacs and Anthony 1993).

Table 4. Success and productivity of bald eagle nest sites, Columbia River estuary, Oregon and Washington, 1980-87 (Anthony et al. 1993).

	1980	1981	1982	1983	1984	1985	1986[a]	1987[a]	Total
Oregon:									
No. occupied sites	5	7	7	7	7	10	9	8	60
% success	0	43	43	57	71	60	67	25	48
No. young produced	0	4	4	6	8	8	9	3	42
Young/occupied site	0.00	0.57	0.57	0.86	1.14	0.80	1.00	0.38	0.70
Washington:									
No. occupied sites	2	3	2	5	10	11	7	12	52
% success	0	33	50	20	20	9	86	27	29
No. young produced	0	1	2	1	3	2	8	4	21
Young/occupied site	0.00	0.33	1.00	0.20	0.30	0.18	1.14	0.36	0.40
Combined:									
No. occupied sites	7	10	9	12	17	21	16	20	112
% success	0	40	44	42	41	33	75	25	39
No. young produced	0	5	6	7	11	10	17	7	63
Young/occupied site	0.00	0.50	0.67	0.58	0.65	0.48	1.06	0.35	0.56

[a]Fresh eggs were collected from 6 and 3 sites in 1986 and 1987, respectively; these sites were not included in any of the above values.

Grubb (1976) found high numbers of active and successful nests among the bald eagles nesting in western Washington and along the Washington coastline. Of the 100 active nests she found, 63 percent were successful and young production averaged 1.37 per successful nests. The researcher stated that these numbers of successful nests were higher than had ever been recorded for Washington and may be due to the excellent nesting habitat along the marine coastline. However, the relatively low numbers of young per successful nest may reflect the persistent presence of pesticide residues.

Isaacs et al. (1983) documented the productivity of nesting bald eagles in Oregon from 1978-1982. Averaged breeding success was 61 percent, although there was substantial fluctuation from year to year. Young fledged per occupied site varied yearly as well, and the study-long

Table 5. Breeding territory data for Columbia River Recovery Zone, Oregon. 1989-1993 (Isaacs and Anthony, 1993).

				Occupied Territories With Known Outcome	Occupied Territories Successful				Young / Occupied Territory		Young / Successful Territory	
Recovery Zone	Year	Territories Surveyed	Occupied Territories		N	%	5-Yr. Ave %	Young	Year	5-Yr. Ave.	Year	5-Yr. Ave.
Columbia River	1989	12	12	12	4	33	37	4	0.33	0.47	1.00	1.29
	1990	11	11	11	5	45	35	6	0.55	0.44	1.20	1.25
	1991	15	15	14	11	79	42	15	1.07	0.52	1.36	1.24
	1992	19	19	18	11	61	51	15	0.83	0.64	1.36	1.26
	1993	22	22	22	11	50	55	16	0.73	0.73	1.45	1.33

mean was 0.92. Productivity values over the four years declined slightly but were within the range considered acceptable for maintaining population stability.

Anthony et al. (1993) found the breeding success of Columbia River estuary to be much lower than statewide averages of 61 or 62 percent. During the study period, 1980-1987, Columbia River estuary bald eagle breeding success rates averaged 39 percent. Productivity was 0.56 young per occupied site. A negative correlation found between breeding success and amount of eggshell thinning ($r = -0.52$) suggests that contaminants may play a role in the lowered productivity of the Columbia River estuary bald eagles. Garrett et al. (1988) observed that productivity on the Washington side were consistently lower than those for the Oregon side of the LCR but the reason for this is not known.

DISCUSSION AND ANALYSIS

Introduction

LCR bald eagles are contaminated by DDE, PCBs, and dioxins at levels higher than any other avian species analyzed in the region (USFWS 1993). The LCR bald eagle population also exhibited depressed productivity relative to other recovering eagle populations in North America, which most likely implicate population health and stability. However, scanty empirical evidence exists to demonstrate causal effects of contaminant presence on reduced productivity, breeding success, and population viability in the LCR. Data show a statistical and experimental relationships between organochlorine pesticides and eggshell thinning which would lead to reduced breeding success (Anthony et al. 1993; Lincer 1975) and the banning of DDT has resulted in recoveries for many bald eagle populations (Green 1985). But a causal relationship between the observed contaminant levels and hypothesized effects tend to be inferential (Gilbertson 1990). In the LCR the depressed breeding success may be attributable to an interaction of factors or to compounding variables, such as both habitat degradation and environmental contamination.

In order to determine the cause of the reduced reproductive success of this population and to determine the extent of the effect, other possible variables influencing productivity trends need to be discussed. These include habitat loss, habitat alteration, predation changes, prey base changes, or natural disaster. For LCR bald eagles, predation and natural disaster do not appear to be important factors. Therefore, important alternative hypotheses would be related to habitat or prey base changes. A difficulty in analyzing the variables of habitat effects, prey effects, and contaminant load effects over time is that these variables are interrelated and interdependent. In the Columbia River Basin, human induced change appears to be the prime mover in the systemic interactions of habitat alteration, contamination, and effects on bald eagle prey base. Certain specific sources of change, such as hydroelectric dams and river shipping, can be identified as important variables which ultimately affect bald eagle populations.

Existing Data and Data Gaps

LCR bald eagles have exhibited highly fluctuating productivity from year to year. In addition to the averages being lower than other regions in the country, the annual productivity have not achieved a consistent trend upwards (or downwards). These fluctuations could be indicative of a highly variable environment, prey base, or contaminant influences and as such could demonstrate the fragile stability of the eagles' ecosystem from year to year.

In the absence of baseline data on what a 'healthy' eagle population would look like or do over time, it is difficult to say if or how the Columbia River population deviates from health. Bald eagle populations in other regions that appear to be relatively free of contaminants have been documented as having fluctuating yearly productivity. Oscillating cycles may be natural for eagle populations (Hansen 1987). A less stable bald eagle populations may be regulated or influenced by qualitatively poor food resources (Dzus and Gerrard 1993). Little research currently exists that evaluates bald eagle populations and population dynamics over time to establish baseline data for healthy dynamics. Future research could focus on populations designated as stable, recovering, and threatened so that a range of dynamics can be considered. Such studies have been conducted to a limited extent in more pristine areas such as Alaska (Dzus and Gerrard 1993; Hansen 1987).

If the fluctuations or lower productivity are indications of the ill health of the LCR bald eagle population, then the question becomes why, more specifically why here and now. The two broad candidates for answers are habitat degradation or loss and sublethal effects of environmental contaminants, both stemming from human presence and activities. Most studies thus far have been concerned with the collection of essential baseline data on eagle ecology in the LCR. One of the most comprehensive recent studies was a report to the U.S. Army Corps of Engineers conducted by Garrett et al. 1988. The results of this report of the ecology of LCR bald eagles focused on habitat criteria, foraging ecology, and contaminant effects on the eagles and their prey (Watson et al. 1991; Anthony et al. 1993; Garrett et al. 1993). In addition, further research is needed on the effects of human disturbance on eagle behavior (McGarigal et al. 1991).

A gap currently existing in the research is the intersection of the variables of habitat degradation and contaminant effects. Examining how these variables interact may be useful in determining what accounts for the relatively poor bald eagle reproductive success in the LCR. Habitat change can affect the pathways of contamination and bioaccumulation of contaminants in bald eagles. It can do so through three mechanisms through: 1) altering bald eagle prey choices, prey availability and foraging behavior, 2) altering the physical dynamics of the river which change the food web structure, accumulation patterns, residency of contaminants in the food web, and accessibility of the contaminants to bald eagles, 3) identifiable, specific human industries which have not one but many effects implicating both habitat degradation and environmental contamination detrimental to bald eagle populations.

Human activities which illustrate this almost perfectly are shipping and dredging and hydroelectric dam construction.

Impact of Habitat Alteration on Prey Choice

Loss of habitat and habitat degradation are problems which in of themselves may impact bald eagle productivity. Garrett et al. (1988) documented that old growth habitat along the LCR essential for nesting and roosting is fast being eliminated by logging and human growth. Forty-one percent of the bald eagle nests they sampled were on private land, implying it is land upon which the continued maintenance of old-growth and other stands is uncertain and primarily determined by economics not ecology.

However, habitat degradation can affect bald eagle breeding success in other ways than loss of roost and nest sites. Recent studies have emphasized that bald eagle food resources appear to be the primary mechanism behind annual population densities and regulation rather than habitat criteria (Dzus and Gerrard 1993; Stalmaster and Prettner 1992; Hansen 1987). Habitat alteration which changes food supply may then be equally, if not more, important in bald eagle population stability than habitat alteration in of itself. Riparian areas are essential for bald eagle foraging (Garrett et al. 1993). Thomas (1983) surveyed changes in the Columbia River estuary over the past one hundred years. The total estuary is three-quarters of what it was in 1870. Almost 37,000 acres have been converted to diked floodplains, uplands, and non-estuarine wetlands. Change has also occurred within the estuary among estuary habitat types. The deep and medium depth water habitats are reduced by 16 percent, while the area of shallows and flats have increased 10 percent. Loss of deep and medium depth water is due primarily to shoaling, which has in turn increased the area of shallows and flats. Shallow and flat area has also been increased by erosion of marshes and swamps by currents and by filling and diking. The tidal swamp area has incurred the most dramatic alterations with a reduction of 77 percent of its area, primarily because of diking.

While bald eagles may benefit from an increase in shallows and flats, habitat change at rates faster than 'natural' tends in general to adversely affect larger biota, whose reproductive rates are too slow to adapt at a species level as fast as the habitat is changing. The deeper water is an important habitat for migrating salmonids and both deep and medium depth water is crucial for organisms which live in the water column, such as zooplankton and creatures which feed on it. The shallows and flats are utilized as a nursery for juvenile fish, are important for primary production, and are common waterfowl habitat. Tidal marshes and swamps are key habitats for primary production and detritus export. The changes in the estuary habitat types, then, have substantial impacts on the biota of the estuary, including juvenile salmonids. Loss of deep water may have impacted the composition of eagles' fish prey base, so that consumption of anadromous salmonids may be replaced by the consumption patterns currently observed where bottom-feeding largescale suckers form the largest part of the eagles' fish diet. This in turn could alter the pathways of bioaccummulation, as different fish accumulate toxins differently and feed on different toxic substances or prey. Bottom-feeders, for instance may be more susceptible to ingesting contaminants directly from sediments and from their benthic and infaunal prey, and resident fish, such as the largescale sucker, accumulate more contaminants from the contaminated waters of the Columbia River than anadromous salmonids (USGS 1992).

In addition to changes within fish species prey choice, habitat change may affect bald eagle choices between fish and avian prey. Bald eagles are opportunistic foragers, meaning they tend to eat what's around and what's ample. In wintering areas, such as the Klamath River Basin in Oregon and California, where fish are not available, large numbers of bald eagles consume waterfowl and small mammals (Frenzel and Anthony 1989). If an area which normally supported fish lost key fish habitat, bald eagles would, in theory, seek out waterfowl, mammals, or leave for greener rivers. Waterfowl have been shown to provide more energy and biomass to wintering bald eagles, even though fish may still be consumed in higher quantities (Stalmaster and Plettner 1992). Stalmaster and Plettner (1992) observed that the presence of hydroelectric dams on a river help create ice-free water, this attracting waterfowl and maiming fish, making both of these prey items more available to bald eagles in winter. This benefit effect could potentially be offset by increased consumption of contaminated prey or upon higher proportional consumption of more contaminated prey, such as other piscivorous birds.

Habitat changes affecting prey choice could be an important variable in contamination effects. PCBs and DDE may accumulate differentially in Columbia River estuary bald eagles through different prey sources (Anthony et al. 1993). In the Klamath Basin eagles, wintering eagles which fed on waterfowl and small mammals actually had lower levels of DDE and PCBs than resident eagles. The consumption of mammalian, terrestrial based prey may then result in lower levels of organochlorine residues in the bald eagles than aquatic prey in some ecosystems. In this study, the high consumption of small mammals may have offset the trophic compounding effects on contaminants thought to be more typical of piscivorous bird consumption.

LCR bald eagles accumulate high contaminant levels because they are at the top of the food chain, consuming fish-eating birds in addition to fish, placing them one trophic level closer to higher contamination levels than other piscivorous birds. LCR eagles are year-long residents and thus gain the opportunity to accumulate toxins year-round from contaminated prey and water, as opposed to the more transient migratory birds in the lower river and estuary. Lastly, eagles are relatively long-lived and reproduce for up to thirty years, so overtime the reproducing adults can accumulate toxins leading to sublethal and lethal effects in embryos. This analysis would indicate that a higher proportion of waterfowl in bald eagle diets would lead to higher contaminant levels and that wintering diets and source of wintering waterfowl as prey are equally critical (Schuler 1994, pers. comm.).

The prevailing assumption is that prey are the source of contamination for bald eagles on the LCR. An analyses of the variables accounting for different contaminant pathways even within the category of prey contamination can not only strengthen the inferential assumption of prey as the source of contamination but can elucidate the details and mechanisms of this source of contamination. Some studies have suggested that for some bald eagle populations migrant waterfowl prey from Central and South America can be a source of pesticide contamination (Frenzel and Anthony 1989). However, in the LCR, it appears that prey contamination is a local issue, as local as the contaminated waters and soil of the river itself (Garrett et al. 1988). Research on deferential contamination resulting from prey choice has been conducted for Great Lakes bald eagles but not for LCR bald eagles (Anthony 1994, pers. comm.; Kozie and Anderson 1991).

Higher fish consumption of LCR bald eagles can be attributed to the presence of tidal flats and shallows in the Columbia River estuary (Watson et al. 1991). While shallows and tidal flats do not currently appear to be the habitats most threatened by human induced change, their future alteration is a possibility. Tidal flats and shallow water increase fish availability, both live and dead. The proportion of fish consumed in the Columbia River estuary is higher than in other areas that have been studied such as the Chesapeake Bay, southern Louisiana, and southeast Alaska. The increase in these habitats over time may have artificially increased the amount of fish consumed, or, conversely their potential decrease may make the eagles more reliant on waterfowl. Watson et al. (1991) state that a decrease in the area of tidal flats could increase competition for foraging and flight distances between nesting and feeding areas. Another possible effect is deferential consumption of prey species based on availability and location.

Temporary human habitat disturbances, such as recreational boating, can affect bald eagle foraging strategies and flight distances on the Columbia River estuary as well (McGarigal et al. 1991). Temporary disturbance would have a proportionally greater effect on temporary eagle activities such as foraging than on more long-term behaviors such as nesting site choice. Much of the Oregon, and some of the Washington, shore is developed and the Columbia River estuary is a major boat thoroughfare for commercial vessels on their way to Portland and other upriver ports. Fishing boats as well occupy open water during gill-net seasons, and during spring and summer -- also coincidentally bald eagle breeding season -- recreational boating activities along the river and estuary outnumber commercial ones.

McGarigal et al. (1991) noted that although human use of the area under study in the Columbia River estuary was high and recreational boating activities outnumbered all other human activities, direct disturbance of eagles rarely occurred and most nest sites were relatively inaccessible to humans. Instead, bald eagles altered their foraging habits in response to temporary disturbances. Eagle foraging activity did not appear to be modified in response to daily fluctuations in human activity but eagle search and capture time within 400 m of high-use areas was slightly greater during increased human activity on weekends, suggesting that human activity might have altered eagle hunting efficiency, possibly because the eagles were preoccupied with avoiding people. Eagle pairs subjected to relatively high boating activity concentrated their foraging more in the early morning than pairs subjected to relatively low boating activity. Eagles also modified their spatial use patterns of the estuary in response to human disturbance. On average, eagles avoided an area within 300-400 m of the boat situated in the center of a previously much used foraging area. This avoidance also reduced eagle foraging efficiency, resulting in more time spent actively hunting.

Alteration of temporal and spatial foraging habits on the part of the bald eagles in response to human activity suggests that this may be a factor in prey choice. Since eagles are opportunistic, they will eat what is available. If they are forced out of their normal foraging areas into more terrestrial ones or areas without stranded fish but with stricken waterfowl or other avian prey, this could potentially alter the kinds, levels, and pathways of contaminants in bald eagles most affected by human disturbance. Since boating activity has been observed to increase as the breeding season progresses (McGarigal et al. 1991), the foraging strategies of adults in response to human activity will affect nestling diets as well. Nestlings in the Columbia River estuary display evidence of organochlorine contamination (Anthony et al. 1993). In addition, Columbia River nestlings diets contain high proportions of juvenile chinook salmon (Anthony et al. 1993), probably because this species is abundant and easy to capture during nestling growth in the spring. If foraging patterns of adults tend to stray away from areas where juvenile chinook are easily captured, this could affect the diets of the new young as well.

Other considerations relative to foraging site alteration are fish prey decomposition rates, intraspecific competition for food, and interspecific competition. Stalmaster and Gessaman (1984) found that bald eagles preferentially fed on gravel-bar stranded salmon carcasses as opposed to waterlogged ones, since salmon carcasses decompose faster in water. They also found that for subadults, intraspecific competition for food in winter may affect survival rates. Dzus and Gerrard (1993) hypothesized that low numbers of nonbreeders and immatures in a Saskatchewan study site indicated a food regulated population where winter food resources did not support the 'extra' birds. The mere presence of humans may disturb spatial partitioning of foraging areas, increasing competition and the foraging distances. The implications of this alteration for contaminant effects could be change in prey species choice.

Impact of Habitat Alteration on Food Web Dynamics and Contaminant Presence

Habitat alteration changes food web structure through impacting community dynamics and trophic interactions. Predators experience the effects of such change through a ripple effect. The problem of quantifying such change and its results is that such things are often more difficult to empirically measure than funding or time allows. However, microhabitat studies may be able to elucidate some of the complexity. Broad inventories of habitat change on the Columbia River and its possible effects have been conducted (Weitcamp 1993; Sherwood et al. 1990; Simenstad et al. 1990b). Such studies are important but they can't help but neglect trophic complexity. Further study might be beneficial which takes, and ultimately exhausts, as its subject microhabitat dynamics. A microscopic focus on study area or habitat type rather than on species may yield insights onto species and community interactions. An example of such a study can be found in the Holton et al. (1984) investigation of benthic infaunal community dynamics in the Columbia River estuary which also provided a wealth of information about a single species (*Corophium salmonis*) which may not have been made available if the focus had been simply the dynamics of that single species throughout the entire estuary.

LCR bald eagles prey mostly on freshwater catostomids and cyprinids, anadromous salmonids and cluepids, waterfowl, seabirds, and small quantities of small mammals (Watson et al. 1991). Avian prey includes mallards, western grebes, cormorants, and gulls. These all in turn prey on lower trophic levels which include benthic invertebrates such as *Corophium salmonis* (the catostomid fish, grebes, and other waterfowl), zooplankton (juvenile salmonids, for instance), and algae and detritus. The invertebrate prey in turn eat phytoplankton, zooplankton, and other detritus.

Habitat change which alters river and estuary physical dynamics will change the microbiotic composition which will change the diets and foraging strategies of bald eagle prey. Dzus and Gerrard (1993) observed two bald eagle populations in Saskatchewan, one stable, one less stable and with lower densities. The habitats of both populations were more or less equivalent but their prey bases differed. While both populations fed predominantly on fish, the fish supporting the more stable and dense population were larger and two species of six sampled were more numerous there. The researchers hypothesized that this could be explained by the higher standing crops of plankton and benthic fauna at the more stable site and cited their study as an example of the extenuation of differences in productivity at the lower end of the food web to the top (i.e. bald eagles). In theory, then, contaminant effects on plankton and benthic fauna in the LCR may be impacting bald eagle breeding success.

Contaminants occur differentially in different microhabitats. Even the three broad regions of the LCR, freshwater, estuarine, and seawater influenced, have vastly different dynamics which may impact contaminant loads, contaminant accumulation in biota, contaminant bioavailability, and contaminant presence in the water column or sediments. While extensive cataloguing has begun on the locations and extent of heavy metals, pesticides, dioxins, and other contaminants (Tetra Tech 1992), no studies have yet addressed the myriad variables affecting such contaminant dynamics. In part, this is because such a study appears very complex. However, the selection of microhabitats in which to carry out detailed study over time might alleviate the prospective headache of quantifying all these interactions. For example, the estuarine mixing zone has received much attention in the literature as a region of extreme dynamism (Simenstad et al. 1990b; Bottom and Jones 1984). It might be profitable to study the trophic interactions along one stretch of this zone over time and with relation to contaminant dynamics and processes.

Habitat alteration which changes river flow rates is one potential source of change in contaminant dynamics. Upriver damming has decreased flow rates in the LCR, as has irrigation and possibly jetty construction (Sherwood et al. 1990a; Simenstad et al. 1990b). Zooplankton are highly affected by river transport rates (Simenstad et al. a; see Alternative Trophic Species results and discussion), as are phytoplankton. River flow rates also affect plankton dynamics and life histories in the estuarine mixing zone. Decreased river flow has also reduced abundances of juvenile salmon flushed to the sea in the spring (Northwest Power Planning Council 1992).

Implications for contamination pathways are complex. Freshwater phytoplankton flushed from upriver hit the zone of salinity intrusion and die off at large rates, becoming detritus and sinking through the water column. Contaminants bound in the plankton would theoretically fall with them. Fish and zooplanktors feeding at different levels in the water column would be differentially affected by the presence of contaminants in varied sites in the water column. Reduced flow rates could alter the mortality rates of phytoplankton as they encounter high salinity. Reduced flow rates could also alter processes of detrital sinking and/or estuarine water column mixing and could alter processes of contaminant deposition. However, dams may also benefit wintering bald eagles through thawing river ice, attracting waterfowl, and killing fish to make them available for consumption (Stalmaster and Plettner 1992). But again, dams are also an instigator of altered physical dynamics, food webs, and diets that may affect pathways of contaminant accumulation.

Zooplankton, endemic to estuarine conditions, such as *Eurytemora affinis*, appear to have specially adapted behavioral mechanisms to retain themselves in the estuarine mixing zone (Hough and Naylor 1992; Hough and Naylor 1991). These mechanisms are not fully understood yet but involve tidal approximation, salinity gauging, and migration through the water column on the part of the organisms. Changes in flow rates seasonally affects salinity in the LCR. Changes in flow rates historically probably do so as well and if the changes occurred faster than the organisms could adapt to then the results could be higher mortalities, different grazing patterns, increased benthic zooplankton flushing or inability to entrain themselves in the mixing zone, or other hypothesized calamities. These effects would of course change food web dynamics and the processes of contaminant entrance and accumulation in the food web. Such effects may also impact benthic organisms proportionally greater in terms of contaminant implications, because sediments tend to retain contaminants longer than flowing water, especially heavy metals and pesticides such as DDT. All of these processes ultimately impact bald eagle prey availability and levels of contaminants in their prey.

Interactions of Habitat Alteration and Contamination Resulting from Human Economic Activities

Shipping and dredging is one example of a human activity in the LCR which affects both habitat change and contaminant loads in the river basin. Dredging, for the purpose of shipping to upriver ports such as Portland, OR and Vancouver, WA, removes bottom sediments from one location and puts them in another, changing the habitat dynamics at both sites. Much of the habitat change in the Columbia River estuary over the past one hundred years has been attributed to dredging (Thomas 1983). In addition, dredging redistributes chemicals and contaminants from deep interment to surface levels and is probably one mechanism of maintaining DDE in the ecosystem.

Compounding the contamination effects of dredging is the presence of the boats that the soils are dredged for. These boats are often coated by anti-fouling biocides, such as tribulytin, which are highly toxic to invertebrates such as Eurytemora affinis (Bushong et al. 1988). Butylins have been found in excessive levels in the LCR (Tetra Tech 1992), but their effects on aquatic biota are still being studied. High toxicities may imply low potential for bioaccumulation (since the organisms die rather than pass on the toxins). Shipping and dredging, however, are still potential threats to the health of the LCR organisms. Dredging, and its 'effect' of shipping,. is one example of how habitat change and contamination are far from separate issues.

Another example is hydroelectric power. Dams along the Columbia River have reduced flow rates substantially over the last century and have drastically affected migrating juvenile salmon (Northwest Power Planning Council 1992). Juvenile salmon are important prey for LCR bald eagles, especially during breeding season (Watson et al. 1993). In addition, dams release toxic substances into the watershed. Bonneville Dam added large quantities of PCBs to the Columbia River following an accident in 1986. While there is potential for dams to benefit bald eagles through the greater prey availability they may provide in winter (Stalmaster and Plettner 1992), even this 'beneficial' habitat alteration merits scrutiny since changing the prey base may change contamination patterns.

Understanding such contaminant, industry, and habitat change interactions are crucial for effective mitigation of the problems affecting bald eagle reproductive success. Sediment bound contaminants would have different food web accumulation patterns than contaminants in the water column, being present in the water column either by binding to or accumulating in the plankton or other organic detritus. For instance, in a study in the Somass River estuary near Vancouver, Canada and adjacent to a pulp and paper mill, researchers found high levels of dioxins in waterfowl and their prey (Vermeer et al. 1993). The prey base of the most contaminated birds was dominated by *Corophium* spp., one of the few infaunal taxa to continue to thrive in the pulp mill waste contaminated waters of the estuary. River flow rates are an important factor in contaminant deposition patterns. Habitat changes affecting flow rates could impact contaminant deposition and subsequent incorporation by *Corophium*, which would then contaminate the waterfowl sampled.

Western grebes were one of the most contaminated birds in the above study. Grebes are also one of the waterfowl eaten by bald eagles in the Columbia River estuary (Watson et al. 1991). One might speculate that Columbia River grebes might be exposed to similar dioxin concentrations, due to the abundance of pulp and paper mills in the region. Dioxin load is abnormally high in the LCR and appears to be a factor in the reduced productivity of the Columbia River estuary bald eagles (USFWS 1994a). This example of the influence of pulp and paper mill dioxin effluent on the trophic interactions in the Somass River estuary could be extrapolated to the Columbia River estuary and up the food chain to bald eagles. The efflux of contaminants and their subsequent uptake may be considerably influenced by estuarine and riverine physical dynamics, such as flow rates.

Migratory double-crested cormorants were another piscivorous bird identified as highly contaminated in the LCR; they were second only to bald eagles in TCDD contamination and also contained high levels of PCBs in their embryos (USFWS 1993). Cormorants are also form a small percentage of bald eagle diets in the Columbia River estuary (Watson et al. 1991). The interactions of species such as cormorants, grebes, and bald eagles, then, is more complex than the observation that they all eat fish. Grebes consume *Corophium* which is a demonstrated accumulator of dioxins, even in contaminated waters with low species diversity (Vermeer et al. 1993). Cormorants have been identified as a useful indicator species for the health of the LCR (USFWS 1994b), however, they also are part of the pathway to contamination for LCR bald eagles. Bald eagles are at risk of contamination because of three compounding factors: their consumption of waterfowl in addition to fish, their year-long residency along the river, and their long reproductive lives. Additional variables may exist to add to the multiplicative effects of these, but they have not been identified. More study needs to be done on such multiplicative interactions as the ones hypothesized above and on interactions not yet identified.

Ongoing and proposed bald eagle studies within the LCR basin

-USFWS: OR/WA -- Environmental contaminants in fish-eating birds from the Columbia and Willamette Rivers

-USFWS: OR -- Impacts of organic contaminants on double-crested cormorants nesting in Lewis and Clark National Wildlife Refuge

-USFWS: OR/WA -- Organochlorine contaminants in aquatic resources from Columbia River estuary

-USFWS: OR/WA: Environmental contaminants in bald eagles nesting along the Columbia River

-Oregon State Cooperative Research Unit productivity surveys, wintering bald eagle counts -- to be eventually incorporated into a major report on bald eagle productivity and breeding biology in the LCR.

Recommendations

-- trophic web study in a microhabitat rather than the entire estuary or river, potentially along the estuarine mixing zone

--studies on factors influencing breeding and wintering eagle prey choices: e.g. energy gained v. energy expended, availability of prey v. abundance, effects of disturbance on prey choice

-- contaminant studies combining laboratory experiments and field observations: e.g. comparative avian physiology and metabolism of varied contaminants and how this changes under varied stresses, e.g. cold, hunger, migration

--studies addressing degrees of magnification of contaminants through food web and variables affecting this

--studies on effects of human industries that are here to stay, e.g. shipping or dams in order to minimize detrimental effects and maximize potential benefits; more focus on physiological effects of various contaminants, including dioxins and PCBs, to get baseline comparative data

EFFECTS OF CONTAMINANTS ON THE MINK (*Mustela vison*) ALONG THE LOWER COLUMBIA RIVER

INTRODUCTION

Mink (*Mustela vison*) are aquatic furbearers distributed across most of North America and all of Canada, south of the tree line. They are carnivores at the top of the food chain and can be exposed to and bioaccumulate environmental contaminants from consumption of prey species. Mink in the Columbia River prey on a variety of species including carp, sucker, largemouth and smallmouth bass, crappie, sculpin, crayfish, birds, small mammals, and a small amount of reptiles and insects.

Studies conducted on species in the Columbia River, particularly studies on raptors, have provided evidence that the Columbia River itself is the source of persistent contamination of its ecosystem. Although eagle and osprey nesting populations have been improving, organochlorine contaminants remain a problem for populations in the lower Columbia River (LCR) (Henny and Anthony 1989). Compared to other populations, DDE and PCB residues have been elevated in the LCR mink population (Henny et al. 1981).

The percentage of Oregon's mink harvest in the two counties bordering the Columbia River decreased from 15.4 percent from 1949-1952 to 9.1 percent in 1973-1976 (Henny et al. 1981). Mink were identified as an indicator of the health of the LCR ecosystem. Mink is a resident carnivore in most non-urban areas of the Columbia River watershed. Essentially the health status of a indicator species; population trends, associated habitat changes and morbidity departing from normal health conditions, growth and behavior, is assumed to be representative of the health status of the entire ecosystem.

A pilot study of mink and river otter in Oregon in 1978-1979 found that PCBs were more frequently encountered in LCR mink than in mink from other areas along, or near the Columbia River (Henny et al. 1981). Mink are among the most sensitive species to toxic effects of TCDD and related compounds, such as PCBs.

Mink have only a 28 day LD_{50} of 4.2 ug/kg TCDD (Hochstein et al. 1988). Levels of TCDD reported in fish collected in the Columbia River were higher than 5 parts per trillion, the level reported by the Canadian Wildlife Service as possibly causing reproductive impairment in mink. This reported level of reproductive impairment in mink does not take into account the cumulative effects of PCBs and other contaminants known to be in the food chain of the Columbia River. Therefore, the mink population in the Columbia River is likely to be at serious risk of eradication caused by the singular and/or compounded effects of environmental contaminants.

The purpose of this study is to compile and summarize available published and unpublished reports related to mink, the LCR and its estuary, and effects of environmental contaminants. Specific objectives are to evaluate existing literature categorically on the basis of data indicating species responses to contaminant levels, habitat alteration, life history, population dynamics and trends, and diet, in the LCR; to analyze and synthesize these data on the basis of study methodology; and to identify weaknesses in the data base.

FINDINGS

Life History

Mink are generally solitary, unsociable animals. As the breeding season approaches general physical activity of mink increases. Males travel widely in search of females for mating. Males and females only associate for brief periods during March or early April (Banfield 1974). Female mink remain in estrus throughout the breeding season, having receptive periods at 7-10 day intervals. Ovulation is induced by the stimulation of mating. Ovulation occurs 33 to 72 hours after coition. If an interval between two matings of a female mink exceeds six days than a second ovulation capable of producing fertilizable ova can occur. Gestation varies from 38 to 85 days. Following ovulation fertilized ova develop to the blastula stage, then enter a quiescent state of variable duration before uterine implantation and further development take place. The duration of the delayed implantation is less in females mated late in the breeding season and appears to be influenced by increasing photoperiod. Litters are born 28 to 30 days after implantation, usually in late April or early May. The average litter size is 4-5 (2-10). Shorter gestation periods may result in larger litters. Newborns weigh 8-10 g and grow rapidly, attaining 40 percent of their body weight and 60 percent of their length by 7 weeks. Young mink's eyes open around three weeks of age, after which they begin to eat solid food. Litters usually break up in early fall. Mink reach sexual maturity during the first year. The life span of a wild mink is about 3 years. The average length of an adult mink is 447-720 mm or 20-30 inches. Males weigh about 3 pounds, twice the weight of females (Linscombe et al. (1982).

No data concerning the period of birth for mink in the Columbia River were obtained by Tabor et al. (1980) and no data pertaining to the breeding period or season of birth specifically for Oregon or Washington were found in the literature search done by Tabor et al. (1980). Marshall (1936) determined mink in Michigan mate in the first two weeks of March, gestation is 52 days and birth occurs in late April to early May. Tabor et al. (1980) assumed mink in the lower Columbia River also give birth in April and May.

Behavior and Movement

Male mink are mainly nocturnal during all seasons, with the level of activity increasing with the length of the night and decreasing temperature. Mink are inactive during periods of low temperature following snow fall. Male activity peaks with peaks in prey activity. The most obvious seasonal movements of males is associated with the mating season. Female have very low activity rates during pregnancy, but increased while caring for litters and is primarily diurnal. Home ranges of adult and juvenile males are similar. Home ranges of female mink are smaller than that of males, but are used more intensely. Because mink are territorial, the greatest movement in mink populations is associated with dispersal of juveniles during summer and fall. However, juveniles are usually more restricted in their travels than adult males and adult females are most sedentary. (Linscombe et al. 1982). Movements between dens are made at night and usually occur in, or along, linear habitat features, such as lake shores, river banks, stream courses, or hedge-rows (Birks and Linn 1982). In Quebec the majority of mink activity was recorded within 3 m of the edges of streams (Allen 1986).

Habitat Requirements

The distribution of mink includes all of the United states, except Arizona, and all of Canada south of the tree line, except Anticosti Island and the Queen Charlotte islands. Mink inhabit many types of wetland areas, including banks of rivers, streams, lakes, ditches, swamps, marshes, and back water areas (Banfield 1974; Mason and MacDonald 1983). In the United States, according to fur harvest records, the most mink pelts are produced in Louisiana, Minnesota, and Wisconsin. A study by Burns (1964) in the tundra of the Yukon-Kushokwim Delta found the highest density of mink occurred in the low, swampy terrain surrounding the largest body of water in the area, and in the extensively interconnected water system with large concentrations of blackfish (*Dallia pectoralis*) and whitefish (*Coregonus* spp). In Louisiana, Arthur (1931) found mink density to be the highest in coastal marshes, followed by deep cypress-tupelo swamp and then backwater bottomland hardwood areas in central and northern portions of the state.

Mink are most often found in close association with wetlands and brush or wooded cover, with abundant woody debris for cover and shallow pools for foraging, immediately adjacent to streams and rivers. Mink usually depend on aquatic prey for a large portion of the year. Transient use of upland cover may occur, particularly during the fall and winter when terrestrial prey becomes increasingly more important. Sufficient vegetative cover interspersed with, or immediately adjacent to water is assumed to provide an adequate source of terrestrial prey species to supplement the aquatic portion of mink diet. When mink move upland in search of food, the edge of habitats are used most because they provide adequate cover and small animals can easily be found (Allen 1986). Mink generally avoid exposed or open areas, hence habitats associated with small streams are preferred to those associated with large, broad rivers (Allen 1986). Decreased diversity in shoreline configuration, elimination of aquatic vegetation, and decreased abundance and diversity of riparian vegetation caused by channelization reduces habitat quality, prey availability and utilization by mink. As an example, the decreased complexity of shoreline habitats along Ontario lake shores, caused by the construction of cottages, is believed to have reduced the amount of shelter available to crayfish resulting in decreased availability of prey for mink (Allen 1986). According to Allen (1986) adequate foraging and cover habitats can be described by the same set of habitat characteristics. Allen (1986) also assumes habitat requirements for den selection are identical to requirements for cover.

Tabor (1976) studied 292 river miles along the Columbia River (beginning at the seaward end of the river mouth jetties to McNary Dam) and observed mink sign in tidal marsh and beachgrass between river mile (RM) 0 and 12, and tidal willow and sitka spruce between RM 12 and 79. Two signs of mink were observed between RM 125 and 133. The greatest abundance of mink (sign noted at 10 locations) appeared to be between RM 152-184.

Between RM 152 and 184, rip-rap, especially those adjacent to embayments appeared to be utilized consistently by mink. Density appeared very low in the riparian vegetation associated with the Deschutes river, especially in the upstream portion. Mink within Tabor's study area along the Columbia river occurred at low densities (Table 1).

Den Sites

The distribution of mink in some cases is related to habitat preference, however the presence of den sites can also be important. Mink dens are usually located close to water and may be in temporary or permanent use. Birks and Linn (1982) observed nearly all dens within 10 meters of the water, most dens observed in this study were within 2 meters of the water. Den sites in Idaho were 5 to 100 meters from water and never farther than 200 meters (Melquist et al. 1981). The tendency of mink to utilize existing den-sites rather than excavating their own and to concentrate foraging activity close to dens, suggests the presence of potential den sites is an important requirement for suitable habitat (Birks and Linn 1982). Therefore the absence of dry den sites may limit the use of some habitats by mink. Seasonal use of dens varies, possibility in response to seasonal changes in prey availability. Birks and Linn (1982) found the majority of den stays are less than a day, dens occupied for longer periods usually contained food surpluses. Den sites of mink are commonly within cavities beneath tree roots at the water's edge, within cavities or piles of rock above the water line, in areas with a large number of deadfalls and stumps, or in fallen branches, brush, and other debris (Allen 1986). Active dens are usually not located on heavily grazed shorelines. Tabor (1976) and Tabor et al. (1980) conducted the only studies on mink distribution and habitat use in the Columbia River. In 1980 only three mink dens were found in shoreline composed of large rocks, the actual vertical and horizontal extent of dens could not be determined.

Foraging Habitats

Habitat quality influences the distribution, density, and reliability of prey, which directly affect mink population and density (Allen 1986). For example, mink populations in Louisiana are believed to cycle with peaks in crayfish and muskrat populations (Allen 1986; Linscombe et al. 1982) and Erlinge (1972) found habitat selection by mink in Sweden indicated changes in summer and winter habitat preference relates to changes in food supply. Shallow water and low flow rates contribute to minks' foraging effectiveness in aquatic habitats therefore habitats with these characteristics are preferred by mink. Melquist et al. (1981) observed log jams are used as shelter for aquatic prey as well as shelter for mink therefore provide excellent foraging cover.

Home Range

Home ranges tend to approximate the shape of the body of water along which a mink lives. Differences in the intensity of use of portions of home ranges are often in response to changes in the distribution of food and foraging habitats. The great variety of habitats

Table 1. Observations of mink/river otter sign and habitat type[1].

RM (River Mile)	Habitat Types	% of Total Miles of Shoreline	No. of Observations	
			Otter	Mink
0-12 (to mouth of Youngs River)	Tidal marsh, beachgrass, sand, grassland, residential, alder, pasture		1	sign observed
12-79	Tidal marsh, shrub willow, willow/cottonwood, large trees, grassland, cottonwood/large trees, willow/large trees		3	sign observed
79-145 (to Bonneville Dam)	Cottonwood/large trees, grassland, willow/cottonwood/large trees, willow, cottonwood/ash, large trees/willow		10 (between 125-133)	2 (between 125-133)
145-192 (Bonneville Pool)	Pasture, grassland, industrial, embayments, oak/ponderosa pine, residential, oak, maple/douglas fir	rock riprap (34%), grassland undifferentiated (11%), willow/large trees (7%), Industrial (5%), oak/ponderosa (5%), rock (5%), willow/small trees (4%), oak (3%)	6 (between 152-181)	10 (between 152-184)
192-215.6 (The Dalles Pool)	Grassland, rock/grassland, rabbitbrush, pasture, rabbitbrush/grassland, embayment, herbaceous types, ag. production, shrub willow	rock rip rap (40%) gravel (14%), rock (9%), sand (8%) shrub willow (4%)	NA	NA
215.6-292 (John Day Pool)	Rabbitbrush, rabbitbrush/sagebrush, field crops, rock cliff/grassland, grassland, grassland/rabbitbrush, sagebrush, bitterbrush	sand (39%), rock rip rap (21%), gravel (12%)	8	sign observed

[1]all data derived from Tabor 1976.

utilized by mink indicate definite adaptability of mink. Birks and Linn (1982) observed the mean home range length for male minks ranged from 1.9-2.9 km, for females length ranged from 1.46-2.87 km. Female mink usually have the smallest and most defined home ranges, while males tend to use a more extensive, less defined home range. Females may be restricted to home ranges within riparian habitats, while males exploit upland areas as well (Allen 1986). This difference in habitat within home ranges of males and females is likely related to differences in size and feeding habits. Females are smaller than males and are able to subsist on smaller aquatic prey, the larger male can easily prey on relatively larger small mammals. Studies in North Dakota observed intrasexual home range overlap was scarce except during the 2 to 3 week breeding season in April (Allen 1986).

Diet

Food habits of mink have been studied by Errington (1943, 1954), Sealander (1943), Wilson (1954), Korschgen (1958), Waller (1962), Erlinge (1969), and Eberhardt (1973). Mink are well adapted for hunting both aquatic and terrestrial prey. Common food items include mammals, fishes, birds, amphibians, crustaceans, insects, and reptiles. No one food item seems to be consistently more important in the mink diet The importance of each prey item varies with the location and the time of year. Mink are opportunistic predators and therefore respond to highly available prey. This behavior results in a diet that is extremely variable by season and location. For this reason only studies addressing mink food habits within the Columbia River are addressed in the results of this report. Refer to Allen (1986) for studies that address food habits in other regions within the distribution of mink.

Seasonal foods of mink along the Columbia River, as determined by Tabor et al. (1980) include; fish, crayfish, waterfowl, and various small mammals. Crayfish occurred most frequently in spring and summer. Of the identifiable species of fish, carp and sculpin were the most frequently occurring species in spring, sculpin were the most frequently occurring fish species in the summer, sucker and large/smallmouth bass were the most frequently fish occurring species in the fall and sucker and crappie/sunfish were the most frequently occurring fish species in the winter. Unidentified species of birds occurred most frequently in the fall and winter; consumption of mammals was inversely proportional to bird consumption and usually less frequent than fish and/or crayfish consumption. Reptiles and insects also occurred, but in very small percent frequencies (Table 2).

Contaminants

The effects of contaminants on mink have been relatively well studied. There is a stock of mink available for experiments, without interfering with the wild mink population. Also, the mink farming industry is concerned that farm raised mink can be exposed to environmental contaminants from feed made from the by- products of cattle and grain.

Table 2. Seasonal foods of mink along the Columbia River. Data are presented in percent frequency of occurrence (Tabor et al. 1980).

Foods	Season			
	Spring (March-May)	Summer (June-Aug)	Fall (Sept-Nov)	Winter (Dec-Feb)
FISH				
Carp	13			
Sucker		6	7	9
Largemouth/smallmouth bass			7	
Crappie				9
Sculpin	13	9		7
Other/unidentified fish spp.	20	19		12
CRAYFISH	44	53	48	33
BIRDS	41	5	68	60
MAMMALS	13	35	20	18
Small rodents	14	34	20	15
Muskrat/rabbit/other		2		2

However, relying on experimental data to determine the effects of environmental contaminants on wild mink neglects the compounded effects of contaminants on mink by natural stresses. Simultaneous exposure to other chemicals, food restriction, temperature change, or disease can exacerbate the effects of exposure to a toxic chemical (Wren 1991). The potential toxicity of environmental chemicals is usually tested on healthy experimental animals that have good access to unlimited food and water. Dosages of environmental contaminants administered to experimental mink are usually lethal and the subjects are usually sacrificed at the end of the experiment for tissue analysis. Therefore, any sublethal doses that are administered are not studied for long term effects.

Polychlorinated biphenyls (PCBs) and dichloro-diphenyl-trichloroethane (DDT) and 2,3,7,8-tetrachlorodibenzon-p-dioxin (TCDD) were identified as persistent contaminants in the lower Columbia River and selected as a focus in the survey of studies on the toxicological effects of environmental contaminants on the mink population in the LCR. PCBs and DDT are similar in their mode of transport in the environment. They have low biodegradation rates and tend to concentrate in fatty tissues of animals as they move through the food web.

Studies testing the biophysiological response of mink to DDT and TCDD have indicated that mink are relatively tolerant to DDT, but is one of the most sensitive species to the toxicological effects of TCDD. Aulerich and Ringer (1970) isolated levels as high as 771 ppm of total DDT from the fat of mink fed 100 ppm DDT plus 50 ppm DDD. These levels indicate the ability of mink to store and concentrate these residues, therefore mink have a high tolerance to DDT, DDD, and DDE. Feeding dieldrin (a compound of DDT) in concentrations of 2.5 ppm was found to be toxic to adult mink. However the long term effects of dieldrin on reproduction are not known. Hochstein et al. (1988) reported no deaths in mink fed 2.5 ug/kg of TCDD, however mink mortalities in the group fed 5.0 ug/kg and 7.5 ug/kg died by the 17th day post-exposure. Therefore mink were determined to be extremely sensitive to the toxicological effects of TCDD and related compounds, having a 28-day LD_{50} (lethal dose of 50 percent of the sample) value of 4.2 ug/kg body weight. The exact mechanism of TCDD toxicity is unclear.

Aulerich and Ringer (1977) found the degree of PCB toxicity effects on mink reproduction to be directly proportional to the total intake of the compound. From experiments comparing the chronic toxicity of Aroclors (chlorinated PCBs) 1016, 1221, 1242, and 1254, only Aroclor 1254 was found to have detrimental effect on mink reproduction, however different Aroclors at varying levels may have adverse effects on ovulation, implantation, and/or gestation. Retention of higher chlorinated PCBs might account for their greater biological effectiveness and may explain the reproductive impairment in mink caused by Aroclor 1254 but not the other lesser chlorinated PCBs. Bleavins et al. (1980) found Aroclor 1016 produced 100 percent mortality in all mink fed diets with 20 ppm and 40 ppm. At 10 ppm 66.7 percent mortality occurred in males and females. At 5 ppm only females died. Aroclor 1242 caused complete reproductive failure in animals that survived long enough to be bred. Aroclor 1016 reduced but did not eliminate reproductive capacity in surviving treated mink. These results indicate that mink are better able to tolerate Aroclor 1016 than Aroclor 1242. The ability of mink to metabolize and excrete PCBs may be dependent on the percentage of the higher chlorinated biphenyls, as well as the overall percentage of chlorination of the PCB mixture. In Platonow and Karstad's study (1973) no live kits were produced and all adult mink died during a 105 day period of consuming daily rations of 3.57 ppm of PCB. Consuming daily rations of 0.64 ppm PCB, one of 12 mink produced 3 kits, all kits died during the first day after birth. Jensen et al. (1977) demonstrated that mink are more sensitive to PCBs than to DDT. Sixty-six daily doses of 11 mg each, did not cause any significant decrease in the number of whelps mink, while 22 such doses of PCB did. At a dose of 11 ppm PCB in the feed, corresponding to a total dose of 218 mg, no whelps were born. The exact mechanism by which PCBs effect the reproductive performance of mammals is not well understood. Except for reduced growth rate and hemorrhagic gastric ulcers, the clinical signs associated with PCB poisoning in mink appear to be non-specific. Platonow and Karstad (1973) and Aulerich and Ringer (1977) observed clinical signs of PCB toxicity as reduced growth, anorexia, bloody stools, fatty infiltration, degeneration of the liver and kidneys and hemorrhagic gastric ulcers. A summary of these studies is found in Table 3.

One study on the levels of contaminants in mink inhabiting the Columbia River was found in this literature search. Henny et al. (1981) conducted a pilot study on concentrations of contaminants in mink and river otter from Oregon. Tissue concentrations were generally higher in mink collected from the lower Columbia River than mink collected from the Blue and Wallowa Mountains and the Klamath Basin. One of five mink from coastal streams contained detectable levels of PCB. In mink collected from the Blue and Wallowa Mountains, DDE was the most common pollutant present (0.12-1.2 ppm). PCBs were

Table 3. A summary of liver and muscle residues from mink given PCBs in laboratory studies (Henny et al. 1981).

Diet (duration)	Number animals	Alive	Dead	PCBs ppm (wet weight) Liver (mean)	PCBs ppm (wet weight) Muscle (mean)	Authority
30% Coho salmon Lake Michigan (6 mo)	3		X	5.21±1.66[a]	4.73±1.80[a]	Ringer et al. (1972)
Basal + 30 ppm PCB (6 mo)	12		X	4.18±0.58[a]	4.88±0.54[a]	Ringer et al. (1972)
3.57 ppm PCB (105 days)	16		X	11.99±11.0[b]	3.31±0.98[b]	Platonow and Karstad (1973)
0.64 ppm PCB (<160 days)	2		X	1.10±0.08[b]	0.62±0.12[b]	Platonow and Karstad (1973)
0.64 ppm PCB[c] (160 days)	4	X		1.23±0.10[b]	0.97±0.51	Platonow and Karstad (1973)
0.64 ppm PCB (160 days)[d]	4	X		0.87±0.15[b]	0.83±0.43[b]	Platonow and Karstad (1973)
0.64 ppm PCB (160 days)[e]	4	X		1.21±0.05[b]	0.77±0.19[b]	Platonow and Karstad (1973)
0.64 ppm PCB (160 days)[f]	2	X		1.33±0.16[b]	0.64±0.09[b]	Platonow and Karstad (1973)

[a]SE
[b]SD
[c]Killed after 160 days on contaminated diet.
[d]After dosage, on clean feed 1 mo.
[e]After dosage, on clean feed 2 mo.
[f]After dosage, on clean feed 3 mo.

detected in only one liver from this area. Mink from the upper Klamath Basin contained extremely low levels of DDE (4 of 10 liver contained 0.12-1.2 ppm). PCBs were detected in only one liver. Mink from the Klamath River were also generally uncontaminated (4 of 24 livers contained 0.10-0.13 ppm of DDE and 0.50-0.68 ppm of PCBs). Six of the 9 mink collected in the Lower Columbia River were contaminated with PCB residues (0.55-2.1 ppm). DDE was detectable in mink from the Lower Columbia River and coastal streams but was less common than PCB residues. PCB residues in mink from the Lower Columbia River were generally not as high as levels in mink that died in laboratory studies. However, several levels (0.92, 1.1 and 1.2 ppm) were within the range detected in mink that survived long-term tests with a diet of 0.64 ppm PCB (0.87-1.33 ppm, Platonow and Karstad 1973). At these levels in the laboratory only 1 of 12 female mink produced a litter and the 3 kits born to that female died during the first day of life.

Because fish are important in the diet of river otter, fish were also collected and analyzed during the same study (Henny et al. 1981). Cutthroat trout, steelhead trout, and Klamath River sculpin were collected from Mack Creek in the Cascade Mountain Wilderness Area, but contained only low levels of contaminant residues (42-430 ppb). PCBs were the most abundant organochlorine contaminant in largescale sucker, chiselmouth, northern squawfish, and smallmouth bass collected from the Columbia River near the Bonneville Dam and the Willamette River, near Portland, Oregon. PCB levels in these fish ranged from 0.24-2.8 ppm, similar to the levels of PCB fed to experimental mink that caused reproductive failure (0.64 ppm, Platonow and Karstad 1973), and PCB concentrations in Columbia River otters (range 4.8 - 23 ppm) were much higher than the concentrations found in livers of experimental mink (mean range of 1.10 - 11.99 ppm, Platonow and Karstad 1973) that died on various dosages of PCB (0.64 and 3.57 ppm). Therefore even though the levels of PCB residues in mink from the Lower Columbia River were generally not as high as levels in mink that died in laboratory studies, wild mink may be exposed to equivalent dietary levels of PCB. However, the diet of wild mink is varied and localized, different trophic levels are utilized to an extent that prey from each of these levels would have to be analyzed in order to determine a realistic dietary exposure to PCBs (Henny et al. 1981).

A study by Foley et al. (1988) also measured and described the contaminant levels in river otter and fish but then also correlated the concentrations found in the fish and otter on the basis of watersheds from which the otter and fish were collected. The fish examined in this study were collected near trapping locations of the mink, and no specifications were made for the collection of fish that are more frequently eaten by mink in each particular watershed. Fish from 14 localized, geographically restricted, upper New York state watersheds had significantly ($p < 0.05$) different concentrations of PCB, DDT and mercury. Foley et al. (1988) observed significant correlations ($p < 0.5$) between fish and mustelid contaminant levels, indicating a relationship between accumulation of PCB and DDT in mustelids and the levels of these chemicals in fish. Habitats where significant PCB contamination occurred produced fish with PCB concentrations greater than 0.64 ug/g. Contamination levels of 3.57 to 20 ug/g produced 100 percent mortality in adult, ranch-reared mink (Platonow and Karstad 1973, Aulerich et al. 1973). In 93 percent of the mink/fish paired comparisons residues were higher in fish. Fat and liver PCB concentrations in mink ranged from 0.4 to 95 ug/g and from 0.03 to 7.9 ug/g respectively. Juvenile mink had higher PCB concentrations (5.9 ug/g) than the 1+, 2+, and 3+ age classes (3.8, 3.5 and 2.7 ug/g, respectively). Older mink had the highest DDE concentrations. Both PCB and DDE levels in mink (correlation coefficient of 0.43 and 0.81, respectively) were significantly correlated ($p < 0.05$) with concentration levels in fish on a restricted watershed scale but not on a major watershed basis. This suggests otter are more susceptible to the influence of local sources of PCB and DDE. Areas where concentrations of these chemicals are high may cause suppression or elimination of a population.

A mean PCB concentration of 0.4 ug/g (wet weight) in the livers of Oregon mink (Henny et al. 1981) was equivalent to that measured in New York. The maximum PCB level in Oregon mink was 3.5 ug/g, about half the maximum found in New York. However, in order to relate the conclusions of Foley et al. (1988) concerning the influence of point or non-point source pollution on the status of the mink population in Oregon, data would have to be collected on the contaminant concentrations in fish from the aquatic habitats of Oregon mink. Then, correlations would have to be made between the concentrations found in the Oregon fish, concentration found in Oregon mink, and the specific geographic locations the fish and otter were taken from.

Heavy metals, such as lead, mercury, are also commonly found toxic substances in the lower Columbia River and selected as a focus in the survey of studies on the toxicological effects of environmental contaminants on the mink population in the lower Columbia River. Although studies have determined mink sensitivity to environmental contaminants, such as polychlorinated biphenyls (Hornshaw et al. 1983) and methyl mercury (Aulerich 1974), the relationship of metals and other contaminants to reported declines of wild mink populations is uncertain.

Lead concentrations of $\geq$10 ug/g have been associated with diagnostic lead toxicosis in experimental mammals. Blus and Henny (1990) and Blus et al. (1986) studied the concentrations of lead in mink from the Coeur d'Alene river system in western Idaho and eastern Washington. Lead concentrations in mink were generally low except in localized areas of the river system where the highest level for this species was recorded. Concentrations ranged from 0.02 ug/g to 34 ug/g. Cadmium levels were detected but were relatively low. Concentrations of lead and cadmium were not significantly intercorrelated ($P > 0.05$) either by time or place of collection or in the total sample. Blus et al. (1986) speculate the unlikelihood of mink survival in certain sections of the Coeur d'Alene River because of high concentrations of metals and the apparent absence of both suitable cover and an adequate prey base. Foley et al. (1988) found mink contained consistently higher mercury residue concentrations than did the fish ($p < 0.05$), however correlations of mercury levels between fish and mink were not significant ($p > 0.05$) for data sets taken in individual watersheds. But a significant correlation ($r = 0.74$, $p < 0.05$) was found by pooling the mercury levels in fish for each watershed and comparing these data with mercury levels in mink. Based on this correlation the researchers concluded that accumulation of mercury is determined by regional influences rather than by local point-source discharges.

Prey Species Contaminants

Presently there are studies being conducted for the Bi-State Water Quality Program in the lower Columbia River analyzing the concentrations of dioxins, furans, organochlorine pesticides, PCBs, and heavy metals in crayfish, largescale sucker, carp, all important prey species of mink in the lower Columbia River. Crayfish, suckers, carp, northern squawfish and largemouth bass are other prey species of the mink which are being analyzed in studies in the Lower Columbia River estuary by the Oregon Cooperative Wildlife Research unit and Oregon State University. However this data is being collected for analysis and comparison to data also collected on the bald eagle. One of the objectives of the study is to compute biomagnification across trophic levels and compare chemical and biological parameters from different locations. The conclusions drawn from these data, although computed in relation to the bald eagle, may be a useful addition to data on the mink in the Lower Columbia River estuary, if extrapolation of data across species is possible.

Results from the reconnaissance surveys for the Bi-State Water Quality Program reported contaminant reference levels were exceeded in tissue samples from largescale sucker and crayfish in six sites along the lower Columbia River. Dioxin and furan reference levels (3.0 ng/kg wet wt.) were exceeded in largescale sucker at Youngs Bay (RM 14), and in crayfish at Elochoman Slough (RM 36). Total PCB reference levels (25, 100, 120, 200, 500 ug/kg wet wt.) were exceeded in largescale sucker at Youngs Bay, Cathlamet Bay (RM 21), Scappoose Bay (RM 88), Bachelor Island Slough (RM 90) and Camas Slough (RM 120).

Populations

Population Dynamics

Mink do not appear to suffer significant mortality from predators other than humans. They may occasionally be prey to fisher, red fox, grey fox, bobcat, lynx, wolf, alligator, and the great horned owl. Parasites, such as trematodes and nematodes, may cause also death in wild mink (Ingles 1965). Parasites and diseases may contribute to the mortality rate of wild mink but have not been known to seriously affect wild mink populations. Data relating to prenatal mortality in wild mink are limited. Studies on ranch mink indicate that losses throughout the gestation period amount to 50-60 percent of the zygotes. Possible causes of prenatal mortality in mink are: reduced sperm viability caused by long intervals between mating and fertilization; high estrogen levels (resulting from the continued growth of ovarian follicles), which in other species prevent nidation; unequal development of ova and blastocysts due to remating; delayed implantation; and variation among uteri ability to support implanted blastocysts (Linscombe et al. 1982).

Residues from environmental pollutants constitute potential biohazards for mink. Mercury pollution of aquatic environments is widespread and mercury poisoning has been reported in wild mink from the consumption of contaminated fish and seabirds. Feeding trials have shown mink are sensitive to methyl mercury, phenylmercuric acetate, and magnesium-bromalkylmercuric chloride (Linscombe et al. 1982), but are comparatively tolerant of mercury in an inorganic form (Aulerich 1974). Studies have also shown mink are highly sensitive to PCBs (Aulerich and Ringer 1977; Bleavins et al. 1980; Jensen et al. 1977; Platonow and Karstad 1973). The widespread persistence of polychlorinated biphenyls in the Columbia River have constituted concern over the effects on mink and river otter populations in the Columbia River (Henny and Anthony 1989; Henny et al. 1981).

Data on the age ratio of mink populations is limited and highly favors juveniles. The juvenile to adult ratio in mink trapped in coastal southeastern Alaska was 1.4:1, in Montana 4:1,and in Ohio 3.14:1. Data concerning sex ratios in mink populations based on harvested mink yields inconsistent results, 81:100 in coastal southeast Alaska and 1:1 in Ohio. It is likely the sex ratio reflects when in the season the majority of the mink were trapped, rather than the actual population structure. An early harvest usually traps more male mink because they have a wider range of movement then females. Females may increase movements in order to find food late in the winter therefore more females may be trapped late in the season. (Linscombe et al. 1982). No data were found relating to recruitment, survivorship, productivity, age structure, or age distribution of the population in the lower Columbia River or similar systems.

Population Trends

Mink distribution and density depends on habitat preference, and is therefore based on the presence of den sites, availability of food and competition with other species. Based on trends in the harvest reports from the last 20 years and the PCB concentrations found in mink by Henny et al. (1981) it is likely the mink population in the lower Columbia River has decreased substantially or has been completely extirpated (Henny pers. comm. 1994).

DISCUSSION AND ANALYSIS

There is not a substantial body of data available on the productivity, abundance, and distribution of mink in the lower Columbia River and its estuary, we found and cited 3 studies specific to the Columbia River, Washington, or Oregon. This lack of data makes it difficult to make any substantive statements concerning its health, morbidity, contaminant burden, or population status in the lower Columbia River. Most conclusions are inferential and based on data collected 10 to 15 years ago, or from other sites. Data gaps are apparent and recommendations for further studies can be made. However, the objective of this discussion is not to recommend how to specifically fill these data gaps, but rather, to point out missing or weak areas of information necessary to assess the role of the mink in the Columbia River ecosystem. Emphasis is put on relating past, current, and future data to form a comprehensive picture of how the mink population is being affected by contaminant burdens and habitat alteration in the lower Columbia River. Ultimately, drawing conclusions based on the measured contaminant levels in mink and their prey species that state "contaminate burdens in the lower Columbia River may have negative effects on the mink population, therefore further research is recommended", should be avoided.

Collecting data on mink in the lower Columbia River is difficult. Mustelids tend to be solitary animals which prefer seclusion from human disturbances, therefore finding individuals of any population for sampling is difficult. If the population in a particular geographic area is low, as it is in the lower Columbia River, then finding individuals to sample is even more difficult. Natural causes, harvest, habitat loss and high levels of chemical contaminants are the three factors attributed to mink mortality found in this literature survey for mink populations in general (Linscombe et al. 1982, Henny et al. 1981). The activities of humans attributing to habitat loss and high contaminant concentrations are hypothesized, in this discussion, to have had the greatest negative impact on the decline of the mink population in the lower Columbia River. This hypothesis is based on the generalizations already referred to and the conclusions of Henny et al. (1981). Unlike populations of fish species and migratory birds whose home ranges may extend across geographic and international boundaries, mustelid populations appear to utilize a relatively restricted geographic home range. If contaminant residues are high in tissues of mink from the lower Columbia River, then the contaminant level in the mink can be assumed to be directly related to the amount of contaminants being discharged in to the Columbia River system and not some other unrelated system. However the contaminant load in the lower Columbia River is not restricted to point-source discharges into the lower Columbia River. Contaminants discharged all along the Columbia and its tributaries ultimately contribute to the contaminant load in the lower Columbia River (Lebovitz and Everson 1994).

Based on this literature survey, three factors, individually and in combination, are likely to affect the mink population in the lower Columbia River; 1) habitat alteration and suitability, 2) prey abundance as determined by habitat suitability and contaminant burdens, and 3) contaminant burdens in the mink as determined by contaminant burdens in prey and habitat. Impacts of harvest on mink populations may be an important variable in the status of the mink population, but to what extent is uncertain. According to the Oregon Department of Fish and Wildlife (ODFW) 1976-1994 Annual Furbearer Harvest Reports, harvest has not been found to cause extensive negative impacts on mink populations in Oregon. However, this is based solely on annual harvest reports which are not necessarily reflective of the annual status of the mink population.

Population Trends of Mink in the Lower Columbia River

Studies and literature surveys by Merker (1983), Tabor (1976), and Tabor et al. (1980) conclude that riparian habitats are critical for mink in the lower Columbia River. Tabor (1976) attempted to determine a population estimate of mink in the lower Columbia River based on utilization of riparian habitat. His study was unable to make a concrete estimate of population size and density because observations occurred in sample sizes too small to statistically analyze.

Harvest reports for the state of Oregon may give some indication of general mink population trends in the counties bordering the lower Columbia River, which in turn, may be attributed to the condition of habitat and water quality in the Columbia River. However, such extrapolations based on harvest reports can be deceptive. Changes in annual harvest success does not necessarily reflect changes in the population. Factors more likely to affect harvest success in any particular year are; pelt price on the world wide market, the effort of the trapper (catch per unit effort), or climatic conditions. The total catch of otter is likely a combined function of the number of trappers seriously trapping otter in a given year and the actual abundance and density of otter in regularly trapped areas. The same may be assumed for the total catch of mink in an area (Rowbotham-Vita 1982).

A relatively steady decline in the percent of mink harvested in Clatsop, Columbia, and Multnomah counties has been observed (Table 4). This decline may be an indication of the steady decline in the mink population in those three counties, which would include the population along the lower Columbia River. Henny et al. (1981) also compared the number of mink harvested in Clatsop and Columbia counties with the number of mink harvested in all of Oregon between 1949 and 1976 and observed a slight decline in the percent of mink harvested in Clatsop and Columbia counties. However, the number of trappers declined as well, which may account for the decline in mink harvest. The cause-effect linkage between the two are speculative. A more detailed analysis of the mink population in Oregon by the Department of Fish and Wildlife (ODFW), including sex ratios and age structure, was not available from the ODFW. Mink are less of a priority than other, more publicly spotlighted, furbearers, therefore less money is allocated by ODFW for research on mink (Larry Copper, ODFW pers. comm. 8/94).

Habitat Suitability and Alteration

As discussed in the findings section of this report, mink are most often found in close association with wetlands, with extensive channels and slow moving water, and the adjacent riparian habitat (Melquist et al. 1981 and Allen 1986). Preference of habitats associated with small feeder streams, rather than those associated with large, broad rivers, is likely to limit the presence of mink in the main body of the lower Columbia River. Definitive studiesof habitat selection and use at sites of varying quality are required to properly identify the relative importance of variables in the habitat which determine use by mink in the lower Columbia River. Correlating these data with the history of habitat alteration of the estuary and lower river in the last 100 years may provide a significant additional explanation for the extremely low density Tabor (1976) observed in the lower Columbia River and its estuary. Dams, dredging and filling for navigational improvements, rapid growth in human population, industrial growth, over fishing, and natural climatic variation are some of the factors that have impacted the habitat in and along the lower Columbia River (Lebovitz and Everson 1994). According to a map produced by Northwest Environmental Advocates

Table 4. Mink harvest[1] data for Clatsop, Columbia and Multnomah (C-C-M) counties as related to state harvest in western Oregon.

Season	Pelt Price	No. Mink Harvested C-C-M / Western Oregon	Percent Mink Harvested in C-C-M
1976-77	$10	137 / 654	21
1977-78	8	170 / 582	29
1978-79	14	83 / 496	17
1979-80	16	140 / 764	18
1980-81	17	135 / 852	16
1981-82	12	122 / 559	22
1982-83	10	123 / 735	17
1983-84	12	114 / 716	16
1984-85	13	132 / 810	16
1985-86	10	84 / 595	14
1986-87	17	90 / 547	16
1987-88	19	108 / 580	19
1988-89	16	70 / 343	20
1989-90	13	25 / 292	9
1992-93	NA	8 / 114	7
1993-94	NA	13 / 140	9

[1]annual harvest reports, Oregon Dept. of Fish and Wildlife (1976-1994).

(NWEA 1992) considerable habitat loss in the lower Columbia and its estuary has occurred since 1870. Another study and map series produced for the Columbia River Estuary Data Development Program (Fox et al. 1984) also describes the alterations of the estuary since 1870.

It is likely mink will use similar habitats as other aquatic furbearers, such as river otter. Henny, using the habitat suitability model as a reference to locate mink and river otter in the lower Columbia River (pers. comm. 8/94), confirmed this likelihood. Additionally, interspecific use of habitats by mink and river otter in Idaho was studied by Melquist et al. (1981). Therefore, results from the determination of the relative intensity of habitat use in the lower Columbia River by muskrats, nutria, beaver and river otter (Frenton and Merker 1983) may be applicable to habitat use by mink in the lower Columbia River. According to Frenton and Merker (1983), tidal swamps, high tidal marshes and low tidal marshes were used for feeding and denning by 20 to >50 percent of these aquatic furbearers. Swamp and low marsh were shown to occur in Cathlamet and Grays Bay (RM 20) through the end of Puget Island at RM 45. Low marshes were also shown to occur in small portions of Trestle Bay (RM 6-8) and Baker Bay (RM 3-5). The most concentrated area of high marsh was between RM 23 and RM 35. High marsh and swamp habitats are described as tidal wetlands receiving irregular tidal inundation, with lower limits of vegetation types between 6.5 and 8.5 feet above the mean lower low water (MLLW, -2 ft.) and an upper limit between 8.0 and 12 feet above MLLW. Low marshes are described as tidal wetlands receiving regular tidal inundation, with lower limits of elevation at 3 feet above MLLW and an upper limit of vegetation type between 6.5 and 8.5 feet above MLLW.

Interspecific competition does not interfere with mutual habitat utilization by mink and river otter. According to Melquist et al. (1981), niche overlap of mink and river otter in Idaho existed in food habits, habitat utilization, and diel activity patterns. However, the degree of overlap was minimized by differences in body size and morphological adaptations. Differences in foraging strategies allowed mink to exploit both aquatic and terrestrial prey, and otter to exploit a wider variety of fishes. Variability in prey selection and activity patterns minimized depletion of available prey and the possibility of aggressive interactions. Finally, morphological differences promoted habitat segregation, confining otter to aquatic habitats, while mink exploited both aquatic and terrestrial habitats.

The tidal inundation of the high and low marshes and tidal swamps may leave pools of shallow, slow moving water, providing excellent foraging habitat for mink. In the last century the surface area of tidal marshes and swamps have been reduced by 65 percent, mainly the result of diking. According to Thomas (1983) the total estuary is three-quarters of what it was in 1870. Almost 37,000 of the total estuarine acreage have been converted to diked floodplains, uplands, and non-estuarine wetlands. The tidal swamp area has been the most dramatically altered with a reduction of 77 percent of its area, again primarily the result of diking. If the mink population in the lower Columbia River ever used swamp highland and lowland marshes in the past, and prefer to utilize those habitats presently, then mink have suffered a substantial loss in habitat quantity and suitability.

Chemical Contamination of Habitat, Prey and Mink

In monitoring of contaminant levels in the mink population it is necessary to analyze contaminant levels in mink from the lower Columbia River as a compounded effect influenced by, not only, direct intake of contaminants from prey, but also by the levels in the water and sediment. Contaminants, once in the mink, may have aggravated effects due to stress induced by reproduction, habitat loss, or increased foraging activity caused by decreased prey availability. All of these factors interrelate to form compounded effects on the health of the mink population. The major chemical contaminant groups addressed in this report; 2,3,7,8-tetrachlorodibenzon-p-dioxin (TCDD), polychlorinated biphenyls (PCBs), dichloro-diphenyl-trichloroethane (DDT), dichloro-dipheny-dichloroethylene (DDE) and heavy metals, are directly discharged into the lower Columbia River from approximately 33 major and minor industries between river miles 0 and 145 (Lebovitz and Everson 1994). Sewage, agricultural run-off and urban run-off are other sources of chemical contamination, particularly in areas with high human population density. Additionally, the lower Columbia River receives contaminant loads from industries, agriculture and urban development in upstream portions of the Columbia River and from tributaries that flow into the Columbia. In general, sources of contaminants are as follows: DDT and DDE from agriculture, dioxins and furans from pulp mills, wood treating facilities, incinerators, and other sources. PCBs are released from aricultural, urban areas, and industrial discharge, and dams. Heavy metals are discharged mainly from industrial sources. Chemicals can be introduced and accumulated in bodies of water by several different pathways: sediment uptake and release, chemical degradation, chemical and photochemical formation, direct absorption, fallout, out sprat transfer, and volatization (Lebovitz and Everson 1994).

Accumulation of these chemicals in the biota of a system usually results from transport in the food chain. Food chains act as a biological amplifier. Total biomass is reduced with each trophic level in the food chain. However, losses of such compounds as DDT and PCBs are relatively small, so concentrations increase with each trophic level (Ehrlich et al. 1977). Accumulation in the mink is likely to result from the following simplified pathway. Fish and other prey accumulate residues from intestinal absorption of residues in their prey, or from dermal absorption of residues in the water. Residues in fish will bioaccumulate to levels higher than ambient water. Mink then eat the fish and crayfish and absorbs the prey's accumulated residues. Residues are then stored in the tissues of the mink and released, in either subtoxic or toxic levels, in times of stress. Additionally, the residues are passed on to the unborn mink through the placenta, and to kits through mammary glands during lactation. If female mink accumulates subtoxic levels of contaminants, it is likely that a kit will have accumulated the same contaminant residues before birth, will accumulate more while nursing, and still more once it begins to prey on fish and other items. Eventually the subtoxic accumulations will begin affecting productivity and kit survivorship. Pathways taken by each type of contaminant will vary, as well as the subtoxic or toxic effects, individually or in combination, on a species. The issue for the mink population in the lower Columbia River, is that the levels of contaminants in the environment and their prey are unknown. While the level of biophysiological sensitivity in mink is known for DDT and PCBs, the combined and subtoxic effects of contaminants, compound by natural stresses, is unknown and warrants further research.

Few data were found in this survey on contaminants levels in prey species of mink in the lower Columbia River. There are studies presently being conducted that are measuring contaminants levels, abundances and distribution of crayfish, sculpin, perch, squawfish and sucker in the lower Columbia River by Tetra Tech, for the Bi-State Water Quality Program, and the U.S Fish and Wildlife Service. Measurements of contaminant concentrations in largescale sucker, crayfish and carp are summarized in Table 5. Data from these studies may be related to contaminant levels, and abundances and distribution of mink in the lower Columbia River, also presently being studied. There is concern over the status of mink in the lower Columbia River based on the contaminant levels measured in mink and reported by Henny et al. in 1981. As stated above, based on the mechanisms of biomagnification, any contaminant residue in prey species of mink will ultimately become residues in mink. However, there are no data from the last 15 years estimating the population, or quantitatively linking the health of the mink population to habitat suitability; the contaminant levels in, or abundances of prey populations; water quality in the lower Columbia River; or the amount of chemical discharge into the river.

Table 5. Measurements of contaminant levels in prey (Tetra Tech 1994).

Contaminant	Crayfish		Largescale Sucker		Carp	
	1991	1993	1991	1993	1991	1993
Lead						
Cancer Range (ng/kg)	0.01-0.05	0.048-0.174	0.02-0.86	0.038-0.376	0.02-0.23	0.116-0.173
Frequency of Detection	30/35		28/34		10/10	
Mercury						
Cancer Range (ng/kg)	0.012-0.078	0.029-0.081	0.022-0.137	0.100-0.264	0.056-0.166	0.145
Frequency of Detection	32/35		34/34		9/10	
P,P-DDT						
Cancer Range (ng/kg)	0.29	NA	NA	NA	0.24	NA
Frequency of Detection	1/33		0/34		1/11	
P,P-DDE						
Cancer Range (ng/kg)	NA	NA	0.44-1.80	NA	0.18-0.39	NA
Frequency of Detection	0/33		9/34		3/11	
Aroclor 1232						
Cancer Range (ng/kg)	NA	NA	NA	NA	NA	0.5
Frequency of Detection	0/33		0/34		1/11	
Aroclor 1254						
Cancer Range (ng/kg)	NA	NA	2.2-10.6	0.5-56.3	1.5-9.6	1.1-1.2
Frequency of Detection	0/33		33/34		7/11	
Aroclor 1260						
Cancer Range (ng/kg)	NA	NA	3.5	0.6-4.0	1.0-2.8	0.5
Frequency of Detection	0/33		10/34		5/11	
2,3,7,8-TCDD						
Cancer Range (ng/kg)	0.018-0.041	0.03-0.10	0.019-0.055	0.01-0.07	0.025-0.085	NA
Frequency of Detection	15/27		14/28		5/7	

Aroclor 1016, 1221, 1242 and 1248 were also measured for but were not detected in any sample.

Table 5a. Locations of maximum metal concentrations (Tetra Tech 1994).

Contaminant Reference Level	Max. Concentration ng/kg (ppm)	Prey Species	River mile Location
Lead - 3.0 ng/kg	0.507	Largescale sucker	23-Svenson Island
	0.444	Crayfish	124-Gary & Flag Island
	0.285	Largescale sucker	120-Camas Slough
	0.204	Crayfish	95-Willow Bar Island
	0.174	Largescale sucker	68-Carrolls Channel
Mercury - 3.0 ng/kg	0.264	Largescale sucker	21-Cathlanel Bay
	0.245	"	14-Youngs Bay
	0.222	"	68-Carrolls Channel
	0.213	"	88-Scappoose Bay
	0.196	"	124-Gary & Flag Island

Table 5b. Locations reference levels were exceeded (Tetra Tech 1994).

Contaminant	Reference Level	Species	Location
Dioxins/Furans	3.0 ng/kg	Largescale Sucker	Youngs Bay
		Crayfish	Elochoman Slough
Total PCBs	110 μg/kg	Largescale Sucker	Youngs Bay
		"	Cathlamet Bay
		"	Scappoose Bay
		"	Bachelor Island
		"	Camas Slough

Table 6 correlates the number of mink observations made by Tabor (1976) in the lower Columbia River (which are assumed to reflect the density of mink in the same area) with the number of industries in the same area (NEA 1992). Foley et al. (1988) concluded that mink are more susceptible to the influence of local sources of PCB and DDE. Therefore, areas where concentrations of PCBs and DDE are high may cause suppression or elimination of a mink population. Attempting to prove the hypothesis that mink density will be low in areas with high levels of contaminants concentrated in proximity to their sources is difficult based on the data from Table 6. The table does indicate where there are industries there are few if any mink, but the sample size is limited. Mink may not inhabit areas near industries where contaminants are more concentrated. The absence of mink may also be because the areas associated with industries support a greater human population, mink will tolerate human disturbances but prefer to avoid them. Prey abundance and habitat suitability are known to determine what habitat a mink uses. If contaminant concentrations in certain areas decreases the abundance of prey species, or causes accumulation of contaminants in prey, then mink will also be adversely affected.

Table 6. Locations of industry discharge and locations of otter/mink observations in Lower Columbia River (RM 0 to RM 145) (Tabor 1976; NEA 1992).

RM (River Mile)	No. of observations		Approximate Number of Industries	Types of Industry	Possible Contaminant Discharges
	Otter	Mink			
0-12	1	1	8	Food Processing	Nutrients - nitrogen & phosphorus, TSS, BOD
12-79	3	1	9	Pulp & Paper Mills Lumber & Plywood Mills Chemical Industries Power Generating Plants	Dioxins, heavy metals, PCBs
79-145			16	Pulp & Paper Mills Lumber & Plywood Mills Chemical Industries/Misc. Toxic Sites	Dioxins, heavy metals, urban runoff, PCBs
125-133	10	2	0		

Data Gaps and Some Suggestions

All of the issues in this discussion here are representative of data gaps and the need for further research. Little is known about the population size, productivity, contaminant burdens, and habitat utilization of mink in the lower Columbia River. Based on this discussion, some of the most fruitful areas for priority work might include:

- An estimate of the current population and its distribution to better assess impacts of contaminants and habitat alteration on the mink population;
- Determination of the biophysiological sensitivity of adult and juvenile mink, kits, fetuses, and embryos to multi contamination and the compounded effects of natural stress;
- Measurement of the contaminant burden in populations of prey species. Analysis of the short and long term effects of contaminants on prey populations;
- Determination of factors influencing habitat preference and selection by mink;
- Comparison of the contaminant levels in the mink and their prey. Determination of the extent of biomagnification of contaminants in the system;
- Correlation of contaminant and population data to habitat type and location within the river;

- Determine how far from the point source is a contaminant concentration dilute enough to permit the sustainment of mink populations and their prey;

- Determine which areas are better suited for support of prey populations and mink populations and which areas may be responsible for the decline in the mink population.

The above obviously indicate substantial data gaps concerning the mink population in the lower Columbia River. Some are in the process of being filled, others may be lined up in future research proposals. A major conclusion derived from this literature survey is that data collected concerning the status of and contaminant concentrations in the mink population in the lower Columbia River will be of little or no use unless the results and conclusions are correlated with the biophysical dynamics of the Columbia River, including the interrelated compounding variables of habitat degradation and environmental contamination.

EFFECTS OF CONTAMINANTS ON RIVER OTTER (*Lutra canadensis*) ON THE LOWER COLUMBIA RIVER

INTRODUCTION

Northern river otter (*Lutra canadensis*) are aquatic furbearers whose distribution historically extended across most of the North American continent. Present distribution of the northern river otter extend north from Florida to Alaska and west from eastern Newfoundland to the Aleutian Islands. Otter are rare in Arizona, Colorado, Indiana, Iowa, Kansas, Kentucky, Nebraska, New Mexico, North Dakota, Ohio, Oklahoma, South Dakota, Tennessee, Utah, and West Virginia. Oregon and Washington populations are stable or increasing (Toweill and Tabor 1982). Northern river otter are carnivores, occupying a top position in the aquatic food web, and are therefore exposed to and bioaccumulate environmental contaminants from consumption of prey species. Otter in the lower Columbia River (LCR) prey on a variety of species including carp, sucker, crappie, sculpin, salmonidae, and crayfish.

Northern river otter have been an important furbearer since Europeans first colonized North America. The total raw pelt values for the northern river otter in North America in 1976-1977 was almost $ 3,000,000 (Toweill and Tabor 1982). Unlike the mink (*Mustela vison*), raising river otter has never become profitable because of its large size and specialized requirements. Therefore, wild otter populations are the only source for the harvest of otter pelts; however, most trappers harvest otter incidental to trapping other species, particularly beaver.

River otter have been identified as an indicator of the health of the LCR ecosystem. They can be found in most non-urban areas, but the present distribution and abundance of river otter along the LCR are not well known, particularly in relation to suitable habitat and known contamination in the river. Compared to other populations that live in areas adjacent to the Columbia River, DDE and PCB residues in river otter have been elevated. Also, in a 1978-1979 pilot study of mink and river otter in Oregon, the levels of 2,3,7,8-tetrachlorodibenzon-p-dioxin (TCDD) in fish collected in the Columbia River were higher than the level reported by the Canadian Wildlife Service as possibly causing reproductive impairment in mink. River otter in the Columbia River contained concentrations of PCBs higher than those in mink. However, the relative sensitivity, the biophysical responses, of the river otter to environmental contaminants is not known (Henny et al. 1981). Most documentation of the effects on environmental contaminants on mink and/or river otter does not take into account the cumulative effects of PCBs and other contaminants known to be in the food web of the Columbia River.

The purpose of this study is to compile and summarize available published and unpublished reports related to river otter in the LCR and its estuary and effects of environmental contaminants. As top carnivores of the aquatic food wed, river otter are assumed to be important indicators of the health of the whole ecosystem. Specific objectives are to evaluate existing literature categorically on the basis of data indicating responses of river otter to contaminant levels, habitat alteration, life history, population dynamics and trends, and diet, in the LCR; to analyze and synthesize these data on the basis of study methodology; and to identify weaknesses in the data base.

FINDINGS

Life History

Female river otter mature sexually and normally mate for the first time at approximately two years and have long reproductive lives (possibly as long as 16 years). Male otter may not be successful breeders until they are 5-7 years old. Otters mate in the late winter to early spring, the peak mating period being in March or April, multiparous females mate soon after parturition. Implantation is delayed for 8 months and is estimated to occur no earlier than February in western Oregon otter (Tabor 1974). The actual gestation time, the period between implantation and birth, is estimated to last 2 months (Rowbotham-Vita 1982). Parturition varies depending on geographical area and may occur from November to May, but was estimated to occur in early April in western Oregon (Tabor 1974).

The average litter size is 2.5 (range 1-6). Tabor and Wight (1977) found each of the 4 females with implanted embryos in their study had the same number of corpora lutea as embryos, indicating that the intrauterine mortality rate was low. The average weight at birth is 130 g . At 6-10 weeks young first venture out of the natal den (Toweill and Tabor 1982; Wren 1991). Otter pups are weaned at about 5 months, but remain with their mother until shortly before her next litter is born. Family groups (female otter and her pups) regularly travel and remain intact during the November and December. Therefore it is likely young otter benefit from the protective guidance of their mother through the trapping season. (Rowbotham-Vita 1982).

Dunn et al. (1984) studied furbearers in the LCR to determine birth and rearing sites of furbearers by den searches and radio telemetry. However, few data were collected on river otter, due to low otter density in the study area and the difficulties in observing and trapping resident otter. Natality data provided by Dunn et al. (1984) on river otter in the lower Columbia River and its estuary were solely based on data collected from river otter in western Oregon (Tabor and Wight 1977). Therefore, LCR female river otter are assumed to begin to give birth during March and April. Data on the termination of the birthing period of river otter in Oregon were not available or obtained from Dunn's study and literature search.

Behavior and Movement

Male otter have been observed to commonly travel an average of 9-10 km a night (Erlinge 1967). Melquist and Hornocker (1983) described the movements of otter in west central Idaho. The maximum distance recorded by Melquist and Hornocker between consecutive day locations for a dispersing yearling male was 42 km. No apparent seasonal trend in the extent of movement for any age class was determined. However, independent juvenile males and family groups moved significantly less in the winter ($p < 0.05$). Lactating females may stay in close proximity to their natal den during the breeding season, whereas nulliparous and parous females without young may travel more extensively with a male during the breeding season than a lactating female.

Dispersal of otter coincided with the breeding season, insuring the isolation preferred by pregnant females during parturition and the postpartum period. However, dispersing otter are not sexually mature, and not all otter dispersed, suggesting an unequal development of the dispersal tendency among individuals. If a females pups have not left the natal den upon parturition of another litter, the female may leave her pups and move to a different stream to obtain privacy. Melquist and Hornocker (1983) hypothesized that since dispersion could not be linked to sexual maturity or density dependent factors, then dispersal may be an inherent trait stimulated by certain physiological changes unrelated to changes accompanying sexual maturity.

Melquist and Hornocker (1983) asserted that food has the greatest influence on the frequency of movement by otters and the distance travelled. Otter would often concentrate activities at spawning beds and remain for as long as 40 continuous days. When the fish became scarce otter returned to a pattern of frequent movement. When movements could not be attributed to changes in food supply, the researchers suggested certain intrinsic factors also may be responsible for movement. The data collected relating to movements unexplained by food availability indicated otter may move because of exploratory urges. Movement within a region was more frequent than movement between regions, behavior believed by the researchers to be a result of physiographic features and a sense of security brought on by familiarity with home range.

Melquist and Hornocker (1983) found patterns of activity are greatest during nighttime and appear rhythmic. Peak activity occurs shortly before or after midnight and at dawn. Activity steadily decreased after dawn, and was at its lowest during midday. Activity patterns appeared to be similar during spring, summer, and fall. Activity was less rhythmic during winter, again otter were most active at dawn but the frequency of activity remained high during the day and dropped after dusk. Otter activity was found to fluctuate from a high of 75 percent in February to a low of 35 percent in September. Generally otter were more active during winter and progressively less active during spring, summer and fall. Increased energy requirements combined with a reduction in abundant food supply may be responsible for increased activity during winter.

Melquist and Hornocker (1983) observed otter to be more sociable and tolerant of conspecifics than previously thought. Only 50 of the 113 visual observations of otter were solitary. Yearlings occasionally associated with family groups, lone adult females, other yearlings and lone juveniles.

Changes in otter behavior have been attributed to human encroachment. Rowbotham-Vita (1982) noted that diurnal European otters have become nocturnal to avoid humans and have been reported to den in woodland, returning to rivers to fish only at dusk.

Habitat Requirements

Otter are generally most abundant along food rich coastal areas, including the lower portions of streams, rivers and estuaries, and in areas with extensive non-polluted waterways and minimal human impact. Otter are scarce in heavily settled areas, in polluted waterways, and in food-poor mountain streams (these are generalization taken from Melquist and Hornocker 1983 and Toweill and Tabor 1982). Northern river otter are almost entirely aquatic mammals whose extensive aquatic home ranges may be associated with a tremendous variety of land types. Otter adaptation to freshwater habitats is determined by barriers in dispersal including; arid areas, mountain ranges, glaciated areas, and salt water straits. However, extensive movements by otter over mountain ranges have been observed (Pohle 1920), as well as the extensive use of estuarine areas by otter, including the Chesapeake Bay, and brackish and marine environments along the northern coast from Washington to the Alexander Archipelago, including the Strait of Juan de Fuca and exposed outer coast (Toweill and Tabor 1982). Most studies on otter habitat are confined to describing the land type adjacent to the otter's aquatic habitat. Few studies have been performed to evaluate the ranges of water quality that river otter will tolerate. Toweill and Tabor (1982) suggest surveying water quality as a means of clarifying the value of particular areas for otter habitat.

Melquist and Hornocker (1983) found river otter in west central Idaho prefer stream related habitats to ponds, lakes, and reservoirs. Habitat preference is based on the availability of adequate escape cover, shelter and sufficient food and minimal human activity. Melquist and Hornocker (1983) observed the availability of food as having the greatest influence on habitat use by otter. For example, otters that used a logjam on Lake Fork Creek where kokanee were spawning left when the kokanee disappeared. Mudflats, marshes, and backwater sloughs were important habitat used by family groups during summer because the shallow water characteristic of these areas provided an abundant supply of slow-moving fish (e.g., sculpins and perch) and invertebrates. Although food supply was indicated as important in habitat selection, the availability of shelter proved vital if an otter is to use a habitat extensively. For example, the Cascade Reservoir had ample food and was easily accessible to otter but was virtually unused because the flat shoreline provided insufficient escape cover and resting sites. Although stream related habitats are preferred by otter, exposed lakes, reservoirs and ponds, were used frequently in the winter when ice cover provided adequate shelter. Additionally, areas with minimal human activity are preferred by otter, but otter were observed to occupy areas in proximity to human establishments if they were not harassed. Therefore the generalization that otter prefer stream related habitats can be made, but otter have shown flexibility in habitat selection provided the availability of food and shelter is adequate and the level human disturbance is minimal.

In the Columbia River, Dunn et al. (1984) observed 26 otter activity sites in association with four different vegetative habitats; Sitka spruce (21), Sitka willow (3), Lyngby's sedge/horsetail (1), and reed canary grass/cat-tail (1). Activity sites did reflect the presence of otter in vegetated habitats but did not indicate otter feeding in the vegetated portions of these habitats. Tabor (1976) studied 292 river miles along the Columbia River (beginning at the seaward end of the river mouth jetties to McNary Dam) during April 1973 - October 1975. Otter sign were observed in tidal marsh near the mouth of the Chinook River on the Washington shore, at three locations within the tidal marsh and tidal shrub willow habitats; near a small tidal channel on the Oregon shore at RM 75, on a dike at Karlson Island (RM 25), at the mouth of the Cowlitz River (RM 68), at 10 locations upstream from Portland between RM 125 and 133, at six locations between RM 152 and 181, in association with the Deschutes River, at the mouth of Rock Creek, on an island at RM 254, at the mouth of Alder Creek, at the east end of Crow Butte, at Blalock Island, Patterson Slough, RM 290.5 on the Washington shore, RM 290.5 on the Oregon shore, and at McNary Wildlife Park.

According to the observations of Dunn et al. (1984) otter activity was more frequent in the sitka spruce habitat (21/26 sightings) throughout the Columbia River estuary. According to the observations of Tabor (1976) otter appeared to be uncommon or low (sign found at 1 - 3 sites) in non-tidal spruce/alder, non-tidal willow/alder and tidal marsh habitats along the Columbia River, and in tidal marsh, tidal shrub and tidal spruce habitats between RM 25 and RM 75. Otter density appeared to be highest upstream from Portland between RM 125 and 133 (sign sighted at 10 locations). Between RM 125 and 133 otter appeared to utilize ponds, sloughs, embayments and small tributaries primarily, but also were observed using the main river channel. In areas near or between Rock Creek and McNary Wildlife Park density (sign observed at 9 locations) appeared to be only slightly less than between RM 125 and 133. Otter density between RM 152 and 181 appeared to be low but consistent throughout the segment (Table 1).

Critical habitat for river otter in the LCR, are believed to be sloughs and tidal creeks associated with willow-dogwood and sitka spruce habitats (Dunn et al. 1984). Sitka spruce was the only habitat in which otter activity was consistently recorded. Aquatic habitats associated with these vegetated habitats may be important feeding sites as they contain substantial populations of crayfish, sculpin, and carp, therefore the concentration of otter sign in this habitat may reflect the importance of that habitat to otter feeding activity. Otter seem to prefer habitats with a complex network of steep-sided tide channels not exposed at high tide. Low water levels in tidal creeks may also provide desirable conditions for otter feeding activity. Low water levels concentrate prey species in tidal pools thus providing a concentrated food source for the otter. The presence of non-vegetated aquatic habitats may be a limiting factor for river otter populations, however aquatic habitats were not examined by Dunn et al. (1984).

Home Range

Most studies concerning home range and movements of otter have been conducted in Sweden. In general, a mother and young extend their home ranges in the course of a year to an area of about 7 km in diameter, while home ranges of male otter average about 15 km in width and are highly variable in length.

Table 1. Observations of mink/river otter sign and habitat type[1].

RM (River Mile)	Habitat Types	% of Total Miles of Shoreline	No. of Observations	
			Otter	Mink
0-12 (to mouth of Youngs River)	Tidal marsh, beachgrass, sand, grassland, residential, alder, pasture		1	sign observed
12-79	Tidal marsh, shrub willow, willow/cottonwood, large trees, grassland, cottonwood/large trees, willow/large trees		3	sign observed
79-145 (to Bonneville Dam)	Cottonwood/large trees, grassland, willow/cottonwood/large trees, willow, cottonwood/ash, large trees/willow		10 (between 125-133)	2 (between 125-133)
145-192 (Bonneville Pool)	Pasture, grassland, industrial, enbayments, oak/ponderosa pine, residential, oak, maple/douglas fir	rock riprap (34%), grassland undifferentiated (11%), willow/large trees (7%), Industrial (5%), oak/ponderosa (5%), rock (5%), willow/small trees (4%), oak (3%)	6 (between 152-181)	10 (between 152-184)
192-215.6 (The Dalles Pool)	Grassland, rock/grassland, rabbitbrush, pasture, rabbitbrush/grassland, embayment, herbaceous types, ag. production, shrub willow	rock rip rap (40%) gravel (14%), rock (9%), sand (8%) shrub willow (4%)	NA	NA
215.6-292 (John Day Pool)	Rabbitbrush, rabbitbrush/sagebrush, field crops, rock cliff/grassland, grassland, grassland/rabbitbrush, sagebrush, bitterbrush	sand (39%), rock rip rap (21%), gravel (12%)	8	sign observed

[1]all data derived from Tabor 1976.

Use and length of otter home ranges are influenced by prey availability, habitat, weather conditions, topography, conspecifics, and the reproductive cycle. In west central Idaho, Melquist and Hornocker (1983) observed river otter home ranges to overlap extensively and vary in seasonal length from 15-50 km during spring, 10-78 km during summer, 8-48 km during fall and 23-50 km during winter. Activity centers are areas an otter spent at least 10 percent of the time during a specific season, located in areas of abundant food and adequate shelter, varying in number within the home range and are not located in any specific portion of the home range. Activity centers determined the limits of seasonal home ranges, while the home range shape was determined by the drainage patterns of rivers. Because otter spent most of their time at activities centers within their home range, home range shape was often longer than necessary to provide adequate food and shelter. Some river otter may cover 50 to 60 miles of stream habitat a year (Liers 1951).

In the LCR, Dunn et al. (1984) observed greatest river otter activity in the sloughs and creeks of the willow-dogwood and sitka spruce habitats in the Cathlamet Bay area. Based on the presence of an abundant food supply and adequate shelter, Melquist and Hornocker (1983) indicate that activity is concentrated in these habitats.

Den and Rest Sites

Otter dens are usually in or directly adjacent to the water's edge, winter dens may be located within 10 m of the water's edge. River otter select den and resting sites based on availability, convenience, shelter and seclusion. Otter used natural formations, man-made structures, and dens built by other animals rather than excavate their own den. Beaver dens were preferred resting site, used 32 percent of the time by otter in Idaho, because of their availability and provision of shelter and an underwater escape route. Logjams were used frequently, 18 percent of the time as resting sites by otter because they offered excellent foraging and cover. Dense riparian vegetation was used primarily when no other sites were available, and most frequently during the spring, possibly because banks normally used were flooded. Other otter den and resting sites used 50 percent of time varied widely, occurring in; beaver lodge (stick), snow/ice cavity, rip-rap, talus rock, brush/log pile, natural undercut bank, fox/coyote den, muskrat bank den, boat dock/raft, abandoned irrigation dam spillway, and a boulder projecting from the water. Canid dens, located farther from the water's edge, may be used as natal dens because they provide seclusion and are frequently located on bluffs out of the danger of flooding during spring runoff (Melquist and Hornocker 1983).

Tabor et al. (1980) found six otter dens along the Columbia River. Only two were suspected to be natal dens. Dens along the Columbia River included an abandoned, floating boat house, a slough adjacent to the Oregon shore, underneath a concrete foundation of an abandoned building approximately 150 m from the river shoreline, and in shorelines composed of boulders. Otter are believed to use inactive beaver dens, but this behavior pattern was not confirmed within the Columbia River during the 1980 study. Dunn et al. (1984) observed only one den, located on the Oregon shore at RM 142.

Foraging Habitat

Aquatic prey are favored and most frequently consumed by otter. Fish and other aquatic prey of otter usually seek cover along undercuts of banks and among logs and other obstructions in waterways, therefore these are the areas where otter forage. Melquist and Hornocker (1983) observed in small shallow streams otter forage along undercut banks and among other obstructions, and in larger streams, with frequent deep pools and shallow riffles, otter generally forage among logjams situated in deep, slow moving water. Family groups were observed probing muddy and weedy bottoms of shallow backwater sloughs, oxbows and ponds for beetles and small sculpins. LCR river otter seem to prefer habitats with a complex network of steep-sided tidal channels not exposed at high tide. Low water levels in tidal creeks concentrate prey species in tidal pools thus providing a concentrated food source and provide desirable conditions for otter feeding activity (Dunn et al. 1984).

Diet

Studies on food habits of the northern river otter have determined the general primary prey of otter consists of fish and crustaceans, with amphibians, insects, birds (particularly carrion waterfowl), and mammals (particularly muskrats *Ondatra zibethicus*, or carrion) comprising otter diet in lesser portions (Larsen 1984; Toweill 1974; Dunn et al. 1984; Melquist and Hornocker 1983; Merker 1983; Stenson et al. 1984). Major foods of otter along the Columbia River, as determined by Tabor et al. (1980) are; carp, crayfish, suckers (*Catostomus* spp.) and centrarchid fishes. Minor prey species included northern squawfish, salmon, birds, mammals, insects, and mollusks. Carp and crayfish were the most frequently eaten foods of otter in all parts of the study where evidence of otter were found (Table 2). Otter are opportunistic predators therefore foraging strategies of river otter result in prey selection specific to the geographic area the otters live in. Prey selection of otter in western Oregon (Melquist and Hornocker 1983) was dependent on the potential prey species most available, or most vulnerable in the habitat being foraged. When an abundant food source is diminished or other prey becomes more available otters either move to a new location or change feeding habits and select the most available prey.

Table 2. Seasonal foods of river otter below Bonneville Dam. Data are presented in percent frequency of occurrence.. (adapted from Tabor et al. 1980)

	Season			
Foods	Spring (March-May)	Summer (June-Aug)	Fall (Sept-Nov)	Winter (Dec-Feb)
Carp	78	54	100	60
Sucker	42	11		15
Unidentified Salmonidae			50	
Sculpin				15
Other fish spp.	23			
Crayfish	25	76	50	8

Dunn et al. (1984) determined river otter feeding habits in the Columbia River estuary by scat analysis and data were presented as percent frequency of major food groups and prey species by season. Seasonal food habits of the otter were based on examination of 126 scats. Only crustaceans and fish were recorded from the scats collected in the Columbia River estuary between May 1980 and June 1981. Fish were the most important food group for all seasons combined (102/127 amount of prey item/total seasonal sample size, 80.3 percent frequency), only in summer did fish (35/55, 63.4 percent) occur less frequently than crustaceans (54/55, 98.2 percent). Crayfish (84/127, 66.1 percent) were the most important food species in all seasons except spring when carp (31/41, 75.6 percent) and sculpin (13/41, 31.7 percent) occurred most often. The most important fall food items were crayfish (17/25, 68 percent) and unidentified fish (4/25, 16 percent). Findings for winter may not be representative due to the small sample of scats examined during this season. Overall, crayfish occurred in over 65 percent of all scats, and sculpin occurred in 45.7 percent of all scats. Dunn et al. (1984) also suggested that mollusks are an important prey item of otter in the Columbia River Estuary. Other studies that analyzed LCR river otter diet made no mention of mollusks. It is possible that mollusks are overlooked when analyzing scat and stomach contents because the soft bodies may be digested faster than other prey types. It is important to determine if mollusks are actually frequently preyed on by LCR river otter because mollusks are known contaminant accumulators.

Prey Abundance

Emmett et al. (1991) determined the abundance of fish and invertebrate species in west coast estuaries, including the Columbia River estuary. Common prey species of LCR river otter included only the sculpin (*Leptocottus armatus*). Sculpin were categorized as abundant (they are often encountered in substantial numbers relative to other species) in the Columbia River estuary as adults and juveniles, and common (generally encountered but not in large numbers, does not imply even distribution) as spawning adults, larvae and eggs. Juvenile sculpin inhabit shallow water and use shallow tidal flats and pools. Older, larger sculpin reside in marine and highly saline estuarine areas. Juvenile sculpins feed on benthic and epibenthic organisms including *Corophium*, large juveniles and adults consume fish and large crustaceans (*Crangon* spp.). In an attempt to reduce predation sculpins will partially bury themselves in sediment. Juveniles preferred substrate is clean sand, while older juveniles and adults are found primarily in sandy habitats. Planktonic larvae and benthic living juveniles and adults are found over substrates ranging from soft mud to rock. Sculpins are not normally found below a depth of 50 m.

Contaminants

Although otter have large home ranges and travel extensively within their home range, they may be considered relatively sedentary in that they will decrease movement and prolong stays at activity sites according to prey abundance in portions of their home range (Melquist and Hornocker 1983). Therefore, analysis of chemical residues in their tissues is likely to yield a diverse contaminant profile unique to the riparian and aquatic habitats within their home range (Somers et al. 1987). Few studies have been conducted to evaluate the range of water quality that river otter will tolerate. Most studies are confined to describing the land type adjacent to the otter's aquatic habitat. While some experiments have demonstrated that simultaneous exposure to other chemicals, food restriction, temperature change, or disease can exacerbate the effects of exposure to a toxic chemical (Wren 1991), the potential toxicity of environmental chemicals is usually tested on healthy experimental animals having access to unlimited food and water. The potential effects of interactions between chemical contaminants and natural stresses are important when considering the risks to wildlife in natural conditions.

The Oregon river otter population appears to be stable, with recruitment balancing mortality. However, only 5-7 percent of Tabor and Wight's (1977) data came from otter in the LCR. More river otter collected from the LCR may change the population status of Oregon river otter.

High PCB residues measured in the livers of otter from the LCR (range 4.8 - 23 ppm) coupled with an overall decreasing harvest in the LCR counties over the last 2 decades (Henny et al. 1981) is cause for concern of the health of the otter population of the LCR. Henny et al. (1981) conducted a pilot study of concentrations of contaminants in mink and river otter from Oregon. All otter collected in the LCR (Clatsop and Columbia Counties) contained detectable residues of DDE and PCBs. PCB concentrations in the livers of LCR river otter (range 4.8 - 23 ppm) were much higher than the concentrations found in livers of experimental mink (mean range of 1.10 - 11.99 ppm, Platonow and Karstad 1973) that died on various dosages of PCB (0.64 and 3.57 ppm). Two male river otters had the highest PCB residues, 23 and 17 ppm.

Because fish are important in the diet of river otter, Henny et al. (1981) also collected and analyzed fish. PCBs were the most abundant organochlorine contaminant in largescale sucker, chiselmouth, northern squawfish, and smallmouth bass collected from the Columbia River near the Bonneville Dam and the Willamette River, near Portland, Oregon. PCB levels in these fish ranged from 0.24-2.8 ppm, similar to the levels of PCB fed to experimental mink, in rations of byproducts made from domestic cows, that caused reproductive failure (0.64 ppm, Platonow and Karstad 1973). However, no laboratory studies have been conducted on the relative biophysical sensitivity of the river otter to PCB concentrations. Additionally, the diet of river otter is varied and localized, therefore different trophic levels are utilized to an extent that prey from each of these levels would have to be analyzed in order to determine a realistic dietary exposure to PCBs (Henny et al. 1981).

Foley et al. (1988) measured and described the contaminant levels in river otter and fish and correlated the concentrations found in the fish and otter on the basis of watersheds from which the otters and fish were collected. The fish examined in this study were collected near trapping locations of the otter, and no specifications were made for the collection of fish that are more frequently eaten by otter in each particular watershed. Fish from 14 localized, geographically restricted, upper New York state watersheds had significantly ($p < 0.05$) different concentrations of PCB, DDT and mercury. Otter contained consistently higher mercury residue concentrations than did the fish ($p < 0.05$), however correlations of mercury levels between fish and otter were not significant ($p > 0.05$) for data sets taken in individual watersheds. But a significant correlation ($r = 0.74$, $p < 0.05$) was found by pooling the mercury levels in fish for each watershed and comparing these data with mercury levels in otter. Based on this correlation the researchers concluded that accumulation of mercury is determined by regional influences rather than by local point-source discharges.

Foley et al. (1988) also observed significant correlations ($p < 0.5$) between fish and mustelid contaminant levels, indicating a relationship between accumulation of PCB and DDT in mustelids and the levels of these chemicals in fish. Habitats where significant PCB contamination occurred produced fish with PCB concentrations greater than 0.64 ug/g. Contamination levels of 3.57 to 20 ug/g produced 100 percent mortality in adult, ranch-reared mink (Platonow and Karstad 1973, Aulerich et al. 1973). Otter contained greater PCB concentrations than mink which suggests either that otter are more tolerant than mink, that exposure is greater for otter than mink, or that otter are less efficient than mink at eliminating PCBs. No significant differences in concentration of PCB (significance level 0.84) or DDE (significance level 0.77) were observed in otter on the basis of age. Unlike correlations found for mercury, both PCB and DDE levels in otter (correlation coefficient of 0.55 and 0.50, respectively) were significantly correlated ($p < 0.05$) with concentration levels in fish on a restricted watershed scale but not on a major watershed basis. This suggests otter are more susceptible to the influence of local sources of PCB and DDE. Areas where concentrations of these chemicals are high may cause suppression or elimination of a population.

Erlinge (1972) stated that otter (*Lutra lutra*) populations in Sweden have declined continuously since 1950. He has related empty habitats previously occupied by otter to direct disturbance by man or to habitat degradation caused by pollution. In studies of otter populations in polluted aquatic systems, Erlinge recorded a correlation between pollution and low otter populations (Jenkins 1983). Rowbotham-Vita (1982) also summarized otter population declines in other regions. A decrease in abundance of otter (*Lutra lutra*) populations in Sweden since 1950 has been attributed to industrial pollution, specifically large quantities of mercury (Erlinge 1972). The decrease of otter populations in southern England since the 1960s has been attributed to an increase in fishing, tourism, riparian clearance and chemical pollution.

Prey Species Contaminants

Presently there are studies being conducted for the Bi-State Water Quality Program to analyze many of the important prey species of river otter (crayfish, largescale sucker, and carp), for dioxins, furans, organochlorine pesticides, PCBs, and heavy metals in the LCR (Tetra Tech 1994). Crayfish, suckers, carp, northern squawfish and largemouth bass are being analyzed for contaminant concentrations in studies in the LCR estuary by the Oregon

Cooperative Wildlife Research unit and Oregon State University. However, these data are being collected for analysis and comparison to data on the bald eagle, to compute biomagnification across trophic levels, and compare chemical and biological parameters for bald eagles from different locations. The conclusions drawn from these data, although computed in relation to the bald eagle, may be a useful addition to data on the river otter in the LCR estuary, if extrapolation of data across species is possible.

Tetra Tech (1994) reported that contaminant reference levels were exceeded in tissue samples from largescale sucker and crayfish in six sites along the LCR (Table 3). Dioxin and furan reference levels (3.0 ng/kg wet wt.) were exceeded in largescale sucker at Youngs Bay (RM 14), and in crayfish at Elochoman Slough (RM 36). Total PCB reference levels (25, 100, 120, 200, 500 ug/kg wet wt.) were exceeded in largescale sucker at Youngs Bay, Cathlamet Bay (RM 21), Scappoose Bay (RM 88), Bachelor Island Slough (RM 90) and Camas Slough (RM 120) (Tetra Tech 1994).

Population Trends

There are few data available on the population structure of the LCR river otter population. The habitat of the Puget Sound basin similar to the LCR and its estuary, in that both areas are marine estuaries along the Pacific Northwest coast, at similar latitudes, with similar climatic conditions. In the Hood Canal-Kitsap Peninsula, otter population there was no significant difference between the mean number of corpora lutea[1] in ovulating 2 year olds and older adults, indicating that ovulation rates of young adult females may be equal to or greater than those of older adults (it is uncertain if older adults refers to all females >2 years or only females ≥ 5 years) (Rowbotham-Vita 1982). However, proportionately fewer 2 year old females ovulated (38 percent) than any other age-class, while all females five years and older ovulated. Therefore, Rowbotham-Vita concluded that it can not be assumed that young and adult females contribute equally to the overall productivity of the population. However, after considering the number of females in each age class; eight 2 year olds, eight adults ≥ 5, and seventeen 3 and 4 year olds, if the productive rate is equal and the number of 2 year olds and the number of females 5 years and older is equal in this population then they would be expected to contribute equally to the population. If the number of 2 and > 5 year olds are compared to the 3 and 4 year olds, then both age-classes would contribute less to the population than the 3 and 4 year olds. The data were not presented clearly, and are difficult to interpret. Tabor and Wight (1977) also found the number of corpora lutea per age-class did not differ significantly ($p>0.25$) in the population of otter in western Oregon. Additionally, it was estimated that the percentage of females ovulating is near maximum in western Oregon and that ovulating female otter are likely to breed every year.

[1]corpora lutea - a ductless gland developed within the ovary following ovulation.

Table 3. Measurements of contaminant levels in prey (Tetra Tech 1994).

Contaminant	Crayfish		Largescale Sucker		Carp	
	1991	1993	1991	1993	1991	1993
Lead						
Cancer Range (ng/kg)	0.01-0.05	0.048-0.174	0.02-0.86	0.038-0.376	0.02-0.23	0.116-0.173
Frequency of Detection	30/35		28/34		10/10	
Mercury						
Cancer Range (ng/kg)	0.012-0.078	0.029-0.081	0.022-0.137	0.100-0.264	0.056-0.166	0.145
Frequency of Detection	32/35		34/34		9/10	
P,P-DDT						
Cancer Range (ng/kg)	0.29	NA	NA	NA	0.24	NA
Frequency of Detection	1/33		0/34		1/11	
P,P-DDE						
Cancer Range (ng/kg)	NA	NA	0.44-1.80	NA	0.18-0.39	NA
Frequency of Detection	0/33		9/34		3/11	
Aroclor 1232						
Cancer Range (ng/kg)	NA	NA	NA	NA	NA	0.5
Frequency of Detection	0/33		0/34		1/11	
Aroclor 1254						
Cancer Range (ng/kg)	NA	NA	2.2-10.6	0.5-56.3	1.5-9.6	1.1-1.2
Frequency of Detection	0/33		33/34		7/11	
Aroclor 1260						
Cancer Range (ng/kg)	NA	NA	3.5	0.6-4.0	1.0-2.8	0.5
Frequency of Detection	0/33		10/34		5/11	
2,3,7,8-TCDD						
Cancer Range (ng/kg)	0.018-0.041	0.03-0.10	0.019-0.055	0.01-0.07	0.025-0.085	NA
Frequency of Detection	15/27		14/28		5/7	

Aroclor 1016, 1221, 1242 and 1248 were also measured for but were not detected in any sample.

Table 3a. Locations of maximum metal concentrations (Tetra Tech 1994).

Contaminant Reference Level	Max. Concentration ng/kg (ppm)	Prey Species	River mile Location
Lead - 3.0 ng/kg	0.507	Largescale sucker	23-Svenson Island
	0.444	Crayfish	124-Gary & Flag Island
	0.285	Largescale sucker	120-Camas Slough
	0.204	Crayfish	95-Willow Bar Island
	0.174	Largescale sucker	68-Carrolls Channel
Mercury - 3.0 ng/kg	0.264	Largescale sucker	21-Cathlanel Bay
	0.245	"	14-Youngs Bay
	0.222	"	68-Carrolls Channel
	0.213	"	88-Scappoose Bay
	0.196	"	124-Gary & Flag Island

Table 3b. Locations reference levels were exceeded (Tetra Tech 1994).

Contaminant	Reference Level	Species	Location
Dioxins/Furans	3.0 ng/kg	Largescale Sucker	Youngs Bay
		Crayfish	Elochoman Slough
Total PCBs	110 μg/kg	Largescale Sucker	Youngs Bay
		"	Cathlamet Bay
		"	Scappoose Bay
		"	Bachelor Island
		"	Camas Slough

Tabor and Wight (1977) determined the age structure of their entire female otter sample (n=113) as follows; 36 percent in age-class 0, 25 percent in age-class 1, 12 percent in age-class 2, 9 percent in age-class 3, and 19 percent in age-class >4. The mean number of corpora lutea per female having corpora lutea (n=43) present was 2.85, 3.09, and 3.12 for age-classes 2,3, and 4-11, respectively. The number of corpora lutea per age-class did not differ significantly ($p>0.25$). The mean number of blastocysts recovered from uteri of females having blastocysts (n=35) was 2.80. The estimated annual survival was 68 percent for age-class 0, 46 percent for age-class 1, and 73 percent for age-class 2-11. Survival was assumed to be at a constant rate for all females of breeding age (Tabor and Wight 1977). The average annual recruitment[2] of female pups (age-class 0) per adult female was estimated to be 1.14. Using a population mode, Tabor and Wight (1977) concluded that the river otter population of western Oregon is stable and that the number of otters in western Oregon can be expected to remain constant if the balance between mortality and recruitment indicated by data in their study is maintained. Rowbotham-Vita does not estimate the population number for otter in the Puget Sound-Hood Canal area. Table 4 shows the age distribution of the

[2]Tabor and Wight (1977) defined recruitment as the number of pups (age-class 0) per adult female (age-class 3-11) at the beginning of the trapping season.

Table 4. Age distribution of otter trapped in western Oregon during 1991-92 season.

County	Age (yr.) No. Male / No. Female									Total Number	
	0.5	1.5	2.5	3.5	4.5	5.5	6.5	7.5	8.5	Males	Females
Clatsop	4/4	3/0	0/1	2/0	2/0					11	5
Columbia											
Multnomah	8/1	4/1	2/3	0/1	1/0		0/1		1/0	16	7

entire otter harvest in western Oregon during 1991-92 trapping season.

The sex ratios of otter populations in the studies of both Rowbotham-Vita (1982) and Tabor and Wight (1977) did not differ significantly from 1:1 ($p>0.05$). Sex ratios on the basis of age-class also did not differ significantly from a 1:1 ratio ($p>0.05$, Rowbotham-Vita 1982). Because population analysis is usually based on data from trapped otters, sex ratios that show a greater male to female ratio is probably a reflection of the tendency of males to roam over larger home ranges resulting in a higher susceptibility to trapping (Rowbotham-Vita 1982).

Otter that are trapped in the state of Oregon are analyzed to determine sex ratio and age distribution. Table 5 summarizes data from 1983 to 1991. Sex ratios from the 1983 to the 1991 trapping seasons range from 1.1 to 2 males per female. Sex ratios for otter populations do not usually differ significantly ($p > 0.05$) from 1:1 (Rowbotham-Vita 1982 and Tabor and Wight 1977). The sample of otters used for the determination of age distribution had more males than females, this is not necessarily a reflection of the entire population. The sample size was small and males are usually more susceptible to trapping than females.

Human activity is cited as the most serious cause of otter mortality through harvest and destruction and modification of required habitat (Melquist and Hornocker 1983; Toweill and Tabor 1982). Habitat destruction includes effects of waterway development, destruction of riparian habitat caused by homesites or farmland, and declines in water quality due to increased siltation or introduction of chemical residues. Mortality of the river otters studied in west central Idaho (Melquist and Hornocker 1983) were strongly related to human activities, accidents on roads and railroads were responsible for 6 of the 9 known otter deaths. Generally otters have few natural enemies, however otters out of water are more susceptible to predation by bobcats, dogs, coyotes, and foxes. It is difficult to determine natural causes of death in a river otter population because of the difficulty in finding and trapping otters (Toweill and Tabor 1982; Melquist and Hornocker 1983). Otter are often accidentally caught in beaver traps. Direct trapping of otter may not negatively impact an otter population if the population is not under the stress of additional impacts from habitat loss or high contaminant levels. Rowbotham-Vita (1982) asserts that it is unlikely the otter population in the Puget Sound basin will be able to continue to withstand the large harvests experienced in the area in past years as trapping efforts become more advanced, particularly as development and recreation increases causing habitat loss. Toweill and Tabor (1982) state that otter are more susceptible to over harvest because they travel extensively in restricted aquatic and riparian habitats.

Table 5. Ratios of otter population in western Oregon.

Year	Male : Female	Proportion of kit (1/2 yr) and yearling (1-1/2 yr) in sample	All sex and age groups Sample size
1983-84	1.9 : 1	0.65	161
1984-85	1.2 : 1	0.71	112
1985-86	1.6 : 1	0.61	127
1986-87	1.7 : 1	0.73	117
1987-88	1.3 : 1	0.74	176
1988-89	1.1 : 1	0.60	157
1989-90	2.0 : 1	0.76	38
1990-91	1.1 : 1	0.70	164
1991-92	1.2 : 1	0.63	136

Population Trends

In 1978, northern river otter were relatively abundant throughout the Pacific Northwest and were listed as stable or increasing Oregon and Washington (Endangered Species Authority 1978). Few researchers have determined numerical estimates of northern river otter population densities, nor have estimates of seasonal activity patterns been determined within the LCR. Based on an extensive survey in Sweden (Erlinge 1968) otter density of one per 4-5 km is considered moderate, and 1 per 8-10 km as low.

Dunn et. al. (1984) determined the greatest density of furbearer dens in the Columbia River estuary occurs in the tall Lyngy's sedge habitat, however this information is based solely on sightings of muskrat dens - no otter dens were observed. Additionally, the density value was extrapolated from a small area of search, therefore the actual importance of this habitat may be overestimated. All of the important furbearer denning habitats were characterized by extensive steep-sided tide channels or tidal creeks, this may be a more critical factor for furbearer denning and reproduction than the vegetative community surrounding den areas. Furbearer rest site densities indicate that different habitats are used more for resting than for denning. The greatest abundance of rest sites were found in orange balsam habitat, and lesser numbers were recorded in Lyngby's sedge, mixed herbaceous, and sitka willow habitats.

DISCUSSION AND ANALYSIS

There is not a substantial body of data available on the productivity, abundance, and distribution of river otter in the LCR. We found and cited 7 studies relating specifically to the Columbia River, Washington, or Oregon. There have been no studies on the biophysical sensitivity of otter to environmental contaminants. This lack of data applying directly to the river otter makes it difficult to make any substantive statements concerning its health, morbidity, contaminant burden, or population status in the LCR. Most conclusions are inferential and based on data collected 10 to 15 years ago, or from other sites. Data gaps are apparent and recommendations for further studies can be made. However, the objective of this discussion is not to recommend how to specifically fill these data gaps, but rather, to point out missing or weak areas of information necessary to assess the role of the otter in the Columbia River ecosystem. Emphasis is put on relating past, current and future data to form a comprehensive picture of how the otter population is being affected by contaminant burdens and habitat alteration in the LCR. Ultimately, drawing conclusions based on the measured contaminants levels in otters and their prey species that state "contaminate burdens in the LCR may have negative effects on the river otter population, therefore further research is recommended", should be avoided.

Collecting data on river otter in the LCR is difficult. Mustelids tend to be solitary animals who prefer seclusion from human disturbances, therefore finding individuals of any population for sampling is difficult. If the population in a particular geographic area is low, as it is in the lower Columbia River, then finding individuals to sample is even more difficult. Natural causes, harvest, habitat loss and high levels of chemical contaminants are the four factors attributed to river otter mortality found in this literature survey for otter populations in the lower Columbia River. The activities of humans attributing to habitat loss and high contaminant concentrations are thought to have had the greatest negative impact on the decline of the otter population in the lower Columbia River. Unlike populations of fish species and migratory birds whose home ranges may extend across geographic and international boundaries, mustelid populations appear to utilize a relatively restricted geographic home range. If contaminant residues are high in tissues of otter from the lower Columbia River, then the contaminant level in the otter can be assumed to be directly related to the amount of contaminants being discharged in to the Columbia River system and not some other unrelated system. Contaminants discharged all along the Columbia and its tributaries ultimately contribute to the contaminant load in the LCR (Lebovitz and Everson 1994).

Population Trends of River Otter in the LCR

Studies and literature surveys by Dunn et al. (1984), Merker (1983), Tabor (1976) and Tabor et al. (1980) conclude that riparian habitats are critical for LCR river otters. Tabor (1976) attempted to determine a population estimate of otters in the LCR based on utilization of riparian habitat, and Dunn et al. (1984) also attempted an estimate of otter density in the LCR in relation to observed habitat use. Neither study was able to make a concrete estimate of population size and density because observations occurred in sample sizes too small to statistically analyze.

Data from a table based on harvest reports in 1980 from western Oregon (Table 6) indicates a otter population density of approximately 0.5 to 1.5 otter per 10 linear stream miles in Columbia and Multnomah counties compared to approximately 3 otter per 10 linear stream miles throughout all of western Oregon. The population density of western Oregon is almost equivalent to Erlinge's (1968) definition of a moderate population density of otter (1 per 2.4-3 mi), whereas the density in Columbia and Multnomah counties is equivalent to Erlinge's (1968) definition of a low population density (1 per 4.8-6 mi) The difference in otter density between counties bordering the LCR and all other counties in western Oregon may suggest otter density is lower in Columbia and Multnomah counties due to greater human density and/or high contaminant levels created by human density and industries. Alternatively, the lower density in these counties may be because habitat is better in other areas. It is likely otter density is lower in Columbia and Multnomah counties for all of the above reasons.

Habitat maps of the LCR show considerable habitat loss since the late 1800's and concentrations of industries along the Columbia River in those two counties (Northwest Environmental Advocates 1992; Fox et al. 1984). Harvest reports for the state of Oregon may give some indication of general otter population trends in the counties bordering the lower Columbia River, which in turn, may be attributed to the condition of habitat and water quality in the Columbia River. However, such extrapolations based on harvest reports can be deceptive. Changes in seasonal harvest success does not necessarily reflect changes in the population. Factors more likely to affect harvest success in any particular season are; pelt price on the world wide market, the effort of the trapper (catch per unit effort), or climatic conditions, such as weather. Total catch is more likely a function of the number of trappers seriously trapping otter in a given year and the actual abundance and density of otters in areas which are regularly trapped (Rowbotham-Vita 1982).

Table 6. Otter population estimates and habitat in western Oregon (1980 Statewide Planning Database).

County	No. Otter	Linear Stream (mi.)	Habitat (sq. mi.)
Clatsop	--	--	--
Columbia	40	1,000	1,003
Multnomah	75	400	400
Total Western Oregon	3,500	13,245	13,291

If data from one area or county are compared to the harvest data over a larger area, more accurate trends may be deciphered. For example, seasonal harvests in Clatsop and Columbia counties between 1949 and 1976 counties did not indicate a declining trend. Therefore, Henny et al. (1981) compared otter harvests (pooled in groups of four seasons each) in Clatsop and Columbia counties to the total otter harvest in Oregon to demonstrate that there was a declining trend in the number of otter harvested in Clatsop and Columbia counties

(Table 7). However, the number of trappers declined as well, which may account for the decline in otter harvest. The cause-effect linkages between the decline in the number of otters harvested and the decline in the number of trappers are purely speculative.
Table 8 similarly relates the number of otter legally harvested in Clatsop, Columbia and Multnomah counties to the number of otter harvested in all of western Oregon between 1976 and 1994 (except 1990 and 1991). The market pelt price was also included to see if fluctuations in the number of otter harvested coincided with changes in pelt price. No trends in the number of otter harvested seasonally were apparent from these data. Changes in the price of pelts did not seem to create drastic fluctuations in the harvest either. The number of trappers each season, which may also have an effect on the number of otter harvested, was neglected. However, since there was no apparent trend in the number of otter harvested, the number of trappers, as with the pelt price, may not offer any enlightening conclusions in this instance.

The productivity rates of otter populations in Idaho (Melquist and Hornocker 1983), the Puget Sound-Hood Canal (Rowbotham-Vita 1982) and western Oregon (Tabor and Wight 1977) were estimated as being relatively stable. However, in all of these studies minimal reference was made to contaminant burdens in the population and their potential effects on productivity. Although Tabor and Wight (1977) estimated that the otter population in western Oregon was stable, including populations in Clatsop, Columbia and Multnomah counties, only 5-7 percent of their data came from otters trapped in the LCR. If the river otter population in the LCR is actually low because of high contaminant burdens, then a larger sample from the LCR may have unbalanced their estimate of the status of the otter population in Western Oregon. Tabor and Wight (1977) stated that the population can be expected to remain constant if the balance between mortality and recruitment indicated in their study is maintained. An obvious question is, what will happen if the balance is not maintained, and what determines that balance?

Table 7. River otter harvest data for Columbia and Clatsop (C-C) Counties as related to state harvest in Oregon (from Annual Reports, Oregon Department of Fish and Wildlife) (Henny et al. 1981).

Trapping Seasons[a]	Number Trappers			Number Otter Harvested	C-C
	Oregon	C-C	% Total	Oregon / C-C	% Otter Harvest
73-76	5,584	382	6.8	1,352 / 76	5.6
69-72	3,072	239	7.8	1,020 / 52	5.1
65-68	3,055	281	9.2	1,274 / 103	8.1
61-64	3,096	279	9.0	1,201 / 144	12.0
57-60	3,921	480	12.2	1,209 / 142	11.7
53-56	4,203	569	13.5	960 / 163	17.0
49-52	6,065	852	14.0	734 / 169	23.0

[a]four trapping seasons pooled.

Table 8. River otter harvest[1] data for clatsop Columbia and Multnomah (C-C-M) counties as related to state harvest in western Oregon.

Trapping Season	Pelt Price	Number Otter Harvested C-C-M / Western Oregon	Percent Otter Harvest C-C-M
1976-77	$57	21 / 379	6
1977-78	45	17 / 227	7
1978-79	57	44 / 371	12
1979-80	51	43 / 489	9
1980-81	41	17 / 321	5
1981-82	22	18 / 236	8
1982-83	33	31 / 281	11
1983-84	35	35 / 156	22
1984-85	18	42 / 357	12
1985-86	23	27 / 310	9
1986-87	29	37 / 362	10
1987-88	27	49 / 402	12
1988-89	27	31 / 234	13
1989-90	27	42 / 266	16
1992-93	NA	18 / 247	7
1993-94	NA	65 / 358	18

[1]annual harvest reports, Oregon Dept. of Fish and Wildlife (1976-1994).

Without an accurate population estimate, or knowledge of mortality and natality rates within the river otter population in the LCR, it is difficult to assess the factors that may influence mortality and natality. However, based on this literature survey three factors, individually and in combination, are likely to affect the balance between mortality and recruitment in an otter population; 1) habitat alteration and suitability, 2) prey abundance as determined by habitat suitability and contaminant burdens, and 3) contaminant burdens in the otters as determined by contaminant burdens in prey and habitat.

Habitat Suitability and Alteration

As discussed in the findings section of this report, otter prefer marshes and sloughs that have complex networks of tidal channels. Otters are usually scarce in heavily settled areas and in polluted waterways. Availability of shelter is as important as availability of food. Therefore, flat shorelines, or shorelines with no undercuts will be virtually unused regardless of prey abundances. Preference for non-estuarine habitats such as small feeder streams, is likely to limit the use of the LCR to sloughs, ponds and embayments along the main river body and will therefore limit the presence of river otter. As observed by Melquist and Hornocker (1983), regional differences in habitat use by otters existed primarily because of differences in habitat composition. Definitive studies of habitat selection and use at sites of varying quality and otter use within the LCR are required to identify the relative importance of variables in the habitat which determine use by otter. Correlating those data with the history of habitat alteration of the estuary and lower river in the last 100 years may in itself provide a significant explanation for the extremely low otter density in the area (density as determined by Dunn et al. 1984 and Tabor 1976).

Frenton and Merker (1983) determined swamps were used for feeding by more than 50 percent of the regional population of otters, and low marshes were used for feeding by less than 20 percent. Between 20 and 50 percent of the regional otter population are believed to use marshes for both feeding and denning and low marshes used by less than 20 percent of the regional otter population. Swamp and low marsh were shown to occur in Cathlamet and Grays Bay (RM 20) through the end of Puget Island at RM 45. Low marshes were also shown to occur in small portions of Trestle Bay (RM 6-8) and Baker Bay (RM 3-5). Swamp habitat is described as tidal wetlands receiving irregular tidal inundation, with lower limits of vegetation types between 6.5 and 8.5 feet above the mean lower low water (MLLW, -2 ft.) and an upper limit between 8.0 and 12 feet above MLLW. Low marshes are described as tidal wetlands receiving regular tidal inundation, with lower limits of elevation at 3 feet above MLLW and an upper limit of vegetation type between 6.5 and 8.5 feet above MLLW. The irregular tidal inundation of the swamp habitat probably provides otter with access to preferred foraging areas e.g., shallow tidal pools with concentrations of prey species and exposed muddy bottoms, for extended periods of time. Whereas the regular inundation of low marsh habitat may provide the same preferred foraging areas, but for shorter periods of time. It may also be possible that differences in the tidal inundation affect the concentrations of prey within the tidal pools.

In the last century the surface area of tidal marshes and swamps have been reduced by 65 percent, mainly the result of diking. The total estuary is three-quarters of what it was in 1870 (Thomas 1983). Almost 37,000 acres of the total estuarine acreage have been converted to diked floodplains, uplands, and non-estuarine wetlands. The tidal swamp area has been the most dramatically altered with a reduction of 77 percent of its area, again primarily the result of diking. If the otter population in the LCR used as much swamp and lowland marshes in the past as thought to be presently used in the estuary, then otters have suffered a substantial loss in habitat quantity and suitability.

Chemical Contamination of Habitat, Prey and Otters

The major chemical contaminant groups addressed in this report; 2,3,7,8-tetrachlorodibenzon-p-dioxin (TCDD), polychlorinated biphenyls (PCBs), dichloro-diphenyl-trichloroethane (DDT), dichloro-diphenyl-dichloroethylene (DDE) and heavy metals, are directly discharged into the LCR from approximately 33 major and minor industries between river miles 0 and 145 (Lebovitz and Everson 1994). Sewage, agricultural run-off and urban run-off are other sources of chemical contamination, particularly in areas with high human population density. Additionally, the LCR receives contaminant loads from industries, agriculture and urban development in upstream portions of the Columbia and from tributaries that flow into the Columbia. In general, sources of contaminants are as follows: DDT and DDE from agriculture and pesticides, dioxins and furans from pulp mills, PCBs from industrial discharge and dams, and heavy metals from industrial discharge. Chemicals can be introduced and accumulated in bodies of water by several different pathways; sediment uptake and release, chemical degradation, chemical and photochemical formation, direct absorption, fallout, out sprat transfer, and volatization (Lebovitz and Everson 1994). Based on the data reviewed in this literature survey, to assess the population status of otter it is necessary to analyze multi-contaminant levels in otters from the LCR whose compounded effects are influenced by, not only, direct intake of contaminants from prey, but also by the levels in the water and sediment. Contaminants, once in the otter, may have aggravated effects due to stress induced by reproduction, habitat loss, or increased foraging activity caused by decreased prey availability.

Accumulation of these chemicals in the biota of a system usually results from transport in the food chain. Food chains act as a biological amplifier. Total biomass is reduced with each trophic level in the food chain and losses of such compounds as DDT and PCBs are relatively small, so concentrations increase with each trophic level (Ehrlich et al. 1977). Accumulation in the river otter is likely to result from the following simplified pathway. Fish and crayfish accumulate residues from intestinal absorption of residues in their prey, or from dermal absorption of residues in the water. Residues in fish will bioaccumulate to levels higher than ambient water. Once an otter eats a contaminated fish or crayfish it absorbs the accumulated residues. Residues are then stored in the tissues of the otter and released, in either subtoxic or toxic levels, in times of stress. Additionally, the residues are passed on to an unborn otter through the placenta, and to kits through mammary glands during lactation. If a female otter accumulates subtoxic levels of contaminants, it is likely that a kit will have accumulated the same contaminant residues before birth, will accumulate more while nursing, and still more once it begins to prey on fish and crayfish. Eventually the subtoxic accumulations will begin affecting productivity and kit survivorship. Pathways taken by each type of contaminant, individually or in combination, will vary, as will the subtoxic or toxic effects on a species. Two issues exist for the otter population in the LCR in relation to contaminant levels that warrant further research. One, the levels of contaminants in the environment and their prey are unknown, and two, the level of biophysiological sensitivity in otter to any one contaminant, or any combination of contaminants.

Few data were found in this literature survey on contaminants levels in prey species of otter in the LCR. There are studies presently being conducted that are measuring contaminants levels, abundance and distribution of crayfish, sculpin, perch, squawfish and sucker in the LCR by Tetra Tech and the U.S Fish and Wildlife Service. Data from these studies may be related to contaminant levels, and the abundance and distribution of LCR river otters, also presently being studied. There is concern over the status of the river otter in the LCR based on the contaminant levels measured in otter and reported by Henny et al. in 1981. As stated above, based on the mechanisms of biomagnification, any contaminant residue in prey species of otter will ultimately become residues in river otter. However, there are no data from the last 15 years estimating the population (besides based on harvest reports), or quantitatively linking the health of the otter population to habitat suitability, the health of prey populations, water quality in the LCR, or the amount of chemical discharge into the river.

Table 9 correlates the number of otter observations made by Tabor (1976) in the LCR (which are assumed to reflect the density of otter in the same area) with the number of industries in the same area (NWEA 1992). Foley et al. (1988) concluded that otter are more susceptible to the influence of local sources of PCB and DDE. Therefore, areas where concentrations of PCBs and DDE are high may cause suppression or elimination of an otter population. Attempting to prove the hypothesis that otter density will be low in areas with high levels of contaminants concentrated in proximity to their sources is difficult based on the data from Table 8. The Table does indicate that where there are industries there are few, if any otter. However, the sample size is limited. Otter may not inhabit areas near industries where contaminants are more concentrated, but how far from the source of the contamination are the concentrations dilute enough to permit otter populations and their prey? The absence of otter may also be because the areas associated with industries support a greater human population, otters will tolerate human disturbances but prefer to avoid them. Prey abundance and habitat suitability are known to determine what habitat an otter uses, how does the contaminant concentrations in certain areas affect prey species?

Table 9. Locations of industry discharge and otter/mink observations in the LCR from RM 0 to RM 145.

RM (River Mile)	No. of observations		Approximate Number of Industries	Types of Industry	Possible Contaminant Discharges
	Otter	Mink			
0-12	1	1	8	Food Processing	Nutrients - nitrogen & phosphorus, T[illegible] BOD
12-79	3	1	9	Pulp & Paper Mills Lumber & Plywood Mills Chemical Industries Power Generating Plants	Dioxins, heavy metals, PCBs
79-145			16	Pulp & Paper Mills Lumber & Plywood Mills Chemical Industries/Misc. Toxic Sites	Dioxins, heavy metals, urban runoff, PCBs
125-133	10	2	0		

Harvest

The impacts of harvest on the otter population in the LCR is questionable and may be another area where further research would be valuable in assessing the population status of otter. Otter are susceptible to over harvest, particularly males, because of their large home ranges. Dunn et al. (1984) concluded that harvest may negatively impact the otter population in the Columbia River estuary. Rowbotham-Vita (1982) doubted the otter population in the Puget Sound-Hood Canal area could sustain under heavy harvesting experienced prior to her study. According to the Oregon Department of Fish and Wildlife (ODFW) 1976-1994 Annual Furbearer Harvest Reports, harvest has not been found to cause extensive negative impacts on otter populations in Oregon. However, the ODFW data only account for otter harvested legally. Illegal harvest may have a significant impact on the otter population, but can not be measured. An alternate analysis of the impacts of harvest in the LCR would be to compare harvest reports with a population estimate obtained through other means besides harvest data. However, such a population estimate does not exist to date.

Data Gaps and Some Suggestions

All of the issues discussed here are representative of data gaps and the need for further research. Little is known about the productivity, contaminant burdens, and habitat utilization of river otter in the LCR. Based on this discussion, some of the most fruitful areas for priority work might include:

- An estimate of the current population to better assess impacts of contaminants and habitat alteration on the otter population;

- Determination of the biophysiological sensitivity of adult and juvenile otter, kits, fetuses, and embryos;

- Measurement of the contaminant burden in the present population to compare with those measured by Henny et al. in 1981. Such a comparison may then offer insights into the contaminant dynamics and burdens in prey species and in the river itself;

- Measurement of the contaminant burden in populations of prey species;

- Determination of factors influencing habitat preference and selection;

- Comparison of the contaminant levels in the otter and their prey and determination of the extent of biomagnification of contaminants in the system;

- Correlation of contaminant and population data to habitat type and location within the river to determine which areas are better suited for support of prey populations and otter populations.

The above are obviously substantial data gaps concerning the LCR river otter population. Some are in the process of being filled, others may be lined up in future research proposals. A major conclusion derived from this literature survey is that data collected concerning the status of contaminants in the otter population in the LCR will be of little or no use unless the results and conclusions are correlated with the biophysical dynamics of the Columbia River, including the interrelated compounding variables of habitat degradation and environmental contamination.

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RESPONSE TO PEER REVIEW COMMENTS

Comments from Washington Department of Ecology and Oregon Department of Environmental Quality staff were incorporated into this final product. Additional comments were received from Al Whitford, Longview Fibre Company, Tech Services, Environmental Section.

Comments from Al Whitford, Longview Fibre Company:

Comment 1. Comments on the cover letter.

Response 1. The cover letter will not be included with the final draft.

Comment 2. Typographical errors in the table of contents.

Response 2. Comments noted.

Comment 3. Did the brief examination of major contaminant sources above Bonneville Dam increase the cost of the project.

Response 3. No.

Comment 4. Page numbering system is confusing.

Response 4. We have tried to improve the format of the report.

Comment 5. The report should mention that metals also occur in the environment naturally.

Response 5. Comment noted.

Comment 6. This draft needs an extensive peer review.

Response 6. Comment noted.

Comment 7. Water quality measurements for some metals are confused with tissue measurements for DDD/DDE and PCBs.

Response 7. Comment noted.

Comment 8. Comments on the largescale sucker executive summary.

Response 8. All the executive summaries were combined summary at the beginning of the document. Other comments noted.