PORTLAND HARBOR RI

APPENDIX G
BASELINE ECOLOGICAL RISK ASSESSMENT
ATTACHMENT 13
DETAILS ON EXPOSURE AND EFFECTS ASSUMPTIONS FOR THE FISH DIETARY LOE

DRAFT

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Prepared for
The Lower Willamette Group

Prepared by
Windward Environmental LLC

TABLE OF CONTENTS

LIST OF TABLES iii

1.0 INTRODUCTION 1

2.0 DIETARY ASSESSMENT EXPOSURE ASSUMPTION DETAILS 2
2.1 Largescale Sucker 4
2.1.1 Body Weight and Daily Food Ingestion Rate 4
2.1.2 Incidental Sediment Ingestion Rate 5
2.1.3 Diet Composition 5
2.1.4 Scale of Exposure Areas 6
2.2 Juvenile White Sturgeon 6
2.2.1 Body Weight and Daily Food Ingestion Rate 6
2.2.2 Incidental Sediment Ingestion Rate 6
2.2.3 Diet Composition 7
2.2.4 Scale of Exposure Areas 8
2.3 Juvenile Chinook Salmon 9
2.3.1 Body Weight and Daily Food Ingestion Rate 9
2.3.2 Incidental Sediment Ingestion Rate 9
2.3.3 Diet Composition 9
2.3.4 Scale of Exposure Areas 10
2.4 Peamouth 11
2.4.1 Body Weight and Daily Food Ingestion Rate 11
2.4.2 Incidental Sediment Ingestion Rate 11
2.4.3 Diet Composition 11
2.4.4 Scale of Exposure Areas 12
2.5 Sculpin 12
2.5.1 Body Weight and Daily Food Ingestion Rate 12
2.5.2 Incidental Sediment Ingestion Rate 13
2.5.3 Diet Composition 13
2.5.4 Scale of Exposure Areas 14
2.6 Smallmouth Bass 14
2.6.1 Body Weight and Daily Food Ingestion Rate 14
2.6.2 Incidental Sediment Ingestion Rate 14
2.6.3 Diet Composition 14
2.6.4 Scale of Exposure Areas 15
2.7 Northern Pikeminnow 16
2.7.1 Body Weight and Daily Food Ingestion Rate 16
2.7.2 Incidental Sediment Ingestion Rate 16
2.7.3 Diet Composition 16
2.7.4 Scale of Exposure Areas 17

3.0 DIETARY ASSESSMENT EFFECT ASSUMPTIONS 18
3.1 Cadmium 18
3.2 Copper 19

3.3 Mercury 19
3.4 Tributyltin 20
3.5 Benzo(a)pyrene 20
3.6 Total PAHs 20

4.0 REFERENCES 22

LIST OF TABLES

Table 2-1. Exposure Parameters use for Fish Dietary Risk Calculations.....2
Table 2-2. Receptor-Specific Prey Species Used to Derive Risk Estimates Based on Single Prey Consumption.....3
Table 2-3. Receptor-Specific Prey Species and Portions Used to Derive Risk Estimates Based on Multiple Prey Consumption.....4
Table 3-1. Fish Dietary Dose TRVs.....18

1.0 INTRODUCTION

Section 7.2 of the baseline ecological risk assessment (BERA) presents the dietary assessment for fish. This attachment presents the details and sources for the exposure parameters and effects thresholds (i.e., toxicity reference values [TRVs]) presented in Sections 7.2.2 and 7.2.3, respectively, of the BERA. Section 2.0 of this attachment presents the details of the exposure assumptions, and Section 3.0 presents the details of the selected fish diet TRVs.

2.0 DIETARY ASSESSMENT EXPOSURE ASSUMPTION DETAILS

Section 7.2 of the BERA presents a summary of the fish exposure parameters used in the dietary assessment for fish. Table 2-1 presents the exposure parameters used in the fish dietary risk calculations. Body weights were based on the average body weights measured in individual fish from the Rounds 1, 2, and 3 sampling efforts. Measured food ingestion rates were not available for the fish receptors and were estimated using the equation presented in Arnot and Gobas (2004):

$$\mathrm{FIR} = (0.022 \times \mathrm{BW}^{0.85}) \times \exp^{(0.06*T)} \qquad \textbf{Equation 2-1}$$

Where:

FIR = food ingestion rate (mg/kg bw/day)
BW = body weight (kg)
exp = 2.71828
T = temperature (degrees Celsius)

Dietary fish ingestion rates were estimated using Equation 2-1 and assuming a temperature of 13.4 ºC, which was based on the average data collected by the Oregon Department of Environmental Quality (ODEQ) from a station within the Study Area (SP&S Railroad Bridge) from 1995 to 2005 (ODEQ 2005). Per US Environmental Protection Agency's (EPA's) Problem Formulation (Attachment 2), fish food ingestion rates were also calculated using the high-end water temperature (16.2 ºC). A comparison of calculated ingestion rates and the impact on hazard quotient (HQ) calculations is presented in the fish dietary assessment of the BERA.

Table 2-1. Exposure Parameters use for Fish Dietary Risk Calculations

Receptor	BW (kg)[a]	FIR (kg ww/day)[b]	SI (%)[c]	SIR (kg dw/day)[d]	% Moisture in Prey	SUF	Exposure Scale
Largescale sucker	0.79	0.040	8%	0.00048	85%[e]	1.0	Site-wide
Juvenile white sturgeon	7.6	0.28	8%; 56%[f]	0.0033; 0.023[f]	85%[e]	1.0	Site-wide
Juvenile Chinook salmon	0.012	0.0011	1%	0.0000024	79%[g]	1.0	Site-wide
Peamouth	0.10	0.0072	5%	0.000075	79%[g]	1.0	Site-wide
Sculpin	0.020	0.0017	5%	0.000018	79%[g]	1.0	0.1 mile
Smallmouth bass	0.40	0.022	1%	0.000058	74%[h]	1.0	1 mile
Northern pikeminnow	0.56	0.030	1%	0.000078	74%[h]	1.0	1 mile

[a] Body weights are based on field-collected data (including Round 3 data).

[b] FIR was calculated based on the equation from Arnot and Gobas (2004): FIR (ww) = (0.022 x $BW^{0.85}$) x ($\exp^{(0.06 \times T)}$); where exp = 2.71828 and T = 13.4°C (average of temperatures collected by ODEQ from 1995 to 2005 from a station near the SP&S Railroad Bridge).

[c] Percent of incidental sediment ingestion of the dry diet.

[d] SIR = FIR x SI. The SIR was calculated as a percent of the FIR on a dw basis. The dw FIR was calculated based on the following equation: FIR (dw) = FIR (ww) x (1 – moisture content of diet).

[e] Average percent moisture of invertebrate tissue analyzed from the Study Area.
[f] Two SI scenarios (8% and 56%) were evaluated for juvenile white sturgeon.
[g] Average percent moisture of invertebrate tissue (excluding laboratory-exposed clams and crayfish) analyzed from the Study Area.
[h] Average percent moisture of fish tissue analyzed from the Study Area.
BW – body weight
dw – dry weight
FIR – food ingestion rate
ODEQ – Oregon Department of Environmental Quality
SI – sediment ingestion
SIR – sediment ingestion rate
SUF – site use factor
ww – wet weight

Tables 2-2 and 2-3 present the dietary prey assumptions used to derive risk estimates based on the ingestion of individual and multiple prey species. Details on how prey assumptions were used are presented in the BERA.

Table 2-2. Receptor-Specific Prey Species Used to Derive Risk Estimates Based on Single Prey Consumption

Prey Species	Large-scale Sucker	Juvenile White Sturgeon	Juvenile Chinook Salmon	Sculpin	Peamouth	Smallmouth Bass	Northern Pike-minnow
Invertebrates							
Clam	X[a]	X[a]	X[a]	X[a]	X[a]		
Worm	X	X	X	X	X	X	X
Crayfish						X	X
Mussel		X					
Epibenthic invertebrates			X		X		
Fish							
Largescale sucker							X
Carp							X
Peamouth							X
Sculpin				X	X	X	X
Northern pikeminnow							X
Stomach Contents							
Juvenile white sturgeon		X					
Juvenile Chinook salmon			X				

[a] Both laboratory clam tissue and field-collected tissue concentrations were evaluated based on single prey consumption.
COPC – chemical of potential concern

Table 2-3. Receptor-Specific Prey Species and Portions Used to Derive Risk Estimates Based on Multiple Prey Consumption

Prey Species	Large-scale Sucker[a]	Juvenile White Sturgeon[a]	Juvenile Chinook Salmon[b]	Sculpin[c]	Peamouth	Small-mouth Bass	Northern Pike-minnow
Invertebrates							
Clam[d]	1.0and 0	1.0 and 0	0.30 and 1.0	1.0 and 0	0.25		
Worm[e]	1.0 and 0	1.0 and 0	0.40 and 1.0	1.0 and 0	0.25	0.05[f]	0.25[f]
Crayfish						0.05[g]	0.30[g]
Epibenthic invertebrates			0.30[h] and 1.0		0.40[h]		
Fish							
Largescale sucker							0.05[i]
Carp							0.05
Peamouth							0.05[i]
Sculpin				1.0 and 0	0.10	0.90	0.25[j]
Northern pikeminnow							0.05[i]

[a] Two scenarios were evaluated for largescale sucker and juvenile white sturgeon: one based on the ingestion of clams and one based on the ingestion on worms.
[b] Two scenarios were evaluated for juvenile Chinook salmon: one based on the ingestion of 30% clams, 40% worms, and 30% epibenthic invertebrates, one based on 100% ingestion of epibenthic invertebrates.
[c] Sculpin prey were each evaluated individually on a sample-by-sample basis.
[d] HQs were calculated using field clam tissue only. Tissues from field clams are more representative of field conditions in the Study Area than are tissues from laboratory-exposed clams.
[e] HQs were calculated using laboratory-exposed worms.
[f] Crayfish were used as a surrogate when no worm tissue data were available.
[g] Worms were used as a surrogate when no crayfish tissue data were available.
[h] Clams and worms were used as a surrogate when no epibenthic inverterbrate tissue data were available.
[i] Sculpin were used as a surrogate when no largescale sucker, peamouth, or northern pikeminnow tissue data were available.
[j] Carp was used as a surrogate when no sculpin tissue data were available.

The following subsections present the receptor-specific exposure assumptions used in the dietary line of evidence (LOE) for fish for each receptor.

2.1 LARGESCALE SUCKER

This section provides the rationale for largescale sucker-specific exposure parameters, as summarized in Tables 2-1 and 2-2.

2.1.1 Body Weight and Daily Food Ingestion Rate

The average body weight reported for individual largescale suckers collected in Round 1 sampling was 794 g. This body weight was used with Equation 2-1 to estimate a food ingestion rate of 0.041 kg ww/day for largescale sucker.

2.1.2 Incidental Sediment Ingestion Rate

The largescale sucker prefers to remain close to the bottom in shallow waters, both as a juvenile and as an adult, and is primarily a bottom-feeder. This native fish is known to consume large amounts of sediment during feeding as an adult (CBFWA 1996), and sediment was noted in the stomach content analyses of four largescale suckers collected during Round 1 sampling (Attachment B8 of Appendix B of the programmatic work plan [Integral et al. 2004]). Up to 30% of the gut content was unidentified organic matter from a largescale sucker from Lake Roosevelt, Washington (Spotts et al. 2002). Charles Lee, a biologist who analyzed the stomach contents of largescale sucker, estimated that based on his observation, 10% or less of the unidentified organic material was likely to be sediment (Lee 2006). Based on the information from Lee (2006) and Spotts et al. (2002), 3% of total diet is a conservative estimate of the dietary sediment fraction. Because sediment consumption has not been measured but large quantities of sediment have been noted in largescale sucker diets, an incidental sediment ingestion of 8% of the dry diet ingestion rate was conservatively assumed. Using the calculated food ingestion rate and an assumption of 85% moisture in the diet (average percent moisture of invertebrate tissue analyzed from the Study Area), the estimated sediment ingestion rate for largescale sucker was 0.00048 kg dw/day.

2.1.3 Diet Composition

Largescale suckers live in close proximity to sediment, and benthic invertebrates are a primary component of their diet. Adult largescale suckers almost exclusively consume benthic organisms, detritus, and plant material associated with the bottom, including macrophytes, algae, diatoms, detritus and organic material, crustaceans, aquatic insects, and small mollusks (Carl 1936; CBFWA 1996; Spotts et al. 2002). The stomach contents of four largescale suckers collected during Round 1 sampling were analyzed, and prey items included bivalves (*Corbicula* sp.), chironomids, oligochaetes, bryozoans, gastropods, algae, sediment, and detritus (Attachment B8 of the programmatic work plan [Integral et al. 2004]). Field benthic invertebrate tissue data were collected from the Study Area (i.e., clams), and benthic invertebrate tissue chemical concentrations were measured in laboratory bioaccumulation tests using Study Area sediment. Plant and algae tissue are not available in the BERA dataset.

For the evaluation of individual prey, the exposure of largescale sucker via dietary ingestion was modeled using field-collected clam tissue, as well as clam and worm tissue from the bioaccumulation testing (based on surface sediment collected from within the Study Area). In the evaluation of multiple prey species, individual prey portions were not assigned to each of the representative prey for largescale sucker because both clam and worms are meant to be representative of benthic invertebrate prey that suckers may ingest. Instead, the dietary exposure of largescale sucker was modeled separately using 100% ingestion of field clam tissue[1] and 100% ingestion of worm tissue from the bioaccumulation testing.

[1] Field clams were assumed to be more representative of chemical concentrations in bivalves from the Study Area than lab clams.

There is some uncertainty associated with the use of the available benthic prey items as the only prey for largescale sucker, inasmuch as suckers also feed on plant material. Dietary prey estimated concentrations may be over or underestimated for largescale sucker. The use of clam and worm tissue chemical concentrations based on 28-day laboratory bioaccumulation testing as representative of benthic prey living in the Study Area is also uncertain. Field and steady-state conditions may not be represented by tissue chemical concentrations measured in laboratory testing conditions. Clam and worm tissue concentrations of neutral organic chemicals of potential concern (COPCs) were adjusted to reflect steady-state concentrations using the process in the Inland Testing Manual (EPA and USACE 1998). Attachment 3 presents the methods used to derive steady-state concentrations. As with the use of any model, the equations (based on McFarland (1995) and assumptions (i.e., K_{OW} value) associated with the steady-state adjusted concentrations are uncertain, and adjusted worm tissue chemical concentrations may over or underestimate steady-state concentrations expected in worms in the Study Area.

2.1.4 Scale of Exposure Areas

The Study Area encompasses approximately a 10-mile (16 km) reach from approximately RM 2.0 to RM 12. This site-wide scale is a reasonable assumption for fish with home ranges larger than or approximately equal to the Study Area (i.e., largescale sucker). Largescale suckers have been recorded to have home ranges up to 37 miles (60 km), and many Columbia River largescale suckers are found to be transients (Dauble 1986).

2.2 JUVENILE WHITE STURGEON

This section provides the rationale for juvenile white sturgeon-specific exposure parameters, as summarized in Tables 2-1 and 2-2.

2.2.1 Body Weight and Daily Food Ingestion Rate

The average body weight reported for individual juvenile (pre-breeding) white sturgeon collected in Round 3 sampling was 7.6 kg. This body weight was used with Equation 2-1 to estimate a food ingestion rate of 0.27 kg ww/day.

2.2.2 Incidental Sediment Ingestion Rate

White sturgeon are benthic feeders and are in frequent contact with sediments. Sturgeon identify prey on the bottom surface using their long barbels and then extend their mouths and suck up the prey item (USFWS 1961). In a study of 42 juvenile sturgeon (ranging from 8.66 to 32.3 inches [22 to 82 cm] fork length) from the lower Columbia River, sand mixed with unidentifiable body parts of invertebrate prey items accounted for about 56% of the total wet mass of stomach contents (Romano et al. 2002). Juvenile sturgeon collected from the Study Area were larger, ranging from 35.8 to 43.3 inches (91 to 110 cm) in fork length. Based on expert judgment, an incidental sediment ingestion rate of 8% was selected.

Based on the above, two sediment ingestion scenarios were developed. In Scenario 1, it was estimated that white sturgeon ingest sediment equal to 8% of the volume of food typically ingested. In Scenario 2, it was estimated that white sturgeon ingest sediment equal to 56%

of the volume of food typically ingested. Using the food ingestion rate and an assumption of 85% moisture in the diet (average percent moisture of invertebrate tissue analyzed from the Study Area), the estimated sediment ingestion rates for pre-breeding white sturgeon were 0.0033 and 0.023 kg dw/day for Scenarios 1 and 2, respectively.

2.2.3 Diet Composition

White sturgeon are opportunistic benthic feeders and may feed on crustaceans (e.g., crayfish), shrimp, mollusks, insects and insect larvae, annelids, and small fish (USFWS 1961; Radtke 1966; Beamesderfer and Farr 1997; Wydoski and Whitney 1979). Juvenile white sturgeon feed on amphipods and mysid shrimp (McCabe et al. 1993; Radtke 1966).

Columbia River white sturgeon have been reported to feed almost exclusively on amphipods (primarily on *Corophium* spp.) (Romano et al. 2002). In another study, Columbia River white sturgeon were found to prey on small bottom-dwelling mollusks, snails, insect larvae, and crustaceans (Bajkov 1949). Larger sturgeon have been found to feed on fish (Bajkov 1949), including large fish such as full-grown sockeye salmon, northern pikeminnow, and sucker (USFWS 1961). Immature white sturgeons in the Lower Columbia River feed primarily on amphipods and occasionally on bivalves (*Corbicula fluminea*), chironomid larvae, and fish eggs (McCabe et al. 1993). In the Bonneville and The Dalles pools of the Columbia River, amphipods (almost exclusively *Corophium* sp.) made up 78 and 72% of subyearling and yearling stomach contents, respectively, based on weight, reported in Beamesderfer and Nigro (1992). In the same study, mysids (*Neomysis mercedis*) made up 15%, and mollusks (*Corbicula* sp.) made up 12% of the yearling white sturgeon stomach contents by weight. Other items in yearling and subyearling white sturgeons included aquatic insects, oligochaetes, and unidentifiable digested material.

The stomach contents of juvenile sturgeon collected from the Study Area during Round 3 sampling were removed, and prey items and digestive material were examined. Prey items in the stomachs were identified to the lowest taxonomy possible. Stomach contents also underwent chemical analysis to estimate chemical concentrations in prey items. Seven of the fifteen sturgeon had identifiable prey remains in their stomachs.[2] Shrimp were the most commonly observed item; they were observed in five of the seven stomachs. Crayfish remains were observed in two of the stomachs, one of which contained the remains of 15 crayfish. Additional prey items included fish remains (observed in three stomach samples), aquatic insects (i.e., *Chironomid* sp. and stoneflies) (observed in three stomach samples), and amphipods (e.g., *Corophium* sp.) (observed in two stomach samples); and other crustaceans (mysids) (observed in one stomach sample).

Benthic invertebrate tissue data, including bivalve tissue (*Corbicula* sp.) and worm tissue, were collected in Round 2 sampling; however, other benthic prey tissue such as insect larvae and amphipods are not available in the BERA dataset. For the evaluation of individual prey, the pre-breeding white sturgeon diet was modeled using worm tissue from the bioaccumulation testing (to represent benthic invertebrates living in sediments), field

[2] Two of the fifteen sturgeon had empty stomachs, and six of the fifteen had unidentifiable remains or biofluid.

clams and clam tissue from the bioaccumulation testing, and mussels to estimate dietary exposure concentrations. Juvenile sturgeon stomach tissues were also used to represent dietary exposure concentrations. In the evaluation of multiple prey species, individual prey portions were not assigned to each of the representative prey for juvenile white sturgeon because both clam and worms are meant to be representative of benthic invertebrate prey that juvenile sturgeon may ingest. Instead, dietary exposure of juvenile white sturgeon was modeled separately using 100% ingestion of field clam tissue[3] and 100% ingestion of worm tissue from the bioaccumulation testing.

There is some uncertainty associated with the use of the available benthic prey items as the only prey for juvenile sturgeon because sturgeon also feed on various other organisms. Dietary prey estimated concentrations may be over or underestimated for juvenile sturgeon. The use of clam and worm tissue chemical concentrations based on 28-day laboratory bioaccumulation testing as representative of benthic prey living in the Study Area is also uncertain. Field and steady-state conditions may not be represented by tissue chemical concentrations measured in laboratory testing conditions. Clam and worm tissue concentrations of neutral organic COPCs were adjusted to reflect steady-state concentrations using the process in the Inland Testing Manual (EPA and USACE 1998). Attachment 3 presents the methods used to derive steady-state concentrations. As with the use of any model, the equations (based on McFarland (1995)) and assumptions (i.e., K_{OW} value) associated with the steady-state adjusted concentrations are uncertain, and adjusted worm tissue chemical concentrations may over or underestimate steady-state concentrations expected in worms in the Study Area.

2.2.4 Scale of Exposure Areas

Some studies suggest that sturgeon can show strong site fidelity (Veinott et al. 1999), while other studies indicate individual sturgeon can have large ranges (DeVore and Grimes 1993). Juvenile (pre-breeding) white sturgeon were targeted for collection from the Study Area based on the assumption that sturgeon in the juvenile lifestage would have the longest exposure time of any sturgeon lifestage because they are migratory fish. The average age of sturgeon collected from the Study Area during Round 3 was 13 years old (ages ranged from 7 to 26 years).[4] The median age at sexual maturity for white sturgeon in the lower Columbia River is 24 years (DeVore et al. 1995).

Although none of the Round 3 pre-breeding sturgeon caught from the Study Area were passive integrated transponder (PIT) tagged, one legal-sized sturgeon that was collected and analyzed from the Study Area in March 2007 as part of Round 3 sampling had been previously captured by Washington Department of Fish and Wildlife (WDFW) and was tagged with a spaghetti wire. The age of this tagged sturgeon based on a pectoral fin ray sample was 7 years old. Per WDFW (2007), the sturgeon was originally tagged on June 6, 2006, at Rocky Point, which is located along the west shore of Grays Bay near the

[3] Field clams were assumed to be more representative of chemical concentrations in bivalves from the Study Area than lab clams.
[4] Age analysis of juvenile sturgeon was determined by Ruth Farr and Michele Weaver at ODFW using pectoral fin ray samples following ODFW protocols (Beamesderfer et al. 1998).

Pacific/Wahkiakum Counties border on the Washington side of the Columbia River. The initial tagging location was approximately 72 miles (116 km) from the location where the sturgeon was collected in the Study Area, supporting a much larger home range assumption for juvenile sturgeon than the 10-mile (16 km) stretch of the Study Area.

The entire 10-mile (16 km) reach of the Study Area was assumed to represent the exposure scale for juvenile sturgeon. However, there is uncertainty with this assumption due to the large home range of sturgeon, even during the juvenile lifestage, as indicated by the literature and by the tagged sturgeon collected during Round 3. Estimated dietary exposure concentrations may be over or underestimated for juvenile sturgeon. Their exposure area is likely much greater than the Study Area, and they may be exposed to additional contaminants outside of the Study Area.

2.3 JUVENILE CHINOOK SALMON

This section provides the rationale for juvenile Chinook salmon-specific exposure parameters, as summarized in Tables 2-1 and 2-2.

2.3.1 Body Weight and Daily Food Ingestion Rate

The average body weight reported for individual juvenile Chinook salmon collected in Round 1 sampling was 12 g. This body weight was used with Equation 2-1 to estimate a food ingestion rate of 0.0011 kg ww/day for juvenile Chinook salmon.

2.3.2 Incidental Sediment Ingestion Rate

Low numbers of benthic invertebrates have been reported in the diet of subyearling Chinook salmon, suggesting that they are likely feeding in the water column (Vile et al. 2005) and are thus infrequently exposed to sediments. In addition, stomach content analysis of large numbers of juvenile Chinook salmon show no evidence of sediment ingestion (Cordell 2001). However, it was conservatively estimated that juvenile Chinook salmon may incidentally ingest sediment equal to 1% of the volume of food typically ingested. Using the food ingestion rate and an assumption of 79% moisture in the diet (average percent moisture of invertebrate tissue [excluding laboratory-exposed clams and crayfish] and small fish analyzed from the Study Area), the estimated sediment ingestion rate for juvenile Chinook salmon was 0.0000024 kg dw/day.

2.3.3 Diet Composition

Based on the stomach contents of juvenile Chinook salmon collected in the Lower Willamette River (LWR) (n = 346), water fleas (*Daphnia* sp.), fish, and amphipods (*Corophium* sp.) made up 43, 32, and 12%, respectively, of their diet based on wet weight (Vile et al. 2005). LWG Round 2 juvenile Chinook stomach content data (n = 20) showed that juvenile Chinook prey principally on daphnids, terrestrial insects, and chironomid larvae (Integral and Windward 2006). In the Columbia River, the diet of juvenile Chinook salmon in riverine habitats is predominately aquatic insects, including midges and caddisflies (CBFWA 1996; Rondorf et al. 1990). Daphnid and amphipod tissue are not available from the ERA dataset; however, multiplate samplers were used to collect

invertebrates, and amphipods (*Corophium* sp.), chironomids, and daphnids (*Daphnia* sp.) were included in the multiplate tissue analysis and were used to represent juvenile Chinook diets in this analysis.

Because juvenile Chinook salmon are primarily pelagic feeders, exposure concentrations in their diet were estimated using multiplate tissue. The analytical results of juvenile Chinook salmon stomach tissues were also used to evaluate dietary exposure. In addition, per EPA (2008), worm and clams were used to estimate dietary exposure concentrations for juvenile Chinook salmon. Thus, for the evaluation of individual prey species, three representative prey species (i.e., epibenthic tissue, worm, and clams) and stomach content tissues were used to estimate dietary exposure concentrations for juvenile Chinook salmon.

In the evaluation of multiple prey species, the dietary portions were assigned to represent a more realistic dietary scenario. Multiplate samplers were used to collect epibenthic invertebrates (including amphipods [*Corophium* sp.], chironomids, and daphnids [*Daphnia* sp.]) that were composited and analyzed to support the dietary evaluation. These tissue data likely best represent juvenile Chinook prey tissue chemical concentrations, and epibenthic tissue was selected to represent 100% of the diet. Clams and worms are likely not representative of the pelagic prey or feeding habits of juvenile Chinook salmon; however, per EPA(2008), the following prey portions, in which epibenthic invertebrates, worms, and field-collected clams[5] each made up 30%, 40%, and 30% of the diet, respectively, were also evaluated in a second dietary scenario for juvenile Chinook salmon.

There is uncertainty associated with the use of benthic organisms to represent the diet of a pelagic feeding fish. The use of benthic tissue as prey may overestimate dietary exposures of juvenile Chinook salmon to sediment-associated contaminants. The use of worm tissue chemical concentrations based on 28-day laboratory bioaccumulation testing as representative of benthic prey living in the Study Area is also uncertain. Field and steady-state conditions may not be represented by tissue chemical concentrations measured in laboratory testing conditions. Worm tissue concentrations of neutral organic COPCs were adjusted to reflect steady-state concentrations using the process in the Inland Testing Manual (EPA and USACE 1998). Attachment 3 presents the methods used to derive steady-state concentrations. As with the use of any model, the equations (based on McFarland (1995)) and assumptions (i.e., K_{OW} value) associated with the steady-state adjusted concentrations are uncertain, and adjusted worm tissue chemical concentrations may over or underestimate actual steady-state concentrations expected in worms in the Study Area.

2.3.4 Scale of Exposure Areas

The Study Area encompasses approximately a 10-mile (16 km) reach from approximately RM 2.0 to RM 12. This site-wide scale was considered to be a representative exposure area for migrating juvenile Chinook salmon.

[5] Field clams were assumed to be more representative of chemical concentrations in bivalves from the Study Area than lab clams.

There is some uncertainty associated with the duration of exposure of migrating juvenile Chinook salmon to Study Area contaminants. Migration rates for juvenile Chinook salmon through the LWR from Willamette Falls to the mouth of the Columbia River ranged from 2 days to 2 months, based on calendar year 2001 Oregon Department of Fish and Wildlife (ODFW) studies (North et al. 2002). Beach seining data collected in 2001 shows that the migration rate of sub-yearling fall Chinook salmon is slower than that of yearling spring Chinook salmon. Preliminary radio telemetry studies found that the range of residence times for sub-yearling fall Chinook was 1.2 to 6.8 days from RM 9.5 to RM 3.5 and 1.6 to 26.8 days from RM 18.5 to RM 3.5 (Ellis Ecological 2001). Residence time of smaller juvenile salmon (<4.25 inches[108 mm]) has not been measured and may vary somewhat from those reported here. Periods of adult salmonid migration through Portland Harbor are not as well documented as downstream movements (Ellis Ecological 2001).

2.4 PEAMOUTH

This section provides the rationale for peamouth-specific exposure parameters, as summarized in Tables 2-1 and 2-2.

2.4.1 Body Weight and Daily Food Ingestion Rate

The average body weight reported for individual peamouth collected in Round 1 sampling was 104 g. This body weight was used to estimate the food ingestion rate. Using Equation 2-1, the estimated food ingestion rate for peamouth was 0.0072 kg ww/day.

2.4.2 Incidental Sediment Ingestion Rate

Peamouth consume some benthic species; and while feeding, they may incidentally ingest sediment. However, benthic species make up only a part of the peamouth diet. Peamouth spend a significant portion of time in the pelagic zone and are not likely to have substantial direct contact with sediment. Therefore, it was estimated that peamouth incidentally ingest sediment equal to 1% of the volume of food typically ingested. Using the food ingestion rate and an assumption of 79% moisture in the diet (average percent moisture of invertebrate tissue [excluding laboratory-exposed clams and crayfish] and small fish analyzed from the Study Area), the estimated sediment ingestion rate for peamouth was 0.000075 kg dw/day.

2.4.3 Diet Composition

Peamouth prey on both benthic and pelagic species. Adult peamouth predominately feed on benthic invertebrates, crustaceans, and small fish (Wydoski and Whitney 1979). Peamouth in the mid-Columbia River were reported to prey most frequently (based on percent occurrence) on periphyton, snails, caddisfly larvae, and midges (Gray and Dauble 2001). Larger peamouth occasionally feed on mollusks but rarely on small fish such as small sculpin (Carl et al. 1959). Terrestrial and aquatic insects were observed in the stomach contents of five peamouth collected and analyzed in Round 1 sampling (Attachment B6 of the programmatic work plan [Integral et al. 2004]). Filamentous algae, sediment, fish remains, bryozoans, and statoblasts were also found in Round 1 peamouth stomach contents.

Based on this analysis, peamouth in the Study Area likely feed on a mixture of benthic invertebrates, pelagic invertebrates, and small fish. For the evaluation of individual prey species, the peamouth diet was modeled using the following species: multiplate tissue (representing pelagic invertebrates), worms and clams (representing benthic invertebrates), and sculpin (representing small fish).

In the evaluation of multiple prey species, the dietary portions were assigned to represent a more realistic dietary scenario. Peamouth likely feed on a mixture of benthic invertebrates, pelagic invertebrates, and small fish; and the following prey portions were assigned to estimate the peamouth diet using multiple prey items: 40% epibenthic invertebrate tissue, 25% field-collected clams,[6] 25% laboratory exposed worms, and 10% sculpin.

There is some uncertainty associated with the use of the available prey items for the diverse diet of peamouth. Dietary prey estimated concentrations may be over or underestimated for peamouth. The use of clam and worm tissue chemical concentrations based on 28-day laboratory bioaccumulation testing as representative of benthic prey living in the Study Area is also uncertain. Field and steady-state conditions may not be represented by tissue chemical concentrations measured in laboratory testing conditions. Clam and worm tissue concentrations of neutral organic COPCs were adjusted to reflect steady-state concentrations using the process in the Inland Testing Manual (EPA and USACE 1998). Attachment 3 presents the methods used to derive steady-state concentrations. As with the use of any model, the equations (based on McFarland (1995)) and assumptions (i.e., K_{OW} value) associated with the steady-state adjusted concentrations are uncertain, and adjusted worm tissue chemical concentrations may over or underestimate steady-state concentrations expected in worms in the Study Area.

2.4.4 Scale of Exposure Areas

The Study Area encompasses approximately a 10-mile (16 km) reach from approximately RM 2.0 to RM 12. Peamouth are assumed to have a large home range, moving in and out of the Study Area. A site-wide scale was identified as the appropriate exposure scale for peamouth.

2.5 SCULPIN

This section provides the rationale for sculpin-specific exposure parameters, as summarized in Tables 2-1 and 2-2.

2.5.1 Body Weight and Daily Food Ingestion Rate

The average body weight reported for individual sculpin collected in Rounds 1 and 3 sampling was 20 g. This body weight was used with Equation 2-1 to estimate the food ingestion rate of 0.0017 kg ww/day for sculpin.

[6] Field clams were assumed to be more representative of chemical concentrations in bivalves from the Study Area than lab clams.

2.5.2 Incidental Sediment Ingestion Rate

Sculpin are small predatory fish that live in close proximity to sediment and are primarily benthic feeders. As adults, sculpin can burrow up to 14.2 inches (36 cm) into gravel to forage (Wydoski and Whitney 1979). Londi Tomaro, a biologist who studied prickly sculpin in Alaska, noted that although she has not measured sediment in stomachs, in her experience quantifying prickly sculpin stomach contents, a very small proportion of sculpin had sediment in their stomachs; and those that did had a very small amount (ODFW 2006). Prickly sculpin are a common sculpin species in the LWR. Based on this information, it was estimated that sculpin incidentally ingest sediment equal to 5% of the volume of food typically ingested. Using the food ingestion rate an assumption of 79% moisture in the diet (average percent moisture of invertebrate tissue [excluding laboratory-exposed clams and crayfish] and small fish analyzed from the Study Area), the estimated sediment ingestion rate for sculpin was 0.000018 kg dw/day.

2.5.3 Diet Composition

Adult sculpin consume crustaceans, aquatic insects, snails, fish, fish eggs, and mollusks, while juvenile sculpin feed on aquatic insect larvae (Wydoski and Whitney 1979). Sculpin may also prey on other sculpin (Armstrong et al. 1995; Pfister 2003). The stomach contents of eight sculpin collected during Round 1 sampling were analyzed; and aquatic insects, as well as amphipods, gastropods, and bryozoans, were the predominant prey items present (Attachment B6 of the programmatic work plan [Integral et al. 2004]). Northcote et al. (1954) studied prickly sculpin in British Columbia lakes. All fish analyzed were less than 3.43 inches (87 mm), and fish (redside shiner) constituted more than 75% of the prickly sculpin diet by volume. Tomaro (2006) observed prickly sculpin that were larger than 4.02 inches (102 mm) from Auke Lake in Alaska, and noted that 40% of the fish fed on aquatic insects, 37% fed on fish, 35% fed on mollusks, and 25% fed on plant matter, however, the proportions of food items in gut contents were not quantified. The amount of fish consumed by sculpin increased with the size of the sculpin; and Dolly Varden, stickleback, and small sculpin were the principal fish prey of large sculpins (> 5.08 inches [129 mm] in length).

Sculpin analyzed in Round 1 and Round 3 tissue samples ranged from 3.27 to 6.89 inches (83 to 175 mm) in length; thus, sculpin in the Study Area are likely to feed on a mixture of benthic invertebrates and small fish. However, fish were not a dominant prey item present in the sculpin stomach contents analyzed from the Study Area. For the evaluation of individual prey species, exposure concentrations in the sculpin diet were estimated using the following prey items: worms (to represent benthic invertebrates living in sediments), clam, and sculpin. Because of the small exposure area scale for sculpin (see Section 2.5.4), each of the individual prey species were evaluated only on a sample-by-sample basis in the BERA.

The use of clam and worm tissue chemical concentrations based on 28-day laboratory bioaccumulation testing as representative of benthic prey living in the Study Area is also uncertain. Field and steady-state conditions may not be represented by tissue chemical concentrations measured in laboratory testing conditions. Clam and worm tissue

concentrations of neutral organic COPCs were adjusted to reflect steady-state concentrations using the process in the Inland Testing Manual (EPA and USACE 1998). Attachment 3 presents the methods used to derive steady-state concentrations. As with the use of any model, the equations (based on McFarland (1995)) and assumptions (i.e., K_{OW} value) associated with the steady-state adjusted concentrations are uncertain, and adjusted worm tissue chemical concentrations may over or underestimate steady-state concentrations expected in worms in the Study Area.

2.5.4 Scale of Exposure Areas

Foraging ranges reported in the literature support small home ranges for sculpin. Sculpin movements of less than 30 feet to more than 200 feet were reported in the literature (Hill and Grossman 1987; Natsumeda 1998, 1999, 2001; Petty and Grossman 2004; Cunjak et al. 2005), and an exposure area of 0.1 mile (500 ft [161 m]) was assumed to be representative of the home range of sculpin. This exposure scale was assumed to be roughly equivalent to the scale over which composite samples were collected.

2.6 SMALLMOUTH BASS

This section provides the rationale for smallmouth bass-specific exposure parameters, as summarized in Tables 2-1 and 2-2.

2.6.1 Body Weight and Daily Food Ingestion Rate

The average body weight reported for individual smallmouth bass collected from within the Study Area during Rounds 1 and 3 was 400 g. This body weight was used to estimate the food ingestion rate. Using the body weight and Equation 2-1, the estimated food ingestion rate for smallmouth bass was 0.022 kg ww/day.

2.6.2 Incidental Sediment Ingestion Rate

Smallmouth bass consume benthic prey and may incidentally consume some sediment. Like peamouth, smallmouth bass spend a significant portion of time in the pelagic zone and are not likely to have substantial direct contact with sediment. Therefore, it was estimated that smallmouth bass may incidentally ingest sediment equal to 1% of the volume of food typically ingested. Using the food ingestion rate and an assumption of 74% moisture in the diet (average percent moisture of fish tissue analyzed from the Study Area), the estimated sediment ingestion rate for smallmouth bass was 0.000058 kg dw/day.

2.6.3 Diet Composition

As a benthopelagic species, smallmouth bass consume fish, crayfish and other crustaceans, mollusks, and worms as adults and insect larvae and zooplankton as juveniles (George and Hadley 1979; Turner 1966; Wydoski and Whitney 1979). In the main stem of the Willamette River, the following proportions (by volume) of prey items were reported based on the identification of the stomach contents of 114 bass (primarily smallmouth bass): 50% sculpin, 26% crayfish, 18% other fish, 2% insects, < 1% salmonids, and 4% other prey items, including snails, worms, and vegetation (ODFW 2001). One study reported stomach

contents of smallmouth bass in fish collected in 2002 and 2003 from the LWR (Pribyl et al. 2005; Vile and Friesen 2005). Crayfish and fish (sculpin) made up 62 and 35% (by wet weight), respectively, of the stomach contents of 48 juvenile smallmouth bass (< 3.94 inches [100 mm]) (Vile and Friesen 2005). Fish (primarily sculpin) made up approximately 90% of the stomach contents of 17 adult smallmouth bass (> 7.87 inches [200 mm]) (by wet weight); and shrimp and crayfish made up 7 and 3%, respectively, of the stomach contents (Pribyl et al. 2005). Crayfish remains were the dominant prey item in the stomach contents of four smallmouth bass collected in Round 1 (Attachment B6 of the programmatic work plan [Integral et al. 2004]). In addition, juvenile Chinook salmon were not found in the digestive tracts of smallmouth bass collected from RM 12 of the LWR during peak juvenile salmonid migration from April to June and do not appear to be a typical prey species of smallmouth bass (Fishman 2001).

Representative crayfish and fish prey tissue data for smallmouth bass are available from the Study Area. Based on the data reported in the reviewed literature, smallmouth bass in the Study Area likely feed primarily on sculpin and other small fish. Because the smallmouth bass collected in Round 1 and Round 3 ranged from 8.66 to 17.0 inches (220 to 432 mm) in length, the prey portions for this receptor were based on the smallmouth bass > 7.87 inches (200 mm) reported in Pribyl et al. (2005), in which small fish (sculpin) made up the majority of the prey items. For the evaluation of individual prey species, exposure concentrations in the smallmouth bass diet were modeled using the following prey items: sculpin, crayfish, and worms.

In the evaluation of multiple prey species, the dietary portions were assigned to represent a more realistic dietary scenario. Sculpin made up 90% of the stomach contents in LWR-collected sculpin based on Pribyl et al. (2005), and shrimp and crayfish made up 7% and 3% of the stomach contents, respectively. Based on these data, the smallmouth bass diet was modeled assuming sculpin represented 10% of the diet, and crayfish and laboratory exposed worms each represented 5% of the diet.

The use of worm tissue chemical concentrations based on 28-day laboratory bioaccumulation testing as representative of benthic prey living in the Study Area is also uncertain. Field and steady-state conditions may not be represented by tissue chemical concentrations measured in laboratory testing conditions. Worm tissue concentrations of neutral organic COPCs were adjusted to reflect steady-state concentrations using the process in the Inland Testing Manual (EPA and USACE 1998). Attachment 3 presents the methods used to derive steady-state concentrations. As with the use of any model, the equations (based on McFarland (1995)) and assumptions (i.e., K_{OW} value) associated with the steady-state adjusted concentrations are uncertain, and adjusted worm tissue chemical concentrations may over or underestimate steady-state concentrations expected in worms in the Study Area.

2.6.4 Scale of Exposure Areas

Foraging ranges and movements reported in the literature and in region-specific studies for smallmouth bass have supported small home ranges that are smaller than the entire length

of the Study Area. Pribyl et al. (2005) conducted a study from 2000 to 2003, in which the movement of predatory resident fish (including smallmouth bass) was tracked using radio-tagged fish in the LWR. Radio-tagged smallmouth bass tended to stay near release points, and the median of the maximum distance traveled by smallmouth bass was 1.4 miles (2.3 km) from the release site over the tracking period; however, most smallmouth bass traveled only 0.25 mile (0.4 km) within 1 month after the release. In addition, all of the radio-tagged smallmouth bass collected from the lower portion of the Willamette River (from RM 0 to RM 22.5) were located within 20% of the width of the river from either shore, suggesting a preference for nearshore habitat. An exposure area of approximately 1 mile (1.6 km) was assumed to be representative of the foraging range of the smallmouth bass.

2.7 NORTHERN PIKEMINNOW

This section provides the rationale for northern pikeminnow-specific exposure parameters, as summarized in Tables 2-1 and 2-2.

2.7.1 Body Weight and Daily Food Ingestion Rate

The average body weight reported for individual northern pikeminnow collected in Round 1 sampling was 558 g. This body weight was used to estimate the food ingestion rate. Using the body weight and Equation 2-1, the estimated food ingestion rate for northern pikeminnow was 0.030 kg ww/day.

2.7.2 Incidental Sediment Ingestion Rate

As with smallmouth bass, northern pikeminnow may occasionally come into contact with sediment when foraging, though they are not likely to have substantial direct contact with sediment. Therefore, it was estimated that northern pikeminnow may incidentally ingest sediment equal to 1% of the volume of food typically ingested. Using the food ingestion rate and an assumption of 74% moisture in the diet (average percent moisture of fish tissue analyzed from the Study Area), the estimated sediment ingestion rate for peamouth was 0.000078 kg dw/day.

2.7.3 Diet Composition

Northern pikeminnow is a benthopelagic species, consuming predominately fish and some insects (Buchanan et al. 1981; Jeppson and Platts 1959). Fish and crayfish were the dominant prey items in the stomachs of northern pikeminnow collected from the mid-Columbia River, making up 32 and 16%, respectively, of the diet by volume (Gray and Dauble 2001). Crayfish and fish have been found to be dominant prey items of the northern pikeminnow in the Willamette River as well. Fish, crayfish, and insects were present in 36, 31, and 26%, respectively, of northern pikeminnow stomachs that contained prey items for fish collected from locations in the Willamette Basin (Buchanan et al. 1981). In the same study, sculpin, salmonids, and other fish (i.e., cyprinids, catostomids, and lampreys) made up 68, 10, and 22%, respectively, of the total northern pikeminnow fish diet based on stomach contents (Buchanan et al. 1981). In another study, crayfish made up approximately 58% (by wet weight) of the digestive tract contents of northern pikeminnow collected in the LWR (n = 20) from 2002 to 2003 (Pribyl et al. 2005). Other prey made up 42% of the

contents in northern pikeminnow digestive tracts. Fish and crayfish remains were found in the stomach contents of four northern pikeminnow collected from the Study Area during Round 1 tissue sampling (Attachment B6 of the programmatic work plan [Integral et al. 2004]).

Based on the site-specific data and the reviewed literature, northern pikeminnow likely feed on a variety of fish species as well as on crayfish and benthic invertebrates in the Study Area. For the evaluation of individual prey species, the northern pikeminnow diet was modeled using crayfish, sculpin, worms, largescale sucker, carp, peamouth, and northern pikeminnow). Sculpin and crayfish likely make up the largest portion of prey in the northern pikeminnow diet.

In the evaluation of multiple prey species, the dietary portions were assigned to represent a more realistic dietary scenario. Based on the regional data presented above, the following prey portions and species were selected: crayfish, worms, and sculpin were selected to represent 30%, 25%, and 25% of the diet, respectively; and largescale sucker, carp, peamouth, and northern pikeminnow were each selected to represent 5% of the total diet.

The use of worm tissue chemical concentrations based on 28-day laboratory bioaccumulation testing as representative of benthic prey living in the Study Area is uncertain. Field and steady-state conditions may not be represented by tissue chemical concentrations measured in laboratory testing conditions. Worm tissue concentrations of neutral organic COPCs were adjusted to reflect steady-state concentrations using the process in the Inland Testing Manual (EPA and USACE 1998). Attachment 3 presents the methods used to derive steady-state concentrations. As with the use of any model, the equations (based on McFarland (1995)) and assumptions (i.e., K_{OW} value) associated with the steady-state adjusted concentrations are uncertain, and adjusted worm tissue chemical concentrations may over or underestimate actual steady-state concentrations expected in worms in the Study Area.

2.7.4 Scale of Exposure Areas

Foraging ranges and movements reported in the literature indicate that northern pikeminnow may forage in stretches that are smaller than the entire length of the Study Area. Pribyl et al. (2005) conducted a study from 2000 to 2003, in which the movement of predatory resident fish (including northern pikeminnow) was tracked using radio-tagged fish in the LWR. Radio-tagged northern pikeminnow tended to stay near release points, and the median of the maximum distance traveled by northern pikeminnow was 0.87 mile (1.4 km) from the release site over the tracking period. In addition, the majority of northern pikeminnow collected from the lower portion of the Willamette River (from RM 0 to RM 22.5) were located within 20% of the width of the river from either shore, suggesting a preference for nearshore habitat. An exposure area of approximately 1 mile (1.6 km) was conservatively assumed to be representative of the foraging range of the northern pikeminnow.

3.0 DIETARY ASSESSMENT EFFECT ASSUMPTIONS

Section 7.6.3 of the BERA presents a summary of the effects data used in the dietary assessment for fish. Effects data were based on fish dietary TRVs. Table 3-1 presents the fish dietary TRVs selected for the fish dietary risk assessment. The following subsections present the details on the selected TRVs.

Table 3-1. Fish Dietary Dose TRVs

COPC	TRV (mg/kg bw/day) NOAEL	LOAEL	Source
Cadmium	0.002[a]	0.01	Kim et al. (2004), Kang et al. (2005)
Copper	0.24	0.48	Murai et al. (1981)
Mercury	0.005	0.013	Matta et al. (2001)
TBT	0.00042[a]	0.0021	Shimasaki et al. (2003)
Benzo(a)pyrene	0.66	1.4	Rice et al. (2000)
Total PAHs	6.1	18	Meador et. al. (2006)

[a] NOAEL extrapolated from LOAEL using an uncertainty factor of 5.

bw – body weight
COPC – chemical of potential concern
LOAEL – lowest-observed-apparent-effect level
NOAEL – no-observed-apparent-effect level
PAH – polycyclic aromatic hydrocarbon
TBT – tributyltin
TRV – toxicity reference value

3.1 CADMIUM

A lowest-observed-adverse-effect level (LOAEL) of 0.010 mg/kg bw/day, derived from Kim et al. (2004) and Kang et al. (2005), was selected to evaluate risks to fish via dietary cadmium exposure. At this LOAEL, juvenile rockfish growth was reduced following dietary exposure to cadmium for 60 days (Kim et al. 2004; Kang et al. 2005). This was the lowest LOAEL derived from the nine studies measuring the toxicity of dietary cadmium that were reviewed (Attachment 14). No no-observed-adverse-effect level (NOAEL) below this selected LOAEL TRV was derived from the reviewed studies, so a NOAEL TRV was extrapolated from the LOAEL TRV using an uncertainty factor (UF) of 5. The resulting NOAEL of 0.0020 mg/kg bw/day was selected at the NOAEL TRV.

There is some uncertainty associated with the selected dietary fish TRVs for cadmium. The selected LOAEL TRV is two to three orders of magnitude lower than the NOAELs reported in six of other studies reviewed (1.0 to 9.4 mg/kg bw/day) and three orders of magnitude lower than the LOAELs reported in the three other studies that reported LOAELs (4.6 to 68 mg/kg bw/day) (Attachment 14). As such, the selected TRVs are conservative because the majority of the toxicological studies reviewed indicate that the selected TRVs may over-predict cadmium toxicity to fish (by several orders of magnitude). There is additional uncertainty associated with the use of a UF to derive a NOAEL TRV.

3.2 COPPER

The LOAEL and NOAEL of 0.48 and 0.24 mg/kg bw/day, respectively, were selected as the dietary TRVs for copper. These TRVs were derived from Murai et al. (1981). At the selected LOAEL, a significant decrease in body weight was reported for channel catfish fingerlings exposed to 16 mg/kg dw of copper as copper sulfate in a prepared diet for 16 weeks compared to the control group. Growth was not significantly affected in fish fed 8 mg/kg dw relative to controls. The selected LOAEL was the lowest LOAEL derived from the 13 studies measuring the toxicity of dietary copper that were reviewed (Attachment 14).

There is high uncertainty associated with the selected dietary TRVs for copper because the sensitivity of channel catfish fingerlings documented by Murai et al. (1981) has not been confirmed in subsequent studies using similar exposures and fish of similar age (Erickson et al. 2003; Gatlin and Wilson 1986) and has been characterized as atypical by other studies of copper in fish (Lorentzen et al. 1998). Gatlin and Wilson (1986) attempted to reproduce the exposure conditions used by Murai et al. (1981) using fingerling catfish that were larger (body weight = 5.5 g) than those used in Murai et al. (1981) (body weight = 1 g). Gatlin and Wilson (1986) reported no difference in weight gain in their highest dietary exposure of 40 mg/kg dw. Likewise, Erickson et al. (2003)[7] reported no differences in weight gain following exposure for 30 days to copper-contaminated prey at dietary concentrations of 157 and 246 mg/kg dw using much smaller fingerling channel catfish (0.2 g/fish). These studies help bracket the size of fingerlings tested and confirm that the Murai et al. (1981) study results are anomalous. The next lowest NOAEL and LOAEL TRVs from the toxicity studies reviewed were 1.0 and 2.0 mg/kg bw/d, respectively, based on Kang et al. (2005). At this LOAEL, a reduced growth rate of rockfish was observed following exposure to dietary copper for 60 days.

3.3 MERCURY

The LOAEL and NOAEL of 0.005 and 0.013 mg/kg bw/day, respectively, were selected as the dietary TRVs for mercury. At the selected LOAEL, a decrease in male mummichog survival was observed; and at the selected NOAEL, no effect on male mummichog survival was observed (Matta et al. 2001). The selected TRVs were the lowest TRVs reported in the six reviewed toxicological studies in which relevant fish endpoints of survival, growth, or reproduction were directly measured (Attachment 14). A lower LOAEL TRV of 0.0091 mg/kg bw/day was derived from Webber and Haines (2003), in which predator avoidance behavior was reduced in golden shiners exposed to dietary methylmercury for 90 days. The authors contend that the observed behavioral changes could affect fish survival; however, this has not been confirmed in field studies. Therefore, the LOAEL based on Webber and Haines was not selected.

[7] Erickson (2003) is currently under review for publication but has not yet been published and was not included in the studies evaluated for TRV derivation in Attachment 14.

3.4 TRIBUTYLTIN

A LOAEL of 0.0021 mg/kg bw/day, derived from Shimasaki et al. (2003), was the selected as the dietary LOAEL TRV for tributyltin (TBT). This selected LOAEL was based on the only study reviewed on the toxicological effects of dietary TBT on fish (Attachment 14). In this study, the effects on reproductive success of Japanese flounder larvae following exposure to dietary TBT were reported. The flounder larvae that were exposed to TBT were generated from two parents that were genetically female; however, one parent was phenotypically and functionally male, and offspring were generated. All offspring produced were genetically female. The female offspring larvae were exposed to TBT as tributyltin oxide for approximately 65 days at two different doses, and the effects on growth and sex reversal were measured relative to a control. A significant reduction in growth was reported at 100 days at 0.0021 mg/kg bw/day. No NOAEL was reported in this study, so a NOAEL was extrapolated from the selected LOAEL using a UF of 5.

An increase in sex reversal was reported in TBT-exposed fish. The actual impact of TBT on sex reversal is unclear because sex reversal was apparent in the parent fish, prior to TBT exposure; and sex reversal did not appear to affect the production of offspring. The ecological significance of the sex reversal endpoint is uncertain. No difference in the concentration was calculated converting the LOAEL TRV for TBT from tributyltin oxide into tributyltin ion (due to significant figures). There is uncertainty associated with the selected TRVs because the literature dataset for the dietary toxicity of TBT to fish is limited to one study. Extrapolating a NOAEL from a LOAEL using a UF is also uncertain.

3.5 BENZO(A)PYRENE

The NOAEL and LOAEL of 0.66 and 1.4 and mg/kg bw/day, respectively, derived from Rice et al. (2000), were selected as the benzo(a)pyrene TRVs for fish. These were the lowest TRVs derived from the four toxicological studies reviewed (Attachment 14). In the selected study, Rice et al. (2000) fed juvenile English sole polychaete worms that were exposed to benzo(a)pyrene-spiked sediments. Adverse effects on body weight were observed in the highest-dose group (at 1.4 mg/kg bw/day). There is uncertainty associated with the selected TRVs for benzo(a)pyrene because the literature set for the toxicity of benzo(a)pyrene is limited.

3.6 TOTAL PAHS

The NOAEL and LOAEL of 6.1 and 18.0 and mg/kg bw/day, respectively, derived from Meador et al. (2006), were selected as the PAH mixture TRVs for fish. In the selected study, Meador et al. (2006) exposed juvenile Chinook salmon to a mixture of 21 PAHs[8] in a prepared diet at a total PAH dose of up to 22 mg/kg bw/day. Adverse effects on weight were observed in the highest dose relative to the control on a wet-weight basis. Adverse

[8] The PAH mixture included the following chemicals: naphthalene, 2-methylnaphthalene, dimethylnaphthalene, dibenzothiophene, acenaphthene, fluorene, 1,8-dimethyl(9H)fluorene, phenanthrene, 9-ethylphenanthrene, 9-ethyl-10-methylphenanthrene, 1-methyl-7-isopropylphenanthrene, anthracene, fluoranthene, pyrene, methyl pyrene, benz[a]pyrene, benzo(k)fluoranthene, benzo(g,h,i)perylene, dibenzanthracene.

effects on weight were also observed at the next highest dose (18.0 mg/kg bw/day) relative to the control when fish weights were considered on a dry-weight basis.

The selected LOAEL TRV was the lowest acceptable LOAEL reported in the three studies reviewed (Attachment 14). A lower LOAEL of 0.12 mg/kg bw/day was derived from Rice et al. (2000), in which juvenile English sole were fed a few polychaete worms that had been previously exposed to field-collected sediments from Eagle Harbor, Washington. However, this LOAEL was not selected because the dietary prey (i.e., worms) was exposed to field-collected sediments with uncharacterized chemicals, and the significance of the effects observed at this concentration were statistically ambiguous.

There is uncertainty associated with the selected TRVs. The PAH mixture used in the selected study was designed to resemble the PAH mixture observed in juvenile Chinook stomach contents collected in the Duwamish River, Seattle (Meador et al. 2006). This mixture may or may not be consistent with PAH mixtures found in the Portland Harbor Study Area. There is additional uncertainty associated with the selected TRVs because the literature dataset for the toxicity of PAH mixtures to fish is limited to only two studies reporting effect-level concentrations.

4.0 REFERENCES

Armstrong JL, Armstrong DA, Mathews SB. 1995. Food habits of estuarine staghorn sculpin, *Leptocottus armatus*, with focus on consumption of juvenile Dungeness crab, *Cancer magister*. Fish Bull 93:456-470.

Arnot JA, Gobas FAPC. 2004. A food web bioaccumulation model for organic chemicals in aquatic ecosystems. Environ Toxicol Chem 23:2343-2355.

Bajkov AD. 1949. A preliminary report on the Columbia River sturgeon. In: Fish Commission research briefs. Vol 2, no. 2. Fish Commission of Oregon, Portland, OR, pp 3-10.

Beamesderfer RC, Nigro AA, eds. 1992. Status and habitat requirements of the white sturgeon populations in the Columbia River downstream from McNary Dam. Final report of research. Vol II - supplemental papers and data documentation. Prepared for Bonneville Power Administration. Oregon Department of Fish and Wildlife, Portland, OR.

Beamesderfer RCP, J.C. Elliott JC, C.A. Foster CA. 1998. Methods for ageing Columbia River white sturgeon from pectoral fin rays. Report A. 1) Description of the life history and population dynamics of subadult and adult white sturgeon in the Columbia River between Bonneville and McNary dams, and 2) Evaluation of the need and identification of potential methods for protecting, mitigating, and enhancing white sturgeon populations in the Columbia River downstream from McNary Dam. In: Nigro AA, ed, Status and habitat requirements of white sturgeon populations in the Columbia River downstream from McNary Dam. Annual progress report to Bonneville Power Administration, Portland, Oregon. Oregon Department of Fish and Wildlife, Portland, OR, pp 5-52.

Beamesderfer RCP, Farr RA. 1997. Alternatives for the protection and restoration of sturgeons and their habitat. Environ Biol Fish 48:407-417.

Buchanan DV, Hooton RM, Moring JR. 1981. Northern squawfish (*Ptychocheilus oregonensis*) predation on juvenile salmonids in sections of the Willamette River basin, Oregon. Can J Fish Aquat Sci 38:360-364.

Carl GC. 1936. Food of the coarse-scaled sucker (*Catostomus macrocheilus* Girard). J Biol Bd Can 3(1):20-25.

Carl GC, Clemens WA, Lindsey CC. 1959. The fresh-water fishes of British Columbia. Department of Education handbook no. 5. British Columbia Provincial Museum, Vancouver, BC.

CBFWA. 1996. Contamination ecology of selected fish and wildlife of the lower Columbia River. Report to the Bi-State Water Quality Program. Columbia Basin Fish & Wildlife Authority, Portland, OR.

Cordell J. 2001. Personal communication on July 9 (e-mail to Matt Luxon, Windward Environmental, Seattle, WA, regarding observations of juvenile chinook and other wildlife in the Lower Duwamish River). Researcher, Department of Fisheries, University of Washington, Seattle, WA.

Cunjak RA, Roussel J-M, Gray MA, Dietrich JP, Cartwright DF, Munkittrick KR, Jardine TD. 2005. Using stable isotope analysis with telemetry or mark-recapture data to identify fish movement and foraging. Oecologia 144:636-646.

Dauble DD. 1986. Life history and ecology of the largescale sucker (*Catastomus macrocheilus*) in the Columbia River. Am Midl Nat 116:356-367.

DeVore JD, Grimes JT. 1993. Migration and distribution of white sturgeon in the Columbia River downstream from Bonneville Dam and in adjacent marine areas. In: Beamesderfer RC, Nigro AA, eds, Status and habitat requirements of the white sturgeon populations in the Columbia River downstream from McNary Dam. Final report of research. Vol I. Prepared for Bonneville Power Administration. Oregon Department of Fish and Wildlife, Portland, OR, pp 83-100.

DeVore JD, James BW, Tracy CA, Hale DA. 1995. Dynamics and potential production of white sturgeon in the unimpounded lower Columbia River. Trans Am Fish Soc 124:845-856.

Ellis Ecological. 2001. Juvenile salmonid residence time in Portland Harbor. Draft. Prepared for Lower Willamette Group. Ellis Ecological Services, Inc., Estacada, OR.

EPA. 2008. Problem formulation for the Baseline Ecological Risk Assessment at the Portland Harbor Site. Report and letter dated February 15, 2008 to Lower Willamette Group (from E. Blischke and C. Humphrey to J. McKenna and R. Wyatt). US Environmental Protection Agency Region 10, Oregon Operations Office, Portland, OR.

EPA, USACE. 1998. Evaluation of dredged material proposed for discharge in waters of the U.S.- testing manual (Inland Testing Manual. EPA-823-B-98-004. US Environmental Protection Agency and US Army Corps of Engineers, Washington, DC.

Erickson RJ, Highland TL, Hocket JR, Leonard EN, Mattson VR, Mount DR. 2003. Effects of dietary copper, zinc, lead, cadmium, and arsenic on growth and survival of juvenile fish using live food organisms. Platform presentation at SETAC 24th annual meeting, Austin TX, 9-13 November 2003. Manuscript in prep.

Fishman. 2001. Eastbank riverfront (phase I) floating walkway fish predation study: final report and data summary, 1999 and 2000 sampling seasons. Prepared for Portland Development Commission. Fishman Environmental Services, Portland, OR.

Gatlin DM, III, Wilson RP. 1986. Dietary copper requirement of fingerling channel catfish. Aquaculture 54:277-285.

George EL, Hadley WF. 1979. Food and habitat partitioning between rock bass (*Ambloplites rupestris*) and smallmouth bass (*Micropterus dolomieui*) young of the year. Trans Am Fish Soc 108:253-261.

Gray RH, Dauble DD. 2001. Some life history characteristics of cyprinids in the Hanford Reach, mid-Columbia River. Northwest Sci 75(2):122-133.

Hill J, Grossman GD. 1987. Home range estimates for three North American stream fishes. Copeia 1986:376-380.

Integral, Windward. 2006. Portland Harbor RI/FS. Round 2 multiplate tissue data summary report report. Draft. Prepared for Lower Willamette Group. Windward Environmental LLC, Seattle, WA.

Integral, Windward, Kennedy/Jenks, Anchor, Groundwater Solutions. 2004. Portland Harbor RI/FS programmatic work plan. Prepared for Lower Willamette Group. April 23, 2004. Integral Consulting, Inc., Mercer Island, WA; Windward Environmental LLC, Seattle, WA; Kennedy/Jenks Consultants, Portland, OR; Anchor Environmental, LLC, Seattle, WA; Groundwater Solutions, Inc., Portland, OR.

Jeppson PW, Platts WS. 1959. Ecology and control of the Columbia squawfish in northern Idaho lakes. Trans Am Fish Soc 88(2):197-202.

Kang J-C, Kim S-G, Jang S-W. 2005. Growth and hematological changes of rockfish, *Sebastes schlegeli* (Hilgendorf) exposed to dietary Cu and Cd. J World Aquacult Soc 36(2):188-195.

Kim S-G, Kim J-W, Kang J-C. 2004. Effect of dietary cadmium on growth and haematological parameters of juvenile rockfish, *Sebastes schlegeli* (Hilgendorf). Aquacult Res 35:80-86.

Lee C. 2006. Personal communication on August 9, 2006 (telephone conversation with Matt Luxon, Windward Environmental, regarding sediment ingestion by large scale sucker and northern pikeminnow). Fish biologist, Department of Natural Resources, Spokane Tribe, Wellpinit, WA.

Lorentzen M, Maage A, Julshamn K. 1998. Supplementing copper to a fishmeal-based diet fed to Atlantic salmon parr affects liver copper and selenium concentrations. Aquacult Nutr 4:67-72.

Matta MB, Linse J, Cairncross C, Francendese L, Kocan RM. 2001. Reproductive and transgenerational effects of methylmercury or Aroclor 1268 on *Fundulus heteroclitus*. Environ Toxicol Chem 20(2):327-335.

McCabe GT, Jr, Emmett RL, Hinton SA. 1993. Feeding ecology of juvenile white sturgeon (*Acipenser transmontanus*) in the Lower Columbia River. Northwest Sci 67(3):170-80.

McFarland VA. 1995. Evaluation of field-generated accumulation factors for predicting the bioaccumulation potential of sediment-associated PAH compounds. Long-term effects of dredging operations program technical report D-95-2. Waterways Experiment Station, US Army Corps of Engineers, Vicksburg, MS.

Meador JP, Sommers FC, Ylitalo GM, Sloan CA. 2006. Altered growth and related physiological responses in juvenile chinook salmon (*Oncorhynchus tshawytscha*) from dietary exposure to polycyclic aromatic hydrocarbons (PAHs). Can J Fish Aquat Sci 63:2364-2376.

Murai T, Andrews JW, Smith RG, Jr. 1981. Effects of dietary copper on channel catfish. Aquaculture 22:353-357.

Natsumeda T. 1998. Home range of the Japanese fluvial sculpin, *Cottus pollux*, in relation to nocturnal activity patterns. Environ Biol Fish 53:295-301.

Natsumeda T. 1999. Year-round local movements of the Japanese fluvial sculpin, *Cottus pollux* (large egg type), with special reference to the distribution of spawning nests. Ichthyol Res 46(1):43-48.

Natsumeda T. 2001. Space use by the Japanese fluvial sculpin, *Cottus pollux*, related to spatio-temporal limitations in nest resources. Environ Biol Fish 62:393-400.

North JA, Burner LC, Cunningham BC, Farr RA, Friesen TA, Harrington JC, Takata HK, Ward DL. 2002. Relationships between bank treatment/nearshore development and anadromous/resident fish in the lower Willamette River. Annual progress report, May 2000-June 2001. Draft. Prepared for City of Portland. Oregon Department of Fish and Wildlife, Clackamas, OR.

Northcote TG. 1954. Observations on the comparative ecology of two species of fish, *Cottus asper* and *Cottus rhotheus*, in British Columbia. Copeia 1954(1):25-28.

ODEQ. 2005. Laboratory analytical storage and retrieval (LASAR) database. Data for Willamette River At Sp&S RR Bridge (Portland) [online]. DEQ Online, Oregon Department of Environmental Quality, Portland, OR. Updated 6/24/05. Available from: http://www.deq.state.or.us/wq/lasar/LasarHome.htm.

ODFW. 2001. Willamette River bass diet study - spring 2000. No. 2001-04. Fish Division, Oregon Department of Fish and Wildlife, Portland, OR.

ODFW. 2006. Personal communication on September 18, 2006 (telephone message from L M Tomaro to Matt Luxon, Windward Environmental) regarding proportion of sediment in prickly sculpin diet. Oregon Department of Fish and Wildlife, Salem, OR.

Petty JT, Grossman GD. 2004. Restricted movement by mottled sculpin (Pisces: Cottidae) in a southern Appalachian stream. Freshwat Biol 49:631-645.

Pfister CA. 2003. Some consequences of size variability in juvenile prickly sculpin, *Cottus asper*. Environ Biol Fish 66:383-390.

Pribyl AL, Vile JS, Friesen TA. 2005. Population structure, movement, habitat use, and diet of resident piscivorous fishes in the Lower Willamette River. In: Friesen TA, ed, Biology, behavior, and resources of resident and anadromous fish in the Lower Willamette River. Final report of research, 2000-2004. Prepared for City of Portland. Oregon Department of Fish and Wildlife, Clackamas, OR, pp 139-183.

Radtke LD. 1966. Distribution of smelt, juvenile sturgeon, and starry flounder in the Sacramento-San Joaquin delta with observations on food of sturgeon. Fish Bull 136:115-129.

Rice CA, Myers MS, Willis ML, French BL, Casillas E. 2000. From sediment bioassay to fish biomarker – connecting the dots using simple trophic relationships. Mar Environ Res 50:527-533.

Romano MD, Rien TA, Ward DL. 2002. Seasonal presence and diet of white sturgeon in three proposed in-river, deep-water dredge spoil disposal sites in the lower Columbia River. In: Exhibit K-1, Evaluation report, white and green sturgeon, pp. 5-23. Columbia River Channel Improvement Project: Final supplemental integrated feasibility report and environmental impact statement (USACE 2003). Oregon Department of Fish and Wildlife, Clackamas, OR.

Rondorf DW, Gray GA, Fairley RB. 1990. Feeding ecology of subyearling chinook salmon in riverine and reservoir habitats of the Columbia River. Trans Am Fish Soc 119:16-24.

Shimasaki Y, Kitano T, Oshima Y, Inoue S, Imada N, Honjo T. 2003. Tributyltin causes masculinization in fish. Environ Toxicol Chem 22(1):141-144.

Spotts J, Shields J, Underwood K, Chichosz T. 2002. Lake Roosevelt fisheries evaluation program, part A: fisheries creel survey and population status analysis. 1998 annual report. Bonneville Power Administration, Portland, OR.

Tomaro LM. 2006. Apparent predation of juvenile coho salmon (*Oncorhynchus kisutch*) by prickly sculpins (*Cottus asper*) is an artifact of trapping methodology. Mar Freshwat Res 57:513-518.

Turner JL. 1966. Distribution and food habits of centrarchid fishes in the Sacramento-San Joaquin delta. In: Turner JL, Kelly DW, eds, Fish Bulletin 136. California Department of Fish and Game, Sacramento, CA, pp 144-153.

USFWS. 1961. Sturgeons. Fishery leaflet 526. Bureau of Commercial Fisheries, US Fish and Wildlife Service, Washington, DC.

Veinott G, Northcote T, Rosenau M, Evan RD. 1999. Concentrations of strontium in the pectoral fin rays of the white sturgeon (*Acipenser transmontanus*) by laser ablation sampling-inductively coupled plasma - mass spectrometry as an indicator of marine migrations. Can J Fish Aquat Sci 56:1981-1990.

Vile JS, Friesen TA. 2005. Description and categorization of nearshore habitat in the Lower Willamette River. In: Friesen TA, ed, Biology, behavior, and resources of resident and anadromous fish in the Lower Willamette River. Final report of research, 2000-2004. Prepared for City of Portland. Oregon Department of Fish and Wildlife, Clackamas, OR, pp 17-62.

Vile JS, Friesen TA, Reesman MJ. 2005. Diets of juvenile salmonids and introduced fishes of the Lower Willamette River. In: Friesen TA, ed, Biology, behavior, and resources of resident and anadromous fish in the Lower Willamette River. Final report of research, 2000-2004. Prepared for City of Portland. Oregon Department of Fish and Wildlife, Clackamas, OR, pp 185-221.

WDFW. 2007. Letter dated March 8, 2007 to T. Do, Windward, from M. Wall, WDFW, regarding information on a tagged white sturgeon. March 8, 2007. Columbia River Sturgeon Project, Washington Department of Fish and Wildlife Region 5, Vancouver, WA.

Webber HM, Haines TA. 2003. Mercury effects on predator avoidance behavior of a forage fish, golden shiner (*Notemigonus crysoleucas*). Environ Toxicol Chem 22(7):1556-1561.

Wydoski RS, Whitney RR. 1979. Inland fishes of Washington. University of Washington Press, Seattle, WA.