PORTLAND HARBOR RI

APPENDIX G
BASELINE ECOLOGICAL RISK ASSESSMENT
ATTACHMENT 16
DETAILS ON EXPOSURE AND EFFECTS ASSUMPTIONS FOR THE WILDLIFE DIETARY AND BIRD EGG LOEs

DRAFT

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Prepared for
The Lower Willamette Group

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TABLE OF CONTENTS

LIST OF TABLES iv

1.0 INTRODUCTION 1

2.0 DIETARY DOSE ASSESSMENT EXPOSURE ASSUMPTION DETAILS 2
- 2.1 Spotted Sandpiper 4
  - 2.1.1 Body Weight and Daily Food Ingestion Rate 4
  - 2.1.2 Incidental Sediment Ingestion Rate 5
  - 2.2.3 Diet Composition 5
  - 2.2.3 Site Use 6
  - 2.2.4 Scale of Exposure Areas 6
- 2.2 Hooded Merganser 7
  - 2.2.1 Body Weight and Daily Food Ingestion Rate 7
  - 2.2.2 Incidental Sediment Ingestion Rate 8
  - 2.2.3 Diet Composition 8
  - 2.2.4 Site Use 9
  - 2.2.5 Scale of Exposure Areas 9
- 2.3 Bald Eagle 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 10
  - 2.3.4 Site Use 11
  - 2.3.5 Scale of Exposure Areas 11
- 2.4 Osprey 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 12
  - 2.4.4 Site Use 12
  - 2.4.5 Scale of Exposure Areas 13
- 2.5 Mink 13
  - 2.5.1 Body Weight and Daily Food Ingestion Rate 13
  - 2.5.2 Incidental Sediment Ingestion Rate 13
  - 2.5.3 Diet Composition 13
  - 2.5.4 Site Use 16
  - 2.5.5 Scale of Exposure Areas 16
  - 2.5.6 Prey Item Abundance Analysis 16
- 2.6 River Otter 19
  - 2.6.1 Body Weight and Daily Food Ingestion Rate 19
  - 2.6.2 Incidental Sediment Ingestion Rate 20
  - 2.6.3 Diet Composition 20
  - 2.6.4 Site Use 21
  - 2.6.5 Scale of Exposure Areas 21

2.6.6 Prey Item Abundance Analysis 22
2.7 Belted Kingfisher 22
2.7.1 Body Weight and Daily Food Ingestion Rate 22
2.7.2 Incidental Sediment Ingestion Rate 23
2.7.3 Diet Composition 23
2.7.4 Site Use 23
2.7.5 Scale of Exposure Areas 23

3.0 BIRD EGG ASSESSMENT EXPOSURE ASSUMPTION DETAILS 24

4.0 DERIVATION OF DIETARY TOXICITY REFERENCE VALUES 29
4.1 Bird TRVs 31
4.1.1 Aluminum 31
4.1.2 Arsenic 31
4.1.3 Cadmium 31
4.1.4 Chromium 32
4.1.5 Copper 32
4.1.6 Lead 33
4.1.7 Mercury 33
4.1.8 Selenium 33
4.1.9 Thallium 34
4.1.10 Zinc 34
4.1.11 Tributyltin 34
4.1.12 Total PCBs 35
4.1.13 2,3,7,8-TCDD 35
4.1.14 Benzo(a)pyrene 36
4.1.15 HPAHs and LPAHs 36
4.1.16 Aldrin 37
4.1.17 DDE 37
4.1.18 Total DDx 37
4.1.19 BEHP 38
4.1.20 Dibutyl Phthalate 38
4.2 Mammal TRVs 39
4.2.1 Aluminum 39
4.2.2 Antimony 39
4.2.3 Copper 40
4.2.4 Lead 40
4.2.5 Mercury 41
4.2.6 Selenium 41
4.2.7 HPAHs 42
4.2.8 Total PCBs 42
4.2.9 2,3,7,8-TCDD 42
4.2.10 Total DDx 43

5.0 DERIVATION OF BIRD EGG TOXICITY REFERENCE VALUES 44

5.1 Dioxins/Furans and Dioxin-Like PCBs 44
5.2 Total PCBs 45
5.3 DDE 45
5.4 Mercury 45

6.0 REFERENCES 47

LIST OF TABLES

Table 2-1. Exposure Parameters Used for Wildlife 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 2-4. Reported Sandpiper Species Home Range 7
Table 2-5. Percent Frequency of Prey Occurrence Reported in Mink Scat 14
Table 2-6. Relative Abundance of Selected Piscivorous Fish Prey Species 17
Table 3-1. Selected Literature-Based BMFs 24
Table 3-2. Summary of BMFs Derived from the Literature 25
Table 4-1. Bird Dietary Dose TRVs 29
Table 4-2. Mammal Dietary Dose TRVs 30
Table 5-1. Bird Egg Tissue-Residue TRVs 44

1.0 INTRODUCTION

Section 8.1 of the baseline ecological risk assessment (BERA) presents the dietary assessment for wildlife. For "readability" purposes, details of the BERA, where negotiated or directed, and supporting information are provided separately in attachments, including this attachment. This attachment presents the details and sources for the exposure parameters and effects thresholds (i.e., toxicity reference values [TRVs]) presented in Sections 8.1.2 and 8.1.3 of the BERA, respectively. Section 2.0 of this attachment presents the details of the exposure assumptions, and Section 3.0 presents the details of the selected wildlife diet TRVs. This level of detail was not provided in the main text of the BERA because much of this information has been discussed extensively with EPA or its use has been directed by EPA.

2.0 DIETARY DOSE ASSESSMENT EXPOSURE ASSUMPTION DETAILS

Section 8.1.2 of the BERA presents a summary of the wildlife exposure parameters used in the dietary assessment for wildlife. Table 2-1 presents the exposure parameters used in the wildlife dietary risk calculations. 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. The following subsections present the receptor-specific exposure assumptions used in the dietary line of evidence (LOE) for wildlife for each receptor.

Table 2-1. Exposure Parameters Used for Wildlife Dietary Risk Calculations

Receptor	BW (kg)	BW Source	FIR (kg ww/day)	FIR Source	SI (%)[a]	% Moisture in Prey	SIR (kg dw/day)[b]	SUF	Exposure Scale
Spotted sandpiper	0.047	Maxson and Oring (1980)[c]	0.055	Nagy (2001)	18[d]	85%[e]	0.0015	1.0	2 miles[f]
Hooded merganser	0.54	Dunning (1993)	0.20	Nagy (2001)	2[g]	NA	0.0011	1.0	1 mile
Bald eagle	4.5	Wiemeyer (1991)[c]	0.54	Stalmaster and Gessaman (1984)[c]	2[g]	74%[h]	0.0028[g]	1.0	1 mile
Osprey	1.9	Poole (1983)[c]	0.40	Poole (1983)[c]	2[g]	74%[h]	0.0021[g]	1.0	1 mile
Mink	0.97	Hornshaw et al. (1983)[c]	0.16	Bleavins and Aulerich (1981)[c]	9.4[d,j]	74%[i]	0.0038[g]	1.0	1 mile
River Otter	7.7	USGS (2004)	0.76	Nagy (2001)	2[g]	NA	0.0047	1.0	3 miles
Belted Kingfisher[k]	0.148	EPA (1993)	0.080	Nagy (1987)	2[g]	79%[l]	0.00033	1.0	1 mile

[a] Percent of incidental sediment ingestion of the dry diet.
[b] 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).
[c] As cited in EPA (1993).
[d] Based on Beyer et al. (1994).
[e] Average percent moisture in benthic invertebrate tissue collected from the Study Area.
[f] Two-mile exposure area for spotted sandpiper includes only beach exposure areas; individual beach exceedances were evaluated to further narrow which COPCs may pose unacceptable risks; however, a 2-mile exposure scale was used to refine risk estimates using a more ecologically relevant scale supported by the literature.
[g] Based on best professional judgment.
[h] Average percent moisture in fish tissue collected from the Study Area.
[i] Average percent moisture in fish and crayfish tissue collected from the Study Area.
[j] Sediment ingestion percent for mink based on raccoon. No data available for mink.
[k] Belted kingfisher was evaluated as part of the uncertainty evaluation.
[l] Average percent moisture in field clams, lab worms and small fish.

BW – body weight
COPC – chemical of potential concern
FIR – food ingestion rate
SI – sediment ingestion
SIR – sediment ingestion rate
SUF – site use factor
NA – not applicable

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

	Birds					Mammals	
Prey Species	**Spotted Sandpiper**	**Hooded Merganser**	**Bald Eagle**	**Osprey**	**Belted Kingfisher**	**Mink**	**River Otter**
Invertebrates							
Crayfish		X				X	X
Clam	X[a]	X[a]			X		X[a]
Worm	X						
Mussel						X	X
Fish							
Largescale sucker			X	X		X	X
Carp			X	X		X	X
Juvenile Chinook salmon					X	X	
Peamouth		X	X		X	X	
Sculpin		X			X	X	X
Smallmouth bass		X[b]		X		X	X
Northern pikeminnow			X	X		X	
Black crappie						X	X
Brown bullhead				X		X	

[a] Both laboratory clam tissue and field-collected tissue concentrations were evaluated.

[b] Per EPA (2008b), smallmouth bass were also used as a representative prey item for hooded mergansers; however, the size range of smallmouth bass collected from the Study Area (8.6 to 18 inches in length) is much greater than the prey size of fish consumed by mergansers, so the use of this prey item to represent hooded merganser dietary concentrations is uncertain.

COPC – chemical of potential concern

EPA – US Environmental Protection Agency

HQ – hazard quotient

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

	Prey Consumption Portion						
	Birds					Mammals	
Prey Species	**Spotted Sandpiper[a]**	**Hooded Merganser**	**Bald Eagle**	**Osprey**	**Belted Kingfisher**	**Mink**	**River Otter**
Invertebrates							
Crayfish		0.05				0.20[b]	0.10
Clam	1.0[c]	0.25[d]			0.1		0.02[d]
Worm	1.0						
Fish							
Largescale sucker			0.45	0.83		0.20[e]	0.04[e]
Carp			0.45	0.06		0.20	0.40
Peamouth		0.05[f]	0.05				
Sculpin		0.65			0.9	0.20[g]	0.40
Smallmouth bass				0.02		0.20	0.04
Northern pikeminnow			0.05[h]	0.07[i]			
Black crappie							
Brown bullhead				0.02[j]			

[a] Two scenarios were evaluated for spotted sandpiper: one based on the ingestion of clams and one based on the ingestion on worms.
[b] Sculpin were used as a surrogate species when no crayfish data were available for the mink diet.
[c] HQs were calculated using laboratory clam tissues only at locations where field-collected tissue concentrations were not available. Field clams are more representative of concentrations in bivalves from the Study Area.
[d] HQs were calculated using only field clam tissue. Field clams are more representative of concentrations in bivalves from the Study Area.
[e] Carp were used as a surrogate species when no largescale sucker data were available for the mink and river otter diet.
[f] Sculpin were used as a surrogate species when no peamouth data were available for the hooded merganser diet.
[g] Smallmouth bass were used as a surrogate species when no sculpin data were available for the mink diet.
[h] Peamouth was used as a surrogate species when no northern pikeminnow data were available for the bald eagle diet.
[i] Smallmouth bass were used as a surrogate species when no northern pikeminnow data were available for the osprey diet.
[j] Smallmouth bass were used as a surrogate species when no brown bullhead data were available for the osprey diet.

2.1 SPOTTED SANDPIPER

This section provides the rationale for spotted sandpiper-specific exposure parameters, as summarized in Tables 2-1 through 2-3.

2.1.1 Body Weight and Daily Food Ingestion Rate

Maxson and Oring (1980), as presented in EPA (1993), reported average adult female and male body weights to be 0.0471 and 0.0379 kg, respectively. Daily food ingestion rates were

estimated as a function of body weight derived from Nagy (2001), who reported the body weight-normalized daily food ingestion rates for common sandpiper as 0.175 g dw/g bw/day. Using the common sandpiper ingestion rate and the female body weight for spotted sandpiper, the calculated food ingestion rate for spotted sandpiper was 0.0082 kg dw/day. The food ingestion rate was converted into wet weight using a percent moisture of 85%, resulting in a female spotted sandpiper food ingestion rate of 0.055 kg ww/day. Eight-five percent moisture was the average percent moisture in shorebird prey (invertebrate tissue) collected from the Study Area during Rounds 1 and 2. The female body weight and ingestion rates of 0.0471 kg and 0.055 kg ww/day, respectively, were the parameters used for the spotted sandpiper.

2.1.2 Incidental Sediment Ingestion Rate

Sandpipers are known to ingest large amounts of sediment while feeding on benthic prey. Beyer et al. (1994) reported that sediment ingestion by four sandpiper species (i.e., stilt sandpiper, semipalmated sandpiper, least sandpiper, and western sandpiper) ranged from 7.3 to 30% of the dry diet, with an average incidental sediment ingestion of 18%. The female dietary sediment ingestion rate of 0.0015 kg dw/day was calculated assuming 18% incidental sediment ingestion.

2.2.3 Diet Composition

Spotted sandpipers feed primarily on terrestrial and aquatic insects (Bent 1929; Csuti et al. 2001). They may occasionally feed on other benthic macroinvertebrates such as crustaceans, mollusks, and worms (Bent 1929; Csuti et al. 2001) or on leeches, small fish, and carrion (Oring et al. 1983). Amphipod or insect tissue was not available from the Lower Willamette River (LWR) Study Area; however, clams and worms were used as a surrogate species to represent the diet of spotted sandpiper.

For the evaluation of individual prey, the sandpiper diet was represented using field clam tissue and 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 spotted sandpiper because both clam and worms are meant to be representative of benthic invertebrate prey that sandpiper may ingest. Instead, dietary exposure of sandpipers was modeled separately using 100% ingestion of field clam tissue[1] and 100% ingestion of worm tissue from the bioaccumulation testing.

There is uncertainty associated with the use of clam and worm tissue chemical concentrations to represent a sandpiper diet of amphipod or insect tissue. 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

[1] Clam tissue from the bioaccumulation testing was used when no field clam tissue data were available.

chemicals of potential concern (COPCs) from laboratory exposures 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 (and clam) tissue chemical concentrations may over or underestimate steady-state concentrations expected in worms and clams in the Study Area.

2.2.3 Site Use

A site use factor of 1.0 was used to estimate the exposure of spotted sandpipers because they are locally common breeders in the Willamette Valley, and some are present year-round (Puchy and Marshall 1993). It has not been confirmed that spotted sandpipers are, in fact, present in the Study Area year-round; therefore, the assumed site use factor of 1.0 may overestimate sandpiper exposure to contaminants in the Study Area.

2.2.4 Scale of Exposure Areas

Potential shorebird foraging beach habitats were based on the beaches identified during the reconnaissance survey conducted during Round 2 in summer 2004 (Saban and Andersen 2004). Twenty-eight individual shorebird beaches were identified, and these beach areas were used to represent potential exposure areas for spotted sandpiper, assuming fidelity to the LWR. Prey tissue and sediment data used to estimate dietary exposure concentrations for shorebirds were limited to the samples collected within these designated beach habitat areas.

No published data are available on the home ranges of spotted sandpipers. Data for other sandpiper species suggests that their home ranges vary from approximately 0.5 to 5 miles (0.80 to 8.05 km) (Table 2-4), when resident or breeding. Incubating buff-breasted sandpiper females were observed within 0.62 mile (1 km) of the nest site, and females and their broods seen within a 0.62- to 1.86- square mile (1- to 3-km^2) area (Lanctot and Laredo 1994). Mean distances from nest of upland sandpiper females and males were 0.54 mile (869 m) and 0.15 mile (241 m), respectively [Ailes and Toepfer (1977), as cited in Houston et al. (2001)]. Stilt sandpipers forage off territory up to 4.97 miles (8 km) away (Klima and Jehl 1998; Jehl 1973). Purple sandpipers have been reported to fly mean distance of 0.40 to 1.0 mile (0.65 to 1.6 km) from the nest during incubation (Pierce 1993). Western sandpipers were reported to move 2.5 to 3.7 miles (4 to 6 km) while foraging on beaches during migratory stopover in the Fraser River estuary in British Columbia (Butler et al. 2002). Similarly, western sandpipers in San Francisco Bay have been noted to use a relatively small area (8.5 square miles [22 km^2]) during their winter and spring base stopover (Warnock and Takekawa 1995). Breeding territory sizes tend to be quite small, ranging from 0.50 to 12.4 square miles (812 to 20,000 m^2), with beach length from 0.01 to 0.25 miles (20 to 400 m) long (Oring et al. 1997). The home range for spotted sandpiper using the Lower Duwamish Waterway in Seattle, Washington, was estimated to be 1 mile (1.6 km) from the nest (Norman 2002) during breeding season.

Table 2-4. Reported Sandpiper Species Home Range

Sandpiper Species	Average Home Range[a] km	Average Home Range[a] miles	Source
Buff-breasted sandpiper	1.2	0.76	Lanctot and Laredo (1994).
Upland sandpiper	0.55	0.34	Ailes and Toepfer (1977), as cited in Houston et al. (2001)
Stilt sandpiper	8	5.0	Klima and Jehl (1998), Jehl (1973)
Purple sandpiper	1.1	0.70	Pierce (1993)
Western sandpiper	4.9	3.0	Butler et al. (2002), Warnock and Takekawa (1995)
Average (*Median*)	**3.2 (*1.2*)**	**2.0 (*0.75*)**	

[a] Home ranges based on non-migratory and/or nesting periods.

A home range of 2 miles (3.2 km) was assumed to represent an appropriate exposure area for spotted sandpiper based on the published home range data reported for other sandpiper species. Risk estimates for shorebird populations evaluated beach foraging areas within 2-mile (3.2-km) segments of the LWR.

2.2 HOODED MERGANSER

This section provides the rationale for hooded merganser-specific exposure parameters, as summarized in Tables 2-1 through 2-3.

2.2.1 Body Weight and Daily Food Ingestion Rate

Dunning (1993) reported adult female hooded merganser body weights ranging from 0.54 to 0.68 kg and adult male body weights ranging from 0.68 to 0.91 kg. The daily food ingestion rate was estimated as a function of body weight using the following allometric equations developed for carnivorous birds (Nagy 2001):

$$FIR = 3.048 \times BW^{0.665}$$ **Equation 2-1**

Where:

FIR = daily food ingestion rate (g ww/day)
BW = body weight (g)

or

$$FIR = 0.849 \times BW^{0.663}$$ **Equation 2-2**

Where:

FIR = daily food ingestion rate (g dw/day)
BW = body weight (g)

Using the lower of the average female body weight reported in Dunning (1993), the calculated female food ingestion rate was 0.20 kg ww/day and 0.055 kg dw/day (using Equations 2-1 and 2-2, respectively). The female food ingestion rate calculated based on wet

weight was used to estimate dietary exposure doses for hooded merganser. The female body weight and ingestion rates of 0.54 kg and 0.020 kg ww/day, respectively, were the parameters used for hooded merganser.

2.2.2 Incidental Sediment Ingestion Rate

Hooded mergansers are likely to ingest a small amount of sediment incidentally while foraging and indirectly through their prey. An incidental sediment ingestion rate of 2% was used when calculating dietary exposure for hooded mergansers, based on best professional judgment. The female dietary sediment ingestion rate of 0.0011 kg dw/day was calculated assuming 2% incidental sediment ingestion.

2.2.3 Diet Composition

Hooded mergansers feed primarily by diving for whatever small fish are abundant, but they will also eat aquatic invertebrates, especially as hatchlings (Csuti et al. 2001). They are also known to feed on crustaceans, aquatic insects, and small fish (Bendell and McNicol 1995). No data are available on the food preferences of the hooded merganser in Oregon (Marshall et al. 2003). The stomach contents of 138 mergansers from around the United States were presented in Cottam and Uhler (1937), as cited in Dugger et al. (1994). Fish or unidentifiable fish fragments made up 44% of the total stomach contents, and crustaceans (including crayfish) and aquatic insects made up 45%. Mollusks made up less than 1% of the total stomach contents. Prey sizes of fish for hooded merganser have been reported to be 2 inches or less (Alexander 2000).

Hooded mergansers were selected to represent fish-eating water birds in the LWR, birds such as diving water fowl, great blue herons, and other species; therefore, small fish are key prey for this receptor. Sculpin are a key representative resident small fish (sculpin collected from the Study Area ranged from 3.3 to 6.9 inches in length[2]) that are likely an important fish prey item for mergansers. Peamouth were used as a surrogate species to represent small resident pelagic fish prey for mergansers. There is some uncertainty associated with the use of peamouth to represent small pelagic fish because the peamouth collected from the Study Area had body lengths that ranged from 7.1 to 10.7 inches. Per EPA (2008b), smallmouth bass were also used as a representative prey item for hooded mergansers; however, the size range of smallmouth bass collected from the Study Area (8.6 to 18 inches in length) is much greater than the prey size of fish consumed by mergansers, so the use of this prey item to represent hooded merganser dietary concentrations is uncertain. Because hooded mergansers are also known to prey on invertebrates, clams and crayfish were also selected to be representative prey. In the evaluation of individual prey species, these five representative prey species (i.e., sculpin, peamouth, smallmouth bass, clams, and crayfish) were used to estimate dietary exposure concentrations for hooded merganser.

In the evaluation of multiple prey species, the dietary portions were assigned to represent a more realistic dietary scenario. Hooded mergansers were selected as a receptor to represent

[2] Data on size ranges of fish collected from the Study Area are presented in Attachment 4.

fish-eating water birds in the LWR; therefore, small fish (i.e., sculpin and peamouth) were assumed to make up 70% of the diet. Sculpin and peamouth were selected to represent 65% and 5% of the total merganser diet, respectively. Sculpin are a representative resident small fish that are likely an important prey item for mergansers. Peamouth were selected to represent resident small pelagic fish and likely represent some portion of the merganser diet. The remaining portion of the merganser diet was represented by field clams[3] (25%) and crayfish (5%).

2.2.4 Site Use

Hooded mergansers are common local breeders in the Willamette Valley with some being year-round residents (Marshall et al. 2003). Therefore, a site use factor of 1.0 was used to estimate the exposure of hooded mergansers in the Study Area. Mergansers may also forage in ponds and other aquatic environments outside of the Study Area; therefore, there is uncertainty associated with the use of a site use factor of 1.0, which may overestimate the exposure of merganser populations to study-area related contaminants.

2.2.5 Scale of Exposure Areas

No published data are available on the home ranges of hooded mergansers. Due to the paucity of information on the foraging range of mergansers, a 1-mile (1.6-km) foraging range was conservatively assumed to represent an exposure scale for hooded mergansers.

2.3 BALD EAGLE

This section provides the rationale for bald eagle-specific exposure parameters, as summarized in Tables 2-1 through 2-3.

2.3.1 Body Weight and Daily Food Ingestion Rate

Wiemeyer (1991), as cited in EPA (1993), reported average adult female and male body weights for bald eagles to be 4.5 and 3.0 kg, respectively. The food ingestion rate was represented as 12% of the body weight on a wet-weight basis, based on a study by Stalmaster and Gessaman (1984), as presented in EPA (1993), of free-flying eagles in Washington. Using the average female bald eagle body weight, the calculated food ingestion rate was 0.54 kg ww/day. The female body weight and ingestion rates of 4.5 kg and 0.54 kg ww/day, respectively, were the parameters used for bald eagle.

2.3.2 Incidental Sediment Ingestion Rate

Data on sediment ingestion rates were not available for bald eagles, but it is likely that bald eagles consume a small amount of sediment while scavenging along the shoreline. An incidental sediment ingestion rate of 2% was used when calculating the dietary exposure for bald eagles based on best professional judgment. The female dietary sediment ingestion rate of 0.0028 kg dw/day was calculated assuming 2% incidental sediment ingestion and 74%

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

moisture in the diet. Seventy-four percent moisture was the average percent moisture in fish prey for bald eagles collected from within the Study Area.

2.3.3 Diet Composition

Bald eagles are opportunistic foragers with site-specific food habits based on available prey species (Buehler 2000; Anthony et al. 1999). In most regions, bald eagles seek out aquatic habitats for foraging and prefer fish (Buehler 2000; Ehrlich et al. 1988). They also eat carrion, various water birds, and small mammals (Csuti et al. 2001). In one study conducted in the lower Columbia River estuary, the diet composition of bald eagles based on direct observation was 90% fish, 7% birds, and 3% mammals (Watson et al. 1991). Other studies conducted in western Washington have reported similar percentages of fish in the bald eagle diet (Knight et al. 1990; Watson and Pierce 1998; Watson et al. 1995). Freshwater fish were the most commonly found prey in bald eagle nests in the Columbia River estuary by Watson et al. (1991). The following proportion of fish prey ingested by bald eagles were reported: 24% Catostomidae (largescale sucker), 20% Clupeidae (American shad), 35.5% Cyprinidae (including common carp and peamouth), 12% Salmonidae (Chinook salmon and steelhead), 4% Centrarchidae (including black crappie), and 7% other fish species. Carp and largescale sucker are likely the most common prey for bald eagle in the Study Area based on the prey reported in the regional literature. Sizes of fish consumed by bald eagle typically are < 12 inches (30 cm), ranging from 4 to 12 inches (10 – 30 cm), although fish > 12 inches are also consumed by eagles, most frequently during breeding (Watson and Pierce 1998; Watson et al. 1995; Buehler 2000).

Exposure concentrations in the bald eagle diet were estimated using 100% fish species as prey. In the evaluation of individual prey species, these four representative resident fish prey (i.e., carp, largescale sucker, peamouth, and northern pikeminnow) were used to determine exposure concentrations in the bald eagle diet. Average fish lengths for carp, largescale sucker, peamouth, and northern pikeminnow collected from the Study Area were 21 inches (54 cm), 16 inches (42 cm), 8.5 inches (22 cm), and 14.5 inches (37 inches), respectively. Average size ranges of Study Area fish are slightly larger than fish typically consumed by bald eagles (4-12 inches) but may represent the sizes of fish that are consumed during breeding (> 12 inches). Chinook salmon tissue collected from the Study Area was not used to represent salmonids because only juvenile tissue was collected, and bald eagles prey on adult salmon. The four selected fish prey species will act as surrogate species, representing an estimate of tissue residues in adult salmon.

In the evaluation of multiple prey species, the dietary portions were assigned to represent a more realistic dietary scenario. Watson et al. (1991) reported primarily Catostomidae (largescale sucker), Clupeidae (American shad), and Cyprinidae (including common carp) fish in Columbia River estuary bald eagle nests, each making up the majority of the nest remains; therefore, carp and largescale sucker were assumed to make up 90% of the bald eagle diet. Carp and largescale sucker were each assigned a prey portion of 45%. Northern pikeminnow and peamouth were each selected to represent 5% of the total bald eagle diet because these fish also belong to the Cyprinidae family.

No tissue data for birds and mammals that may serve as prey for bald eagles are available. There is uncertainty associated with using 100% fish to represent the bald eagle diet (and excluding birds and mammals), and dietary exposure from the Study Area may be over or underestimated for bald eagle.

2.3.4 Site Use

Bald eagles are known to nest up and down the Willamette River. The closest known nest to the Study Area is on Ross Island at RM 15, though two old nests are located on Sauvie Island (Isaacs and Anthony 2001). Eagles are year-round residents in western Oregon, with some eagles from further north over-wintering in the area (Csuti et al. 2001). Therefore, a site use factor of 1.0 was used to determine dietary exposure for breeding bald eagles. Because non-breeding bald eagles may forage in nearby aquatic and terrestrial environments in addition to the Study Area, there is uncertainty associated with the use of a site use factor of 1.0, which may overestimate the exposure of eagles to study-area related contaminants.

2.3.5 Scale of Exposure Areas

Shoreline foraging ranges from the literature indicate that bald eagles may forage in sediment reaches that are smaller than the entire length of the Study Area. Garrett et al. (1993) determined that home ranges of breeding bald eagles in the lower Columbia River estuary extended from 5.9 to 47.3 km^2 (with an average home range of 21.7 km^2). The average shoreline range of these home ranges in breeding bald eagles was 3.5 miles (5.6 km). Garrett et al. (1993) also reported breeding areas of 0.11 square miles (0.3 km^2) for 25% of the birds studied, 0.5 square mile (1.3 km^2) for 50% of the birds studied, and 8.5 square miles (22 km^2) for nearly all (95%) of the birds studied. Per EPA's Problem Formulation (Attachment 2), a foraging range of 1 mile (1.6 km) was conservatively assumed for estimating the exposure of bald eagles based on the data reported in the literature regarding bald eagle foraging ranges in the Columbia River.

2.4 OSPREY

This section provides the rationale for osprey-specific exposure parameters, as summarized in Tables 2-1 through 2-3.

2.4.1 Body Weight and Daily Food Ingestion Rate

Poole (1983), as cited in EPA (1993), reported average adult female and male osprey body weights during courtship to be 1.88 and 1.48 kg, respectively. The food ingestion rate during the courtship period were represented as 21% of the body weight on a wet weight basis, based on studies of adult female osprey in Massachusetts (Poole 1983), as presented in EPA (1993). Using the average female osprey body weight, the calculated food ingestion rate was 0.40 kg ww/day.

2.4.2 Incidental Sediment Ingestion Rate

Data on osprey sediment ingestion were not available, but osprey may consume a small amount of sediment while scavenging along the shoreline (which occurs very rarely).

Therefore, a sediment ingestion of 2% of the dry diet will be assumed for calculating the dietary exposure for osprey. The female dietary sediment ingestion rate of 0.0021 kg dw/day was calculated assuming 2% incidental sediment ingestion and 74% moisture in the diet. Seventy-four percent moisture was the average percent moisture in fish prey for osprey collected from within the Study Area during Rounds 1 and 2.

2.4.3 Diet Composition

Osprey tend to feed solely on fish, primarily on slow-moving fish that swim near the water surface (Poole et al. 2002; Csuti et al. 2001). They may occasionally eat other types of vertebrate prey such as birds, reptiles, and small mammals, and they only rarely feed on invertebrates. Size classes of fish caught by osprey in Idaho were reported by Van Daele and Van Daele (1982): 89% of fish preyed on by osprey were 4.3 to 11.8 inches (11 to 30 cm) long, suggesting a preference for medium-sized fish by osprey. Henny et al. (2003) investigated the diet of a migratory population of osprey nesting along the main stem of the Willamette River and the McKenzie River in 1993. The following proportions (by biomass) of fish prey were reported based on the identification of prey in osprey nests: largescale suckers (83%), northern pikeminnow (7%), common carp (6%), largemouth and smallmouth bass (2%), and invertivorous fish (2%) (includes mountain whitefish, black crappie, white crappie, bullhead species, and bluegill). Based on these data, largescale sucker likely make up the largest portion of the diet of osprey foraging in the Study Area. In the evaluation of individual prey species, exposure concentrations in the osprey diet were estimated using five representative fish prey: largescale sucker, northern pikeminnow, carp, smallmouth bass, and brown bullhead.

In the evaluation of multiple prey species, the dietary portions were assigned to represent a more realistic dietary scenario. The following prey portions were assigned to estimate the osprey diet based on the portions reported from observing osprey nests along the main stem of the Willamette River and the McKenzie River in 1993 (2003): 83% largescale sucker, 7% northern pikeminnow, 6% common carp, 2% smallmouth bass, and 2% brown bullhead.

2.4.4 Site Use

Ospreys are present from March until September, with several breeding pairs nesting in or near the Study Area (Henny et al. 2003). In 2001, 10 osprey nests were observed between RM 0 and RM 26, several of which were located within the Study Area boundaries (Henny et al. 2008). Each fall, they migrate south to western Mexico and Central America (Martell et al. 2001). Because osprey breed in the Study Area, a site use factor of 1.0 was used to estimate exposure inasmuch as osprey could be exposed to Study Area contaminants during a critical life stage. Because non-breeding osprey forage outside the Study Area during the migratory season, there is uncertainty associated with the use of a site use factor of 1.0. However, Elliott et al. (2007) reported that a wintering site had no significant effect on organochlorine contaminant concentration in sample eggs, including ospreys from Lower Columbia River, which supports the assumption of a site use factor equal to 1.0.

2.4.5 Scale of Exposure Areas

Foraging behavior of osprey based on the literature suggests that osprey may forage in areas smaller than the 10-mile (16-km) Study Area. Breeding adults can have a home range of 6.2 to 8.7 miles (10 to 14 km) from nesting sites (Poole et al. 2002); however, smaller foraging ranges have been reported. Reported foraging areas for osprey range from 0.6 to 3.1 miles (1 to 5 km) (Greene et al. (1983) and Prevost (1982), as cited in Poole et al.(2002)). Per EPA (2008b), osprey have been known to forage in approximately a 1-mile (1.6-km) area along the Willamette River. The exposure scale of 1 mile (1.6 km) was assumed for estimating the exposure of osprey based on the reported foraging ranges.

2.5 MINK

This section provides the rationale for mink-specific exposure parameters, as summarized in Tables 2-1 through 2-3.

2.5.1 Body Weight and Daily Food Ingestion Rate

Hornshaw et al. (1983), as presented in EPA (1993), reported average farm-raised adult female and male body weights for mink in the summer to be 0.974 and 1.734 kg, respectively. The daily food ingestion rate was estimated as 16 and 12% of body weight on a wet weight basis, based on studies of farm-raised female and male mink, respectively, in Michigan (Bleavins and Aulerich 1981), as presented in EPA (1993). Using the female mink parameters, the calculated food ingestion rate for females was 0.16 kg ww/day.

2.5.2 Incidental Sediment Ingestion Rate

Data were not available on the amount of sediment consumed by mink while feeding. Beyer et al. (1994), as presented in EPA (1993), reported a soil ingestion rate of 9.4% of the dry diet for raccoons. The female dietary sediment ingestion rate of 0.0038 kg dw/day was calculated assuming 9.4% incidental sediment ingestion and 74% moisture in the diet. Seventy-four percent moisture was the average percent moisture in crayfish and fish prey for mink collected from within the Study Area. The incidental sediment ingestion rate assumed for mink based on soil ingestion by raccoons is conservative (i.e., may overestimate sediment ingestion) and may overestimate incidental exposure of mink to Study Area-related contaminants in sediments.

2.5.3 Diet Composition

Mink are opportunistic feeders and consume a range of prey, including muskrats, fish, frogs, crayfish, small mammals, and birds found near water (Csuti et al. 2001). The diet of mink depends largely on prey availability (Melquist et al. 1981; Racey and Euler 1983; Ward et al. 1986; Wise et al. 1981), and portions of fish in the mink diet vary widely across field studies. Mink diet composition has been reported in several studies (WDG 1980; Wise et al. 1981; Melquist et al. 1981; Alexander 1977; Korschgen [1958] and Sealander [1943], both as reported in EPA 1993). Limited data were identified regarding the percentages of fish prey in the mink diets based on weight or volume of prey. Alexander (1977) reported that fish made up 85% of the mink diet (56% of which was trout), based on the wet weight of stomach

contents in mink from a Michigan river. Crustaceans, amphibians, and birds/mammals made up 4, 3, and 6%, respectively, of mink stomach contents by wet weight (Alexander 1977). EPA (1993) reported diets composed of 5 to 20% fish, 2 to 9% crayfish, 6 to 15% birds, 38 to 54% mammals, and 25% amphibians and reptiles based on stomach contents of mink from non-stream habitat in Michigan and from throughout Missouri.

Regional data were available from the Columbia River drainage basin. The Washington Department of Game (WDG; now the Washington Department of Fish and Wildlife [WDFW]) analyzed the percent frequency of prey in 259 mink scats from eight reaches of the Columbia River from the Chief Joseph Dam (at RM 546 of the Columbia River) to Vancouver, Washington (RM 107) (WDG 1980). These data do not necessarily reflect the relative biomass of prey; however, because biomass data are not available, percent frequency was assumed to correspond to the relative importance of each prey type. The data collected in the area between the Bonneville to Vancouver reach (from RM 146 to 107; n=2) may be most relevant to Portland Harbor mink because this reach of the Columbia River is the closest to the Study Area; the mouth of the Willamette River is located at RM 101 of the Columbia River. The frequency of prey in scat from this reach, as well as the data from the entire study, is presented in Table 2-5. Fish, crayfish, birds, and mammals were reported as key prey, and the most frequently eaten fish throughout the year that were identified in the Columbia River study were carp, sculpin, suckers, and centrarchids (WDG 1980).

Table 2-5. Percent Frequency of Prey Occurrence Reported in Mink Scat

	Percent Frequency of Occurrence			
Prey Item	**Lower Columbia River Reach (RM 107 – RM 146)[a]**	**Columbia River (RM 107 – RM 546)[a]**	**North Fork Payette River, Idaho[b]**	**Webburn River, England[c]**
Fish	**50%**	**32%**	**59%**	**24.8%**
Carp	50%	3%	0%	NR
Largemouth/smallmouth bass	50%	2%	0%	NR
Largescale sucker	0%	0%	0%	NR
Mottled sculpin	0%	0%	7%	NR
Brown bullhead	0%	0%	0%	NR
Salmonidae[d]	0%	1%	10%	NR
Invertebrates	**0%**	**47%**	**0%**	**1.8%**
Crustaceans (crayfish)	0%	47%	0%	0%
Other	0%	0%	0%	1.8%
Birds	**0%**	**29%**	**19%**	**4.8%**
Mammals	**50%**	**26**	**43%**	**57%**
Reptiles/Amphibians	**0%**	**1%**	**2%**	**9.6%**
Insects (aquatic and	**0%**	**9%**	**24%**	**0%**

Table 2-5. Percent Frequency of Prey Occurrence Reported in Mink Scat

Prey Item	Percent Frequency of Occurrence			
	Lower Columbia River Reach (RM 107 – RM 146)[a]	Columbia River (RM 107 – RM 546)[a]	North Fork Payette River, Idaho[b]	Webburn River, England[c]
terrestrial)				
Plants	**0%**	**0%**	**0%**	**2%**

[a] As reported in WDG (1980).
[b] As reported in Melquist et al. (1981).
[c] As reported in Wise et al. (1981).
[d] Includes mountain whitefish, kokanee, and other unidentified fish species.
NR – not reported
RM – river mile

A similar range of fish percentages in the mink diet were reported in two additional field studies (Table 2-5). Melquist et al. (1981) analyzed the percent frequency of prey in 659 mink scats from the North Fork Payette River, Idaho. In this study, the frequency of occurrence of fish (primarily cyprinids) in the mink diet was 77%. Melquist et al. (1981) reported on the feeding behavior of mink; mink observed in the study would travel along floating logs or logjams to dive for food in the water. It was also noted that sculpin were less available to mink because they live on stream bottom. Wise et al. (1981) analyzed the percentage of prey in mink scats from the Webburn River in England. The results indicated a seasonal variation in the diet composition; mink tend to eat more fish in the winter than in the summer, and more birds in the summer and fall. Salmonids were present in 22% of scats and were caught more frequently in the winter. There was no seasonal variability in the overall mammal consumption. Wise et al. (1981) reported that mink do not appear to select fish prey from a particular size range. Melquist et al. (1981) reported that mink preyed primarily on fish ranging from 7 to 12 cm but opportunistically ate dead and dying salmonids and other fish ranging up to 30 cm.

Because of the opportunistic nature of mink and based on the available data, in the evaluation of individual prey species, crayfish, mussels, and all fish species collected from the Study Area (i.e., largescale sucker, carp, sculpin, smallmouth bass, peamouth, juvenile Chinook salmon, northern pikeminnow, brown bullhead, and black crappie) were used to represent potential prey for mink and estimate exposure concentrations.

In the evaluation of multiple prey species, the dietary portions were assigned to represent a more realistic dietary scenario. The mink diet assumed 100% ingestion of fish and crayfish, although birds and mammals have been known to make up a portion (occurring in 29% and 26% of scats, respectively) of Columbia River mink diets (WDG 1980). Mink are opportunistic feeders, and fish species that are most abundant in the Study Area are likely to be more frequently preyed upon by mink than less abundant species. An evaluation was conducted (Section 2.5.6) to determine the relative abundance of potential fish prey collected from the Study Area and estimate the relative contribution of fish species to the mink diet. The most frequently eaten fish throughout the year that were identified in the Columbia River

study were carp, sculpin, suckers, and centrarchids; and crayfish were found in almost half of Columbia River mink scat remains (WDG 1980). Based on the semi-quantitative evaluation of the relative abundance of fish in the Study Area (Section 2.5.6), largescale suckers were the most abundant fish, and carp, sculpin, and smallmouth bass have relative similar abundances in the Study Area. Assuming equal portions of these five prey species (i.e., crayfish, largescale sucker, carp, sculpin, and smallmouth bass), a prey portion of 20% was assigned to each of these prey species. No prey portions were assigned to peamouth, northern pikeminnow, juvenile Chinook salmon, brown bullhead, or black crappie because these species were either not commonly reported as prey species and/or do not appear to be commonly abundant in the Study Area (Section 2.5.6).

No tissue data are available for birds, mammals, and amphibians that may serve as prey for mink. There is uncertainty associated with using only fish and crayfish to represent the mink diet, and dietary exposure concentrations from the Study Area may be over or underestimated for mink. Uncertainties associated with the dietary assumptions for mink are evaluated in the risk characterization section of the BERA.

2.5.4 Site Use

Mink are semi-aquatic mammals that are closely associated with water and riparian habitats. They are active year-round and are primarily nocturnal. A site use factor of 1.0 was assumed for estimating exposure concentrations in the mink diet.

2.5.5 Scale of Exposure Areas

Mink have linear home ranges along rivers and move considerable distances in streams and on river banks (Csuti et al. 2001). Environment Canada (2003) reported mink home ranges as generally restricted to areas adjacent to bodies of water, with American mink males ranging over 1.6 to 3.4 miles (2.5 to 5.5 km) of shoreline, and female mink ranging over a more restricted 0.3 to 1.7 miles (0.5 to 3 km) of shoreline. Therefore, an exposure scale of 1 mile (1.6 km) was assumed for estimating the exposure of mink based on reported home ranges.

2.5.6 Prey Item Abundance Analysis

Mink (and river otter as discussed in Section 2.6.6) are opportunistic feeders, and fish species that are most abundant in the Study Area are likely to be more frequently preyed upon by these piscivorous mammals than less abundant species. To provide a basis for selecting prey items that were most abundant in the Study Area and for varying prey portions in the uncertainty analysis, a semi-quantitative evaluation was conducted to determine the relative abundance of potential fish prey collected from the Study Area and estimate the relative contribution of fish species to the mink and river otter diet. The relative abundance of fish species in the Study Area was determined based on available data.

The Study Area is approximately a 10-mile (16-km) stretch (from RM 1.9 to RM 11.8). Three studies that compared the relative abundance of fish species in the lower Willamette River, from the confluence of the Willamette River with the Columbia River (RM 0) to RM 12, were identified:

- Hughes and Gammon (1987) collected fish from approximately RM 1.2 to RM 176 of the Willamette River in August of 1983 using a boat electroshocker. Two of the sampling sites (at RM 1.2 and RM 3.1) were located in the Study Area.
- Farr and Ward (1993) collected fish from the mouth of the LWR (approximately RM 0) to RM 17 in 1987 through 1990 during the months of expected juvenile salmonid migration (March though June). Fish were sampled through a variety of methods (i.e., gillnets, electroshocking, trap nets, beach seines, and purse seines); however, abundance data were only presented for bottom gillnets and electrofishing because catch using the other gear was almost entirely juvenile salmon. Eight of the sampling sites were located within the Study Area (at RM 0, RM 1.2, RM 5, RM 5.1, RM 6.8, RM 7.6, RM 9.8, and RM 9.9).

Tetra Tech (1995) collected fish using a boat electroshocker from 26 stations along the entire mainstem stretch of the Willamette River in October of 1992, 1993, and 1994. Three sites were located within the Study Area at RM 1, RM 6, and RM 9.

Data from these three studies provide an indication of the relative abundance of potential fish prey that have been sampled in the Study Area for piscivorous receptors. Nine potential fish prey species have been sampled from the Study Area: largescale sucker, carp, brown bullhead, peamouth, sculpin, juvenile Chinook salmon, black crappie, smallmouth bass, and northern pikeminnow. Table 2-6 presents the number and relative abundance of fish caught in the three studies for those nine fish prey species that have been sampled from the Study Area. Table 2-6 also presents a potential dietary frequency range that each possible fish prey species comprises in the mink and river otter fish portion diet.

Table 2-6. Relative Abundance of Selected Piscivorous Fish Prey Species

	RM 1.2 – RM 3.1 (August 1983)[a]		RM 0 – RM 9.9 (1987 – 1990)[b]		RM 1 – RM 9 (1992 – 1994)[c]			
Species	**Total Fish Caught**	**Percent Total**	**Total Fish Caught**	**Percent Total**	**Total Fish Caught**	**Percent Total**	**Median Percent Total**	**Dietary Frequency Range[d]**
Largescale sucker	56	39%	1,918	48%	99	38%	39%	35 – 50%
Northern pikeminnow	36	25%	623[e]	16%	12	5%	16%	5 – 25%
Common carp	26	18%	ND[f]	ND	21	8%	13%	5 – 20%
Peamouth	5	3%	1,139	29%	32	12%	12%	0 – 30%
Sculpin	22[g]	15%	ND[h]	ND	15[i]	6%	11%	5 – 15%
Smallmouth bass	0	0%	77	2%	80	31%	2%	0 – 35%
Chinook salmon	0	0%	50	1%	2	1%	1%	0 – 5%
Black crappie	0	0%	135	3%	0	0%	0%	0 – 5%
Brown	0	0%	27[j]	1%	0	0%	0%	0 – 5%

Table 2-6. Relative Abundance of Selected Piscivorous Fish Prey Species

	RM 1.2 – RM 3.1 (August 1983)[a]		RM 0 – RM 9.9 (1987 – 1990)[b]		RM 1 – RM 9 (1992 – 1994)[c]			
Species	**Total Fish Caught**	**Percent Total**	**Total Fish Caught**	**Percent Total**	**Total Fish Caught**	**Percent Total**	**Median Percent Total**	**Dietary Frequency Range[d]**
bullhead								
Total	**145**	**100%**	**3,969**	**100%**	**261**	**100%**	**94%**	

[a] Based on reported results of Hughes and Gammon (1987). Fish were caught using a boat electroshocker.

[b] Based on reported results of Farr and Ward (1993). Fish were caught by gillnets and electrofishing in March through June in 1987, 1988, 1989, and 1990.

[c] Based on reported results of Tetra Tech (1995). Fish were caught using a boat electroshocker in October of 1992, 1993, and 1994.

[d] Minimum percent total was rounded down to nearest 5% and maximum percent total was rounded up to nearest 5%.

[e] Sizes of northern pikeminnow were ≥ 7.87 inches (200 mm); fish < 7.87 inches (200 mm) were not counted.

[f] Carp were reported to be common in gillnets and throughout the area sampled but were not counted.

[g] Prickly sculpin was the only sculpin species caught at RM 3.1. Eighteen prickly sculpin and one reticulate sculpin were caught at RM 1.2.

[h] Sculpin were reported to be common throughout the area sampled but were not counted. Prickly sculpin constituted approximately 85% of sculpin identified to species in this study.

[i] Paiute sculpin, prickly sculpin, and reticulate sculpin were caught between RM 6 and RM 9 in 1993; prickly sculpin was the only sculpin species caught between RM 6 and RM 9 in 1994.

[j] Brown bullhead were observed in study, but not all of the fish collected were counted.

ND – no data

RM – river mile

Largescale sucker were the most abundant fish collected from within the Study Area based on all three studies, ranging from 38 to 48% of the total fish catch abundance. This species was also the most abundant species collected throughout the entire mainstem of the Willamette River (Tetra Tech 1995). Northern pikeminnow also were relatively abundant based on Hughes and Gammon (1987) and Farr and Ward (1993), making up 25 and 16% of the total catch, respectively. While pikeminnow were infrequently caught during the sampling conducted in the lower river stations (from RM 0 to RM 9.9) during October in 1992 to 1994, they were the second most abundant species collected throughout the entire mainstem of the Willamette River (Tetra Tech 1995), indicating that some seasonal movement of northern pikeminnow out of the Study Area may occur. Although specific counts are not reported in Farr and Ward (1993), carp were commonly caught in the Study Area, making up 8 to 20% of the total catch.

The relative abundance of sculpin, smallmouth bass, and peamouth were variable across the three studies. Smallmouth bass were commonly caught during the October sampling in 1992 to 1994, making up almost a third (31%) of the total catch (Tetra Tech 1995); whereas, smallmouth bass were rarely collected in the other two studies. Peamouth abundance varied from 3 to 29% of the total catch. The seasonal and catch method differences among the three studies may account for the variability in the abundance of smallmouth bass and peamouth. Sculpin abundance was also variable, and the catch methods likely explain this variability in abundance because sculpin are difficult to catch using a boat electrofisher, unless they are explicitly targeted.

All studies indicated that black crappie and brown bullhead abundances are relatively low. Juvenile Chinook salmon were rarely caught in two of the studies (Tetra Tech 1995; Hughes and Gammon 1987) but were abundant when salmon-specific sampling methods (e.g., beach seine and vertical gillnet samples) were used (1993). Other salmon-specific sampling efforts have shown juvenile Chinook salmon to be seasonally abundant in the LWR.

2.6 RIVER OTTER

This section provides the rationale for river otter-specific exposure parameters, as summarized in Tables 2-1 through 2-3.

2.6.1 Body Weight and Daily Food Ingestion Rate

Average adult female and male river otter body weights in western Oregon and Washington have been reported for trapped otters submitted to the US Geological Survey (USGS 2004). Body weights were reported without pelts, and weights were adjusted to estimate body weight with pelts using the following equation, as agreed upon by EPA, EPA's partners, and LWG.

$$BW_{unpelted} = BW_{pelted} + \frac{0.5748}{0.8437} \qquad \textbf{Equation 2-3}$$

Where:

$BW_{unpelted}$ = body weight of river otter with skin (kg)
BW_{pelted} = body weight of river otter without skin (kg)

Estimated pelted body weights for adult female and male river otters were 9.46 and 11.15 kg, respectively. Estimated pelted body weights for juvenile female and male river otters were 7.7 and 8.48 kg, respectively. The juvenile female body weight was used to estimate the exposure of river otter. This body weight is similar to adult body weight in other regions; Melquist and Hornnocker (1983), as presented in EPA (1993), reported average female and male river otter body weights in Idaho as 7.9 and 9.2 kg, respectively. The daily food ingestion rate for river otter was estimated as a function of body weight using the following allometric equation developed for carnivorous mammals (Nagy 2001):

$$FIR = 0.102 \times BW^{0.864} \qquad \textbf{Equation 2-4}$$

Where:

FIR = daily food ingestion rate (g dw/day)
BW = body weight (g)

or

$$FIR = 0.348 \times BW^{0.859} \qquad \textbf{Equation 2-5}$$

Where:

FIR = daily food ingestion rate (g ww/day)
BW = body weight (g)

Using the female juvenile river otter body weight, the calculated female food ingestion rate was 0.23 kg dw/day and 0.76 kg ww/day. The female food ingestion rate calculated based on wet weight was used to estimate dietary exposure doses for river otter.

2.6.2 Incidental Sediment Ingestion Rate

No data concerning sediment ingestion for the river otter are available. River otters may ingest a small amount of sediment incidentally while foraging and indirectly through their prey. Therefore, an estimated sediment ingestion rate of 2% of the dry diet will be used when calculating the dietary exposure dose for river otter. The female dietary sediment ingestion rate of 0.0047 kg dw/day was calculated assuming 2% incidental sediment ingestion.

2.6.3 Diet Composition

River otters are opportunistic carnivores that take advantage of food that is most abundant and easiest to catch, although fish are their primary prey (EPA 1993). Slower-moving fish such as suckers, carp, chubs, and bullheads are generally eaten most frequently (Wise et al. 1981; Kurta 1995). Other components of the river otter's diet may include aquatic invertebrates (including crayfish, mussels, clams, and aquatic insects), frogs, snakes, turtles, and occasionally scavenged small mammals and birds (Coulter et al. 1984; Csuti et al. 2001). Field studies in British Columbia, western Oregon, Idaho, and Alaska identifying the prey contents in river otter scat or stomach contents have found that fish remains make up the majority of the prey, occurring more than 85% of the time (Stenson et al. 1984; Larsen 1984; Toweill 1974; Melquist et al. 1981). Fish remains were found in 80 to 83% of river otter scat along the Columbia River (WDG 1980, 1984).

River otters specialize in fish predation. This is supported by the results of Toweill (1974) in which prey were identified in river otter stomachs. In addition to fish prey, crustaceans (most commonly crayfish) were observed in 20% of otter stomachs analyzed, with amphibians and birds occurring less frequently. Mollusks were observed in only 2% of the stomachs examined. The proportion of prey types consumed by river otter will be based on that reported by Larsen (1984) for southeastern Alaskan river otter: 86% fish, 10% crab, 2% invertebrates other than crab, 1% birds, and 1% mammals and plant material. This study was the only study from the Pacific Northwest that reported remains in scat on a volume basis rather than as a frequency of occurrence or as a qualitative observation.

Fish in the river otter diet were represented by those fish commonly observed in region-specific field studies. The primary prey of river otter in the Columbia River in the summer were carp, crayfish, suckers and centrarchids (sunfishes, including bass and perch). Other prey (including American shad, northern pikeminnow, salmon, birds, mammals, insects, and mollusks) were eaten less frequently and were considered of minor importance (WDG 1984). Henny et al. (1996) and the Columbia Basin Fish and Wildlife Authority (CBFWA) (1996) reported carp and crayfish as the most frequently eaten prey of river otter in the summer in

areas within the Lower Columbia River. Another study reported that carp and sculpin were most frequently preyed upon by river otter in the Columbia River estuary year-round (WDG 1984).

In the evaluation of individual prey species, tissue chemical concentrations in clams, crayfish, mussels, and five representative fish prey (i.e., carp, sculpin, largescale sucker, smallmouth bass, and black crappie) were used to determine potential dietary exposure concentrations for river otter.

In the evaluation of multiple prey species, the dietary portions were assigned to represent a more realistic dietary scenario based on the proportion of prey types reported by Larsen (1984) for southeastern Alaskan river otter: 88% fish, 10% crayfish, and 2% field clams.[4] Carp and sculpin were reported as common fish prey in Columbia River river otter diets (CBFWA 1996; Henny et al. 1996). Other prey that have been observed for river otter include largescale sucker and smallmouth bass (WDG 1984). Based on the semi-quantitative evaluation of relative abundance of fish in the Study Area (Section 2.5.6), largescale suckers were the most abundant fish; black crappie abundance is relatively low; and carp, sculpin, and smallmouth bass have relative similar abundances in the Study Area. However, carp and sculpin are more commonly reported as river otter prey. Assuming the majority of the fish portion of the river otter diet is composed of carp and sculpin, a prey portion of 40% was assigned to each of these prey species. Smallmouth bass and largescale sucker were assumed to each make up 4% of the diet. Black crappie was not assigned a prey fraction because it is not a commonly reported prey species and doesn't appear to be commonly abundant.

2.6.4 Site Use

Like mink, river otters are closely associated with water and riparian habitats and are active year-round. A site use factor of 1.0 was assumed for estimating exposure concentrations in the river otter diet.

2.6.5 Scale of Exposure Areas

Foraging behavior of otter based on the literature suggests that river otter may forage in areas smaller than the 10-mile (16.1-km) Study Area. During the breeding season, river otters may travel considerable distances over land (Csuti et al. 2001). A study of river otters in Idaho found the home range length along streams for juvenile, yearling, and adult river otter averaged 24.7 miles (39.8 km) (Melquist and Hornocker 1983). River otters range over an area sufficiently large enough for foraging and reproduction (Melquist and Dronkert 1987); however, they are typically found in a limited number of activity centers within their overall range. At any given time, river otters generally occupy only a few kilometers of stream but often move from one area to another (Nebraska Game and Parks Commission 2000).

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

A radio-tracking study of relocated river otters was conducted as part of the New York State Department of Environmental Conservation river otter reintroduction program. This study showed that river otter ranges were between 0.9 and 14 miles (1.5 and 22.4 km) long, with an average distance of 6 miles (10 km) for individuals monitored in western New York State (Spinola et al. 1999, as cited in EPA 1993). In Alberta, Canada, the annual home range of females with young was reported to be 7.2 square miles (15.8 km^2), and the shoreline foraging lengths of females was as low as 15 miles (33.6 km) (Reid et al. 1994). A foraging range of 3 miles (4.8 km) was conservatively assumed for estimating the exposure of river otter based on EPA's Problem Formulation (Attachment 2); however, the literature supports that river otter home ranges may be larger than this assumed foraging range.

2.6.6 Prey Item Abundance Analysis

Mink and river otter are opportunistic feeders, and fish species that are most abundant in the Study Area are likely to be more frequently preyed upon by these piscivorous mammals than less abundant species. To provide a basis for selecting prey items that were most abundant in the Study Area and for varying prey portions in the uncertainty analysis, a semi-quantitative evaluation was conducted to determine the relative abundance of potential fish prey collected from the Study Area and estimate the relative contribution of fish species to the mink and river otter diet. This relative abundance evaluation was presented in Section 2.5.6.

2.7 BELTED KINGFISHER

An assessment of risks to the belted kingfisher was conducted as part of the uncertainty section of the wildlife dietary risk evaluation. This section provides the rationale for belted kingfisher-specific exposure parameters, as summarized in Tables 2-1 through 2-3. Belted kingfisher assumptions were based on EPA (2008b); however, the belted kingfisher sediment ingestion rate was corrected to reflect 2% incidental ingestion.

2.7.1 Body Weight and Daily Food Ingestion Rate

An average adult body weight of 0.148 kg for belted kingfisher was derived from EPA (1993). The daily food ingestion rate was estimated as a function of body weight using the following allometric equation developed for all birds (Nagy 1987):

$$FIR = 0.648 \times BW^{0.651} \qquad \textbf{Equation 2-6}$$

Where:

FIR = daily food ingestion rate (g dw/day)
BW = body weight (g)

Using a body weight of 0.148 kg, the calculated food ingestion rate was 0.017 kg dw/day. This food ingestion rate was converted into a wet weight ingestion rate, assuming 79% moisture in fish and invertebrate prey. The food ingestion rate calculated based on wet weight, 0.080 kg ww/day, was used to estimate dietary exposure doses for belted kingfisher.

2.7.2 Incidental Sediment Ingestion Rate

Belted kingfishers are likely to ingest a small amount of sediment incidentally while foraging and indirectly through their prey. An incidental sediment ingestion rate of 2% was used when calculating dietary exposure for belted kingfishers, based on best professional judgment. The dietary sediment ingestion rate of 0.00033 kg dw/day was calculated assuming 2% incidental sediment ingestion.

2.7.3 Diet Composition

The diet of belted kingfishers is predominately composed of fish (Hamas 1994). Small fish from 3 to 4 inches (7.6 to 10.2 cm) in length are reported to make up about 90% of the diet of kingfisher (Csuti et al. 2001; Imhof 1962; as cited in Hamas 1994). Fish greater than 5 inches (12.7 cm) are thought to be difficult to swallow for this species [(Salyer and Lagler (1946) as cited in Hamas (1994)]. Small fish collected from the Study Area, such as sculpin (ranging from 3.3 to 6.9 inches in length) and juvenile Chinook salmon (ranging from 3.5 to 4.7 inches in length), represent the length of fish consumed by kingfishers. Peamouth were used as a surrogate species to represent small resident pelagic fish prey for kingfishers. There is some uncertainty associated with the use of peamouth to represent small pelagic fish because the peamouth collected from the Study Area had body lengths that ranged from 7.1 to 10.7 inches. The diet of belted kingfishers can be variable based on the abundance of potential prey items in their foraging area, and they may consume prey other than fish such as crayfish or other prey items (e.g., mollusks, crustaceans, small reptiles and amphibians, insects, small birds, and berries) (Hamas 1994).

Per EPA (2008b), portions of four representative prey items (i.e., sculpin, peamouth, juvenile Chinook salmon, and field-collected clams) were varied to determine estimated dietary exposure concentrations for belted kingfisher. Prey portions were assigned to each of the selected prey items (i.e., 90% sculpin and 10% clams) to further characterize risk estimates in the dietary risk characterization section.

2.7.4 Site Use

Belted kingfishers are common permanent residents throughout Oregon (Marshall et al. 2003), and some are reported year-round residents (Puchy and Marshall 1993). Therefore, a site use factor of 1.0 was used to estimate the exposure of belted kingfishers in the Study Area.

2.7.5 Scale of Exposure Areas

Belted kingfishers generally feed within 1 mile (1.6 km) of their nesting sites but may have a foraging range up to 5 miles (8 km) (Csuti et al. 2001). Home ranges of kingfisher range from 0.5 to 4.9 miles (0.8 to 7.9 km) from the nest site (Cornwell 1963), and territory sizes of breeding pairs of kingfishers range from 0.6 to 1.4 miles (1.0 to 2.2 km) of shoreline (Brooks and Davis 1987; as cited in EPA 1993). The exposure scale of 1 mile (1.6 km) based on reported foraging ranges was assumed for estimating the exposure of belted kingfishers.

3.0 BIRD EGG ASSESSMENT EXPOSURE ASSUMPTION DETAILS

Literature-based biomagnification factors (BMFs) were used to estimate toxicity thresholds from egg tissue residues. Table 3-1 presents the BMFs used in the bird egg assessment. Details on the BMF literature search is presented below.

Table 3-1. Selected Literature-Based BMFs

COPC	BMF	Source
Metals		
Mercury	3.0	Buck (2004)
PCBs		
Total PCBs	8.4	Henny et al. (2008)
PCB TEQ	10[a]	Henny et al. (2003)
Dioxins/Furans		
Dioxin/furan TEQ	10[a]	Henny et al. (2003)
Total TEQ	10[a]	Henny et al. (2003)
Pesticides		
Sum DDE	79	Henny et al. (2008)

[a] Based on reported BMF for 2,3,7,8-TCDD.
BMF – biomagnification factor
COPC – chemical of potential concern
PCB – polychlorinated biphenyl
TCDD – tetrachlorodibenzo-*p*-dioxin
TEQ – toxic equivalent

Seven studies that measured the relationship between chemical concentrations in prey fish tissue and the tissue chemical concentrations in piscivorous bird eggs were reviewed. BMFs were compiled for each of the COPCs based on five piscivorous bird species (i.e., bald eagle, osprey, blue heron, herring gulls, and brown pelican). BMFs were available from four major regions: the Willamette River (Henny et al. 2003; Thomas and Anthony 1999), the Lower Columbia River (USGS 2004; Thomas and Anthony 1999), the Great Lakes region (Braune and Norstrom 1989; Giesy et al. 1995; Kubiak and Best 1991, as cited in Giesy et al. 1995); and South Carolina (Blus et al. 1977). Table 3-2 presents a summary of all the BMFs compiled from the reviewed literature sources.

Table 3-2. Summary of BMFs Derived from the Literature

Chemical	BMF	Bird Species	Fish Tissue	Location	Source
Dioxins/Furans					
2,3,7,8-TCDD	2, 23	Great blue heron	Fish tissue regurgitated or rejected cast from nests; prey collected opportunistically; fish species not weighted as proportion in diet	Lower Columbia River	Thomas and Anthony (1999)
2,3,7,8-TCDD	8	Great blue heron	Fish tissue regurgitated or rejected cast from nests; prey collected opportunistically; fish species not weighted as proportion in diet	Willamette River	Thomas and Anthony (1999)
2,3,7,8-TCDD	10[a]	Osprey	Fish tissue composites of three resident species: largescale sucker, mountain whitefish, and northern pikeminnow	Willamette River	Henny et al. (2003)
2,3,7,8-TCDD	16[b], 20[b]	Bald eagle	Common carp, peamouth chub, and largescale sucker (data collected by Tetra Tech (1993))	Lower Columbia River	Buck (2004)
TEQ	19	Osprey	Unknown	Michigan	Kubiak and Best (1991) as cited in Giesy et al. (1995)
2,3,7,8-TCDD	21[c]	Herring gull	Alewife fish tissue	Lake Ontario	Braune and Norstrom (1989)
PCBs					
Total PCBs	5, 4, 73	Great blue heron	Fish tissue regurgitated or rejected cast from nests; prey collected opportunistically; fish species not weighted as proportion in diet	Lower Columbia River	Thomas and Anthony (1999)
Total PCBs	13, 4	Great blue heron	Fish tissue regurgitated or rejected cast from nests; prey collected opportunistically; fish species not weighted as proportion in diet	Willamette River	Thomas and Anthony (1999)
Total PCBs	8.4[d]	Osprey	Fish tissue composites of three resident species: largescale sucker, mountain whitefish, and northern pikeminnow	Willamette River	Henny et al. (2008)
Total PCBs	11[a]	Osprey	Fish tissue composites of three resident species: largescale sucker, mountain whitefish, and northern pikeminnow	Willamette River	Henny et al. (2003)
Total PCBs	13	Great blue heron	Fish tissue regurgitated or rejected cast from nests; prey collected opportunistically; fish species not weighted as proportion in diet	Puget Sound reference site	Thomas and Anthony (1999)
PCBs	23	Brown pelican	Regurgitated menhaden	South Carolina, 1973	Blus et al. (1977)

Table 3-2. Summary of BMFs Derived from the Literature

Chemical	BMF	Bird Species	Fish Tissue	Location	Source
Total PCBs	28	Bald eagle	Various whole-body fish tissue prey by percentage of diet	Great Lakes	Giesy et al. (1995)
Total PCBs	32[c]	Herring gull eggs	Alewife fish tissue	Lake Ontario	Braune and Norstrom (1989)
Total PCBs	50[b], 113[b]	Bald eagle	Common carp, peamouth chub, and largescale sucker (data collected by Tetra Tech (1993))	Lower Columbia River	Buck (2004)
Sum DDE					
p,p′-DDE	9, 3, 143	Great blue heron	Fish tissue regurgitated or rejected cast from nests; prey collected opportunistically; fish species not weighted as proportion in diet	Lower Columbia River	Thomas and Anthony (1999)
p,p′-DDE	16, 24	Great blue heron	Fish tissue regurgitated or rejected cast from nests; prey collected opportunistically; fish species not weighted as proportion in diet	Willamette River	Thomas and Anthony (1999)
p,p′-DDE	22	Bald eagle	Mean concentration of whole-body fish tissue by percentage of diet; concentration collected in fish and bald eagle eggs collected from inland or coastal areas	Great Lakes	Giesy et al. (1995)
DDE	31	Brown pelican	Menhaden fish tissue	South Carolina 1973	Blus et al. (1977)
DDE	34[c]	Herring gull	Alewife fish tissue	Lake Ontario	Braune and Norstrom (1989)
p,p′-DDE	41	Great blue heron	Fish tissue regurgitated or rejected cast from nests; prey collected opportunistically; fish species not weighted as proportion in diet	Puget Sound reference site	Thomas and Anthony (1999)
DDE	75[b], 141[b]	Bald eagle	Common carp, peamouth chub, and largescale sucker (data collected by Tetra Tech (1993))	Lower Columbia River	Buck (2004)
DDE	79[d]	Osprey	Fish tissue composites of three resident species: largescale sucker, mountain whitefish, and northern pikeminnow	Willamette River	Henny et al. (2008)
DDE	87[a]	Osprey	Fish tissue composites of three resident species: largescale sucker, mountain whitefish, and northern pikeminnow	Willamette River	Henny et al. (2003)

Table 3-2. Summary of BMFs Derived from the Literature

Chemical	BMF	Bird Species	Fish Tissue	Location	Source
Mercury					
Mercury	1	Bald eagle	Mean concentration of whole-body fish tissue by percentage of diet	Great Lakes	Giesy et al. (1995)
Mercury	2.0[b], 3.0[b]	Bald eagle	Common carp, peamouth chub, and largescale sucker (data collected by Tetra Tech (1993))	Lower Columbia River	Buck (2004)

[a] Average lipid content (%) egg:fish = 4.3:5.0.
[b] BMFs based on geomean of BMFs from three river mile segments of the Lower Columbia River.
[c] Average lipid content (%) egg:fish = 7.7:2.8.
[d] Average lipid content (%) egg:fish = 4.2:5.6.
BMF – biomagnification factor
PCB – polychlorinated biphenyl
TCDD – tetrachlorodibenzo-p-dioxin
TEQ – toxic equivalent

In accordance with EPA's comments on the Ecological PRE (EPA 2006), BMFs were selected from the reviewed literature to be most representative of the Study Area region (from the Willamette River) and selected piscivorous bird receptor species (i.e., bald eagle and osprey). Therefore, the BMFs for all COPCs (except mercury) were based on osprey data from the Willamette River reported by Henny et al. (2003) or more recent data from Henny et al. (2008). For mercury, no Willamette-specific data were available, and the selected BMF was based on bald eagle data from the Lower Columbia River (Buck 2004). The selected BMF for mercury based on Columbia River bald eagles (BMF = 3.0) is a conservative estimate for predicting osprey egg concentrations, inasmuch as Henny et al. (2008) reported that mercury concentrations did not biomagnify in Willamette River osprey.

4.0 DERIVATION OF DIETARY TOXICITY REFERENCE VALUES

Effects data were based on wildlife dietary TRVs. Tables 4-1 and 4-2 present the bird and mammal dietary TRVs selected for the wildlife dietary risk assessment, respectively. The following subsections present the details on the selected TRVs for birds and mammals.

Table 4-1. Bird Dietary Dose TRVs

	TRV (mg/kg bw/day)		
COPC	**NOAEL**	**LOAEL**	**Source**
Aluminum	157	NA	Carriere et al. (1986)
Arsenic	2.24[a]	4.5[b]	Eco SSL (EPA 2005b)
Cadmium	1.47[a]	6.34[b]	Eco SSL (EPA 2005c)
Chromium	2.66[a]	15.6[b]	Eco SSL (EPA 2005d)
Copper	4.05[a]	12.1[c]	Eco SSL (EPA 2007a)
Lead	1.63[a]	3.26[c]	Eco SSL (EPA 2005e)
Mercury	0.0064[e]	0.064	Heinz (1975; 1979)
Selenium	0.29[a]	0.579[c]	Eco SSL (EPA 2007d)
Thallium	0.48[d]	24	Hudson et al. (1984)
Zinc	66.1[a]	171[b]	Eco SSL (EPA 2007e)
TBT	6.8	16.9	Schlatterer et al. (1993)
Benzo(a) pyrene	0.28[f]	1.4	Hough et al. (1993)
LPAHs	40	NA	Patton and Dieter (1980)
HPAHs	40	NA	Patton and Dieter (1980)
Total PAHs	40	NA	Patton and Dieter (1980)
BEHP	1.1	11[g]	Peakall (1974)
Di-n-butyl phthalate	0.11[e]	1.1	Peakall (1974)
Total PCBs	0.29	0.58	Britton and Huston (1973)
PCB TEQ	1.4×10^{-5}	1.4×10^{-4}	Nosek et al. (1992)
Dioxin/furan TEQ	1.4×10^{-5}	1.4×10^{-4}	Nosek et al. (1992)
Total TEQ	1.4×10^{-5}	1.4×10^{-4}	Nosek et al. (1992)
Total DDx	0.227[a]	2.27[c]	Eco SSL (EPA 2007b)
Sum DDE	0.032[e]	0.32	Mendenhall et al. (1983)
Aldrin	0.008[f]	0.04	DeWitt (1956)

[a] NOAEL was derived from the Eco SSL reports are based on the chemical-specific Eco SSL.
[b] LOAEL is based on a geometric mean derived using the same data used to calculate the Eco SSL.
[c] LOAEL is based on the same study as the NOAEL used as the basis for the Eco SSL.
[d] NOAEL was extrapolated from LOAEL using a UF of 50.
[e] NOAEL was extrapolated from LOAEL using a UF of 10.
[f] NOAEL was extrapolated from LOAEL using a UF of 5.

[g] LOAEL was extrapolated from NOAEL using a UF of 10 based on SREL (1999; as cited in EPA 2008a).
BEHP – bis(2-ethylhexyl) phthalate
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
Eco SSL – ecological soil screening level
EPA – US Environmental Protection Agency
NA – not available (no TRV selected)
TEQ – toxic equivalent
Total DDx – sum of all six DDT isomers (2,4′-DDD; 4,4′-DDD; 2,4′-DDE; 4,4′-DDE; 2,4′-DDT; and 4,4′-DDT)
TRV – toxicity reference value
UF – uncertainty factor

Table 4-2. Mammal Dietary Dose TRVs

COPC	TRV (mg/kg bw/day) NOAEL	TRV (mg/kg bw/day) LOAEL	Source
Aluminum	34.4	75.8	Ondreicka et al. (1966), Golub et al. (1987)
Antimony	0.059[a]	0.59[b]	Eco SSL (EPA 2005a)
Copper	5.6[a]	9.34[b]	Eco SSL (EPA 2007a)
Lead	4.7[a]	8.9[b]	Eco SSL (EPA 2005e)
Mercury	0.02	0.07	Dansereau et al. (1999)
Selenium	0.143[a]	0.215[b]	Eco SSL (EPA 2007d)
Benzo(a)pyrene	2.0[c]	10	MacKenzie and Angevine (1981)
HPAHs	0.615[a]	3.07[b]	Eco SSL (EPA 2007c)
Total PCBs	0.0074[c]	0.037	Restum et al. (1998)
PCB TEQ	2.2×10^{-7}[d]	2.2×10^{-6}	Tillitt et al. (1996)
Dioxin/furan TEQ	2.2×10^{-7}[d]	2.2×10^{-6}	Tillitt et al. (1996)
Total TEQ	2.2×10^{-7}[d]	2.2×10^{-6}	Tillitt et al. (1996)
Total DDx	0.147[a]	0.735[b]	Eco SSL (EPA 2007b)

[a] NOAEL is based on the chemical-specific Eco SSL.
[b] LOAEL is based on the same study as the NOAEL used as the basis for the Eco SSL.
[c] NOAEL was extrapolated from LOAEL using a UF of 5.
[d] NOAEL was extrapolated from LOAEL using a UF of 10.
bw – body weight
COPC – chemical of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
Eco SSL – ecological soil screening level
EPA – US Environmental Protection Agency
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
Total DDx – sum of all six DDT isomers (2,4′-DDD; 4,4′-DDD; 2,4′-DDE; 4,4′-DDE; 2,4′-DDT; and 4,4′-DDT)
TRV – toxicity reference value

4.1 BIRD TRVS

This section presents a summary of the basis for and the uncertainty associated with the selected TRVs used to estimate risks for bird receptors for all COPCs. A single NOAEL and LOAEL TRV were selected for each COPC. TRVs for PCBs were selected for both total PCBs and dioxin-like PCB congeners (PCB TEQ) and for 2,3,7,8,-tetrachlorodibenzo-*p*-dioxin (TCDD) (as TEQs for dioxins and furans [dioxin/furan TEQ]).

4.1.1 Aluminum

No Eco SSL has been developed for avian exposure to aluminum. Instead, bird dietary TRVs for aluminum were based directly on the toxicological literature. The selected NOAEL TRV of 157 mg/kg bw/day was based on Carriere et al. (1986). This was the higher of the two NOAELs reported in the reviewed literature (Attachment 14). No LOAEL was selected because none was reported in the two studies reviewed. There is high uncertainty associated with this selected NOAEL because it is an unbounded NOAEL based on exposure to the ionic form of aluminum. Toxicity measured for ionic aluminum is not comparable to the toxicity associated with the colloidal, particulate, and complex forms of aluminum present in the environment, including the Study Area. The application of a TRV based on the toxicity of ionic aluminum to measured total aluminum concentrations is highly uncertain and likely overestimates risks associated with aluminum exposure.

4.1.2 Arsenic

The Eco SSL for arsenic, 2.24 mg/kg bw/day, was selected as the NOAEL TRV. This was the lowest NOAEL reported for effects on reproduction, growth, or survival in data presented in the arsenic Eco SSL document (EPA 2005b). The selected LOAEL TRV of 4.5 mg/kg bw/day was based on a geometric mean of the available LOAELs for reproduction, growth, or survival reported in the arsenic Eco SSL document (EPA 2005b).

These TRVs are an order of magnitude lower than the LWG-recommended TRVs from a review of the toxicological literature presented in Attachment 14. The recommended literature-based LOAEL and NOAEL of 40 and 10 mg/kg bw/day, respectively, was based on reproductive effects in mallard ducks as reported by Stanley et al. (1994). The selected LOAEL was the lowest LOAEL based on exposure to arsenic only (a LOAEL of 6.8 mg/kg bw/day was derived from USFWS (1969), as cited in Sample et al. (1996)), in which 20% mortality in brown-headed cowbirds was reported following dietary exposure to arsenic for 7 months; however, birds were exposed to a chemical mixture of copper acetoarsenite.

4.1.3 Cadmium

The Eco SSL for cadmium, 1.47 mg/kg bw/day, was selected as the NOAEL TRV. This was the geometric mean NOAEL reported for effects on reproduction and growth in data presented in the cadmium Eco SSL document (EPA 2005c). The selected LOAEL TRV of 4.5 mg/kg bw/day was based on a geometric mean of the available LOAELs for reproduction and growth reported in the cadmium Eco SSL document (EPA 2005c).

These TRVs are similar to the LWG-recommended LOAEL and NOAEL TRVs based on the reviewed toxicological literature presented in Attachment 14. The recommended literature-based LOAEL (4.0 mg/kg bw/day) was based on observed growth effects in Japanese quail chicks as reported by Richardson et al. (1974) and was the lowest dose-response LOAEL based on a clearly measured adverse effect reported in the reviewed toxicological literature. The LWG-recommended literature-based NOAEL of 1.6 mg/kg bw/day was reported in (Cain et al. 1983), in which no effect on young mallard growth was observed.

4.1.4 Chromium

The Eco SSL for chromium, 2.66 mg/kg bw/day, was selected as the NOAEL TRV. This was the geometric mean NOAEL reported for effects on reproduction and growth in data presented in the chromium Eco SSL document (EPA 2005d). The selected LOAEL TRV of 15.6 mg/kg bw/day was based on a geometric mean of the available LOAELs for reproduction and growth reported in the chromium Eco SSL document (EPA 2005d).

The selected NOAEL TRVs is similar to the LWG-recommended NOAEL TRVs, and the LOAEL TRV is slightly higher than the LOAEL recommended by LWG based on the reviewed toxicological literature presented in Attachment 14. The recommended literature-based LOAEL and NOAEL of 5.0 and 1.0 mg/kg bw/day, respectively, were based on offspring survival of adult black ducks treated with trivalent chromium over a chronic period (10 months) (Haseltine et al. (unpublished) as cited in Sample (1996)). The LWG-recommended LOAEL TRV was the lowest dietary LOAEL calculated from the literature reviewed, and the LWG-recommended NOAEL TRV was the lowest acceptable dietary NOAEL calculated. There are a limited number of toxicological studies reporting adverse effects, and there was high variability in these effect levels (Attachment 14).

4.1.5 Copper

The Eco SSL for copper, 4.05 mg/kg bw/day, was selected as the NOAEL TRV. This was the lowest NOAEL below the lowest bound LOAEL for effects on reproduction, growth, or survival in data presented in the copper Eco SSL document (EPA 2007a). The selected LOAEL TRV of 12.1 mg/kg bw/day was based on a the same study, as reported in the copper Eco SSL document (EPA 2007a).

The selected LOAEL and NOAEL TRVs are lower than the LWG-recommended LOAEL and NOAEL TRVs (62 and 47 mg/kg bw/day, respectively), which were based on the reviewed toxicological literature presented in Attachment 14. The recommended literature-based TRVs were based on chick growth and mortality over 10 weeks of exposure to dietary copper (Mehring et al. 1960).

The LWG-recommended literature-based TRVs were the only chronic TRVs reported in the seven studies reviewed on the toxicity of dietary copper (lower LOAELs were reported for subchronic growth effects; however, chronic exposures were preferred as the most relevant exposure condition because it is unknown whether the adverse effects on growth during a subchronic duration [e.g., 4 weeks] would persist over time [see Attachment 14]).

4.1.6 Lead

The Eco SSL for lead, 1.63 mg/kg bw/day, was selected as the NOAEL TRV. This was the lowest NOAEL lower than the lowest bound LOAEL for effects on reproduction, growth, or survival in data presented in the lead Eco SSL document (EPA 2005e). The selected LOAEL TRV of 3.26 mg/kg bw/day was based on a the same study, as reported in the lead Eco SSL document (EPA 2005e).

The selected NOAEL TRV is similar to the LWG-recommended NOAEL TRV, and the selected LOAEL TRV is an order of magnitude higher than the LOAEL recommended by LWG based on the reviewed toxicological literature presented in Attachment 14. The LWG-recommended literature-based LOAEL and NOAEL of 20 and 2.0 mg/kg bw/day, respectively, were based on egg hatchability in Japanese quail (Edens et al. 1976). The LWG-recommended LOAEL was the lowest LOAEL based on an appropriate endpoint reported in the 14 individual toxicological studies reviewed (lower LOAELs were reported for egg production in domestic quail; however, egg production in domestic species was not considered a relevant endpoint for assessment (see Attachment 14).

4.1.7 Mercury

No Eco SSL has been developed for avian exposure to mercury. Instead, the bird dietary TRVs for mercury were based directly on the toxicological literature. The LOAEL TRV derived from Heinz (1975; 1979) was recommended both by EPA and LWG for evaluating risks. In this study, offspring production was reduced, and avoidance response and maternal response behavior in ducklings was impaired in mallard ducks fed 0.064 mg/kg bw/day mercury for multiple generations (Heinz 1975, 1979). This was the lowest LOAEL derived from the 14 studies measuring the toxicity of dietary mercury via injection or oral ingestion that were reviewed (Attachment 14). No dietary dose that was lower than this selected LOAEL was derived from the literature, so a dietary NOAEL TRV was extrapolated from the selected LOAEL using an UF of 10. The resulting NOAEL TRV was 0.0064 mg/kg bw/day.

The selected LOAEL represented the most conservative number available from the literature reviewed; however, it is uncertain because the literature did not always report adverse effects at this level; no effects on reproduction were reported Heinz (Heinz 1974, 1976) in birds fed the same dose level. Therefore, the selected LOAEL may or may not always result in an adverse effect. There is also uncertainty with the use of a UF to extrapolate a chronic NOAEL TRV from a chronic LOAEL TRV.

4.1.8 Selenium

The Eco SSL for selenium, 0.29 mg/kg bw/day, was selected as the NOAEL TRV. This was the highest NOAEL lower than the lowest bounded LOAEL value for reproduction, growth or survival endpoints, as presented in the selenium Eco SSL document (EPA 2007d). The selected LOAEL TRV of 0.579 mg/kg bw/day was based on a the same study, as reported in the selenium Eco SSL document (EPA 2007d).

These TRVs are similar to the LWG-recommended LOAEL and NOAEL TRVs based on the reviewed toxicological literature presented in Attachment 14. The recommended literature-based LOAEL and NOAEL of 0.82 and 0.42 mg/kg bw/day, respectively, were based on offspring survival and growth measured in mallards fed selenomethionine for about 100 days (Heinz et al. 1989). This was the lowest LOAEL derived from the five toxicological studies reviewed on the dietary toxicity of selenium (Attachment 14).

4.1.9 Thallium

No Eco SSL has been developed for avian exposure to thallium. Instead, the bird dietary TRVs for thallium were based directly on the toxicological literature. The selected LOAEL TRV of 24 mg/kg bw/day was based on Hudson et al. (1984). This was the lowest LOAEL reported in the reviewed literature (Attachment 14); however, there is high uncertainty associated with this selected LOAEL because high mortality was observed over a short exposure period. No NOAEL was reported below the selected LOAEL in the literature reviewed, so a NOAEL TRV was extrapolated from the selected LOAEL TRV using a UF of 50. The resulting NOAEL TRV was 0.48 mg/kg bw/day. There is high uncertainty associated with the literature dataset for thallium toxicity to birds due to the paucity of studies available.

4.1.10 Zinc

The Eco SSL for zinc, 66.1 mg/kg bw/day, was selected as the NOAEL TRV. This was the geometric mean NOAEL reported for effects on reproduction and growth in data presented in the zinc Eco SSL document (EPA 2007e). The selected LOAEL TRV of 171 mg/kg bw/day was based on a geometric mean of the available LOAELs for reproduction and growth reported in the zinc Eco SSL document (EPA 2007e).

These TRVs are similar to the LWG-recommended LOAEL and NOAEL TRVs based on the reviewed toxicological literature presented in Attachment 14. The recommended literature-based LOAEL and NOAEL of 124 and 82 mg/kg bw/day, respectively, were based on white rock chick growth were measured following dietary exposure to zinc for 5 weeks (Roberson and Schaible 1960).This was the lowest bounded LOAEL derived from the six toxicological studies reviewed on the dietary toxicity of selenium (Attachment 14).

4.1.11 Tributyltin

No Eco SSL has been developed for avian exposure to tributyltin (TBT). Instead, the bird dietary TRVs for TBT were based directly on the toxicological literature. The NOAEL and LOAEL TRVs, 6.8 and 16.9 mg/kg bw/day, respectively, based on Schlatterer et al. (1993) as derived by Sample et al. (1996), were recommended by EPA and selected as the dietary TBT TRV for birds. At the LOAEL, egg production, egg fertility, hatchability, offspring body weight, and 14-day offspring survival were all significantly affected in Japanese quail fed TBT for 6 weeks.

The selected TRVs are higher than the LWG-recommended TRVs presented in Attachment 14 that were based on the same study. The LWG recommended LOAEL and NOAEL TRVs

based on a review of the toxicological literature were 6.8 and 2.7 mg/kg bw/day, respectively. The reproductive effects following exposure to TBT were consistently observed in birds fed 16.9 mg/kg bw/day; however, in birds fed 6.8 mg/kg bw/day, the number of hatched eggs per pair was significantly reduced (Schlatterer et al. 1993), as was embryo mortality and hatchability (Coenen et al. 1992). No effect on these endpoints was observed in either study in birds fed 2.7 mg/kg bw/day.

There is some uncertainty associated with the selected TRVs because the literature dataset for TBT toxicity to birds is limited to two studies reporting the effects in Japanese quail only. The sensitivity of other bird species to TBT is unknown.

4.1.12 Total PCBs

No Eco SSL has been developed for avian exposure to PCBs. Instead, the bird dietary TRVs for PCBs were based directly on the toxicological literature. The selected LOAEL of 0.58 mg/kg bw/day was derived from Britton and Huston (1973), in which the reproductive success (i.e., hatchability) of chickens was significant impaired. This was the lowest LOAEL reported in which adverse reproductive effects were clearly measured of the 23 reviewed studies on the oral toxicity of PCB Aroclors to birds (Attachment 14). A LOAEL of 0.35 mg/kg bw/day was calculated from Lowe and Stendell (1991) but was not selected because the impact of eggshell thinning on kestrel reproductive success was not reported. The NOAEL of 0.29 mg/kg bw/day, calculated from the same study (Britton and Huston 1973) as the selected LOAEL, was selected as the NOAEL TRV.

Chickens appear to have a higher sensitivity to PCBs than do other avian species in the laboratory and may not be representative of Portland Harbor avian receptors. Thus, there is some uncertainty associated with the selected TRVs for PCBs in birds. However, the selected TRVs represent conservative values reported in the available literature.

4.1.13 2,3,7,8-TCDD

The dioxin/furan TEQ and PCB TEQ exposure doses for each bird receptor species was compared to the TRVs selected for 2,3,7,8-TCDD. No Eco SSL has been developed for avian exposure to 2,3,7,8-TCDD. Instead, the bird dietary TRVs for 2,3,7,8-TCDD were based directly on the toxicological literature.

The LOAEL and NOAEL of 0.14 and 0.014 µg/kg bw/day, respectively (Nosek et al. 1992), were selected as the dietary TRVs for 2,3,7,8-TCDD in birds because they represent the lower-effect-level concentration of the two studies that were reviewed in which birds were exposed to 2,3,7,8-TCDD (Attachment 14). In addition, the selected TRVs are based on a study measuring adverse effect over a chronic exposure during a critical life stage; the other study reviewed was not conducted during a critical life stage or over a chronic period. At the selected LOAEL, a reduction in survival, body weight, egg production, and embryo survival was observed in ring-necked pheasants given 10 weekly injections of 1.0 µg/kg 2,3,7,8-TCDD over a critical life stage (reproduction). The calculated daily dose from this study was 0.14 µg/kg/day.

There is high uncertainty associated with the selected TRVs because they are based on injection exposure. Intraperitoneal injection is not directly relevant to the dietary exposure pathway for birds, and the selected TRVs may not accurately predict dietary dioxin/furan toxicity.

4.1.14 Benzo(a)pyrene

No Eco SSL has been developed for avian exposure to individual polycyclic aromatic hydrocarbons (PAHs), including benzo(a)pyrene. The bird dietary TRVs for benzo(a)pyrene were based directly on the toxicological literature. The LOAEL of 1.4 mg/kg bw/day benzo(a)pyrene was selected as the LOAEL TRV. This was the lowest LOAEL based on the two toxicological studies reviewed (Attachment 14) and was selected as the LOAEL TRV for benzo(a)pyrene in birds. A NOAEL TRV for benzo(a)pyrene, 0.28 mg/kg bw/day, was extrapolated from the selected LOAEL TRV using a UF of 5.

There is high uncertainty associated with the selected PAH TRVs for birds. For benzo(a)pyrene, there is uncertainty associated with the selected LOAEL due to the route of exposure (intramuscular injection). Injection is not directly relevant to the dietary exposure pathway for birds, and the selected TRVs may not accurately predict dietary PAH toxicity to birds. In addition, there is uncertainty associated with the frequency of exposure (weekly) and the impact of the measured endpoints (fertility and ovarian appearance) on actual reproductive success at the selected LOAEL. It is unknown whether the effects on fertility would reduce the reproductive potential of pigeons at the population level. Finally, there is uncertainty with the literature dataset due to the limited number of studies, the high variability of the effect levels reported, and the use of a UF to extrapolated a chronic benzo(a)pyrene NOAEL TRV from a chronic LOAEL TRV.

4.1.15 HPAHs and LPAHs

No Eco SSL has been developed for avian exposure to a mixture of PAHs. The bird dietary TRVs for high-molecular-weight PAHs (HPAHs) and low-molecular-weight PAHs (LPAHs) were based directly on the toxicological literature. Only one study was available on the dietary toxicity of PAH mixtures to adult birds (Attachment 14). No LOAEL was reported in this study (Patton and Dieter 1980). The lowest NOAEL of 40 mg/kg bw/day was selected as the NOAEL TRV. At 40 mg/kg bw/day, growth was adversely affected in mallards fed an petroleum hydrocarbon mixture containing PAHs[5] combined with paraffin wax at 3 months; however, at 7 months, growth was recovered and change in body weight was not significantly different than the control group (Patton and Dieter 1980). The reduction observed in growth at 3 months was attributed to food avoidance of the aromatic hydrocarbon mixture feed because of the noxious odor of the petroleum hydrocarbons.

[5] Aromatic hydrocarbon mixture contained the following: ethylbenzene, 1,2,3,4-tetrahydronaphthalene, dimethylnaphthalene, 2,3,3-trimethylindolenine, acenaphthylene, acenaphthlene, phenanthrene, 2-methylbenzothiazole, dibenzothiophene, and 2,6-dimethylquinoline.

There is high uncertainty associated with the selected PAH TRVs for birds. The selected NOAEL TRV is based on exposure to a petroleum hydrocarbon mixture that contained some individual PAHs. Therefore, birds were exposed to a PAH mixture in addition to other chemicals. In addition, food avoidance was observed in the PAH mixture study, no LOAEL TRV was reported in the reviewed study, and there are limited toxicological data on bird exposure to PAHs.

4.1.16 Aldrin

No Eco SSL has been developed for avian exposure to aldrin. The bird dietary TRVs for aldrin were based directly on the toxicological literature. The selected LOAEL for aldrin was 0.04 mg/kg bw/day, based on 97.5% mortality observed after 127 days following dietary exposure of quails to aldrin (DeWitt 1956). This was the lowest LOAEL derived from the toxicological data based on the two studies that were reviewed (Attachment 14). A NOAEL TRV for aldrin was extrapolated from the selected LOAEL TRV using a UF of 5. The resulting NOAEL TRV, 0.008 mg/kg bw/day, was selected for aldrin. There is some uncertainty associated with the selected LOAEL TRV because the literature dataset for aldrin toxicity in birds is limited to two studies reporting the effects in domestic species (quail and pheasant) only, with high variability in the dose levels resulting in adverse effects. The sensitivity of wild bird species to aldrin is unknown. There is additional uncertainty with the use of a UF to extrapolate a chronic NOAEL TRV from a subchronic NOAEL TRV.

4.1.17 DDE

No Eco SSL has been developed for avian exposure to DDE. The bird dietary TRVs for DDE were based directly on the toxicological literature. The LOAEL of 0.32 mg/kg bw/day (calculated from Mendenhall et al. 1983) was selected as the LOAEL TRV. This was the lowest LOAEL derived from the 17 studies evaluated (Attachment 14). At 0.32 mg/kg bw/day, eggshell thinning, reduced eggshell strength, and nestling mortality were observed in barn owls fed DDE in their diet for 2 years (Mendenhall et al. 1983).The selected LOAEL is a conservative threshold effect level based on the most sensitive species tested in the reviewed literature. The NOAEL TRV for DDE was extrapolated from the selected LOAEL TRV using a UF of 10, per recommendation by EPA.

This extrapolated NOAEL is an order of magnitude lower than the LWG-recommended NOAEL derived directly from the literature. The LWG-recommended NOAEL of 0.12 mg/kg bw/day was calculated from Lincer (1975) and was the highest NOAEL derived from the toxicological literature below the selected LOAEL based on reproduction (Attachment 14). At this NOAEL, no effect on American kestrel reproduction was observed.

4.1.18 Total DDx

The Eco SSL for DDT and its metabolites, 0.227 mg/kg bw/day, was selected as the NOAEL TRV for total DDx (sum of all six DDT isomers [2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT]). This was the highest bounded NOAEL lower than the lowest bounded LOAEL for reproduction, growth, or survival, as reported in the DDT Eco SSL

document (EPA 2007b). The selected LOAEL TRV of 2.27 mg/kg bw/day was selected from the same study as the NOAEL, as presented in the DDT Eco SSL document (EPA 2007b).

These TRVs are similar to the LWG-recommended LOAEL and NOAEL TRVs based on the reviewed toxicological literature presented in Attachment 14. The recommended literature-based LOAEL and NOAEL of 1.8 and 0.18 mg/kg bw/day, respectively were based on measured eggshell thinning in mallards fed technical DDT for 11 months (Davison and Sell 1974). This was the lowest dose-response LOAEL based on a clearly measured adverse effect reported in the nine reviewed toxicological studies on the dietary toxicity of technical DDT or some other mixture of DDT and its metabolites (Attachment 14). There is uncertainty associated with the selected TRVs for total DDx because the impact of eggshell thinning on reproductive success at a population level is unknown. Eggshell thinning was statistically different from the control group with a difference of about 6% (Davison and Sell 1974). Reproductive effects on field populations of birds have been documented when eggshell thinning has reached 15 to 20% (Lincer 1975; Peakall et al. 1975; Anderson and Hickey 1972 as cited in White et al. 1984). The LWG-recommended LOAEL represents a very conservative threshold value and may overestimate risk to birds at the population level.

4.1.19 BEHP

No Eco SSL has been developed for avian exposure to bis(2-ethylhexyl) phthalate (BEHP). The bird dietary TRVs for BEHP were based directly on the toxicological literature. The selected NOAEL for BEHP was 1.11 mg/kg bw/day, based on no measured effect on eggshell thickness in ringed doves exposed to dietary BEHP for 4 weeks (Peakall 1974). A LOAEL TRV for BEHP (11 mg/kg bw/day) was extrapolated from the selected NOAEL TRV using a UF of 10, per recommendation by EPA.

There is high uncertainty associated with the selected LOAEL TRV because it was extrapolated from an unbounded NOAEL. There is high uncertainty with this LOAEL; the extrapolation of a LOAEL from a NOAEL is unprecedented. In addition, the extrapolated LOAEL (11 mg/kg bw/day) is an order of magnitude lower than the LWG-recommended LOAEL derived directly from the literature. The LWG-recommended LOAEL of 329 mg/kg bw/day was calculated from Ishida et al. (1982) was the only LOAEL reported in the three toxicological studies reviewed (Attachment 14). At this LOAEL, egg production ceased in domestic chickens following 230 days of exposure (also during a critical life stage) (Ishida et al. 1982). There is uncertainty associated with the literature-derived LOAEL as well because the literature dataset for BEHP toxicity to birds is limited to three studies using highly variable dose concentrations.

4.1.20 Dibutyl Phthalate

No Eco SSL has been developed for avian exposure to di(n)butyl phthalate. The bird dietary TRVs for di(n)butyl phthalate were based directly on the toxicological literature. The selected LOAEL for di(n)butyl phthalate was 1.1 mg/kg bw/day. At this LOAEL, the eggshell thickness index and egg permeability in ringed doves exposed to 10 mg/kg di(n)butyl phthalate in the diet for 4 weeks was significantly reduced compared to the control

group (Peakall 1974). A NOAEL TRV of 0.11 mg/kg bw/day was extrapolated from the selected LOAEL TRV using a UF of 10.

There is uncertainty associated with the recommended LOAEL TRV for di(n)butyl phthalate because the impact of eggshell thinning on reproductive success at a population level is unknown. Eggshell thinning was statistically different from the control group with a difference of about 10% (Peakall 1974). Reproductive effects on field populations of birds have been documented when eggshell thinning has reached 15 to 20% (Anderson and Hickey 1972, as cited in White et al. 1984; Lincer 1975; Peakall et al. 1975). This recommended LOAEL represents a conservative threshold value and may overestimate risk to birds at the population level. There is additional uncertainty because the literature dataset for di(n)butyl phthalate toxicity to birds is limited to only one study. The use of a UF to extrapolate a chronic NOAEL TRV from a chronic LOAEL TRV is also uncertain.

4.2 MAMMAL TRVS

This section presents a summary of the basis for and the uncertainty associated with the selected TRVs used to estimate risks for mammal receptors for all COPCs. A single NOAEL and LOAEL TRV were selected for each COPC. TRVs for PCBs were selected for both total PCBs and dioxin-like PCB congeners [PCB TEQ] and for 2,3,7,8,-TCDD (as TEQs for dioxins and furans [dioxin/furan TEQ]).

4.2.1 Aluminum

No Eco SSL has been developed for mammal exposure to aluminum. Instead, mammal dietary TRVs for aluminum were based directly on the toxicological literature (Attachment 14). The selected LOAEL TRV, 75.8 mg/kg bw/day, was based on the only dietary LOAEL reported in the literature reviewed. At this concentration, maternal and offspring body weight and length were significantly reduced following dietary exposure to aluminum of adult mice during gestation (Golub et al. 1987). The selected NOAEL, 34.4 mg/kg bw/day, was the lowest dietary NOAEL in which no effect on body weight, the number of litters, or number of offspring per litter was observed in mice fed aluminum for multiple generations (Ondreicka et al. 1966).

There is high uncertainty associated with the selected TRVs because they are based on exposure to the ionic form of aluminum. Toxicity measured for ionic aluminum is not comparable to the toxicity associated with the colloidal, particulate, and complex forms of aluminum present in the environment, including the Study Area. The application of a TRV based on the toxicity of ionic aluminum to measured total aluminum concentrations is highly uncertain and likely overestimates risks associated with aluminum exposure.

4.2.2 Antimony

The Eco SSL for antimony, 0.059 mg/kg bw/day, was selected as the NOAEL TRV. This was the highest bounded NOAEL reported below the lowest bounded LOAEL based on reproduction, growth, or survival endpoints as presented in the antimony Eco SSL document

(EPA 2005a). The selected LOAEL TRV of 0.59 mg/kg bw/day was selected from the same study as the NOAEL, as presented in the DDT Eco SSL document (EPA 2005a).

The NOAEL TRV is four orders of magnitude lower than the LWG-recommended TRVs based on a review of the toxicological literature presented in Attachment 14. The LWG-recommended literature-based NOAEL of 1,489 mg/kg bw/day was the only dietary exposure study available from the five toxicological studies reviewed. At this NOAEL, no effect on rat growth was observed following exposure to dietary antimony (1,489 mg/kg bw/day) for 90 days. No LOAEL was reported. There are no LOAELs based on a dietary exposure route in the toxicological studies reported in the antimony Eco SSL document for growth, reproductive, or mortality endpoints. The selected TRVs from the Eco SSL document are based on a drinking water study.

Thus, there is uncertainty associated with the selected TRVs because they are based on a drinking water exposure route. Drinking water ingestion involves a method of uptake and absorption that is different than the selected dietary pathway for wildlife receptors being assessed. The selected TRVs may overestimate oral exposure risk to mammals from antimony. Furthermore, the toxicological literature based on the LWG review (Attachment 14) and those reported in the Eco SSL document do not report any LOAELs associated with dietary exposure to antimony.

4.2.3 Copper

The Eco SSL for copper, 5.6 mg/kg bw/day, was selected as the NOAEL TRV. This was the highest NOAEL lower than the lowest LOAEL for reproduction or growth as presented in the copper Eco SSL document (EPA 2007a). The selected LOAEL TRV of 9.34 mg/kg bw/day was selected from the same study as the NOAEL, as presented in the copper Eco SSL document (EPA 2007a).

These TRVs are an order of magnitude lower than the LWG-recommended TRVs based on the reviewed toxicological literature presented in Attachment 14. The LWG-recommended literature-based LOAEL and NOAEL of 26 and 18 mg/kg bw/day, respectively, were based on kit mortality and litter mass in the offspring of adult mink fed dietary copper as copper sulfate following 1 year of exposure during a critical life stage (Aulerich et al. 1982). These TRVs represent concentrations protective of mink, which is a selected receptor species for Portland Harbor.

4.2.4 Lead

The Eco SSL for lead, 4.7 mg/kg bw/day, was selected as the NOAEL TRV. This was the highest bounded NOAEL reported below the lowest bounded LOAEL for reproduction, growth, or survival as presented in the lead Eco SSL document (EPA 2005e). The selected LOAEL TRV of 8.9 mg/kg bw/day was selected from the same study as the NOAEL, as presented in the lead Eco SSL document (EPA 2005e).

These TRVs are an order of magnitude lower than the LWG-recommended TRVs based on the reviewed toxicological literature presented in Attachment 14. The LWG recommended

literature-based LOAEL and NOAEL of 90 and 11 mg/kg bw/day, respectively, were based on body weight and kidney effects in offspring of parents fed dietary lead as lead acetate over 2 years (Azar et al. 1973). The LWG-recommended literature-based TRVs were based on the only dietary exposure study available from the 11 toxicological studies reviewed (Attachment 14).The selected TRVs from the Eco SSL document are based on a drinking water study.

Thus, there is uncertainty associated with the selected TRVs because they are based on a drinking water exposure route. Drinking water ingestion involves a method of uptake and absorption that is different than the selected dietary pathway for wildlife receptors being assessed. The selected TRVs may overestimate oral exposure risk to mammals from lead.

4.2.5 Mercury

No Eco SSL has been developed for mammal exposure to mercury. Instead, the mammal dietary TRVs for mercury were based directly on the toxicological literature. The LOAEL and NOAEL of 0.02 and 0.07 mg/kg bw/day, respectively, were selected as the dietary TRVs for mercury in mammals. The selected TRVs were calculated based on a study in which for several generations mink were fed a diet made up of 20% mink feed, 40% eviscerated chicken carcasses (contaminant free), and 40% field-collected fish from the Robert Bourassa Reservoir, Quebec, and effects on whelping success and adult mortality were measured (Dansereau et al. 1999). These selected TRVs represent the lowest TRVs reported for mink in the seven studies reviewed (Attachment 14). A lower LOAEL (0.0084 mg/kg bw/day) was reported in Verschuuren et al. (1976); however, this study was not used to derive the selected TRV because low-effect doses were reported for mink, one of the selected mammal receptors in Portland Harbor, exposed over a chronic duration for multiple generations through an ecologically relevant exposure (via partial fish consumption).

There is uncertainty associated with the selected TRVs because the field-collected fish (making up 40% of the prepared diet) may have contained other, uncharacterized organic chemicals that could have contributed to the reproductive toxicity reported in mink.

4.2.6 Selenium

The Eco SSL for selenium, 0.143 mg/kg bw/day, was selected as the NOAEL TRV. This was the highest NOAEL lower than the lowest LOAEL for reproduction or growth as presented in the selenium Eco SSL document (EPA 2007d). The selected LOAEL TRV of 0.215 mg/kg bw/day was selected from the same study as the NOAEL, as presented in the selenium Eco SSL document (EPA 2007d).

These TRVs are an order of magnitude higher than the LWG-recommended TRVs based on the reviewed toxicological literature presented in Attachment 14. The LWG-recommended literature-based LOAEL and NOAEL of 0.08 and 0.055 mg/kg bw/day, respectively, were the lowest TRVs calculated from the six studies reviewed (Attachment 14). At the LWG-recommended literature-based LOAEL, growth was significantly reduced in rats fed 0.08 mg/kg bw/day selenium as sodium selenite or seleniforous wheat for 6 weeks, and no

effect on growth was reported in rats fed 0.055 mg/kg bw/day over a subchronic (6-week) period (Halverson et al. 1966).

4.2.7 HPAHs

The Eco SSL for HPAHs, 0.615 mg/kg bw/day, was selected as the NOAEL TRV. This was the highest bounded NOAEL lower than the lowest LOAEL for reproduction, growth, or survival, as presented in the PAH Eco SSL document (EPA 2007c). The selected LOAEL TRV of 3.07 mg/kg bw/day was selected from the same study as the NOAEL, as presented in the PAH Eco SSL document (EPA 2007c). The selected TRVs for HPAHs is based on the toxicity of benzo(a)pyrene. The use of an individual PAH to evaluate a sum of individual HPAHs is uncertain.

4.2.8 Total PCBs

No Eco SSL has been developed for mammalian exposure to PCBs. Instead, the mammal dietary TRVs for PCBs were based directly on the toxicological literature. The LOAEL and NOAEL of 0.037 and 0.0074, respectively, were selected as the dietary TRVs for total PCBs in mammals. At the selected LOAEL, adverse reproductive effects (including reduced kit body weight, delay in the onset of estrus, and reduced whelping success) were observed in mink fed field-collected carp from the Great Lakes region over a chronic period (Restum et al. 1998). This LOAEL was selected because of the 14 toxicological studies reviewed, it represents the most conservative LOAEL reported in the literature based on a chronic mink study in which a relevant dietary fish exposure was used (Attachment 14). A NOAEL TRV was extrapolated from the selected LOAEL TRV using a UF of 5.

There is uncertainty associated with using a study relying on field-collected fish (Restum et al. 1998) to represent the LOAEL TRVs for total PCBs because field-collected fish likely contained other uncharacterized, organic chemicals that could have contributed to the reproductive toxicity reported in mink. This study only has quantitative relevance to mink exposed to chemical mixtures similar those found in the Great Lakes fish. In addition, there is uncertainty associated with these LOAELs because the field-collected fish contained other organic chemicals (e.g., dioxins/furans, DDE, DDD, chlordane) that likely could have contributed to the reproductive toxicity reported in mink. However, LOAELs based on exposure to laboratory diets in which reproductive effects could be attributed exclusively to PCB exposure result in an only slightly higher LOAEL. There is additional uncertainty with the use of a UF to extrapolate a chronic NOAEL TRV from a chronic LOAEL TRV.

4.2.9 2,3,7,8-TCDD

The dioxin/furan TEQ and PCB TEQ exposure doses for each mammal receptor species was compared to the TRVs selected for 2,3,7,8-TCDD or a TEQ. No Eco SSL has been developed for mammalian exposure to 2,3,7,8-TCDD or a TEQ. Instead, the mammal dietary TRVs for TEQs were based directly on the toxicological literature.

The LOAEL and NOAEL of 2.24 and 0.224 ng/kg bw/day, respectively, were selected as the dietary TRVs for TEQs in mammals. At the selected LOAEL, adverse reproductive effects

(including reduced kit body weight and kit survival) were observed in mink fed field-collected carp from Saginaw Bay, Lake Huron, for 182 days (Tillitt et al. 1996). This LOAEL was selected because of the 11 toxicological studies reviewed, it represents the most conservative value reported in the literature based on a chronic mink study in which a relevant dietary fish exposure was used (Attachment 14). A NOAEL TRV was extrapolated from the selected LOAEL TRV using a UF of 10.

There is uncertainty associated with the selected TRVs for 2,3,7,8-TCDD. The selected LOAEL TRV is based on a toxicological study in which adverse effects on reproduction were observed in mink fed field-collected carp from Saginaw Bay, Lake Huron. Mink were fed a prepared diet containing various percentages of field-collected carp (10, 20, and 40%). Effects on reproduction were observed in all mink fed any percentage of the field-collected carp; therefore, this study only has quantitative relevance to mink exposed to chemical mixtures similar those found in the Saginaw Bay fish. In addition, there is uncertainty associated with these LOAELs because the field-collected carp likely contained other uncharacterized, organic chemicals that could have contributed to the reproductive toxicity reported in mink. Field-collected carp also had reported concentrations of total PCBs. However, a LOAEL based on exposure to a laboratory diets in which reproductive effects could be attributed exclusively to PCB exposure result in essentially the same LOAEL. There is additional uncertainty with the use of a UF to extrapolate a chronic NOAEL TRV from a chronic LOAEL TRV.

4.2.10 Total DDx

The Eco SSL for DDT and its metabolites, 0.147 mg/kg bw/day, was selected as the NOAEL TRV. This was the highest bounded reported NOAEL lower than the lowest bounded LOAEL for reproduction, growth, or survival, as presented in the DDT Eco SSL document (EPA 2007b). The selected LOAEL TRV of 0.735 mg/kg bw/day was selected from the same study as the NOAEL, as presented in the DDT Eco SSL document (EPA 2007b).

These TRVs are an order of magnitude lower than the LWG-recommended TRVs based on the reviewed toxicological literature presented in Attachment 14. The LWG-recommended literature-based LOAEL and NOAEL were 1.3 and 1.2 mg/kg bw/day, respectively (Attachment 14). The selected LOAEL represents the most conservative effect-level concentration reported in the 15 studies reviewed on the toxicity of technical DDT or some other mixture of DDT and its metabolites. At the LWG-recommended literature-based LOAEL (1.3 mg/kg bw/day), litter size was reduced in mice fed a DDT mixture during a critical life stage (gestation) over 120 days (Ware and Good 1967). The LWG-recommended NOAEL (1.2 mg/kg bw/day) was the highest NOAEL below the selected LOAEL based on reproduction derived from the reviewed literature. At this NOAEL, no effect on rat reproduction was observed following exposure to dietary DDTs for multiple generations (Duby et al. 1971).

5.0 DERIVATION OF BIRD EGG TOXICITY REFERENCE VALUES

Effects data were based on literature-based bird egg TRVs. Table 5-1 presents the bird egg TRVs selected for the wildlife risk assessment. The following subsections present the details on the selected TRVs.

Table 5-1. Bird Egg Tissue-Residue TRVs

COPC	TRV (mg/kg bw/day) NOAEL	TRV (mg/kg bw/day) LOAEL	Source
Metals			
Mercury	0.5	0.74	Wiemeyer et al. (1984), Heinz and Hoffman (2003)
PCBs			
Total PCBs	3.0	4.5	Wiemeyer et al. (1993), Wiemeyer et al. (1984)
PCB TEQ	2.3×10^{-6}	3.198×10^{-5}	Henny et al. (2003), Anthony et al. (1993)
Dioxins/Furans			
Dioxin/furan TEQ	2.3×10^{-6}	3.198×10^{-5}	Henny et al. (2003), Anthony et al. (1993)
Total TEQ	2.3×10^{-6}	3.198×10^{-5}	Henny et al. (2003), Anthony et al. (1993)
Pesticides			
Sum DDE	1.3	3.5	Wiemeyer et al. (1984)

bw – body weight
COPC – chemical of potential concern
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TEQ – toxic equivalent

5.1 DIOXINS/FURANS AND DIOXIN-LIKE PCBS

The LOAEL and NOAEL of 31.98 and 2.3 mg/kg ww, respectively, were selected as the bird egg TRVs for TEQs. At the selected LOAEL, reduced productivity and eggshell thinning were observed in bald eagles nesting in the Columbia River estuary from 1980 to 1987 (Anthony et al. 1993). This LOAEL TRV for TEQ was selected because it was the lowest LOAEL reported for a representative piscivorous bird species from the Willamette River based on the 17 reviewed toxicological studies on egg tissue residues of dioxins/furans and/or dioxin-like chemicals associated with no effect and adverse effect levels (Attachment 14). At the selected NOAEL, no effect on osprey productivity in the Willamette River was observed in 1993 (Henny et al. 2003) This selected NOAEL was the highest NOAEL below the selected LOAEL based on a representative Willamette bird species in the studies reviewed (Attachment 14).

There is uncertainty associated with using LOAEL TRVs based on field-collected data. Bird egg tissues in the field contained other uncharacterized, organic chemicals (e.g., PCBs, DDE) that could have contributed to the reproductive toxicity reported in bald eagles. However, field-collected data allow for the selection of Willamette-specific data to derive toxicological data to protect Willamette receptors.

5.2 TOTAL PCBS

The LOAEL and NOAEL of 4.5 and 3.0 mg/kg ww, respectively, were selected as the bird egg TRVs for total PCBs. The selected LOAEL TRV for total PCBs was based on the lowest LOAEL TRV reported for a representative piscivorous bird species from the Willamette River in the 27 reviewed laboratory and field studies that reported PCB concentrations and associated effects (Attachment 14). Bald eagle egg tissue PCB concentrations as low as 4.5 µg/g ww were associated with impaired 5-year productivity in bird populations from 14 states in the United States (Wiemeyer et al. 1984). At the selected NOAEL, no effect on 5-year productivity was reported with bald eagle egg tissue PCB concentrations as low as 3.0 µg/g ww was observed (Wiemeyer et al. 1993).

There is uncertainty associated with using LOAEL TRVs based on field-collected data. Bird egg tissues in the field contained other uncharacterized, organic chemicals (e.g., PCBs, DDE) that could have contributed to the reproductive toxicity reported in bald eagles. However, field-collected data allow for the selection of Willamette-specific data to derive toxicological data to protect Willamette receptors.

5.3 DDE

The LOAEL and NOAEL of 3.5 and 1.3 mg/kg ww, respectively, were selected as the bird egg TRVs for DDE. Egg TRVs were selected only for DDE because birds appear to be most sensitive to DDE. The selected LOAEL TRV of 3.5 mg/kg ww DDE was based on Wiemeyer et al. (1984). At this egg chemical concentration (reported in field-collected bald eagle eggs), offspring productivity was below the level necessary to maintain a stable population (Wiemeyer et al. 1984). Thus, the selected LOAEL of 3.5 mg/kg ww DDE represents a conservative effect threshold based on the most sensitive bird species measured in the 31 reviewed laboratory and field studies that reported DDE or DDT concentrations in eggs and associated reproductive effects (Attachment 14). Wiemeyer et al. (1984) also reported a NOAEL of 1.3 µg/g ww, which was selected as the NOAEL TRV.

There is uncertainty associated with using LOAEL TRVs based on field-collected data. Bird egg tissues in the field contained other uncharacterized, organic chemicals (e.g., dioxins/furans, PCBs) that could have contributed to the reproductive toxicity reported in bald eagles. However, field-collected data allow for the selection of Willamette-specific data to derive toxicological data to protect Willamette receptors.

5.4 MERCURY

The LOAEL and NOAEL of 0.5 and 0.74 mg/kg ww, respectively, were selected as the bird egg TRVs for mercury. The selected LOAEL TRV was the lowest acceptable

LOAEL derived from the eight toxicological studies reviewed (Attachment 14). At the selected LOAEL TRV, the embryo development of mallards was affected following exposure to dietary methylmercury chloride during breeding (Heinz and Hoffman 2003). The selected NOAEL TRV is 0.5 µg/g ww, at which no adverse effects on bald eagle productivity are expected, as reported by Wiemeyer et al. (1984). The selected NOAEL represents the highest NOAELs reported below the selected LOAEL reported in the toxicological studies reviewed (Attachment 14).

There is uncertainty associated with using LOAEL TRVs based on field-collected data. Bird egg tissues in the field contained other uncharacterized, organic chemicals (e.g., dioxins/furans, PCBs) that could have contributed to the reproductive toxicity reported in bald eagles. However, field-collected data allow for the selection of Willamette-specific data to derive toxicological data to protect Willamette receptors.

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