United States Environmental Protection Agency
Region 9 & Office of Water
EPA-xxx-x-xx-xxx
November 2018

DRAFT

Aquatic Life and Aquatic-Dependent Wildlife Selenium Water Quality Criterion for Freshwaters of California

(xx November 2018)

U.S. Environmental Protection Agency Region 9
Water Division
San Francisco, CA

U.S. Environmental Protection Agency
Office of Water
Office of Science and Technology
Washington, D.C.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

ACKNOWLEDGEMENTS

Technical Analysis Leads

Amanda Jarvis, U.S. EPA, Office of Water, Office of Science and Technology, Health and Ecological Criteria Division, Washington, DC
jarvis.amanda@epa.gov (primary contact person)

Julianne McLaughlin, Ph.D., U.S. EPA, Office of Water, Office of Science and Technology, Standards and Health Protection Division, Washington, DC
mclaughlin.julianne@epa.gov

Daniel R. Oros, Ph.D., U.S. EPA Region 9, Water Quality Assessment Office, Water Division; Ecosystem Branch, San Francisco, California
oros.daniel@epa.gov

U.S. EPA Office of Water Reviewers

Joseph Beaman
Colleen Flaherty
Kathryn Gallagher
Karen Kesler

Intra-Agency Panel Peer Reviewers

Diane Fleck, U.S. EPA Region 9, San Francisco, CA

Karen Kessler, U.S. EPA, Office of Water, Office of Science and Technology, Standards and Health Protection Division, Washington, DC

Jim Lazorchak, U.S. EPA, Office of Research and Development, National Exposure Research Laboratory, Cincinnati, OH

Technical support was provided by Great Lakes Environmental Center

ACRONYMS

AE	Assimilation Efficiency
ALC	Aquatic Life Criteria
AWQC	Ambient Water Quality Criteria
BAF	Bioaccumulation Factor
CF	Conversion Factor
CFR	Code of Federal Regulations
CTR	California Toxics Rule
CWA	Clean Water Act
dw	Dry Weight
ECx	Effect Concentration at X Percent Effect Level
EF	Enrichment Factor
ESA	Endangered Species Act
EO	Egg-Ovary
FCV	Final Chronic Value
FR	Federal Register
fww	Fresh Wet Weight
GMCV	Genus Mean Chronic Value
GSD	Genus Sensitivity Distribution
IR	Ingestion Rate
k_e	Rate of selenium loss
LCL	Lower Confidence Limit
LOEC	Lowest Observed Effect Concentration
M	Muscle
MATC	Maximum Acceptable Toxicant Concentration (expressed mathematically as the geometric mean of the NOEC and LOEC)
NTR	National Toxics Rule
NPDES	National Pollutant Discharge Elimination System
NOEC	No Observed Effect Concentration
OLS	Ordinary Least Squares
SMCV	Species Mean Chronic Value
T&E	Threatened and Endangered
TMDL	Total Maximum Daily Load
TRAP	EPA's Statistical Program: Toxicity Relationship Analysis Program
TSD	Technical Support Document
TTF	Trophic Transfer Factor
UCL	Upper Confidence Limit
WB	Whole body
WQBELS	Water Quality-based Effluent Limitations
WQC	Water Quality Criteria
WQS	Water Quality Standards
ww	Wet Weight

EXECUTIVE SUMMARY

This document sets forth U.S. Environmental Protection Agency's (EPA) basis for and derivation of the selenium water quality criterion for the inland surface waters, enclosed bays, and estuaries of California to protect aquatic life and aquatic-dependent wildlife, including federally listed threatened and endangered species. This assessment relies on EPA's Section 304(a) *Aquatic Life Ambient Water Quality Criterion for Selenium – Freshwater 2016* for the aquatic life portion of the criteria (U.S. EPA 2016a). In addition, this assessment provides a critical review of all data identified in EPA's literature search quantifying the toxicity of selenium to aquatic-dependent wildlife, and provides a basis for a criterion that will assure protection of aquatic-dependent wildlife species found in California from the chronic toxic effects of selenium.

EPA previously derived the aquatic life chronic tissue-based criterion elements for egg-ovary, whole body and/or muscle concentrations in fish. The aquatic-dependent wildlife chronic tissue-based criterion element for bird eggs is derived herein. These tissue-based criterion element concentrations were developed to protect aquatic life and aquatic-dependent wildlife taxa from reproductive impairments associated with dietary exposure and maternal transfer to eggs resulting in mortality, teratogenicity, and decreased hatchability. The tissue-based criterion elements can then be used to derive site-specific chronic water-column based criterion for both lentic and lotic waters using the Performance Based Approach (PBA) discussed in *Translation of Selenium Tissue Criterion Elements to Site-Specific Water Column Criterion Elements for California* that are protective of both aquatic life and aquatic-dependent wildlife.

To provide an example that is comparable to the previously derived aquatic life selenim criterion, EPA utilized the translation equation from the independently peer-reviewed and validated Ecosystem Scale Selenium Model (Presser and Luoma 2010), which is the mechanistic model approach laid out in the PBA discussed in *Translation of Selenium Tissue Criterion Elements to Site-Specific Water Column Criterion Elements for California*, to estimate protective selenium concentrations in the water column from corresponding bird egg criterion element concentrations. This same approach was previously used for estimating protective selenium concentrations in the water column from fish egg-ovary criterion concentrations (U.S. EPA 2016a) and are summarized in Part 5. The translation equation uses species-specific food web

models, species-specific bioaccumulation parameters (conversion factor (*CF*) and trophic transfer factors (*TTF*), and a site-specific selenium enrichment factor (*EF*), which describes the enrichment of selenium concentrations from water to particulate matter (plankton, detritus, and sediment), to calculate a site-specific water column concentration element from the fish egg-ovary and bird egg criterion elements. All modeling incorporated site-specific ecosystem variables (e.g. fish species, *EFs*, and water body type) on a national scale to calculate selenium water column-based criterion elements for lentic and lotic freshwater systems and an intermittent water column-based criterion element that may be appropriate for California. In this analysis, EPA found that the selenium water column-based criterion elements previously derived by U.S. EPA (2016a) would also be protective of aquatic-dependent wildlife. The available data and modeling results indicate that aquatic life and aquatic-dependent wildlife would be protected from the toxic effects of selenium in California by applying the following multi-element criterion:

Table 1-1. Summary of the Proposed California Selenium Ambient Chronic Water Quality Criteria for Protection of Aquatic Life and Aquatic-Dependent Wildlife.

Media Type	Bird Tissue	Fish Tissue[1]		Water Column[4]	
Criterion Element	**Bird Egg[2]**	**Egg-Ovary[2]**	**Fish Whole Body or Muscle[3]**	**Monthly Average Exposure**	**Intermittent Exposure[5]**
Magnitude	11.2 mg/kg dw	15.1 mg/kg dw	8.5 mg/kg dw whole body or 11.3 mg/kg dw muscle (skinless, boneless filet)	Derived on a site-specific basis using the methodology described in *Translation of Selenium Tissue Criterion Elements to Site-Specific Water Column Criterion Elements for California*	$WQC_{int} = \frac{WQC_{30-day} - C_{bkgrnd}(1 - f_{int})}{f_{int}}$
Duration	Instantaneous measurement[6]	Instantaneous measurement[6]	Instantaneous measurement[6]	30 days	Number of days/month with an elevated concentration
Frequency	Not to be exceeded	Not to be exceeded	Not to be exceeded	Not more than once in three years on average	Not more than once in three years on average

1. Fish tissue elements are expressed as steady-state.
2. Fish Egg-Ovary supersedes any whole-body, muscle, or translated water column element for that taxon when fish egg-ovary are measured. Bird Egg supersedes translated water column elements for that taxon when both are measured.
3. Fish whole-body or muscle tissue supersedes the translated water column element when both fish tissue and water concentrations are measured.
4. Translated water column values ($WQC_{30\text{-day}}$) will be based on dissolved total selenium in water and will be derived using the methodology described in *Translation of Selenium Tissue Criterion Elements to Site-Specific Water Column Criterion Elements for California*. Water column values are the applicable criterion element in the absence of bird tissue data and steady-state condition fish tissue data.
5. Where $WQC_{30\text{-day}}$ is the water column monthly element derived using the methodology described in *Translation of Selenium Tissue Criterion Elements to Site-Specific Water Column Criterion Elements for California*, C_{bkgrnd} is the average background selenium concentration, and f_{int} is the fraction of any 30-day period during which elevated selenium concentrations occur, with f_{int} assigned a value ≥ 0.033 (corresponding to 1 day).
6. Fish tissue and bird tissue data provide instantaneous point measurements that reflect integrative accumulation of selenium over time and space in bird or fish population(s) at a given site.

EPA is proposing the freshwater selenium criterion depicted in Table 1-1 in California (except San Francisco Bay and Delta).[1] The EPA is proposing the 2016 CWA section 304(a) selenium criterion for freshwater with the addition of a bird tissue criterion element and the replacement of the 304(a) selenium monthly average exposure water column criterion element with a performance-based approach for translating the tissue elements into a corresponding water-column criterion element on a site-specific basis. This approach maximizes the flexibility for dischargers and the State to derive site-specific water-column criterion elements based on site-specific data for each waterbody. Available data indicate that applying the criterion in Table 1-1 would be protective of aquatic life and aquatic-dependent wildlife from the toxic effects of selenium, recognizing that the fish tissue elements and the bird element supersede any translated site-specific water column elements (except in special situations, see footnote 4 in Table 1-1) and that the fish egg-ovary element supersedes all other fish tissue elements.. Two of the tissue criterion elements are based on the concentration of selenium in fish tissue andone element is based on the concentration of selenium in bird eggs. The proposed elements are: (1) a fish egg-ovary element; (2) a fish whole body and/or muscle element; and (3) a bird egg element. The fish egg-ovary and bird egg criterion concentrations are derived from analysis of the available selenium toxicity data for aquatic life and aquatic-dependent wildlife species, respectively. The fish whole body and fish muscle tissue criterion element concentrations are derived from a combination of directly measured toxicity values and the fish egg-ovary toxicity values that have been converted using concentration ratios among tissues. The proposed performance-based approach (PBA) consists of a methology to translate the tissue criterion elements into a site-specific water column criterion element. The EPA is also proposing an intermittent exposure element. The EPA is proposing that the bird tissue element be independently applicable from and equivalent to the fish tissue elements, but that all tissue elements will supersede the translated water column criterion elements for the specific taxon when both are measured. The selenium criterion, expressed as a single criterion composed of multiple elements, is expected to protect the most sensitive aquatic life and aquatic-dependent wildlife from potential chronic effects of selenium.

1 https://www.gpo.gov/fdsys/pkg/FR-2016-07-15/pdf/2016-16266.pdf

Part 1 INTRODUCTION AND BACKGROUND

The purpose of this document is to provide the U.S. Environmental Protection Agency's (EPA's) scientific rationale for this proposed selenium water quality criterion for California waters. This criterion is designed to protect aquatic life and aquatic-dependent wildlife, including federally listed threatened and endangered species, and is based solely on the best available data and best professional scientific judgements on the toxicological effects of selenium in egg-laying fish and birds. This criterion was developed following the general approach outlined in the EPA's "*Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses*" (Stephan et al. 1985). Pursuant to Clean Water Act section 303(c) and EPA's implementing regulations at 40 CFR § 131.11(a), water quality criteria must be based on sound scientific rationale and must contain sufficient parameters or constituents to protect designated uses; for water with multiple designations, the criteria must support the most sensitive designated uses. The selenium criterion for California is intended to be protective of the State's established aquatic life and wildlife designated uses in fresh waters which include: migration of aquatic organisms; spawning, reproduction and early development of fish; estuarine habitat; warm and cold freshwater habitats; wildlife habitat; and rare, threatened or endangered species. The criterion presented herein is EPA's best estimate of the maximum concentrations of selenium, with associated frequency and duration specifications, that will support protection of sensitive aquatic life and aquatic-dependent wildlife from unacceptable chronic effects in California.

The information provided herein does not substitute for the Clean Water Act or EPA regulation, nor is this document a regulation itself. Thus, this document cannot, and does not, impose any legally binding requirements on EPA, the State of California, authorized Tribes, the regulated community, or any other party.

1.1 Early Selenium Efforts

National Ambient Water Quality Criteria (AWQC) recommendations are established by the EPA under section 304(a) of the Clean Water Act (CWA). As provided for by the Clean Water Act, EPA reviews and from time to time revises 304(a) AWQC to ensure the criteria are consistent with the latest scientific information. Section 304(a) aquatic life criteria (ALC) serve as recommendations to states and tribes in defining ambient water concentrations that will

protect against adverse ecological effects to aquatic life and aquatic-dependent wildlife resulting from exposure to a pollutant found in water from direct contact or ingestion of contaminated water and/or food. Aquatic life criteria address the CWA goals of providing for the protection and propagation of fish and shellfish. When adopted into state or tribal water quality standards (WQS), these criteria can become a basis for establishing National Pollutant Discharge Elimination System (NPDES) program permit limits, the basis for listing impaired waters under Section 303(d) and establishing Total Maximum Daily Loads (TMDLs).

In 1980, EPA first published numeric aquatic life criteria for selenium in freshwater (acute criterion 260 µg/L and chronic criterion 35 µg/L). These criteria were based on water-only exposure (no dietary exposure). In order to address the lack of consideration of bioaccumulation in the 1980 selenium criteria, EPA published updated selenium criteria in 1987 to address field-based toxicity observed in aquatic ecosystems at concentrations below the existing criteria values. The 1987 criteria were field-based and accounted for both the water column and dietary uptake pathways manifested at Belew's Lake, NC, a cooling water reservoir that had been affected by selenium loads from a coal-fired power plant. At that time, EPA also provided an acute criterion of 20 µg/L derived from a reverse application of an acute-chronic ratio obtained from conventional water-only exposure toxicity tests applied to the 5 µg/L chronic value based on dietary and water column exposure in Belew's Lake.

In 1998, EPA held a peer consultation workshop to evaluate new science available for selenium relevant to the selenium aquatic life criterion. EPA concluded, and the peer reviewers agreed, that fish tissue values better represent chronic adverse effects of selenium than the conventional water concentration approach used by EPA to protect aquatic life, because chronic selenium toxicity is primarily based on the food-chain bioaccumulation route, not a direct waterborne route. During the following years (1998–2016) and through multiple criterion iterations, EPA worked with technical experts to develop a final selenium criterion for fish tissue that would be protective of all aquatic life (See Section 1.1 of U.S. EPA (2016a) for more details).

EPA used the scientific principles established in the 2009 Pellston scientific workshop on the ecological risk assessment of selenium (Chapman et al. 2009, 2010) and additional data generated since 2009 to develop the 2014 draft criterion that was reviewed by an expert external peer review panel. In EPA's 2016 final recommended freshwater chronic criterion for selenium,

revisions reflected consideration of the public and external expert peer reviews of the 2014 draft, public comments on the 2015 draft, data and information from additional studies provided by the public and peer reviewers, and additional scientific analyses. EPA's final 2016 criterion reflected the latest scientific consensus (e.g., Chapman et al. 2010) on the reproductive effects of selenium on aquatic life and their measurement in aquatic systems and supersedes all previous national aquatic life water quality criteria for selenium.

In 2016, EPA recommended a national selenium criterion expressed as four elements. All elements are protective against chronic selenium effects in aquatic life, and account for both short term and longer term exposure to selenium. Two elements are based on the concentration of selenium in fish tissue (eggs and ovaries, and whole body or muscle) and two elements are based on the concentration of selenium in the water column (two 30-day chronic values (lentic and lotic) and two intermittent values (lentic and lotic). EPA derived the 30-day chronic water column element from the egg-ovary element by modeling selenium bioaccumulation in food webs of lotic and lentic aquatic systems. EPA recommended the intermittent values to address short-term exposures that could contribute to chronic effects through selenium bioaccumulation in either lotic or lentic systems. EPA derived the intermittent element based on the chronic 30-day water column element and the fraction of any 30-day period during which elevated selenium concentrations occur. These water column criterion elements apply to the total of all oxidation states (selenite, selenate, organic selenium, and any other forms; See Appendix L in U.S. EPA 2016a for Analytical Methods for Measuring Selenium). Aquatic communities are expected to be protected by EPA's chronic criterion from any potential acute effects of selenium. Chapman et al. (2009) noted that selenium acute toxicity has been reported rarely in the aquatic environment.

EPA has not established national selenium criteria recommendations for the protection of aquatic-dependent wildlife. However, EPA has been involved in two separate efforts dealing with wildlife criteria for selenium. On December 12, 2011, EPA approved a selenium wildlife criterion for Gilbert Bay of the Great Salt Lake.[2] The approved criterion was 12.5 mg/kg dry weight (dw) in bird egg tissue that is a geometric mean over the nesting season to be applied to Gilbert Bay of the Great Salt Lake. On June 30, 2016, EPA proposed to revise the current federal Clean Water Act selenium water quality criteria applicable to the San Francisco Bay and Delta to

2 https://deq.utah.gov/legacy/programs/water-quality/selenium/index.htm

ensure that the criteria are protective of aquatic life and aquatic-dependent wildlife. Within the analysis that supports the proposed rule, EPA reviewed avian toxicity studies and ensured that the most "at risk" birds in this system would be protected by the proposed criteria (U.S. EPA 2016b).[3]

1.2 California Toxics Rule

On May 18, 2000, EPA promulgated *Water Quality Standards; Establishment of Numeric Criteria for Priority Toxic Pollutants for the State of California* at 65 FR 31681 (hereafter referred to as the California Toxics Rule or CTR).[4] The CTR established numeric water quality criteria for priority toxic pollutants for inland surface waters and enclosed bays and estuaries within California. EPA promulgated the CTR after California rescinded its water quality control plans containing pollutant objectives (criteria). The criteria that EPA previously promulgated for California in the National Toxics Rule (NTR), together with the criteria promulgated in the CTR and California's designated uses and anti-degradation provisions, set water quality standards for priority toxic pollutants for inland surface waters and enclosed bays and estuaries in California.

As required by section 7 of the Endangered Species Act (ESA) (16 U.S.C. 1531 et seq.), EPA consulted with the U.S. Fish and Wildlife Service (U.S. FWS) and the U.S. National Marine Fisheries Service (NMFS); (collectively, the Services) concerning EPA's rulemaking actions for California. EPA initiated consultation in 1994, and in March 2000, the Services issued a final Joint Biological Opinion. The final Joint Biological Opinion required that EPA revise its 1987 recommended criteria values for selenium to ensure the protection of species listed as threatened or endangered, and later update the criteria for California consistent with the revised recommendations. In response, EPA reserved the acute freshwater selenium criterion from the final May 2000 CTR, meaning that there was no acute criterion promulgated in the CTR. In addition to reserving the acute criteria, EPA agreed to several follow up actions, and the Services included several Terms and Conditions, which implement Reasonable and Prudent Measures, in the Biological Opinion. For selenium, the Terms and Conditions included reevaluating and revising selenium criteria for the protection of aquatic life in California waters. The CTR (40 CFR § 131.38) as amended, includes selenium water quality criteria, for the

3 https://www.gpo.gov/fdsys/pkg/FR-2016-07-15/pdf/2016-16266.pdf

4 https://www.gpo.gov/fdsys/pkg/FR-2000-05-18/pdf/00-11106.pdf

protection of aquatic life for fresh and marine waters in California.[5] Currently, the CTR freshwater criterion continuous concentration (chronic, 4-day average total recoverable selenium) is 5 µg/L and criterion maximum concentration (acute) is reserved (i.e., there is not an acute criterion in place). The saltwater criterion continuous concentration is 71 µg/L (chronic, 4-day average, total dissolved selenium) and criterion maximum concentration (acute) is 290 µg/L.

In 2013, two organizations filed a legal complaint against EPA, based in part on the fact that work on updating the reserved acute freshwater selenium criterion from the 2000 CTR had not yet been completed. EPA had previously determined, in the proposed CTR, that the criterion was among those necessary to implement section 303(c)(2)(B) of the CWA (62 FR 42160, August 5, 1997).[6] EPA ultimately consented to a court-ordered resolution of these claims. Under the terms of the court order, EPA committed to developing updated selenium aquatic life and aquatic-dependent wildlife criteria for the California waters covered by the original CTR by November 30, 2016. Pursuant to the terms of the court-ordered resolution, EPA proposed "*Water Quality Standards; Establishment of Revised Numeric Criteria for Selenium for the San Francisco Bay and Delta, State of California*" on June 30, 2016, which automatically extended EPA's deadline to propose selenium criteria for the rest of California to November 30, 2018.

Since research documented in U.S. EPA (2016a) demonstrates that the most significant exposure pathway of selenium to species of concern is through diet, the currently applicable freshwater criteria for selenium from the CTR, based solely on direct water column toxicity, is considered to be not adequately protecting species in California because direct water column toxicity is known not to be a major route of toxicity to oviparous (egg-laying) aquatic and aquatic-dependent vertebrate species (Chapman 2010; U.S. EPA 2016a). The current technical support document (TSD) provides a scientifically-defensible revised selenium water quality criterion based on dietary exposures to selenium for California under section 304(a)(1) of the CWA. CWA Section 304(a)(1) requires EPA to develop criteria for water quality that accurately reflect the latest scientific knowledge. These criteria are based on the best available data and best professional scientific judgments on the toxicological effects of selenium. The criteria herein rely

5 https://www.gpo.gov/fdsys/pkg/CFR-2018-title40-vol24/pdf/CFR-2018-title40-vol24-sec131-38.pdf

6 https://www.gpo.gov/fdsys/pkg/FR-1997-08-05/pdf/97-20173.pdf

heavily on the documented science supporting EPA's 2016 final recommended freshwater chronic criterion for selenium (U.S. EPA 2016a) as well as the overarching guidance outlined in the EPA's "*Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses"* (Stephan et al. 1985).

Part 2 PROBLEM FORMULATION

2.1 Overview of Selenium Sources and Occurrence in California

Selenium is a naturally occurring element present in sedimentary rocks and soils. It is present in the aquatic environment as selenite and selenate, as organic forms such as selenium methionine through transformation by algae (Simmons and Wallschlager 2011 and LeBlanc and Wallschlager 2016), or as methyl derivatives of selenium through methylation by bacteria (Ranjard et al. 2003). There are around 40 known selenium-containing minerals, some of which can have as much as 30% selenium, but all are rare and generally occur together with sulfides of metals such as copper, zinc, and lead (Emsley 2011). Sedimentary rocks, particularly shales, have the highest naturally occurring selenium content (Burau 1985). The distribution of organic-enriched, sedimentary shales, petroleum source rocks, ore deposits, phosphorites, and coals, in which selenium typically co-occurs, is well characterized in the United States (Presser et al. 2004). Natural weathering of geologic strata containing selenium can lead to selenium leaching into groundwater and surface water. Two major categories of anthropogenic activities are known to cause increased selenium mobilization and introduction into aquatic systems. The first is the mining of metals, minerals, and refinement and combustion of fossil fuels; the second is irrigation of selenium-rich soils. Atmospheric emissions of selenium can originate from several sources including power plants and other facilities that burn coal or oil, selenium refineries that provide selenium to industrial users, base metal smelters and refineries, resource extraction industries, milling operations, and end-product manufacturers (e.g., semiconductor manufacturers, ATSDR (2003)). Airborne selenium particles can settle either on surface waters or on soils from which selenium can be further transported and deposited into water bodies through ground or surface water conveyances or runoff.

Mining activities bring selenium-enriched deposits to the surface, where they are exposed to physical weathering processes. The release of selenium related to resource extraction activities is most common in the phosphate deposits of southeast Idaho and adjacent areas of Wyoming,

Montana, and Utah, and in coal mining areas in portions of West Virginia, Kentucky, Virginia, and Tennessee (Presser et al. 2004). Where selenium-containing minerals, rocks, and coal are mined, selenium can be mobilized when rock overburden and waste materials are crushed, increasing the surface area and exposure of the material to weathering processes. Selenium contamination of surface waters can also occur when sulfide deposits of iron, uranium, copper, lead, mercury, silver, and zinc are released during the mining and smelting of these metal ores. Additionally, when coal is burned for power production, selenium can enter surface waters as drainage from fly-ash ponds and fly-ash deposits on land (Gillespie and Baumann 1986). Fly ash deposits have a high surface area to volume ratio, resulting in rates of selenium in leachate several times higher than from the parent feed coal (Fernández-Turiel et al. 1994). The refining and burning of crude oil containing high levels of selenium can also be a major source of loading in certain water bodies via direct discharge and atmospheric deposition, respectively (Maher et al. 2010).

High selenium concentrations are found in phosphoritic sedimentary rock such as marine shales and sulfide ore bodies (Mayland et al. 1989). Cretaceous marine sedimentary deposits have weathered to produce high selenium soils in many areas of the western United States (Lemly 1993b). In California, areas with Tertiary and Cretaceous marine sedimentary deposits are known to have elevated selenium (Seiler et al. 1999). Watersheds in these areas may have elevated selenium levels in water, especially if human disturbances to the geological sedimentary deposits in these areas are high (Figure 2-1). For instance, human disturbances have included expanding the width and depth of open drainage channels for flood control purposes in agricultural and urbanized areas, and conducting construction activities in the upland hills that contain marine shales, such that these activities have disrupted and exposed the underlying selenium-bearing marine sedimentary deposits subjecting them to erosion, weathering, and transport to downslope areas in the watershed.

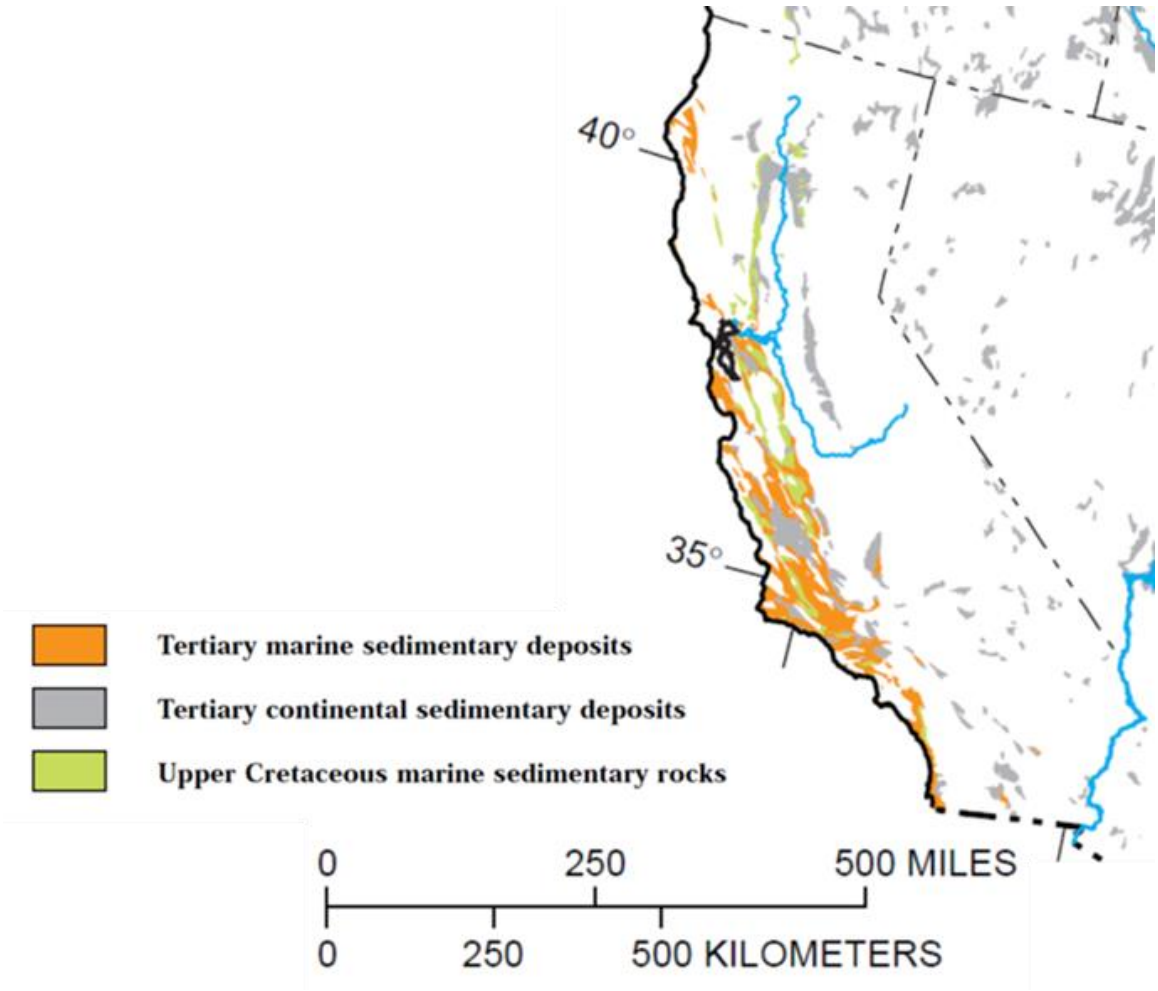


Figure 2-1. Areas of California with seleniferous marine geology.
Modified from: Seiler et al. (1999).

Irrigation of selenium-rich soils for crop production in arid and semi-arid regions of the country can mobilize selenium and move it off-site in drainage water that has leached through soil. Where deposits of Cretaceous marine shales occur, they can weather to produce high selenium soils; such soils are present in many areas of the western United States (Lemly 1993b). Selenium is abundant in the alkaline soils of the Great Plains, and some ground waters in California, Colorado, Kansas, Oklahoma, South Dakota, and Wyoming contain elevated concentrations of selenium due to weathering of and leaching from rocks and soils. In semi-arid areas of the West, irrigation water applied to soils containing soluble selenium can leach selenium. The excess water (from tile drains to irrigation return flow) containing selenium can be discharged into basins, ponds, or streams. For example, elevated selenium levels at the Kesterson

Reservoir in California originated from agricultural irrigation return flow collected in tile drains that discharged into the reservoir (Ohlendorf et al. 1986). Areas of California susceptible to selenium contamination from agricultural irrigation are shown in Figure 2-2.

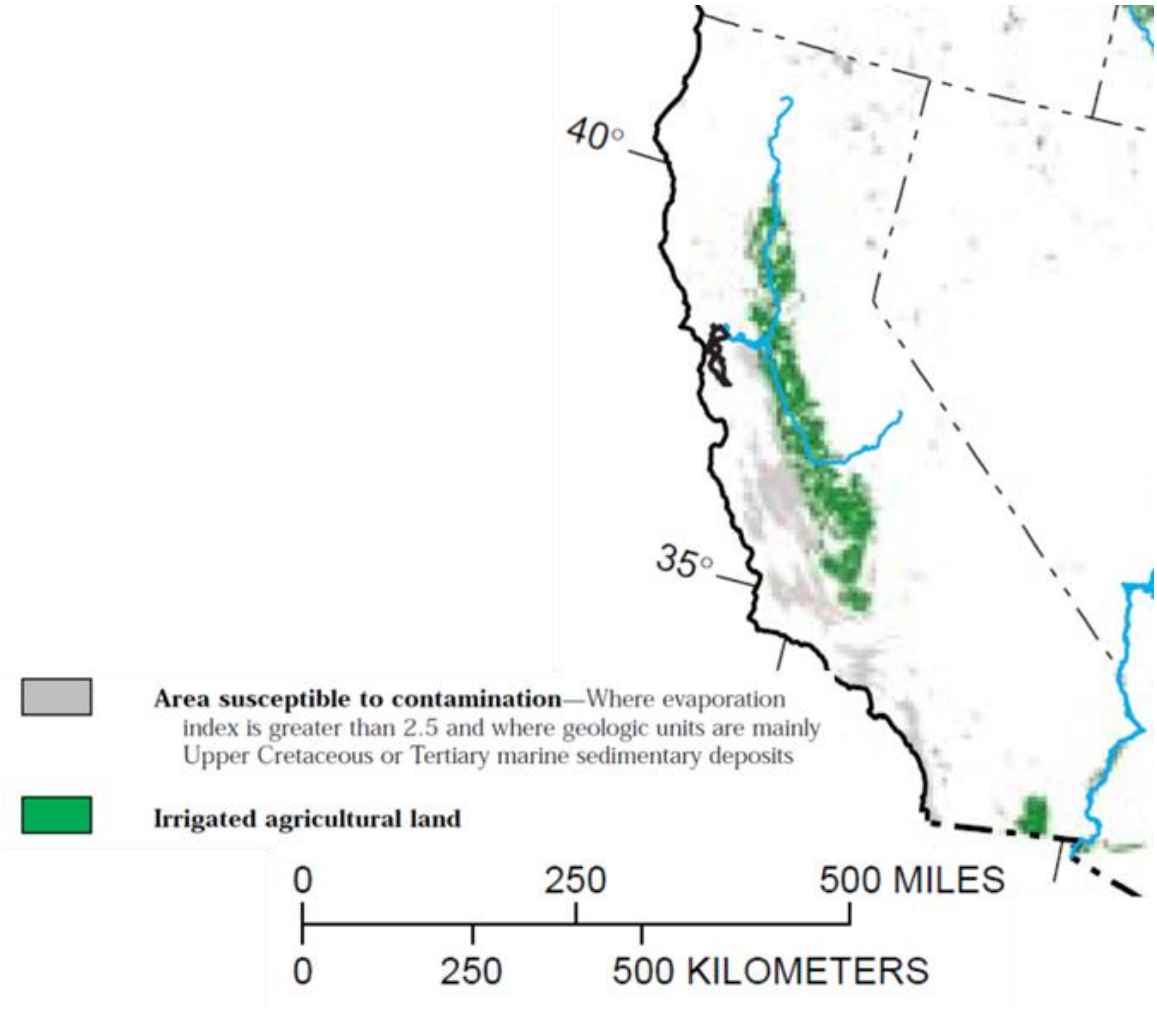


Figure 2-2. Areas of California susceptible to selenium contamination (gray) and where agricultural land is irrigated (green).
Overlap of gray and green show areas susceptible to selenium discharge from irrigation. Modified from: Seiler et al. (1999).

Figure 2-3 shows the distributions and abundances of total selenium concentrations in California water bodies collected over a 10-year period from October 5, 2004 to June 3, 2014 (CEDEN 2015). The total selenium concentration data included 270 water bodies (94% lotic and 6% lentic) and 11,290 water samples collected throughout California. The samples were collected and analyzed by multiple organizations that conduct water quality monitoring in

California. The data results are uploaded into the California Environmental Data Exchange Network (CEDEN) database by those monitoring organizations. The concentration distributions that are binned in the map shown in Figure 2-3 show the data results in relation to the California Toxics Rule (CTR) selenium chronic water quality criterion of 5 µg/L, which applied as the regulatory water quality criterion over the 10-year sampling period. The map shows that most of the field sampling occurred in the central San Joaquin Valley, Central Coast, Los Angeles, and San Diego areas. These same sampling areas also had the largest share of exceedances of the 5 µg/L selenium chronic water quality criterion. As previously noted, these are areas of California that have seleniferous marine and continental sedimentary deposits.

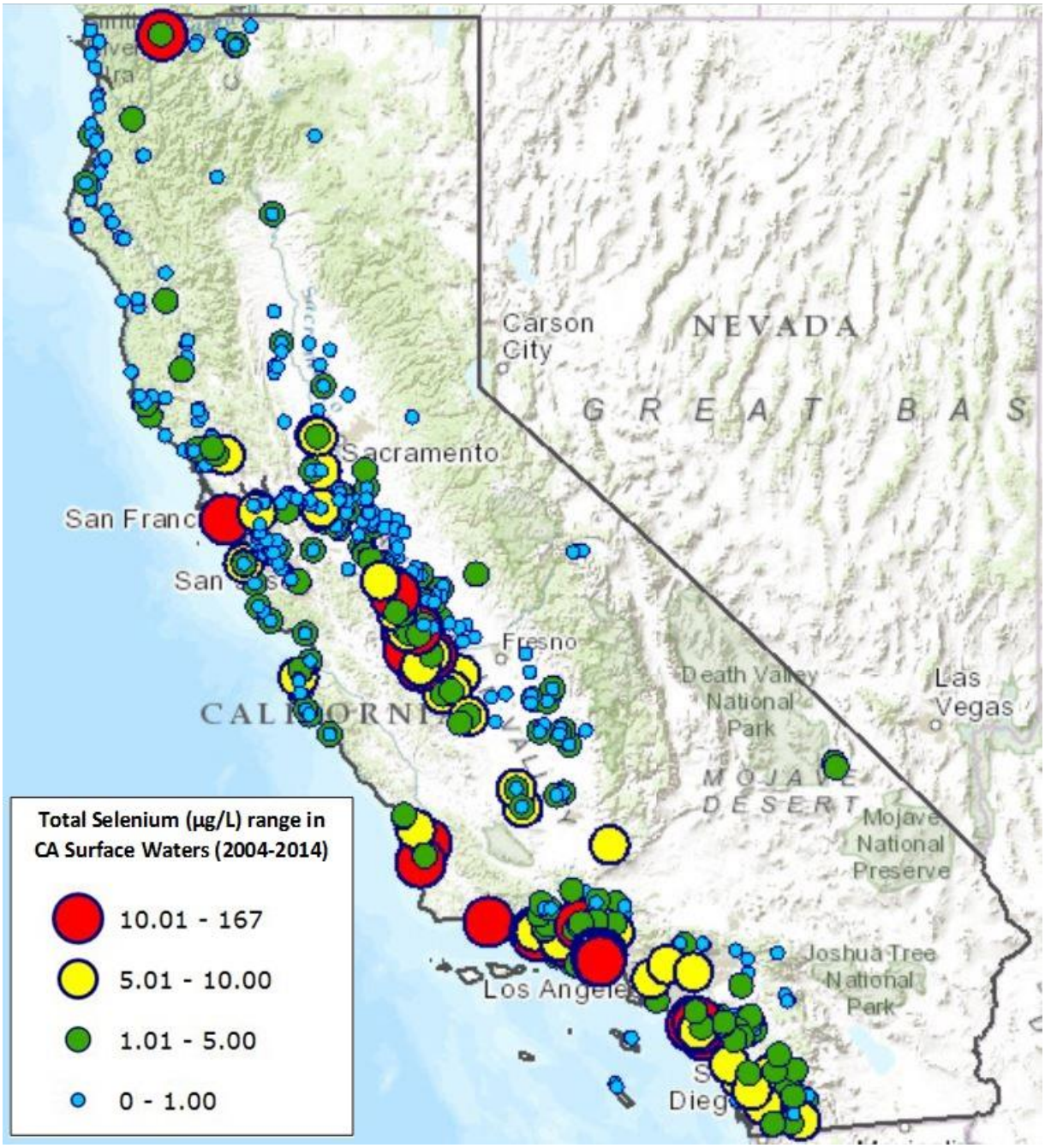


Figure 2-3. Distributions and abundances of total selenium concentrations (µg/L) in surface water samples collected from October 5, 2004 through June 3, 2014.

The California Toxics Rule (CTR) water quality criterion for selenium is currently set at 5 µg/L. The data were accessed from the California Environmental Data Exchange Network website (**CEDEN: www.ceden.org/**) on February 4, 2015.

2.2 Selenium Speciation in Aquatic Systems

The fate and transport of selenium in aquatic systems is affected by the distribution of selenium species and their transformations in water, sediment, and biota. These transformations include the assimilation and conversion of inorganic selenium to organic selenium species in plants and microbes that are transferred to higher trophic level consumer species throughout the aquatic food web.

Aquatic organisms are exposed to a combination of predominantly organic selenium species present in the food web, which result from transformation of inorganic species entering aquatic environments. Organisms accumulate selenium via trophic transfer throughout their life history reaching steady-state when elimination equals uptake. Effects to reproductive stages reflect the integrated exposures to transformed inorganic and organic species of selenium. The bioavailability and toxicity of selenium depend on both its concentration and speciation (Cutter and Cutter 2004; Meseck and Cutter 2006; Reidel and Sanders 1996). Selenium exists in four oxidation states (VI, IV, 0, -II) and in a wide range of chemical species across these oxidation states (Doblin et al. 2006; Maher et al. 2010; Meseck and Cutter 2006). In the effects assessment that follows, we have correlated the adverse effects on aquatic life with total dissolved selenium.

In oxygenated surface waters, the primary dissolved selenium species are selenate (SeO_4^{2-}, or Se[VI]) and selenite (SeO_3^{2-}, or Se[IV]), as well as dissolved organic selenides (-II) formed from fine particulate organic matter (e.g., Doblin et al. 2006; Meseck and Cutter 2006). The relative abundance of selenate and selenite may depend on relative contributions from the geologic and anthropogenic sources of selenium to the receiving waters if there is negligible inter-conversion between the two species (e.g., Maher et al. 2010), or may be influenced by interconversions (Simmons and Wallschlager 2011). Aqueous selenite is more abundant than selenate when the majority of selenium originates from discharges from coal fly ash tailings or oil refineries (e.g., Cutter 1989; Huggins et al. 2007). Particulate species in the water column include selenate, selenite, and elemental selenium [Se(0)] bound to resuspended sediments and organic particles, as well as particulate organic selenium species incorporated into suspended detritus (e.g., Cutter and Bruland 1984; Meseck and Cutter 2006).

In sediments, selenate and selenite can be reduced to iron selenides or elemental selenium under abiotic or biotic processes; elemental selenium and selenides can be converted to selenate under oxidizing conditions (Maher et al. 2010). For example, selenate can be reduced to

elemental selenium in sediments (e.g., Oremland et al. 1990) in the presence of iron oxides (Chen et al. 2008) and iron sulfides (Breynaert et al. 2008). Elemental selenium and organic selenides are produced by selenate-reducing microbes in sediments. Overall, the reduction of selenate and particularly selenite in sediments increases with increasing sediment organic matter (Tokunaga et al. 1997). Selenite in particular is readily bound to iron and manganese oxy-hydroxides (Maher et al. 2010), and is readily adsorbed to inorganic and organic particles, particularly at a lower pH range (e.g., McLean and Bledsoe 1992; Tokunaga et al. 1997). Microbial reduction of selenite to organic forms (via methylation) increases the solubility and bioavailability of selenium (Simmons and Wallschläger 2005). Plants and algae produce volatile selenium species by biomethylation of excess selenium, which upon reaching the sediment can be transformed to a more bioavailable species or deposited in the sediments and effectively removed from the system (Diaz et al. 2009). Depending on environmental conditions, the reduction processes described above are largely reversible, as elemental selenium and selenides in sediments can be oxidized to selenate through microbial or abiotic transformations (e.g., Maher et al. 2010; Tokunaga et al. 1997).

The most important aspect of selenium chemistry, with respect to its toxicity to aquatic organisms, is in the uptake and transformation of dissolved inorganic selenium in the tissues of primary producers at the base of the food web. The main route of entry of selenium into aquatic food webs is from the consumption of selenium incorporated in the tissue of primary producers (particulate), and to a lesser degree, from the consumption of sediments (Doblin et al. 2006; Luoma and Presser 2009). For algae, dissolved species of selenite and organic selenides are more bioavailable than selenate (Baines et al. 2001; Luoma et al. 1992). In vascular plants, selenate uptake is greater than for the other dissolved species, as most selenium uptake occurs in the roots, and selenate is more easily transported to the shoots and leaves than selenite or organic selenides (Dumont et al. 2006). Following uptake, selenium is metabolized into a variety of organic species that are assimilated into plant tissues. Selenium metabolism in plants is analogous to sulfur metabolism (e.g., Dumont et al. 2006; Ouerdane et al. 2013). Selenate is reduced to selenite, which is then reduced to selenide in a process involving reduced glutathione (Dumont et al. 2006). Selenide is converted to selenocysteine, which is then converted to selenomethionine (Dumont et al. 2006). In addition to selenocysteine and selenomethionine, a variety of other organic selenium species can be formed; however, selenocysteine, and

particularly selenomethionine are toxicologically important because these amino acids nonspecifically replace cysteine and methionine in proteins and are more bioavailable to higher trophic level consumers (Fan et al. 2002; Freeman et al. 2006).

The chemical form of selenium that dominates a location is usually dependent on its sources, effluent treatments, and biogeochemical processes in the receiving waters. Irrigation activities in areas with seleniferous soils typically mobilize selenate (SeO_4^{2-}, or Se[VI]) (Seiler et al. 2003). Combustion of coal for power generation creates predominantly selenite (SeO_3^{2-}, or Se[IV]) in the fly ash waste due to the temperatures, pH, and redox conditions involved with the process (Huggins et al. 2007). Similar conditions during refinement of crude oil can also result in high concentrations of selenite relative to selenate, as was observed in the San Francisco Bay estuary in the 1980s (Cutter 1989). Although selenite is the dominant species in the discharges resulting from crude oil refining and coal burning using conventional technologies, the implementation of alternative treatment technologies can alter the relative concentrations of selenate and selenite. For example, in scrubbers with forced oxidation systems that produce strong oxidizing conditions and high temperatures, most of the discharged selenium is in the form of selenate (Maher et al. 2010). For flue gas desulfurization systems that are the inhibited oxidation type, the selenium chemistry is more complex, and selenite may not be the primary form emitted (Petrov et al. 2012). Table 2-1 shows the predominant chemical forms of selenium that are associated with different activities and industries.

Table 2-1. Predominant Chemical Forms of Selenium in Discharges Associated with Different Activities and Industries.

Selenium Form	Sources
Selenate	Agricultural irrigation drainage Treated oil refinery effluent Mountaintop coal mining/ valley fill leachate Copper mining discharge
Selenite	Oil refinery effluent Fly ash disposal effluent Phosphate mining overburden leachate
Organoselenium	Treated agricultural drainage (in ponds or lagoons)

Source: Cutter and Diego-McGlone 1990; Presser and Ohlendorf 1987; Zhang and Moore 1996.

2.3 Bioaccumulation of Selenium in Aquatic Systems

Dissolved selenium uptake by animals is slow, whatever the chemical form, such that under environmentally relevant conditions, dissolved selenium in the water column makes little or no direct contribution to bioaccumulation in animals (Lemly 1985; Ogle and Knight 1996), but does influence the concentration of selenium in particulate matter. Selenium bioaccumulation in aquatic organisms occurs primarily through the ingestion of food (Fan et al. 2002; Luoma et al. 1992; Maher et al. 2010; Ohlendorf et al. 1986; Presser and Ohlendorf 1987; Presser et al. 1994; Saiki and Lowe 1987). However, unlike other bioaccumulative contaminants such as mercury, the single largest step in selenium accumulation in aquatic environments occurs at the base of the food web where algae and other microorganisms accumulate selenium from water by factors ranging from several hundred to tens of thousands (Luoma and Presser 2009; Orr et al. 2012; Stewart et al. 2010). Bioaccumulation and trophic transfer through aquatic food webs are the major biogeochemical pathways of selenium in aquatic ecosystems. Dissolved selenium oxyanions (selenate, selenite) and organic selenides are assimilated into the tissues of aquatic primary producers (trophic level 1 organisms), such as periphyton, phytoplankton, and vascular macrophytes; and subsequently biotransformed into organoselenium. These organisms, together with other particle-bound selenium sources, constitute the particulate selenium fraction in the water column. Selenium from this particulate fraction is then transferred to aquatic primary consumers such as zooplankton, insect larvae, larval fish, and bivalves (trophic level 2), and then to predators such as fish and birds (trophic level 3 and higher). In addition to the presence of selenium in the water, the process of selenium bioaccumulation in aquatic life residing in freshwater systems depends on several factors specific to each aquatic system. These factors include:

Water residence time. Residence time is a measure of the average time a water molecule will spend in a specified region of space. Residence time influences both the proportion of selenium found in particulate and dissolved forms and the predominant chemical form of selenium. Organisms in waters with long residence times such as lakes, ponds, reservoirs, wetlands, or estuaries will tend to bioaccumulate more selenium than those living in waters with shorter residence times such as rivers and streams (ATSDR 2003; EPRI 2006; Luoma and Rainbow 2005; Simmons and Wallschläger 2005). Several interrelated factors affect selenium's greater bioaccumulation potential in slow moving systems including food web complexity and

the organic content and reduction/oxidation potential of underlying sediments. Therefore, selenium toxicity in flowing waters with shorter residence times may only be apparent downstream of their selenium sources, whereas waters with longer residence times are more likely to exhibit selenium toxicity near their sources (Presser and Luoma 2006).

Distribution of selenium between particulate and dissolved forms. Selenium is found in both particulate and dissolved forms in water, but direct transfer of selenium from water to animals is only a small proportion of the total exposure. The proportion of selenium found in particulate matter (algae, detritus, and sediment) is important because it is the primary avenue for selenium entering into the aquatic food web (Luoma and Rainbow 2005; Luoma et al. 1992; Ohlendorf et al. 1986; Presser and Luoma 2006; Presser and Ohlendorf 1987; Presser et al. 1994; Saiki and Lowe 1987).

Bioaccumulation in prey. Trophic level 1 organisms such as periphyton and phytoplankton, as well as other forms of particulate material containing selenium, such as detritus and sediment, are ingested by trophic level 2 organisms such as mollusks, planktonic crustaceans, and many insects, increasing the concentration of selenium in the tissues of these higher-level organisms. Differences in the physiological characteristics of these organisms result in different levels of bioaccumulation. Also, based on the limited toxicity data available, selenium effects on invertebrates typically appear to occur at concentrations higher than those that elicit effects on vertebrates (e.g., fish and birds) that prey upon them (Janz et al. 2010). Additionally, certain molluscan taxa such as mussels and clams can accumulate selenium to a much greater extent than planktonic crustaceans and insects (although the levels do not seem to be toxic to the mussels) due to higher ingestion rates of both particulate-bound (algae) and dissolved selenium from the water column through filter feeding, as well as the lower rate at which they eliminate selenium (Luoma and Rainbow 2005; Stewart et al. 2013).

Trophic transfer to predators. Bioaccumulation of selenium by higher trophic level organisms, such as trophic level 3 and 4 fish and birds, is highly influenced by the specific food web of the aquatic environment that they inhabit. Prey selection influences the amount of selenium bioaccumulated by predatory fish and birds (Ackerman and Eagles-Smith 2009; Luoma and Presser 2009; Ohlendorf et al. 1986; Stewart et al. 2010). For example, fish and birds that primarily consume freshwater mollusks (e.g., redear sunfish and lesser scaup) will exhibit greater selenium bioaccumulation than fish and birds that consume primarily insects or crustaceans from

waters with the same concentration of dissolved selenium because mollusks tend to accumulate selenium at higher concentrations than other trophic level 2 organisms, as noted above (Luoma and Presser 2009; Stewart et al. 2004).

Because egg-laying (oviparous) vertebrates such as fish and birds are the most sensitive vertebrates to selenium effects, (Janz et al. 2010), these vertebrate consumers are also the most vulnerable groups to the potentially harmful effects of selenium, such as reproductive impairments, and selenium poisoning and therefore are the focal point of most selenium environmental assessments and criteria derivations (Ogle and Knight 1996; Stewart et al. 2010).

2.4 Effects and Biota

2.4.1 Mode of Action and Toxicity of Selenium

Selenium is a naturally occurring chemical element that is also an essential micronutrient. Trace amounts of selenium are required for normal cellular function in animals. However, selenium at amounts not much above nutritional levels can have toxic effects, making it one of the most toxic of the biologically essential elements (Chapman et al. 2010). Egg-laying vertebrates have a lower tolerance than do mammals, and the transition from levels of selenium that are biologically essential to those that are toxic for these species occurs across a narrow range of exposure concentrations (Chapman et al. 2009, 2010; Haygarth 1994; Luckey and Venugopal 1977; U.S. EPA 1987, 1998).

As a member of the Group 16 nonmetallic elements, selenium displays similar characteristics to sulfur. Selenium can replace sulfur in two amino acids, the seleno-forms being selenomethionine and selenocysteine. It has been a long-standing hypothesis that the cause of malformations in egg-laying vertebrates is due to the substitution of selenium for sulfur in these amino acids and their subsequent incorporation into proteins, which causes disruption of the structure and function of the protein. When present in excessive amounts, selenium is erroneously substituted for sulfur, resulting in the formation of a triselenium linkage (Se-Se-Se) or a selenotrisulfide linkage (S-Se-S), either of which was thought to prevent the formation of the normal disulfide chemical bonds (S-S). The result was thought to be distorted, dysfunctional enzymes, and protein molecules that impaired normal cellular biochemistry (Diplock and Hoekstra 1976; Reddy and Massaro 1983; Sunde 1984).

More recent research, however, suggests that selenium's role in oxidative stress plays a part in embryo toxicity, whereas selenium substitution for sulfur does not. Contrary to what was

previously hypothesized, the substitution of selenomethionine for methionine does not appear to affect either the structure or function of proteins (Egerer-Sieber et al. 2006; Mechaly et al. 2000; Yuan et al. 1998). The reason is apparently due to selenium not being distally located in selenomethionine; a terminal methyl group on this amino acid insulates the protein from selenium's effect on its tertiary structure and its function. Selenocysteine is present in several enzymes (e.g., glutathione peroxidases) and unlike selenomethionine its incorporation into proteins is highly regulated (Stadtman 1996). Selenium's incorporation into proteins either as selenomethionine or selenocysteine therefore does not affect their functional and structural properties. The role of selenium-induced oxidative stress in embryo toxicity and teratogenesis appears to be related to glutathione homeostasis. A review of bird studies by Hoffman (2002) showed exposure to selenium-altered concentrations and ratios of reduced to oxidized glutathione and increasing measurements of oxidative cell damage. Palace et al. (2004) suggested oxidative stress due to elevated selenium levels results in pericardial and yolk sac edema in rainbow trout embryos. In addition to oxidative stress, Kupsco and Shlenk (2014) found selenomethionine may disrupt endoplasmic reticulum (ER) homostasis in the Japanese medaka which could result in teratogenesis and embryo lethality. Evidence for the role of oxidative and ER stress in selenium toxicity is growing but mechanistic studies are still needed to better understand its effects on egg-laying vertebrates. For a more in-depth discussion on the mechanism of toxicity at the cellular level, including the evidence against sulfur substitution as a cause and the role of oxidative stress, see Janz et al. (2010).

2.4.2 Narrow Margin between Sufficiency and Toxicity of Selenium

Selenium is an essential nutrient that is incorporated into functional and structural proteins as selenocysteine and selenomethionine. Several of these proteins are enzymes that provide cellular antioxidant protection. Selenomethionine is readily oxidized, and its antioxidant activity arises from its ability to deplete reactive oxygen species. Selenomethionine is required as a mineral cofactor in the biosynthesis of glutathione peroxidases. All the classic glutathione peroxidases contain selenium and are found to be involved in the catalytic reaction of these many enzymes (Allan et al. 1999). The major functions of the glutathione peroxidases involve the reduction of hydrogen peroxide to water at the expense of the oxidation of glutathione, the enzyme's cofactor, an important antioxidant process at normal dietary levels, and the detoxification of lipid hydroperoxides. Selenium has a narrow range encompassing what is

beneficial for biota and what is detrimental. This margin between essentiality and toxicity of selenium is the narrowest of all trace elements, making the risk of negative impacts from environmental contamination extremely high (Luoma and Rainbow 2008).

Aquatic and terrestrial organisms require low levels of selenium in their diet to sustain metabolic processes, whereas excess concentrations of selenium that are approximately an order of magnitude greater than the required level have been shown to be toxic to fish and birds (Palace et.al. 2004; Ohlendorf and Heinz 2011). Dietary requirements in fish have been reported to range from 0.05 to 1.0 mg Se/kg dw (Watanabe et al. 1997). Selenium requirements for optimum growth and liver glutathione peroxidase activity in channel catfish were reported as 0.25 mg Se/kg dw (Gatlin and Wilson 1984). Estimated selenium dietary requirements in hybrids of striped bass, based on selenium retention, were reported as 0.1 mg Se/kg dw (Jaramillo 2006). Studies in rainbow trout were the first to identify the narrow range margin between essentiality and toxicity of selenium, with toxicity occurring at between seven and 30 times greater dietary exposure than essential levels (Hilton and Hodson 1983; Hodson et al. 1980). In birds, egg selenium concentrations lower than 0.66 mg Se/kg dw may indicate inadequate selenium in the diet, resulting in poor adult health and reproduction. In areas without selenium contamination, background concentrations of selenium in bird eggs are 3 to 4 mg Se/kg dw, with maximum individual values usually <5 mg Se/kg dw (Ohlendorf and Heinz 2011; Ohlendorf et al. 1986; Skorupa et al. 1996; U.S. DOI 1998). Selenium deficiency has been found to affect humans (U.S. EPA 1987), sheep and cattle (U.S. EPA 1987), deer (Oliver et al. 1990), fish (Thorarinsson et al. 1994; Wang and Lovell 1997; Wilson et al. 1997; U.S. EPA 1987), aquatic invertebrates (Audas et al. 1995; Caffrey 1989; Cooney et al. 1992; Cowgill 1987; Cowgill and Milazzo 1989; Elendt 1990; Elendt and Bais 1990; Harrison et al. 1988; Hyne et al. 1993; Keating and Caffrey 1989; Larsen and Bjerregaard 1995; Lim and Akiyama 1995; Lindstrom 1991; U.S. EPA 1987; Winner 1989; Winner and Whitford 1987), and algae (Doucette et al. 1987; Keller et al. 1987; Price 1987; Price et al. 1987; Thompson and Hosja 1996; U.S. EPA 1987; Wehr and Brown 1985). The predominance of research on selenium deficiency in invertebrates and algae is related to optimizing the health of test organisms cultured in the laboratory.

2.4.3 Adverse Effects of Selenium in Fish and Birds

The best documented, overt, and severe toxic symptoms in fish are reproductive teratogenesis and larval mortality. Egg-laying vertebrates appear to be the most sensitive taxa,

with toxicity resulting from maternal transfer to eggs. Selenomethionine is incorporated into vitellogenin in fish liver and then transfered to eggs during vitellogenesis where it is cleaved into distinct yolk proteins. In fish, the yolk proteins lipovitellin and phosvitin have been shown to contain selenium (Janz et al. 2010; Janz 2011). In studies involving young organisms exposed through transfer of selenium from adult female fish into their eggs, the most sensitive diagnostic indicators of selenium toxicity in vertebrates occur when developing embryos metabolize organic selenium that is present in egg albumen or yolk. It is then further metabolized by larval fish after hatching. Enzymes such as P450 or flavin monoxygenase can biotransform organoselenium compounds into selenoxides (Palace et al. 2004).

A variety of lethal and sublethal deformities (terata) can occur in the developing fish exposed to selenium, affecting both hard and soft tissues (Lemly 1993a). Developmental malformations are among the most conspicuous and diagnostic symptoms of chronic selenium poisoning in fish and have been used to identify impacts of selenium on fish populations (Lemly 1997; Maier and Knight 1994). Deformities in fish that affect feeding or respiration can be lethal shortly after hatching. Terata that are not directly lethal, but distort the spine and fins, can reduce swimming ability and overall fitness. Because the rate of survival of deformed young would be less than that for normal young, the percentage of deformed adults observed during biosurveys will likely underestimate the underlying percentage of deformed young, although quantitation of the difference is ordinarily not possible.

The most sensitive indicators of selenium toxicity in fish larvae are effects modulated through the reproductive process and exhibited in fish larvae as teratogenic deformities such as skeletal, craniofacial, and fin deformities, and various forms of edema that result in mortality (Lemly 2002). The toxic effect generally evaluated is the reduction in the number of normal healthy offspring compared against the initial number of eggs. In studies of young organisms exposed to selenium solely through their own diet rather than via maternal transfer, reductions in survival and/or growth are the effects that are generally evaluated.

Movement of selenium through the aquatic food web (e.g., aquatic plants, invertebrates and fish) has been shown to lead to selenium bioaccumulation in aquatic-dependent wildlife, which results in reproductive impairments and malformations (Hoffman et al. 1988; Hothem and Ohlendorf 1989; Ohlendorf et al. 1986; Skorupa and Ohlendorf 1991). For birds, diet and subsequent maternal transfer represent the critical selenium exposure route. Most of the selenium

found in bird eggs is mobilized exogenously from the maternal diet rather than endogenously from maternal tissue (DeVink et al. 2008; Ohlendorf and Heinz 2011). Thus, the most direct means of determining the potential for toxic effects of selenium in birds is through measuring egg selenium concentrations (Adams et al. 1998; Fairbrother et al. 1999; Ohlendorf and Heinz 2011). Additionally, given the rapid patterns of selenium accumulation and loss observed in birds, selenium concentrations measured in eggs will also likely represent contamination of the local environment.

Bird embryos are very sensitive to selenium (Moxon and Olson 1974; NAS 1976; Ort and Latshaw 1977, 1978). The more sensitive chronic effects identified in birds are related to reproductive impairments. Reproductive impairment is a general term including decreased fertility, reduced egg hatchability (embryo mortality), and increased incidence of deformity in embryos (Ohlendorf and Heinz 2011). Selenium exposure may cause multiple overt deformities in bird embryos including hydrocephaly, missing eyes, twisted bills, and deformed limbs (Hoffman and Heinz 1988; Hoffman et al. 1988; Ohlendorf and Heinz 2011). Toxicity studies on birds show that thresholds for reduced egg hatchability are usually below those for teratogenic effects (Ohlendorf 2003).

In 1983, incidents of mortality, congenital deformities, and reproductive failures in aquatic birds were documented at Kesterson Reservoir (Merced County, CA), a U.S. Department of Interior (DOI) National Wildlife Refuge located in the western San Joaquin Valley, California. The Reservoir consisted of a series of twelve ponds within the Kesterson National Wildlife Refuge (NWR) that were used for disposal of subsurface drainage from agricultural fields. The analyses of food chain biota (such as plants, aquatic invertebrates, and fish) and bird tissues or eggs showed that selenium was the only chemical found at concentrations high enough to cause the adverse effects on bird health and reproduction that were observed (Ohlendorf 2002). Field studies, supported by findings from laboratory studies, revealed relationships between exposure to high selenium diets, tissue selenium concentrations, and adverse effects (Heinz et al. 1988, 1989, 1990; Hoffman and Heinz 1988). For example, the mean selenium concentrations in bird eggs at Kesterson Refuge were usually 20 to 30 times higher than the reference site at Volta Refuge, which did not receive agricultural subsurface drainage discharge (Ohlendorf and Hothem 1995). All bird species mean egg concentrations at Volta were less than 3 mg/kg dw, which is typical of normal background, whereas mean egg concentrations at

Kesterson were measured up to 69.7 mg/kg dw (Ohlendorf 2002). Similar occurrences of impaired bird reproduction were subsequently observed elsewhere in the western U.S., including in the Tulare Basin of California (Skorupa 1998a; Skorupa and Ohlendorf 1991).

2.5 Assessment Endpoints

Assessment endpoints are defined as "explicit expressions of the actual environmental value that is to be protected" and are defined by an ecological entity (species, community, or other entity) and its attribute or characteristics (U.S. EPA 1998). Assessment endpoints may be identified at any level of organization (e.g., individual, population, community). In the context of the Clean Water Act, aquatic life and aquatic-dependent wildlife criteria for toxic pollutants are typically determined based on the results of toxicity tests with aquatic and aquatic-dependent organisms in which unacceptable effects on growth, reproduction, or survival occurred. This information is typically compiled into a sensitivity distribution based on genera and representing the impact on taxa across the aquatic community. Criteria are intended to be protective of most aquatic organisms in the community (i.e., approximately the 95^{th} percentile of tested aquatic organisms or aquatic-dependent wildlife representing the aquatic community).

Thus, the health of the aquatic ecosystem may be considered as an assessment endpoint indicated by survival, growth, and reproduction. For more details on aquatic life assessment endpoints for selenium see Section 2.6 in EPA's 2016 "*Aquatic Life Ambient Water Quality Criterion for Selenium – Freshwater, 2016*." This previously published EPA aquatic life criterion was developed using a genus sensitivity distribution (GSD) which represents the impact on taxa across the aquatic community but focused on reproductive effects on the most sensitive aquatic taxa, oviparous fish. For aquatic-dependent wildlife, there are significantly fewer toxicity studies available that are focused on the assessment endpoints of survival, growth, and reproduction. For this criterion, EPA relied on toxicity studies from the most sensitive aquatic-dependent wildlife species (mallard) tested to date to develop the aquatic-dependent wildlife assessment endpoint based on mallard hatchability, a reproductive endpoint.

2.6 Measures of Ecological Effect

Each assessment endpoint requires one or more "measures of ecological effect," which are defined as changes in the attributes of an assessment endpoint itself or changes in a surrogate

entity or attribute in response to chemical exposure. Ecological effects data are used as measures of direct and indirect effects to growth, reproduction, and survival of aquatic organisms.

The amount of toxicity testing data available for any given pollutant varies significantly, depending primarily on whether any major environmental issues have occurred. An in-depth evaluation of available data and subsequent review for data acceptability of selenium aquatic life studies has been performed by EPA (U.S. EPA 2016a; see Stephan et al. 1985 for additional detail on data acceptability).

In conventional chronic tests used in many EPA aquatic life criteria documents, organisms are exposed to contaminated water but fed a diet grown in uncontaminated media not spiked with the toxicant prior to introduction into the exposure chambers. Such tests are not suitable for deriving a criterion for a bioaccumulative pollutant unless (1) effects are linked to concentrations measured in appropriate tissues, and (2) the route of exposure does not affect the potency of residues in tissue. For selenium, the first condition might be met, but the second condition is not, because the route of selenium exposure appears to influence the potency of a given tissue residue (Cleveland et al. 1993; Gissel-Nielsen and Gissel-Nielsen 1978). Consequently, water-only exposure tests (and any tests not relying on dietary exposure) were not included in EPA's 2016 aquatic life criteria for selenium (U.S. EPA 2016a) and are not included in this assessment for determining criteria protective of aquatic-dependent wildlife.

Selenium toxicity in aquatic life and aquatic-dependent wildlife is primarily manifested as reproductive impairment due to maternal transfer, resulting in embryo mortality and teratogenicity. Measurements of fish tissue and bird tissue, such as eggs, are most closely linked to the chronic adverse effects of selenium (Chapman et al. 2010), since chronic selenium toxicity is based on the food chain bioaccumulation route, not a direct waterborne route. The following parts of this TSD describe the approaches used to establish selenium effect concentrations in fish tissue (U.S. EPA 2016a), and in bird egg, and to relate the concentrations in fish tissue and bird egg to concentrations in water.

2.7 Selenium Effects Concentrations in Fish Tissues and Bird Eggs

Chronic measures of effect concentrations are the $EC_{10,}$ $EC_{20,}$ No Observed Effect Concentration (NOEC), Lowest Observed Effect Concentration (LOEC), and Maximum Acceptable Toxicant Concentration (MATC). The EC_{10} is the concentration of a chemical that is estimated to result in a 10 percent effect in a measured chronic endpoint (e.g., growth,

reproduction, and survival); the EC_{20} corresponds to 20 percent effect. The NOEC is the highest chemical concentration at which none of the observed effects are statistically different from the control, as determined by hypothesis testing. The LOEC is the lowest test concentration at which observed effects are found to be statistically different from the control. Finally, the MATC is calculated as the geometric mean between the NOEC and the LOEC.

For selenium, in all cases the effect endpoint used in the estimation of chronic values (e.g., EC_{10} values) is an effect on offspring (with exposure via maternal transfer) from parents exposed to selenium via diet. For fish and birds, selenomethionine was used exclusively in dietary exposures in the lab, whereas field-exposed females would be exposed to a combination of forms of selenium as a function of the selenium in their prey items. When considering the use of the EC_{10} versus the EC_{20}, an EC_{10} was determined to be a more appropriate measure of effect concentration for tissue-based criteria given the nature of exposure and effects for this bioaccumulative chemical. Historically, EC_{20} values have been used in the derivation of EPA criteria applicable to the water medium. While water concentrations may vary rapidly over time, tissue concentrations of bioaccumulative chemicals are expected to vary gradually over time. Thus, where concentrations of selenium in bird eggs and fish tissue are used as an effect threshold, there is potential for sustained impacts on aquatic systems, relative to non-bioaccumulative chemicals. Furthermore, it was found that the dose-response curves for selenium across a broad range of fish genera are very steep compared to most toxicants, such that a small change in selenium tissue concentration yielded a large increase in observed adverse effect. These characteristically steep dose-response curves were also observed for mallards, and are likely present across additional bird genera (Ohlendorf 2003). Thus, selection of a more protective effect endpoint level (EC_{10}) as the criterion basis was deemed appropriate. For more information on methods used in EPA's derivation of effects concentrations for aquatic life, see EPA's 2016 "*Aquatic Life Ambient Water Quality Criterion for Selenium – Freshwater, 2016*." This approach is consistent with EPA's recent recommendations to States and Tribes for setting selenium water quality criteria for aquatic life (U.S. EPA 2016a). In this document, chronic values are presented as tissue concentrations (either fish egg-ovary, whole body fish tissue, muscle fish tissue, or bird egg) of total selenium in units of mg/kg dry weight (dw).

2.7.1 Water

The EPA is proposing the performance-based approach (PBA) for translating the fish and bird tissue elements into a corresponding water-column criterion element on a site-specific basis. Using the PBA, the State will derive water-column criterion elements on a site-specific basis that are translated from the fish and bird tissue criterion elements and therefore correspond to the concentration of selenium in fish and bird tissue estimated to result in a 10 percent effect (lotic or lentic water bodies as described below in Part 5.5). This water criterion element would be subordinate to the bird and fish tissue criterion elements. As in U.S. EPA (2016a), it would be derived by modeling transfer of selenium through the food web resulting in the fish and bird tissue concentrations that yield the chronic reproductive effects of concern. In Part 5, EPA discusses the translation of the tissue elements into water-column concentrations using the mechanistic modeling approach and presents a translation for birds (described below) that is comparable to the water-column translation for the fish tissue criterion elements in the 2016 304(a) selenium criterion. This water-column translation from the bird egg showed that the water column element translated from fish tissue is also protective of aquatic-dependent birds.

As described in U.S. EPA (2016a), EPA previously collaborated with the United States Geological Survey (USGS) to develop a model (later published in Presser and Luoma 2010) that relates the concentration of selenium in fish tissue to the water column. The approach models bioaccumulation and trophic transfer of selenium through aquatic food webs. Model parameters are calculated using both field and laboratory measurements of selenium in water, particulate material (algae, detritus and sediment), invertebrates, fish whole body, and fish egg-ovary. Although EPA and USGS use the same model to relate the concentration of selenium in fish tissue to water, EPA starts with selenium in the fish egg-ovary (reproductive effects criterion) whereas USGS starts with selenium in the fish whole body. The EPA approach therefore has the additional step of converting the concentration of selenium in the egg-ovary to whole body or muscle tissue concentrations using a conversion factor. This model is described in more detail in Section 3.2.1 of EPA's 2016 "*Aquatic Life Ambient Water Quality Criterion for Selenium – Freshwater, 2016*," as well as Parts 5.2, 5.3, and 5.4 of this document. Additionally, for the purpose of developing water column criteria that would also be protective of aquatic-dependent wildlife, EPA used the model with appropriate parameters to relate the concentration of selenium

in bird eggs to water (Part 5.5.2). This additional analysis showed that the water column criteria derived from fish tissue concentrations (Part 5.5.1) are protective of aquatic-dependent wildlife.

2.7.2 Summary of Assessment Endpoints and Measures of Effect

The typical assessment endpoints for aquatic life and aquatic-dependent wildlife criteria are based on effects on growth, deformity rates, reproduction, or survival of the assessed taxa. These measures of effect on toxicological endpoints have potential consequences to populations-are provided by results from toxicity tests with aquatic life and aquatic-dependent wildlife. The toxicity values (i.e., measures of effect expressed as genus means) are used in the genus sensitivity distribution of the aquatic community to derive the aquatic life criteria. For the aquatic-dependent wildlife, the tissue-based criterion is an EC_{10} for mallard hatchability (a sensitive endpoint for a sensitive species) exposed to selenomethionine and calculated from three combined mallard toxicity studies. The tissue-based criterion was derived from toxicity data for one species since the current literature does not include sufficient toxicity data for other species. However, mallard is the most sensitive species for which there is selenium toxicity data (see Parts 4.2 and 4.6). Endpoints considered and used in this assessment are listed in Table 2-2.

Table 2-2. Summary of Assessment Endpoints and Measures of Effect Used in Criteria Derivation for Selenium.

Assessment Endpoints	Measures of Effect
Fish: Survival, growth, and reproduction/teratogenesis of freshwater fish, other freshwater vertebrates, and invertebrate effects	For effects from chronic exposure: 1. EC_{10} concentrations in egg and ovary for offspring mortality and deformity. 2. Measured or estimated reproductive EC_{10} in whole body and muscle. Note: The chronic criterion is expected to be protective of acute effects.
Birds: Reproduction in birds (hatchability, teratogenesis, chick survival, and growth)	For effects from chronic exposure: 1. EC_{10} concentrations in bird egg for hatchability. Note: The chronic criterion is expected to be protective of acute effects.

2.7.3 Conceptual Model of Selenium Effects on Aquatic Life and Aquatic-Dependent Wildlife

A conceptual model depicts the relationship between a chemical stressor and ecological compartments, linking exposure characteristics to ecological endpoints. The conceptual model provided in Figure 2-4 summarizes potential pathways of selenium exposure for aquatic life and aquatic-dependent wildlife.

Selenium initially enters the aquatic environment through runoff, leachate, and wastewater discharges from mining, oil refineries, disturbance and excavation in Cretaceous marine shales, and agricultural activities. Selenium entering the aquatic environment occurs as selenate, selenite, and selenides in dissolved and particle-bound forms and readily sorbs to surfaces, such as sediment and particulate matter in the water column, which is depicted in the conceptual model (Figure 2-4). Exposure pathways for the biological receptors of concern (i.e., non-target aquatic-dependent wildlife) and potential effects (e.g., reproductive impairment by reduced hatch, deformities, and mortality) in those receptors are represented in the conceptual model (Figure 2-4). Both direct (i.e., exposure from the water column which is represented by *) and indirect (i.e., bioconcentrated by producers and bioaccumulated by consumers in higher trophic levels represented by **) pathways are represented in the conceptual model (Figure 2-4).

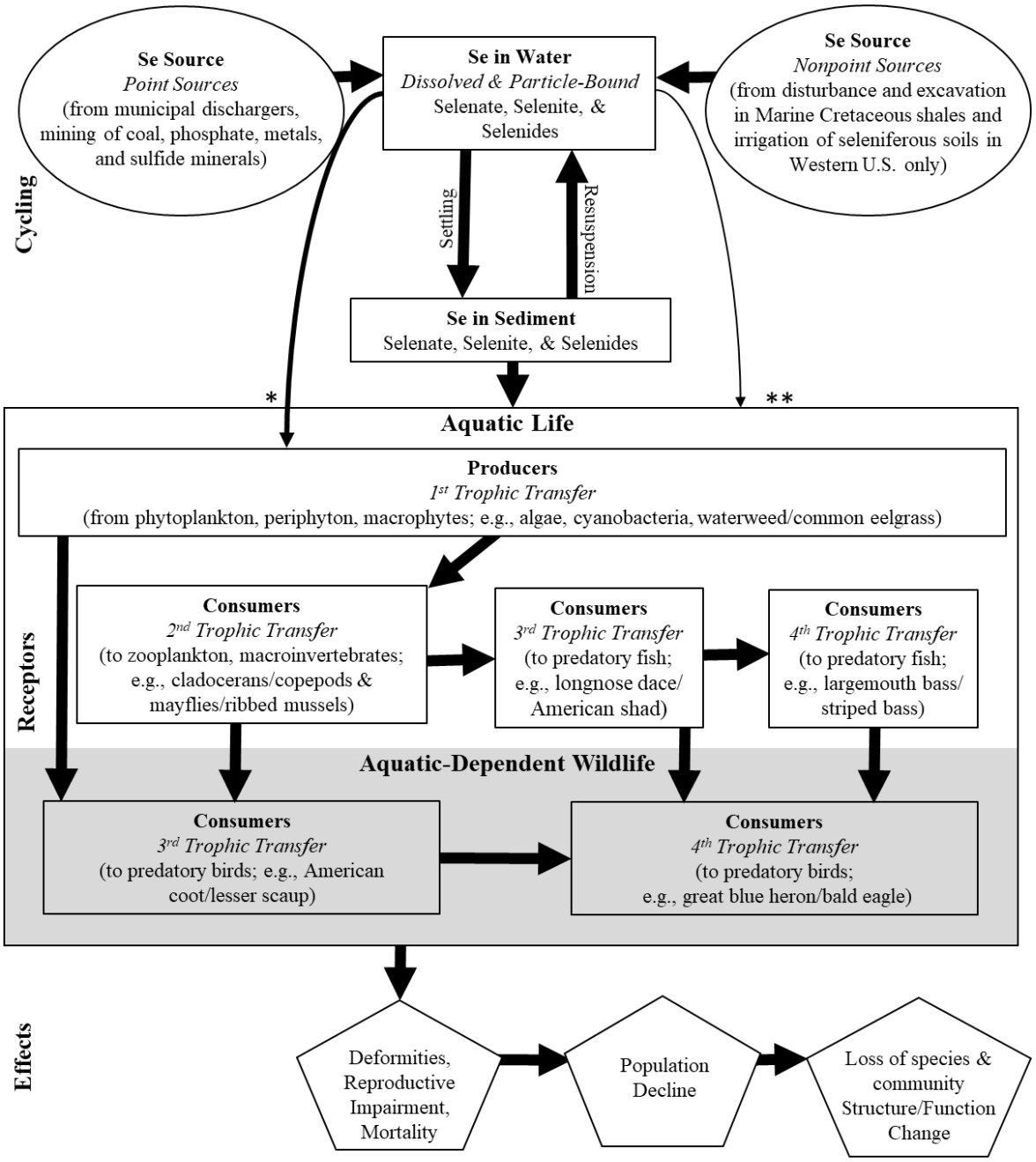


Figure 2-4. Conceptual model diagram of sources, compartmental partitioning, and trophic transfer pathways of selenium in the aquatic environment and bioaccumulation and effects in aquatic-dependent wildlife.

Selenium sources represented in ovals, compartments within the aquatic ecosystem represented by rectangles, and effects (on trophic levels of aquatic-dependent wildlife, represented by shaded box) in pentagons. Examples of organisms in each trophic transfer provided as freshwater/marine. Weighted arrows indicate relative proportion of selenium from each source. Movement of selenium from water indicated by two separate pathways: bioconcentration by producers (*) and direct exposure to all trophic levels within box (**). Relative proportion of selenium transferred between each trophic level is dependent on life history characteristics of each organism.

Part 3 EFFECTS ANALYSIS FOR FRESHWATER AQUATIC ORGANISMS

3.1 Purpose

The purpose of this chapter is to summarize EPA's 2016 "*Aquatic Life Ambient Water Quality Criterion for Selenium – Freshwater, 2016*," which was finalized and published in June of 2016 (EPA 822-R-16-006).[7] EPA is proposing the national recommended 2016 selenium aquatic life criterion as the aquatic life criterion for California. The tissue-based criterion element concentrations were developed to protect against reproductive impairment in aquatic life due to maternal transfer of selenium to offspring, resulting in mortality and teratogenicity, and will be briefly summarized in this chapter. The national recommended criterion has four elements: two fish tissue based elements and two water column based elements. The fish tissue elements consist of an egg or ovary tissue final chronic value of 15.1 mg Se/kg dw, and whole body or muscle tissue final chronic values of 8.5 and 11.3 mg Se/kg dw, respectively. The water column elements are described in detail in Part 5 of this Technical Support Document (TSD).

3.2 Overview of Effects Analysis for Freshwater Aquatic Organisms

In EPA's 2016 "*Aquatic Life Ambient Water Quality Criterion for Selenium – Freshwater, 2016*," data were obtained primarily by search of published literature using EPA's public ECOTOX database. The most recent ECOTOX database search extended to July 2013; this document also reflects data either gathered or received by EPA based on information from the 2014 public comment period and 2014 external expert peer review of the "External Peer Review Draft" published in May 2014, as well as information gathered based on public comments on the 2015 draft criterion. All available, relevant, and reliable chronic toxicity values were incorporated into the appropriate selenium AWQC tables and used to recalculate the final chronic value (FCV), as outlined in detail in the EPA Ambient Water Quality Criteria Guidelines. The chronic values derived for the reproductive effects (survival, deformities, and edema) endpoints are based on the concentration of selenium in the eggs or ovary, the tissues most directly associated with the observed effects.

Data used to derive the FCV were differentiated based on the effect (reproductive and non-reproductive effects). Acceptable chronic toxicity data on fish reproductive effects are

7 https://www.epa.gov/sites/production/files/2016-07/documents/aquatic_life_awqc_for_selenium_-_freshwater_2016.pdf

available for 10 fish genera. Acceptable chronic toxicity data on non- reproductive effects are available for 7 fish genera and 3 invertebrate genera. The fish non- reproductive effects data were not used to calculate tissue criterion elements because they were more variable and less reproducible than the data on reproductive effects. The genus sensitivity distribution is predominantly populated with data on fish species because field evidence demonstrated that fish communities were affected in situations having no observable change in the accompanying diverse array of invertebrate communities. As a result, decades of aquatic toxicity research have focused primarily on fish. The studies that have been done with invertebrates have shown them to be somewhat more tolerant than most of the tested fish species. Table 3-1 summarizes the effect concentrations obtained from all acceptable reproductive studies with fish.

Also, while amphibians are potentially sensitive due to physiologic similarities to fish, effects clearly attributable to selenium are not well-known (Hopkins et al. 2000; Janz et al. 2010; Massé et al. 2016; Unrine et al. 2007). Hopkins et al. (2000) reported that amphibian larvae at sites receiving coal combustion wastes appear to efficiently accumulate selenium in their tissues and have exhibited axial malformations (possibly due to selenium). In a recent laboratory exposure, Massé et al. (2015) determined an EC_{10} of 44.9 mg Se/kg for the African clawed frog (*Xenopus laevis*) suggesting that this species is similarly sensitive to the less sensitive fish species.

This section presents a summary of reproductive studies included in the selenium data set and how they were used to derive the tissue criterion elements for egg-ovary, whole body and muscle. For a detailed review of each reproductive study used to derive the criterion, see Section 3.1.1 Acceptable Studies of Fish Reproductive Effects for the Four Most Sensitive Genera in EPA's 2016 aquatic life criterion (ALC) document. Other reproductive and non-reproductive studies that support the derivation of the tissue criterion are provided in Section 6 of the 2016 ALC document, Effects Characterization.

Table 3-1. Maternal Transfer Reproductive Toxicity Studies.

Species	Reference	Exposure Route	Toxicological Endpoint	Chronic Value mg Se/kg dw[a]	SMCV mg Se/kg dw	GMCV mg Se/kg dw
Salvelinus malma Dolly Varden	Golder Associates 2009	dietary and waterborne (field: Kemess Mine NW British Columbia)	EC_{10} for total deformities	56.2 E	56.2 E	56.2 E
Esox lucius northern pike	Muscatello et al. 2006	dietary and waterborne (field: Saskatoon, Sask.)	EC_{24} larval deformities	34.0 E	34.0 E	34.0 E
Cyprinodon macularius desert pupfish	Besser et al. 2012	dietary and waterborne (lab)	Estimated EC_{10} for offspring survival	27 E	27 E	27 E
Micropterus salmoides largemouth bass	Carolina Power & Light 1997	dietary (lab)	EC_{10} for larval mortality & deformity	26.3 O	26.3 O	26.3 O
Pimephales promelas fathead minnow	Schultz and Hermanutz 1990	dietary and waterborne (mesocosm: Monticello)	LOEC for larval edema and lordosis	<25.6 E[b]	NA[c]	NA
Oncorhynchus mykiss rainbow trout	Holm 2002; Holm et al. 2003, 2005	dietary and waterborne (field: Luscar River, Alberta)	EC_{10} for skeletal deformities	24.5 E[b]	24.5 E	25.3 E
Oncorhynchus clarkii lewisi Westslope cutthroat trout	Rudolph et al. 2008	dietary and waterborne (field: Clode Pond, BC)	EC_{10} for alevin mortality	24.7 E	26.2 E	
Oncorhynchus clarkii lewisi Westslope cutthroat trout	Nautilus Environmental 2011	dietary and waterborne (field: Clode Pond & Fording River, BC)	EC_{10} for survival at swim-up	27.7 E		
Salmo trutta brown trout	Formation Environmental 2011; AECOM 2012	dietary and waterborne (field: Lower Sage Creek & Crow Creek, ID)	EC_{10} for larval survival	21.0 E	21.0 E	21.0 E

Species	Reference	Exposure Route	Toxicological Endpoint	Chronic Value mg Se/kg dw[a]	SMCV mg Se/kg dw	GMCV mg Se/kg dw
Lepomis macrochirus bluegill	Doroshov et al. 1992	dietary (lab)	EC_{10} larval edema	22.6 E	20.6 E	20.6 E
Lepomis macrochirus bluegill	Coyle et al. 1993	dietary and waterborne (lab)	EC_{10} for larval survival	26.3 E		
Lepomis macrochirus bluegill	Hermanutz et al. 1992, 1996	dietary and waterborne (mesocosm: Monticello)	EC_{10} for larval edema	14.7 O[b]		
Acipenser transmontanus white sturgeon	Linville 2006	dietary (lab)	EC_{10} for combined edema and deformities	15.6 E	15.6 E	15.6 E

E–concentration reported in egg; O–concentration reported in ovary.
SMCV–species mean chronic value; GMCV-genus mean chronic value.

[a] All chronic values reported in this table are based on the measured concentration of selenium in egg-ovary tissues.

[b] Tissue value converted from ww to dw. See U.S. FWS (2017) for conversion factors.

[c] SMCV not calculated due to variability in the observations among replicates in Schultz and Hermanutz (1990). The chronic value is presented in this table to show it is in the range of selenium effect concentrations. See U.S. FWS (2017) for detail. Also, see Appendix E of U.S EPA (2016a) for an additional study with fathead minnow.

3.2.1 Fish Egg-Ovary Criterion Element Concentration

The lowest four GMCVs for fish reproductive effects as measured in eggs or ovaries are presented below in Table 3-2. With n = 15 GMCVs (see Section 3.1.6 in U.S. EPA 2016a), the 5th percentile projection yields an egg-ovary criterion element concentration of 15.1 mg Se/kg dw egg-ovary, lower than the most sensitive fish species tested, white sturgeon (*A. transmontanus*).

Table 3-2. Four Lowest Genus Mean Chronic Values for Fish Reproductive Effects (U.S. EPA 2016a).

Relative Sensitivity Rank	Genus	GMCV (mg Se/kg dw egg-ovary)
4	*Oncorhynchus*	25.3
3	*Salmo*	21.0
2	*Lepomis*	20.6
1	*Acipenser*	15.6

3.2.2 Fish Whole Body Criterion Element Concentration

Whole body reproductive chronic values were calculated directly from whole body tissue concentrations measured in the study or by applying an egg-ovary (EO) to whole body (WB) conversion factor (*CF*) described in Section 3.2.2.2 of U.S. EPA (2016a). Direct calculations were done when whole body measurements were available in the study and the data were amenable to an effect level determination. The final EO/WB *CF* applied to each taxon was determined using a hierarchical approach based on taxonomic relatedness, and is described in Section 3.2.2 and Appendix B of U.S. EPA (2016a). The four most sensitive reproductive-effect fish whole body GMCVs are shown in Table 3-3. Because the factors used to convert egg-ovary to whole body concentrations vary across species, the whole body rankings differ from the egg-ovary rankings. With n = 15 GMCVs, the 5th percentile projection yields a whole body criterion element concentration of 8.5 mg Se/kg dw whole body, slightly lower than the most sensitive fish species tested, white sturgeon (*A. transmontanus*).

Table 3-3. The Lowest Four Reproductive-Effect Whole Body GMCVs.

Relative Sensitivity Rank	Genus	GMCV (mg Se/kg dw whole body)
4	*Salmo*	13.2
3	*Oncorhynchus*	11.6
2	*Lepomis*	9.9
1	*Acipenser*	9.2

3.2.3 Fish Muscle Criterion Element Concentration

Reproductive chronic values for muscle tissue were calculated directly from muscle tissue concentrations measured in the study or from the egg-ovary to muscle conversion factors (described in Section 3.2 of U.S. EPA 2016a). Direct calculations were made when muscle measurements were available in the study and the data were amenable to an effect level determination. The final EO/M *CF* applied to each taxon was determined using a hierarchical approach based on taxonomic relatedness, consistent with the approach used to calculate EO/WB CFs. The four most sensitive reproductive-effect fish muscle GMCVs are shown in Table 3-4. Because the factors used to convert egg-ovary to muscle concentrations vary across species based on empirical data, the whole body rankings differ from both the egg-ovary rankings and the muscle rankings. With n = 15 GMCVs, the 5^{th} percentile projection yields a muscle criterion element concentration of 11.3 mg Se/kg dw muscle, lower than the muscle value for the most sensitive fish species tested, white sturgeon (*A. transmontanus*).

Table 3-4. The Lowest Four Reproductive-Effect Fish Muscle GMCVs.

Relative Sensitivity Rank	Genus	GMCV (mg Se/kg dw muscle)
4	*Salmo*	18.5
3	*Lepomis*	15.9
2	*Oncorhynchus*	14.3
1	*Acipenser*	11.9

Part 4 EFFECTS ANALYSIS FOR AQUATIC-DEPENDENT WILDLIFE

4.1 Purpose

For the derivation of this aquatic life criterion, species that rely on aquatic prey as a major food source were considered aquatic-dependent (see Part 4.2 for more detailed definition). This tissue-based criterion was developed to protect against the adverse effects associated with elevated exposure to selenium to aquatic-dependent wildlife, such as mortality, altered growth, and reproductive impairment. Birds appear to be the most sensitive aquatic-dependent taxa to selenium exposure (Ohlendorf 2003; Janz et al. 2010), therefore the chronic tissue-based criterion element was derived using birds. Similar to previous assessments focused on the effects of bioaccumulative contaminants on aquatic-dependent wildlife (U.S. EPA 1995, 1997, 2011a), the derivation of this bird egg criterion element was based on toxicity data from the most sensitive tested bird species (mallard), as this approach is expected to be protective of aquatic-dependent wildlife including endangered species living in California. The tissue-based criterion element was then translated to a protective water concentration, considering the different diets and other life history traits of individual avian species. The resulting water concentration is approximately equal to the chronic water column based criterion element for aquatic life (Part 5.5.2), which demonstrates that the chronic water column based criterion for aquatic life is also protective of aquatic-dependent wildlife.

4.2 Chronic Toxicity to Aquatic-Dependent Wildlife

All available data relating to the chronic toxicological effects of selenium on aquatic-dependent wildlife were considered in the derivation of this selenium criterion for the state of California. Data meeting the quality objectives and test requirements that were utilized in deriving this criterion for aquatic-dependent wildlife are presented in Table 4-1.

Data for possible inclusion in this California selenium criterion were obtained from published literature reporting chronic exposures of selenium that were associated with mortality, growth, and reproduction. This set of published literature was identified by both EPA's public ECOTOX database and additional literature searches. Studies with dietary and/or maternal transfer selenium exposures were considered for possible inclusion. In developing this selenium aquatic-dependent wildlife criterion for the state of California, only taxa that depend on aquatic prey (e.g., fish and emergent aquatic insects) as a major food source were considered aquatic-

dependent. The dietary composition of the taxa considered in this criterion consisted of ≥75% aquatic prey, including fish, aquatic invertebrates, amphibians, and other aquatic-dependent wildlife (birds). Additionally, studies utilizing taxa that are not considered aquatic-dependent (e.g., members of the order Galliformes such as chickens and pheasant) were not considered for possible inclusion unless the taxa could be a surrogate for an aquatic-dependent species within the same or closely related order (e.g., studies focused on American kestrel were included as other members of this order such as peregrine falcon are aquatic-dependent). Lastly, only studies that utilized organic selenium, such as selenomethionine, were considered for possible inclusion. Selenomethionine has been shown to be highly toxic to birds and appears to be the chemical form most likely to bioaccumulate in tissues including bird eggs (Heinz et al. 1987; Hoffman and Heinz 1988), and therefore is important to consider in evaluating potential risks from natural exposures experienced by wild birds (Ohlendorf and Heinz 2011). Results based on dosing with selenite and/or selenate were not utilized in the derivation of this criterion due to differences in toxicity when compared to organic selenides (Heinz et al. 1987; Hoffman and Heinz 1988).

The studies meeting these inclusion criteria were screened for data quality generally as described by Stephan et al. (1985) in the 1985 Guidelines and in EPA's Office of Chemical Safety and Pollution Prevention (OCSPP)'s Ecological Effects Test Guidelines (U.S. EPA 2012). These toxicity data were further screened to ensure that the observed effects could be primarily attributed to exposure to selenium. Both controlled laboratory experiments and field studies were included. When available, measured selenium concentrations were used; however, for several studies measured dietary selenium concentrations were not reported, and nominal concentrations were utilized if a dose-response relationship was observed in another media (e.g., blood or eggs).

The studies meeting the inclusion criteria described above were used to derive a reproductive effect-based EC_{10}, which is the basis for this aquatic-dependent wildlife criterion element for the state of California. As discussed in Part 2.7 above, due to the bioaccumulative nature of selenium and the dietary pathway of exposure, the derivation of the criterion was based on an effect concentration that impacted a small percentage of the study organisms (e.g., a 10% effect concentration [EC_{10}]; U.S. EPA 2016a).

4.2.1 Summary of Selenium Reproductive Toxicity Studies Used to Derive the Aquatic-Dependent Wildlife Criterion

Data for chronic selenium toxicity were available for eleven bird species, representing nine families and six orders. Mallards (*Anas platyrhynchos*) were the most sensitive species tested and hatchability was consistently the most sensitive endpoint. In contrast, red-winged blackbird (*Agelaius phoeniceus*) appears to be the least sensitive species with selenium toxicity data in the current literature. Harding (2008) reports adverse effects on hatchability at selenium egg concentrations of approximately 22.0 mg/kg dw (see Part 4.6.1 for additional details).

Six of the mallard toxicity studies described below (Heinz et al. 1987; Heinz et al. 1989; Heinz and Hoffman 1996; Heinz and Hoffman 1998; Stanley et al. 1994; Stanley et al. 1996) had a similar test design in which seleno-DL-methionine was fed to breeding pairs in artificial diets. Data from three of these studies (Heinz et al. 1987; Heinz et al. 1989; Stanley et al. 1996) were combined into a single concentration-response relationship for hatchability versus selenium concentrations in eggs. This concentration-response relationship was used to derive the aquatic-dependent wildlife criterion (see Part 4.3). The other three studies (Heinz and Hoffman 1996 and 1998; Stanley et al. 1994) were not included in the combined dataset mentioned above and were not used quantitatlively to derive the aquatic-dependent wildlife criterion. See Part 4.3 below for details on the qualitative use of these studies. A summary of the studies including dietary concentrations, control hatchability, and observed effects is in Part 4.6.1.

Below is a brief description of the three mallard toxicity studies used in the derivation of the present criterion for the state of California, including a synopsis of the experimental design, test duration, relevant test endpoints, and other critical information. Data are summarized in Table 4-1, and more detailed study summaries are included in Appendix Table A-1.

All three mallard toxicity studies used to derive the bird egg criterion (Heinz et al. 1987, 1989; Stanley et al. 1996) were conducted at the Patuxent Environmental Science Center, Laurel, Maryland under similar test conditions. Each study exposed breeding pairs of mallards (between one and two years old) to a commercial diet supplemented with varying concentrations (between 1 and 16 mg/kg) of selenium as seleno-DL-methionine (Table 4-1). To delay the onset of egg laying, females were kept in indoor pens for three to four weeks at eight hours of light per day. The females were fed their assigned diet (control or selenium treated) prior to being paired with males and placed in outdoor pens (1 m^2). The dietary treatments and the number of breeding pairs per treatment for each study are listed in Table 4-1. Nests were monitored daily, eggs were

numbered sequentially, and either the eighth (Heinz et al. 1989; Stanley et al. 1996) or tenth (Heinz et al. 1987) egg was collected to measure whole egg weight, length, width; shell weight and thickness; and weight of egg contents. The contents of each of these eggs were saved for selenium analyses. Additional eggs were collected throughout the breeding period from one extra breeding pair by Heinz et al. (1987), as the first, fifth, ninth, thirteenth, seventeenth, twenty-first, twenty-fifth, twenty-ninth, and thirty-third eggs and and from three extra breeding pairs by Heinz et al. (1989) as the first, fourth, seventh, tenth, thirteenth, and sixteenth eggs to demonstrate that selenium concentrations varied little across the clutch. In Stanley et al. (1996), females incubated their own clutch of ≤ 20 eggs. In Heinz et al. (1987, 1989) eggs were selected for incubation, labeled according to pen, and stored at 10°C, until placement in an incubator maintained at 37.6°C and at a relative humidity of 60-68%.

In addition to selenium, Stanley et al. (1996), included dietary treatments that exposed the birds to boron. To avoid complications of potential interactions with boron, only those treatments to which selenium alone was added to the diet were included in the effects analysis for this study.

Heinz et al. (1987) and Heinz et al. (1989) included dietary treatments with chemical forms of selenium (Se) other than seleno-DL-methionine. Heinz et al. (1987) included dietary treatments of selenium as sodium selenite at 1, 5, 10, 25, and 100 mg/kg. Heinz et al. (1989) included a dietary treatment of 16 mg/kg selenium as seleno-DL-cystine. As stated in the previous section, the chemical form of selenium determined to be suitable for the effects analysis was selenomethionine because of its toxicity and bioavailability. An example of its greater bioavailability was observed in Heinz et al. (1987) where the dietary treatment of 10 mg/kg selenium as selenite resulted in 0.53 mg Se/kg wet weight (ww) in eggs, whereas 10 mg/kg selenium as selenomethionine yielded 4.6 mg Se/kg ww in eggs. Selenomethionine was also found to be much more bioavailable than selenocystine in Heinz et al. (1989) where 16 mg/kg dietary treatments of both forms of selenium resulted in the respective egg selenium concentrations of 18 and 0.57 mg/kg ww. Additionally, eggs collected from extra breeding pairs fed the selenium treated diets in Heinz et al. (1987) and Heinz et al. (1989) showed little intra-clutch variability in measured selenium concentrations.

These three mallard toxicity studies looked at endpoints such as mortality and body weight in the parents and offspring as well as hatchability, egg weight, embryo deformity, fertility, and growth. The addition of 10 mg/kg selenium as selenomethionine to the diet did not

have any effects on adult survival or weight at sacrifice (mean weights of 1,120 g for males and 1,114 g for females) compared to those in the control group (mean weights of 1,046 g for males and 1,141 g for females) in Heinz et al. (1987). And while the percent hatch of fertile eggs and duckling weight at twenty-one days old were reduced in the selenium treatment group (30.9% and 297 g, respectively) compared to the control group (65.7% and 371 g, respectively), these reductions were not statistically significantly different from controls. Alternatively, an 18.3% increase in abnormal embryos was observed in the selenium treatment group as were reductions in the percent of healthy hatchlings surviving twenty-one days of age (50%) when compared to controls (98.7%).

Similarly, Heinz et al. (1989) did not observe any effects on adult survival or signs of selenium intoxication. The study authors reported statistically significant reductions in percent hatch of fertile eggs in the 16 mg/kg selenium dietary treatment group (2.2% hatch of fertile eggs) and a statistically significant reduction in nestling weight in the 8 mg/kg selenium dietary treatment group (58 g) compared to controls (59.6% hatch of fertile eggs and 72 g, respectively). Of embryos that did not hatch, 6.8 and 67.9% contained malformed embryos in the 8 and 16 mg/kg selenium treatment groups, respectively, compared to 0.6% in the control group. The results of the deformity analysis in the Heinz et al. (1987, 1989) were reported and discussed in Hoffman and Heinz (1988). For a summary of the deformity findings reported by Hoffman and Heinz (1988), see Part 4.6.1.

Lastly, Stanley et al. (1996) did not observe any effects of selenium on adult weight. However, reductions in fourteen-day old duckling weight were observed in the 7 mg/kg selenium treatment group (mean weight of 130.1 g) compared to controls (mean weight of 145.1 g), but these reductions were not statistically significant. A statistically significant decrease in hatching success was observed in the 7 mg/kg selenium treatment group (41% hatch) compared to the control (62% hatch).

Hatchability was the reproductive endpoint in all three studies (Heinz et al. 1987, 1989; Stanley et al. 1996). Duckling weight, growth, and production were all equally sensitive to hatching success in Stanley et al. (1996), and the number of normal hatchlings and nestling weight were also similar in sensitivity to hatchability in Heinz et al. (1989). Therefore, as hatchability was one of the most sensitive endpoints reported, was consistently observed, and

was comparable across all three studies, the bird egg criterion element was based on hatchability data reported by Heinz et al. (1987, 1989) and Stanley et al. (1996).

Table 4-1. Effect of Dietary Selenium (as Selenomethionine) on Hatchability of Mallard Eggs and the Associated Concentration of Selenium in Eggs.

Modified from Table 17.1 in Ohlendorf (2003).

Diet Se mg/kg[a] Nominal	N (hens)	Egg Hatchability %[b]	% Hatchability as % Control	Percent Moisture	Egg Se, mg/kg dw	Reference
Control	11	64.4	100	71	0.17	Heinz et al. 1987
10	5	34.6	54	71	15.9	Heinz et al. 1987
Control	32	57.3	100	70	0.60	Heinz et al. 1989
1	15	65.0	114	70	2.77	Heinz et al. 1989
2	15	59.6	104	70	5.33	Heinz et al. 1989
4	15	54.3	95	70	11.3	Heinz et al. 1989
8	15	42.3	74	70	36.7	Heinz et al. 1989
16	9	7.4*	13	70	60.0	Heinz et al. 1989
Control	33	62	100	71	0.93	Stanley et al. 1996
3.5	29	61	98	71	12.1	Stanley et al. 1996
7	34	41*	66	71	24.5	Stanley et al. 1996

[a] Selenium concentrations in diet are presented as nominal. Control diets typically contained 0.4 mg Se/kg dw.

[b] Asterisks indicate hatchability determined by respective authors to be significantly different than control following post hoc means comparison testing.

4.3 Derivation of Bird Egg Criterion Element

The data outlined in Table 4-1 from the three mallard toxicity studies summarized above in Part 4.2.1 (Heinz et al. 1987, 1989; Stanley et al. 1996) were analyzed using the statistical software program R (version 3.4.3) and the associated dose-response curve (drc) package to calculate a bird egg EC_{10} of 11.2 mg Se/kg dw with a lower 95% confidence limit of 7.4 mg Se/kg dw and a 95% upper confidence limit of 15.0 mg Se/kg dw (Figure 4-1). All parameters in this model yielded significant p-values ($P \leq 0.05$). This selenium EC_{10} was derived from a four-parameter model (Equation 4-1) and is the basis for the aquatic-dependent wildlife criterion element.

$$\pi(x) = c + \frac{d - c}{1 + \exp^{b(\log(x) - e))}}$$

(Equation 4-1)

where:

x	=	Selenium concentration
$\pi(x)$	=	Probability egg hatches at concentration x
b	=	Slope of the dose response curve at EC_{50}
c	=	Lower horizontal asymptote
d	=	Upper horizontal asymptote
e	=	EC_{50} concentration

The approach used to derive the bird egg EC_{10} of 11.2 mg Se/kg dw was similar to the meta-analysis conducted by Ohlendorf (2003) described in detail in Part 4.4 below. The meta-analysis by Ohlendorf (2003) included data from three mallard toxicity studies not included here by EPA (Heinz and Hoffman 1996, 1998; Stanley et al. 1994) to calculate a selenium mallard EC_{10} of 12.5 mg Se/kg dw based on hatchability. The Ohlendorf (2003) bird egg EC_{10} of 12.5 mg Se/kg dw serves as the basis for the selenium standard in the Great Salt Lake of Utah (CH2M Hill 2008). Two of the three mallard toxicity studies used in Ohlendorf (2003) but not in this meta-analysis had control hatchability below 52% (Heinz and Hoffman 1996, 1998) and therefore did not meet EPA's test guidelines (U.S. EPA 2012). In contrast to Ohlendorf (2003), data in this California selenium criterion analysis were not control normalized prior to analysis,

and a Fisher's exact test was performed to determine if statistically significant differences existed in hatchability across the control groups. As a result, the data from Stanley et al. (1994) were removed from the meta-analysis because the high control hatchability in this study was determined to be statistically different from the other control groups in the meta-analysis (91.4% in Stanley at al. 1994, compared to 57-64.4% in the remaining studies) and resulted in a poor goodness of fit. The bird egg EC_{10} derived from the remaining three studies (Heinz et al. 1987, 1989; Stanley et al. 1996) was 11.2 mg Se/kg dw.

In addition to removing three of the studies for reasons described above, the data used to derive the EC_{10} of 11.2 mg Se/kg dw here differ from those analyzed by Ohlendorf (2003) in the following respects. First, selenium concentrations used in the EC_{10} calculation were converted from wet weight to dry weight using whole egg percent moisture contents provided by the authors in the respective studies, in contrast to the average value of 70% whole egg moisture content used by Ohlendorf (2003). The difference was negligible. Second, for data from Heinz et al. (1987) and Heinz et al. (1989), the arithmetic mean percent hatchabilities were determined from raw data provided by the lead author instead of mean concentrations reported in the respective publications in order to be consistent with the remaining study. Mean hatchabilities reported in Heinz et al. (1987) and Heinz et al. (1989) had been back calculated from arcsine square root transformed values, which were slightly different than the original measured values (G. Heinz, pers. comm.).

The modeling approach used to derive the bird egg EC_{10} value of 11.2 mg Se/kg dw for this selenium aquatic-dependent wildlife criterion was selected because it is conceptually similar to the approach used by Ohlendorf (2003), which is a widely accepted EC_{10} for selenium and serves as the basis for the selenium standard in the Great Salt Lake of Utah (CH2M Hill 2008). The bird egg EC_{10} of 11.2 mg/kg dw calculated for this aquatic-dependent wildlife selenium criterion is considered preferable to the Ohlendorf (2003) EC_{10} because the corrections to the dataset described above ensure that EPA's data quality guidelines are met and that the observed effects on egg hatchability reflect selenium exposure.

Figure 4-1. Logistic regression model of mallard hatchability in relation to egg selenium concentrations.

Mallard egg EC_{10} for selenium of 11.2 mg Se/kg dw. Gray shaded area surrounding the fitted curve represents 95% confidence interval.

4.4 Previously Calculated Selenium Thresholds (as EC_{10}) for Mallard Hatchability

Meta-Analysis of Six Mallard Toxicity Studies – Ohlendorf (2003)

As mentioned above in Part 4.3, Ohlendorf (2003) calculated an egg EC_{10} of 12.5 mg Se/kg dw for mallard egg hatchability based on a meta-analysis of six different laboratory studies using logistic regression (Heinz and Hoffman 1996, 1998; Heinz et al. 1987, 1989; Stanley et al. 1994, 1996). Data from the six studies were normalized to their respective controls and combined in a single dataset prior to analysis. The resulting EC_{10} for mallard egg hatchability was 12.5 mg Se/kg dw, with a 5% lower confidence limit of 6.4 mg Se/kg dw and a 95% upper confidence limit of 16.5 mg Se/kg dw (Figure 4-2). At around the same time, Adams et al. (2003) using five of the above six studies (excluding Heinz et al. 1987), had calculated EC_{10}s in

the range of 12-15 mg Se/kg dw (rounded to two digits) using logit, probit, and piece-wise linear curves.

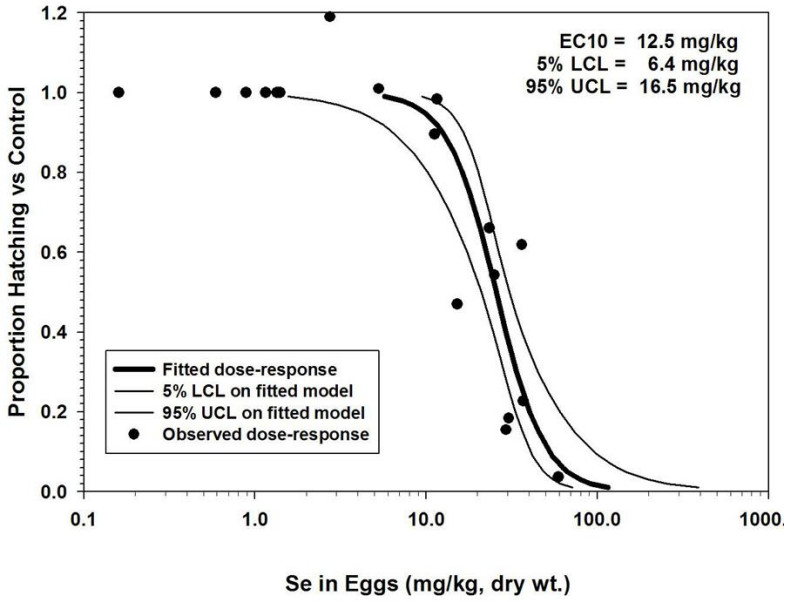


Figure 4-2. Mallard egg hatchability as a function of selenium concentration in eggs.
Source is Figure 17.2 from Ohlendorf (2003).
The data were normalized to their respective control hatchability values. LCL = lower confidence limit. UCL = upper confidence limit.

2011 EPA Reanalysis of the Six Mallard Toxicity Studies

EPA performed an independent evaluation of the mallard data used in the Ohlendorf (2003) analysis during its review of the selenium standard for the Great Salt Lake in Utah (U.S. EPA 2011a; CH2M Hill 2008). The values used in the U.S. EPA (2011a) reanalysis were adjusted from those in Ohlendorf (2003) after accounting for author-reported percent moisture content in eggs, potential arsenic exposure in the Stanley et al. (1994) treatment mean, and differences in mean percent hatchabilities from Heinz et al. (1987, 1989) resulting from back calculation of arcsine square root transformed values as described in Part 4.3. Collectively, these adjustments had a minor influence on the results but improved the accuracy of the dataset. In U.S. EPA (2011a), three EC_{10} values were calculated using different models (tolerance distribution and nonlinear regression models) and data (all six studies vs. only the four studies with control hatchability greater than 52%). The egg EC_{10} values were calculated using the U.S. EPA Toxicity Relationship Analysis Program (TRAP; U.S. EPA 2011b), and ranged from 9.7-

12.7 mg Se/kg dw. The concentration range of these tests supported the results of the Ohlendorf (2003) analysis, which serves as the basis for the Great Salt Lake selenium standard (CH2M Hill 2008).

The first egg EC_{10} was calculated from the six study Ohlendorf (2003) mallard dataset without normalizing egg hatchability to controls. Some authors have suggested that control normalization is inappropriate because control responses themselves contain variability, and that control normalization effectively removes this estimation error from the control values (OECD 2006). The resulting egg EC_{10} was 12.3 mg Se/kg dw using a tolerance distribution model.

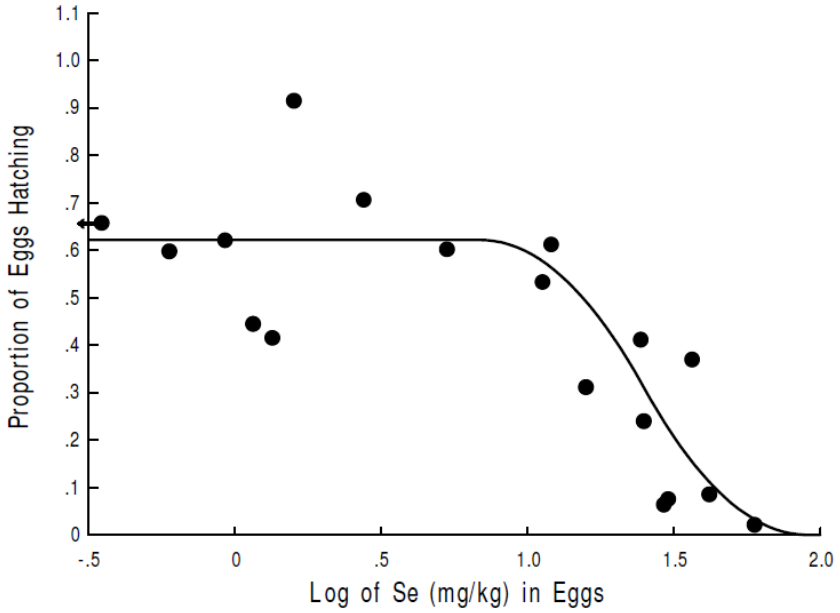


Figure 4-3. Mallard egg hatchability in the six studies, fitted using a tolerance distribution model, without normalization to control values. Source is Figure 2 in U.S. EPA (2011a). Confidence intervals surrounding the EC_{10} were not included in the source document.

A second egg EC_{10} with no control normalization was calculated from the combined mallard dataset after excluding the two studies with low (<52%) control hatchability (Heinz and Hoffman 1996, 1998). The resulting egg EC_{10} was 12.7 mg Se/kg dw, and was also calculated using a tolerance distribution model.

In a third egg EC_{10} calculation, EPA derived an EC_{10} for all six mallard studies after first normalizing egg hatchability from each study to their respective controls. EPA calculated an egg EC_{10} of 9.7 mg Se/kg dw using a logistic nonlinear regression model. Although this estimate of

an egg EC_{10} is different than the 12.5 mg Se/kg dw in Ohlendorf (2003), EPA could find no scientific basis for concluding that a logistic nonlinear regression fit was more or less appropriate than a tolerance distribution fit. In the absence of any meaningful scientific justification to prefer the one approach over the other, the different values derived from the application of these two models to the same data are both scientifically defensible.

EPA further evaluated the effects of selenium in egg tissue below the EC_{10} of 12.5 mg Se/kg dw. Figure 4-4 shows the percent hatch in the six control treatments, and the selenium exposed treatments for those studies (U.S. EPA 2011a). The egg EC_{10} of 12.5 mg Se/kg dw is represented by the vertical line. Hatchability at all treatment concentrations less than 12.5 mg Se/kg dw are within the range of the controls and the lower 95% confidence range of the control mean, which is shown by the lower horizontal dashed line. By contrast, all treatment concentrations greater than the egg EC_{10} of 12.5 mg Se/kg dw yielded hatchability below the lower confidence bound for the control mean and below the hatchability of any control. These data suggest that the hatchability associated with the egg EC_{10} of 12.5 mg Se/kg dw was statistically similar to the that of the control mean, and that selenium concentrations up to 12.5 mg Se/kg dw would not be expected lead to additional reductions in hatchability beyond natural conditions based on the limited available data.

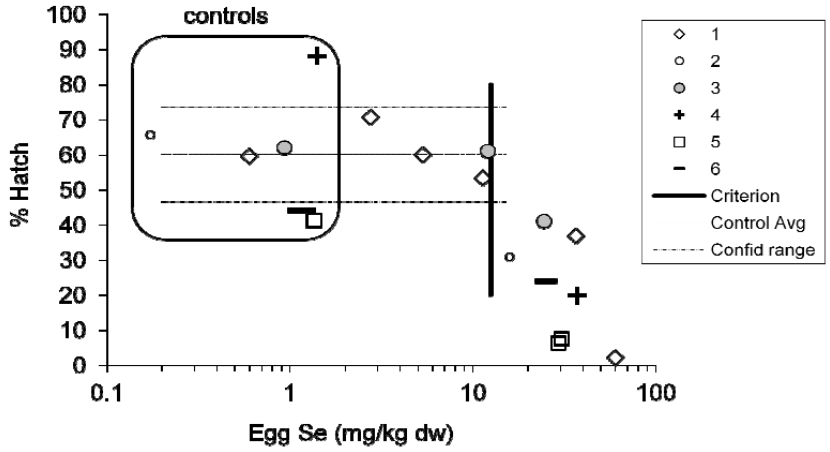


Figure 4-4. Mallard percent hatch v. egg concentration for six studies values.
Source is Figure 3 of U.S. EPA (2011a).
Raw data without normalization to control values.
1 = Heinz et al. (1989); 2 = Heinz et al. (1987); 3 = Stanley et al. (1996); 4 = Stanley et al. (1994); 5 = Heinz and Hoffman (1996); 6 = Heinz and Hoffman (1998).

For this current aquatic-dependent wildlife selenium criterion for the state of California, EPA again reanalyzed the mallard toxicity data to calculate an egg EC_{10} value for selenium from the dataset described above in Part 4.3 based on the three mallard toxicity studies that met EPA data quality guidelines and did not have outliers when combined into a single dataset (Heinz et al. 1987, 1989; Stanley et al. 1996). The selenium egg EC_{10} of 11.2 mg Se/kg dw is similar to those calculated in the 2011 EPA reanalysis of the mallard toxicity studies detailed above.

A notable difference is regarding the model used to calculate the EC_{10} value. As noted above, the EC_{10} values calculated in the 2011 EPA reanalysis of the mallard toxicity studies were calculated with the use of TRAP (U.S. EPA 2011b). However, TRAP was not designed to work with data pooled from multiple studies. Therefore, in this current reanalysis and derivation of the selenium aquatic-dependent wildlife criterion for the state of California EPA used a generalized linear model to calculate an egg EC_{10} of 11.2 mg Se/kg dw, which is believed to be a better statistical fit to the mallard toxicity data compared to earlier meta-analyses.

Mallard Biphasic Dose-Response Analysis Study – Beckon et al. (2008)

Beckon et al. (2008) applied biphasic modeling in their description of the biphasic dose-response behavior of selenium in biological samples. A biphasic model has both a rising and falling limb, and is applied to datasets where both low and high concentrations of a substance can negatively impact an organism. Beckon et al. (2008) calculated an egg EC_{10} of 7.7 mg Se/kg dw (Figure 4-5B) for reduced egg hatchability when applying a biphasic model to the mallard egg hatchability data reported by Heinz et al. (1989).[8] Beckon et al. (2008) fit these same data to two other models, a conventional log-logistic concentration-response model (with an egg EC_{10} of 28.6 mg Se/kg dw, Figure 4-5A), and a second model with a rising and falling limb, the Brain-Cousens (Brain and Cousens 1989) model (with an egg EC_{10} of 3.4 mg Se/kg dw, Figure 4-5C). Beckon et al. (2008) note that the Brain-Cousens model provides a poor fit, and that the conventional log-logistic model is inappropriate if the relationship between selenium and hatchability is biphasic.

8 The EC_{10} for the biphasic model is reported as 7.7 mg/kg in the text of Beckon et al. (2008) and as 7.3 in Figure 5 of Beckon et al. (2008).

EPA has previously evaluated the biphasic relationship between selenium and egg hatchability during its review of the selenium standard for the Great Salt Lake in Utah (CH2M Hill 2008; U.S. EPA 2011a),and concluded that the relationship cannot be modeled as biphasic. The six mallard toxcicity studies were not designed to study selenium deficiency and included no treatment that was intentionally selenium deficient. Consequently, implicit in fitting the biphasic model to these data is a belief that the control diet (i.e., the culture diet) was unintentionally deficient. If unintentionally deficient in selenium, there is little reason to suspect the deficiency was limited to selenium – several other nutrients may have been involved. This implies that the responses at all treatment levels could have been confounded by multiple stresses involving such deficiencies (U.S. EPA 2011a). In addition, control hatchability among the six mallard toxicity studies was high. If data from the six mallard studies are combined and fit to a biphasic model, and the EC_{10} for selenium excess and deficiency are calculated relative to the average control hatchability of the six studies, the EC_{10} for excess selenium would be 11.8 mg Se/kg dw, which is within 10% of the Ohlendorf (2003) EC_{10} of 12.5 mg Se/kg dw (U.S. EPA 2011a), and is similar to the current calculated EC10 of 11.2 mg Se /kg dw.

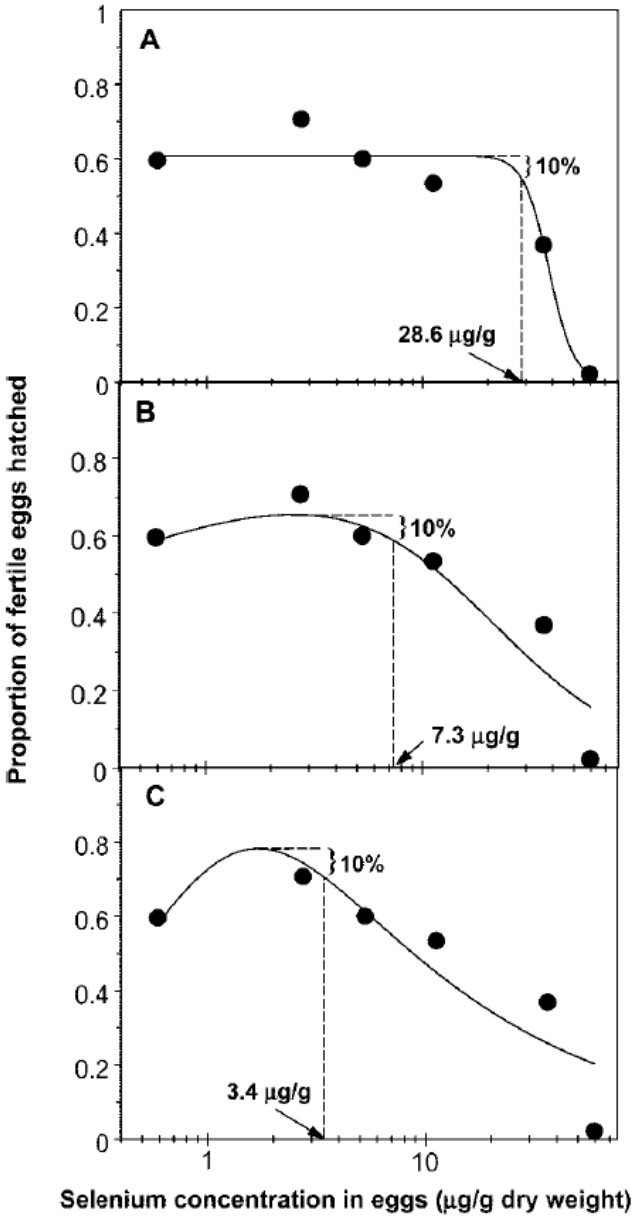


Figure 4-5. Data relating the hatchability of mallard eggs to the selenium concentration in the eggs from Heinz et al. 1989.

Source is Figure 5 from Beckon et al. (2008). Confidence intervals surrounding the EC_{10}s were not included in the source document. Curves fitted by least-squares nonlinear regression are (A) the log-logistic model, (B) a log-logistic^2 model, and (C) the Brain-Cousens model. The estimated egg EC_{10} based on the log-logistic model is more than eight times higher (less protective) than the estimate based on the Brain-Cousens model, while log-logistic^2 model yields an intermediate estimate.

Table 4-2. Previously Calculated and Current Selenium EC_{10} values for Mallard Hatchability.

Common Name	Scientific Name	Toxicological Endpoint[a]	Mean Egg Se Effect Threshold (mg Se/kg egg dw)	Reference
mallard	*Anas platyrhynchos*	EC_{10}s for post-hatch survival based on control normalized results of five laboratory studies, using various curve shapes	12 - 15	Adams et al. (2003)
mallard	*Anas platyrhynchos*	EC_{10} for egg hatchability based on control normalized results of six laboratory studies with mallards, using logistic regression analysis	12.5 (95% CI = 6.4 - 16.5)	Ohlendorf (2003)
mallard	*Anas platyrhynchos*	EC_{10} for egg hatchability based on results of six laboratory studies with mallards, using TRAP	12.3	U.S. EPA (2011a)
mallard	*Anas platyrhynchos*	EC_{10} for egg hatchability based on results of four laboratory studies with mallards, using TRAP	12.7	U.S. EPA (2011a)
mallard	*Anas platyrhynchos*	EC_{10} for egg hatchability based on control normalized results of six laboratory studies with mallards, using TRAP	9.7	U.S. EPA (2011a)
mallard	*Anas platyrhynchos*	EC_{10} for egg hatchability based on results of Heinz et al. (1989), assuming hormetic effects; reanalysis using biphasic model regression	7.7	Beckon et al. (2008)

Common Name	Scientific Name	Toxicological Endpoint[a]	Mean Egg Se Effect Threshold (mg Se/kg egg dw)	Reference
mallard	*Anas platyrhynchos*	EC_{10} for egg hatchability based on results of three mallard studies (Heinz et al. 1987, 1989; Stanley et al. 1996) using logistic regression analysis. This model serves as the basis for the egg tissue criterion element.	11.2	Part 4.3 of this Current Draft Document

[a] An effect concentration (EC) can be specified at different levels of effect and for different endpoints. ECs are the concentrations of selenium that adversely affect a certain percentage of the test organisms, i.e., an EC_{10} level affects 10% of the test organisms.

TRAP is the Toxicity Relationship Analysis Program U.S. EPA (2011b).

4.5 Chronic Egg Selenium Criterion Element Concentration

Table 4-2 shows the effect concentrations obtained from maternal transfer reproductive toxicity studies conducted with mallards. Mallard toxicity studies form the basis for the most reliable bird thresholds to date. Based on an analysis described above in Part 4.3 of three mallard toxicity studies meeting EPA's data quality guidelines (summarized in Part 4.2), a selenium egg EC_{10} of 11.2 mg Se/kg dw was derived for the most sensitive bird species studied and was based on the most sensitive endpoint (hatchability) measured. EPA is proposing a mallard egg EC_{10} of 11.2 mg Se/kg dw as an appropriately conservative aquatic-dependent wildlife criterion for protecting aquatic-dependent birds. As selenium concentrations appear to vary little within a single clutch and thus are not influenced by laying sequence (DeVink et al. 2008; Heinz et al. 1987, 1989; Weech et al. 2012), a sampling effort to measure egg selenium concentrations would not be dependent on egg laying sequence to reduce differences caused by intra-clutch variability. As discussed in Part 2.7, an EC_{10} was determined to be an appropriate effect concentration for tissue-based criteria given the nature of exposure and effects for this bioaccumulative chemical.

In Part 5 of this TSD, EPA translated the mallard egg EC_{10} value of 11.2 mg Se/kg dw to a selenium water column concentration based on the diets of a number of bird species to provide an example translation from bird egg to water that is comparable to the previously derived 2016

national aquatic life selenium criterion. In this analysis, EPA found that the translated selenium water column concentration for aquatic-dependent wildlife is approximately equal to the 2016 national aquatic life selenium water column criterion element of 1.5 µg/L fir lentic and 3.1 µg/L for lotic systems..

4.6 Summary of Selenium Toxicology Studies Used Qualitatively in the Criterion Derivation

Several studies were identified as either not meeting EPA's data quality guidelines for inclusion in the criterion calculations or would not support the derivation of an EC_{10}. However, these studies showed similar effects and ranges of toxicity to the studies presented in Part 4.2.1 above and demonstrate that mallard is the most sensitive species to selenium exposure. To provide additional evidence of the observed toxicity and effects of selenium, including the relative sensitivity of the bird species studied compared to mallards, these studies are presented below, divided into those with reproductive effects and non-reproductive effects and grouped by order. NOEC and LOEC values are provided in several of the following studies as representative effect concentrations for comparison to the EC_{10} value calculated for mallards. The NOEC/LOEC values were not used in any quantitative analysis toward the determination of the final chronic value for aquatic-dependent birds. Summary tables for the qualitative reproductive and non-reproductive studies described below are included in Appendix A.

4.6.1 Reproductive Studies Used Qualitatively in the Criterion Derivation

Anseriformes (Ducks, Geese, and Swans)

Hoffman and Heinz (1988) primarily described deformity endpoints for mallards that were measured, but not reported, in two separate studies (Heinz et al. 1987, 1989) described in Part 4.2.1 above. Both studies were conducted at the Patuxent Environmental Science Center, Laurel, Maryland, where breeding pairs of mallards were exposed to a commercial feed diet supplemented with different chemical forms of selenium (sodium selenite and seleno-DL-methionine). Heinz et al. (1987) divided mallard breeding pairs into six groups: one control group of eleven pairs, ten pairs fed 1, 5, 10, or 25 mg/kg selenium as sodium selenite, and five pairs fed 10 mg/kg selenium as seleno-DL-methionine. Corresponding selenium concentrations in eggs by group were 0.17, 0.10, 0.60, 1.77, 4.3, and 15.3 mg Se/kg dw. Heinz et al. (1989) divided mallard breeding pairs into six groups: one control group of thirty-five pairs, fifteen pairs

fed 1, 2, 4, or 8 mg/kg selenium as seleno-DL-methionine, and ten pairs fed 16 mg/kg selenium as seleno-DL-methionine. Corresponding selenium concentrations in eggs by group were 0.60, 2.77, 5.33, 11.3, 36.7, and 60.0 mg Se/kg dw. In Heinz et al. (1987), the percentage of abnormal embryos and day one to seven-day embryo mortality was significantly higher in the 10 mg/kg sodium selenite treatment relative to controls. Abnormal embryos included all individuals with any physical malformations, as well as edema and stunted growth, and are considered more sensitive than deformity endpoints. No significant differences in malformed embryos were observed for any of the sodium selenite treatments, but percent malformations were significantly higher for the 10 mg/kg selenium as seleno-DL-methionine treatment. Embryo mortality was significant when measured for all eggs per treatment, but not for eggs per nest per treatment. In Heinz et al. (1989), the percentage of malformed embryos was significantly higher in the 8 mg/kg seleno-DL-methionine treatment relative to controls. The authors concluded that comparable dietary concentrations of seleno-DL-methionine were more toxic than sodium selenite, most likely because seleno-DL-methionine is more readily incorporated into tissues and transferred to offspring.

Heinz and Fitzgerald (1993a) exposed forty breeding pairs of mallards to 15 mg Se/kg ww (16.7 mg/kg dw) dietary selenium, as selenomethionine, for twenty-one weeks through winter until soon after the first females began laying eggs, at which time dosing ceased. Controls consisted of twenty breeding pairs. The study was designed to determine the length of time needed for recovery from effects after selenium exposure ended. Selenium concentrations in eggs laid in the control treatment group ranged from 0.4-0.6 mg Se/kg ww. After roughly a week of receiving the selenium treated diet, eggs laid in the selenium treatment group ranged from 10-20 mg Se/kg ww. Treatment group females had statistically significant lower body weight at pairing and took longer to lay the first egg after pairing. The hatching success of the first eggs laid by selenium treated females was statistically significantly reduced (44% after one week off selenium diet and 50% after two weeks off selenium diet) compared to controls (70.5% throughout experiment) and four of these early eggs contained deformed embryos. Selenium concentrations in the series of eggs subsequently laid decreased over time following the end of the exposure period. Two weeks after selenium treatments ceased reproductive success in the selenium treatment group returned to levels comparable to controls. The authors concluded that for birds migrating from contaminated to uncontaminated areas, reproductive performance would

return to control levels within about two weeks after leaving the contaminated site. For this study, based on the single treatment level and its effect on delaying onset of egg-laying, the LOEC would be 15 mg Se/kg ww or 16.7 mg Se/kg dw in diet.

Heinz and Hoffman (1996) fed ten breeding pairs of adult mallards a control diet, and fed fifteen breeding pairs diets containing 10 mg/kg selenium as either seleno-DL-methionine, seleno-L-methionine or selenized yeast, respectively for approximately fourteen days. The average selenium concentrations in the eighth egg of each clutch were 0.41, 9.2, 8.9, 6.6 mg Se/kg ww for the control, seleno-DL-methionine, seleno-L-methionine, and selenized yeast treatments, respectively. No effects on adult mallards were observed. Several endpoints showed significant differences between the selenium treatments and the control including percent hatch of fertile eggs and the number of six-day old ducklings produced per hen. For instance, hatching of fertile eggs was significantly lower for females in both selenomethionine treatments (7.6% and 6.4% for seleno-DL-methionine and seleno-L-methionine, respectively) compared to controls (41.3%). Also, the number of six-day old ducklings produced per female was significantly lower for mallards fed seleno-DL-methionine (0.47 ducklings/female) and seleno-L-methionine (0.13 ducklings/female) compared to controls (6.10 ducklings/female). However, no significant differences in reproductive endpoints were observed between the two forms of selenomethionine. As noted above, the data from this study were not used in the derivation of the bird egg criterion as test validity requirements in EPA's Ecological Effects Test Guidelines for Avian Reproduction Tests (U.S. EPA 2012) states that control hatchability should be greater than 52% in mallard toxicity studies and the authors of this study report that the control hatchability was 41.3% (Heinz and Hoffman 1996). The results of this study do provide qualitative support of the bird egg criterion in that the percent hatch of fertile eggs for the two selenium treatments fall within the dose response curve for egg selenium (see Part 4.3).

In a separate study, Heinz and Hoffman (1998) fed adult mallard breeding pairs a control diet or diets containing 10 mg/kg selenium, 10 mg/kg mercury, or 10 mg/kg mercury plus 10 mg/kg selenium, respectively, for fifty-six to seventy days. Average selenium concentrations measured in the eleventh egg from each clutch was 0.35 mg Se/kg ww for controls, and 7.6, 0.39, 9.3 mg Se/kg ww for each of the three treatment groups listed above, respectively. No effects were observed on adults in the selenium treatment group of 10 mg/kg selenium in diet with 7.6 mg Se /kg ww in the eleventh egg. Also, no significant differences in the number of

days between eggs laid, percentage of eggs laid outside the nest box, whole egg weight, egg-shell thickness, or fertility of eggs were observed among the treatments. The combination of 10 mg/kg mercury and 10 mg/kg selenium in the diet had greater toxic effects on hatching success and survival of ducklings (1.4% and 0.2%, respectively) compared to diets containing either mercury (11.3% and 1.1%, respectively) or selenium (24.0% and 2.8%, respectively) alone. The percent hatchability and survival of ducklings in the 10 mg/kg selenium only diet was low (24% and 2.8%, respectively), but was not significantly different than the control (44.2% and 7.6%, respectively). Similar to Heinz and Hoffman (1996), the low control percent hatchability did not meet the test validity for mallards (U.S. EPA 2012) as the percent hatch of fertile eggs in the control was 44.2%, so these data were not used in the derivation of the bird egg criterion element. However, the hatchability data do provide qualitative support of the wildlife criterion (see Part 4.3).

Stanley et al. (1994) examined the independent and interactive effects of dietary selenium and arsenic on adult mallard breeding pairs at the Patuxent Environmental Science Center, Laurel, Maryland. Birds received a commercial diet spiked with one of eight dietary treatments. One of two dietary concentrations of selenium, a control diet and a 10 mg Se/kg diet (as selenomethionine) were crossed with four arsenic dietary concentrations: control, 25, 100, and 400 mg As/kg, respectively in a 4 x 2 factorial design. Measured selenium concentrations were 0.35 mg Se/kg dw in the control diet and 6.5 mg Se/kg dw in the selenium amended diet. Birds were fed treated diets for four weeks before pairing, and diets were maintained throughout the study (115-124 days). The eighth egg from every clutch was measured for selenium and arsenic concentrations. Eggs were incubated by hens, hatchlings were placed on the same diet as their parents. Adult and hatchling weights and survival were measured. At the end of the study, selenium and arsenic was measured in adult and hatchling tissues. No effects on adult weight or survival were observed when breeding pairs were fed selenium treated diets. Alternatively, decreased hatching success was observed in the 10 mg/kg selenium treatment group (8.5%) compared to the control group (91.4%). The occurrence of embryo deformities and duckling mortality was high in the selenium only treatment group (57.5% and 90%, respectively) compared to controls (0% and 17.5%, respectively). The independent effects of arsenic were less pronounced than those of selenium. Hatching success decreased from 91.4% to 74.5%, duckling mortality increased from 17.5% to 56.7%, and embryo deformities were similar (0% and 2.9%)

between the control and the 400 mg As/kg treatment level. The co-occurrence of arsenic mitigated the effects of selenium on hatchling success at 400 mg As/kg treatment level (59.7%), but effects on embryo deformities and duckling mortality were minor (0% and 63.8% respectively).

Pelecaniformes (Pelicans, Herons, Ibises, and Allies)

Smith et al. (1988) exposed ten breeding pairs of black-crowned night-herons (*Nycticorax nycticorax*) to seleno-DL-methionine at dietary concentrations of 0 (control), 10, and 30 mg Se/kg ww (9% moisture in diet) each over a period of ninety-two days. Average selenium concentrations measured in eggs were 0.56, 3.3 and 9.2 mg Se/kg fresh wet weight (fww for the control, 10 and 30 mg/kg ww dietary treatments, respectively). All groups lost weight during the test, but male and female herons fed the 30 mg Se/kg diet lost more weight than herons fed the 10 mg Se/kg or control diets. The authors attribute this to a possible aversion to the selenium-treated diets or perhaps illness caused by the selenium treatment. None of the herons died or showed signs of selenium toxicosis. Hatching success of fertile eggs laid by the 10 mg Se/kg diet group (43.9%) did not differ significantly from controls (32.2%). Nor did they show soft tissue, external, or skeletal deformities, although three day-old hatchlings in the 10 mg Se/kg ww dietary treatment group had statistically significantly ($P \leq 0.05$) shorter femur (15.1 mm) and radius-ulna lengths (10.6 mm) compared to controls (15.7 and 11.3 mm, respectively). Hematology of hatchlings appeared unaffected by selenium treatment; however, liver concentrations of malondialdehyde were higher in hatchlings from the 10 mg Se/kg ww selenium group (26.2 nmol/g) compared to controls (18.3 nmol/g). Because only two pairs of herons on the 30 mg Se/kg ww diet produced eggs, the authors did not emphasize these results. The authors, experienced with mallard studies described in this document, observed none of the mallard teratogenic effects that had occurred at the equivalent 10 mg Se/kg ww diet, such as hydrocephaly, bill and eye defects, and malformations of the legs, feet, and toes. Based on the absence of such effects, and the absence of a reduction in hatching success, the authors conclude that black-crowned night-herons are less sensitive to selenium toxicity than mallards (Smith et al. 1988). Egg concentrations of the 10 mg Se/kg ww diet group (yielding no effects) averaged 3.3 mg Se/kg fresh wet weight (n = 5; range 2.7-3.6 mg Se/kg fw). Given the absence of effects, a

threshold cannot be ascertained, but assuming 82.4% moisture content (Sotherland and Rahn 1987) the dry-weight NOEC would be 18.75 mg Se/kg dw egg.

Strigiformes (Owls)

Wiemeyer and Hoffman (1996) administered selenium in the form of seleno-DL-methionine to the diets of adult Eastern screech owls through the breeding season at the Patuxent Environmental Science Center, Laurel, Maryland. Adults were divided into three groups of breeding pairs, and received either a control (<0.21-0.34 mg Se/kg dw), a low (average 8.81 mg Se/kg dw), or a high (average 30.0 mg Se/kg dw) selenium diet. Adults were monitored for changes in weight and survival. Hatchability, growth, and liver enzyme levels were measured. Adult weights at the end of the study were statistically significantly lower in the high dietary treatment than the control and low dietary treatment. Egg selenium concentrations in control, low, and high dietary treatments averaged 0.26 mg Se/kg ww egg, 2.57 mg Se/kg ww egg, and 7.44 mg Se/kg ww egg, respectively. No nestlings survived to five days in the high selenium treatment; however, nestling survival and average body mass were similar between the control (2.4 five-day old nestlings per pair and 47.3 g, respectively) and low selenium treatment (3.3 five-day old nestlings per pair and 46.2 g, respectively). Statistically significant differences between nestlings in the control and low selenium treatments were observed for several liver enzymes (and in femur lengths (20.0 and 18.6 mm in controls and low selenium treatment group, respectively), but not among other measured bone lengths. For the conventional endpoints of nestling survival and adult weight, the NOEC was 8.81 mg Se/kg dw diet or 2.57 mg Se/kg ww egg, and the LOEC was 30.0 mg Se/kg dw diet or 7.44 mg Se/kg ww egg.

Charadriiformes (Plovers, Sandpipers, and Allies)

Hoffman et al. (2002) collected American avocet and black-necked stilt eggs from three sites with varying levels of selenium and hatched them in the laboratory. Fifteen, twenty-six, and seventeen avocet eggs were collected from Tulare Lake Drainage District–north Kings County (TLDD-N, water 2.5 µg/L Se), TLDD–south Kern and Kings Counties (TLDD-S, water 8.6 µg/L Se) and Westfarmers Kern County (WF, water 190 µg/L Se), respectively. Sixteen, twenty-two and seventeen stilt eggs were collected from these same respective locations. Geometric mean egg selenium concentrations in dry weight for avocets were 3.3 mg Se/kg (TLDD-N), 6.7 mg

Se/kg (TLDD-S) and 31.4 mg Se/kg (WF). Geometric mean egg selenium concentrations (dw) for the stilt eggs were 2.3 mg Se/kg (TLDD-N), 8.4 mg Se/kg (TLDD-S), and 20.5 mg Se/kg (WF). No meaningful effects were observed in the stilts, which had an overall lower selenium exposure compared to the avocets. There were no significant reductions in hatching success or malformations in the avocets, which had comparatively higher exposures (31.4 mg Se/kg dw egg at WF). There was, however, a small (7%) but significant reduction in chick weight (without yolk sac) for avocets at the high exposure relative to the reference. The NOEC and LOEC for this avocet chick weight endpoint were 6.7 mg Se/kg dw egg and 31.4 mg Se/kg dw egg, respectively. This study was not recommended for quantitative use because of the lack of effects in hatching success and malformations, the relatively small (7%) difference in yolk sac-free chick weights, and the large difference in concentrations between the moderately high and high exposure sites.

Harding et al. (2005) investigated the effects of selenium on spotted sandpipers (*Actitis macularia*) in areas of elevated selenium stream concentrations and in reference areas in the Elk River watershed of British Columbia. The average spotted sandpiper egg selenium concentration was 7.3 mg Se/kg dw in the exposed areas compared to 3.8 mg Se/kg dw in the reference areas. Fledglings per nest was 3.0 (standard error = 0.2, n = 27) in the exposed areas and 3.5 (standard error = 0.13, n = 27) in the reference areas. The authors note that despite the slightly reduced hatchability in sandpipers, overall productivity was higher than regional averages. In addition, no teratogenic effects were detected in any embryos or juveniles (nestlings) observed. Among the exposed sites, the degree of variation in concentrations and fledglings per nest suggest that averaging all of the exposed area observations might be problematic. The study does not provide a firm basis for estimating an effect threshold.

Black necked stilts (*Himantopus mexicanus*) are one of the few species with sufficient selenium exposure data from which to calculate an EC_{10} that can be compared to mallards. Adams et al. (2003) analyzed field data relating nest inviability in black-necked stilts to selenium exposure originally presented in Skorupa (1998b). A nest was considered inviable if at least one egg from a nest was inviable, making nest-wise, or clutch-wise, inviability a more sensitive endpoint than egg inviability. Skorupa (1998a) applied a weighted average to stilt nests with egg concentrations ranging from 4-9 mg/kg dw, and concluded that the upper bound of safe exposure for stilt eggs was around 6 mg/kg dw. Using a logistic model, Adams et al. (2003) calculated an

EC_{10} of 16.0 mg/kg dw egg for stilt nest inviability across the full range (approximately 2-75 mg/kg dw) of field egg concentrations presented in Skorupa (1998b). In addition, Adams et al. (2003) used an empirically calculated equation reported in Skorupa (1998b) to convert the probability of an inviable clutch to the probability of an inviable egg, so that the stilt field data would be more comparable to the mallard laboratory data. Adams et al. then grouped inviable egg data across the full range of selenium concentrations using a variety of binning schemes, and calculated egg-inviability EC_{10}s using hockey stick regression ranging between 20.9-31.0 mg/kg dw egg depending on the binning scheme. Based on these results, Adams et al. (2003) concluded that black necked stilts were less sensitive than mallards when similar endpoints were compared.

Passeriformes (Perching Birds)

Weech et al. (2012) examined selenium concentrations in invertebrates and bird eggs of several species, including tree swallows, in an environment receiving effluent from the Key Lake uranium mill in northern Saskatchewan, and in nearby reference areas. Hatching success and nestling health of tree swallows (*Tachycineta bicolor*) were also examined. Measured tree swallow egg selenium concentrations had a maximum of 13.3 mg Se/kg dw. The authors found no significant relationships between tree swallow egg selenium concentrations and hatchability or clutch size. There was also no difference in the growth of tree swallow nestlings among study areas. The study therefore does not provide a quantifiable threshold for effects. However, of interest for studies of mobile birds in environments of heterogeneous degrees of contamination, the authors noted high intra-clutch variability of selenium concentrations in both tree swallows and mallards. They suggest that in contaminated areas a single egg randomly removed for selenium measurement may not be representative of the concentrations in other eggs in the same nest observed for hatching success and nestling health.

Walls et al. (2015) studied tree swallow reproduction in Watts Bar Reservoir, Tennessee, in 2009-2010 following the spill of coal fly ash from the Kingston Fossil Plant in 2008. Tree swallows were exposed to ash-related contaminants via their diet of emergent aquatic insects, whose larval forms can accumulate constituents from submerged river sediments. Reproduction of 471 tree swallow nests was assessed over a two-year period. Egg concentrations of mercury and selenium in the impacted sites were somewhat elevated compared to reference sites. Average selenium concentrations measured in eggs ranged from 3.15-4.75 mg Se/kg dw egg among six

impacted sites across two years and 2.79-3.04 mg Se/kg dw across the two years at the reference site. Hatching success at ash-impacted sites (average of 87.4%) was statistically significantly lower reference sites (98.5%), but female fledglings produced per nesting female (2.10 and 2.22 for ash-impacted and reference sites, respectively) were not significantly different likely due to larger clutch sizes in the impacted colonies. Even for hatching success, the authors indicate that no combination of twenty-six potential contaminants measured (including selenium) in the eggs was predictive in a multiple regression analysis. Therefore, the study does not provide a basis for establishing an egg concentration threshold for effects.

Harding (2008) evaluated the effects of selenium on the reproductive success in red-winged blackbirds (*Agelaius phoeniceus*) at a coal mining site in southeastern British Columbia, Canada. Nests were monitored at reference sites and sites with elevated selenium for productivity, hatching success, egg failure, egg size and health, mortality, glutathione peroxidase, and malformations. Mean egg selenium across sites ranged from 2.96 to 21.7 mg Se/kg dw with concentrations in individual eggs as high as 40 mg Se/kg dw. The only effect observed to be related to selenium was hatchability; a quadratic model found a significant relationship between hatchability and egg selenium ($P < 0.001$, $n = 116$). The authors noted that the point of the downward inflection of the reverse U-shaped curve indicated adverse effects on hatchability at approximately 22 mg Se/kg dw egg. Because of the amount of scatter in the hatchability data, this value is considered qualitative rather than quantitative support of the aquatic-dependent wildlife criterion.

Ratti et al. (2006) collected reproductive data on 298 nests (from 152 reference and 146 mining sites) of American robin (*Turdus migratorius*) and 325 nests (from 166 reference and 159 mining sites) of red-winged blackbird (*Agelaius phoeniceus*) in Idaho. Twelve reproductive endpoints were measured, including nest success, clutch size, hatching success, fledging success, egg weight, and neonate weight. Average egg selenium concentrations were somewhat higher at the mining sites (4.48 mg Se/kg dw and 7.18 mg Se/kg dw in robin and blackbird, respectively) compared to the reference sites (3.17 mg Se/kg dw and 2.73 mg Se/kg dw in robin and blackbird, respectively). However, they did not often exceed concentrations that might have been expected to cause effects (none of the robin eggs exceeded 10 mg Se/kg dw; 13% of blackbird eggs exceeded 10 mg Se/kg dw). The authors did not observe any reductions in reproductive success. With no effects observed, the species LOECs are deemed greater than the mining site reported

average concentrations: >4.48 mg Se/kg dw egg for American robin and >7.18 mg Se/kg dw egg for red-winged blackbird.

4.6.2 Non-Reproductive Studies Used Qualitatively in the Criterion Derivation

Anseriformes (Ducks, Geese, and Swans)

The effects of dietary selenium concentrations as selenomethionine and sodium selenite on newly hatched mallard ducklings were examined by Heinz et al. (1988) at the Patuxent Wildlife Research Center in Laurel, MD. One-day old ducklings (n = 40) were assigned to one of ten treatments, and fed commercial starter mash containing 0, 10, 20, 40, or 80 mg Se/kg in the chemical form of either sodium selenite or selenomethionine for six weeks. Mortality, weight, and food consumption were monitored daily throughout the study. Food consumption decreased significantly in the 20, 40, and 80 mg/kg sodium selenite treatments by week one, and in the same selenomethionine treatments by week three. Duckling weights were reduced significantly in the 40 and 80 mg/kg sodium selenite treatments by week one, and in the same selenomethionine treatments by week two. Significant mortality was observed in the 80 mg Se/kg treatments for both selenium forms by week one. Mortality decreased significantly in the 40 mg/kg sodium selenite treatments by week two, and in the same selenomethionine treatments by week three. After six weeks, mortality was 97.5% in the 80 mg/kg sodium selenite treatment and 100% in the 80 mg/kg selenomethionine treatment. Six-week mortality was 25% in the 40 mg/kg sodium selenite treatment and 12.5% in the 40 mg/kg selenomethionine treatment. Selenium concentrations in livers among surviving ducklings reached an asymptote of 10 mg/kg among the sodium selenite treatments, but continued to increasingly bioaccumulate with concentration levels among the selenomethionine treatments. The dietary LOEC of 40 mg Se/kg observed for both growth and mortality endpoints in this study was higher than the range of dietary LOEC values (7 to 16 mg Se/kg) determined for egg hatchability (Heinz et al. 1989; Stanley et al. 1994, 1996). This finding supports the use of egg hatchability in maternal transfer studies as a sensitive toxicity endpoint that will be protective of birds.

Hoffman et al. (1992) examined the independent and interactive effects of dietary selenium and protein levels following a 3 x 3 factorial design, where three levels of dietary selenium as selenomethionine (control, 15 mg Se/kg, and 60 mg Se/kg) were crossed with three levels of dietary protein (11% - low, 22% - adequate, and 44% - high), and fed to one-day old

mallard ducklings for 28 days. A separate 2 x 3 factorial design was conducted using the same three levels of dietary selenium crossed with the control and low protein diets described above, where all treatments received supplemental dietary methionine (0.42% in the control diet and 0.21% in the low protein diet). The study was conducted at the Patuxent Environmental Science Center, Laurel, Maryland. Reduced 28-day weights were observed in the 15 mg Se/kg high protein treatment, and in the 60 mg Se/kg control protein treatment. No ducklings receiving 60 mg Se/kg and a low protein diet survived to 28 days. Reduced 28-day tarsal lengths and survival were observed in both the 60 mg Se/kg low protein and high protein treatments. There were no statistically significant independent effects of supplemental methionine, although for the 22% protein diet, survival in the 60 mg Se/kg treatment with the methionine supplement was slightly higher than the 60 mg Se/kg treatment with no methionine supplement.

Heinz (1993) acclimated ten adult male mallards to either zero (control) or 15 mg Se/kg as selenomethionine in a nearly dry diet (10% moisture content) for twenty-one weeks. There were no effects at either dose. After this acclimation period, all birds received the control diet for an additional 12 weeks. After this period of no exposure, the birds received either zero or 100 mg/kg selenomethionine for 5 weeks in their diet. The acclimation period was found not to influence mortality (14-15%) or weight reduction (39-41%) during the 5-week 100 mg Se/kg exposure. From the acclimation period results, it can be concluded that the NOEC is greater than 15 mg Se/kg in diet.

Heinz and Fitzgerald (1993b) exposed ten adult male mallards to dietary selenium concentrations of 10, 20, 40, and 80 mg Se/kg ww in a commercial diet, corresponding to 11.3, 22.6, 45.2, and 90.4 mg Se/kg dw (in addition to the control), for sixteen weeks over the winter. Mortality was monitored for an additional sixteen weeks after the exposure ended. No mortality was observed at 11.3 mg Se/kg dw diet, 25% was observed at 22.6 mg Se/kg dw diet, and 95-100% was observed at 45.2-90.4 mg Se/kg dw diet. The dietary dry weight NOEC (0% mortality) is 11.3 mg Se/kg dw and the LOEC (25% mortality) is 22.6 mg Se/kg dw. The data are too sparse to confidently estimate an EC_{10}, but they do suggest a steep concentration-response slope, with a dietary EC_{10} of approximately 19 mg Se/kg dw. These results indicate that reduction in overwintering survival of adult mallards begins at dietary concentrations higher than those yielding reductions in mallard egg hatchability.

Albers et al. (1996) fed one-year old male mallards a mash diet supplemented with 0, 10, 20, 40, 80 mg Se/kg ww as seleno-DL-methionine. Each treatment consisted of twenty-one ducks that were fed the selenium-spiked diets for sixteen weeks in outdoor pens. All the ducks died in the highest dietary treatment (80 mg Se/kg ww), with no significant mortality observed in any other treatment. The most sensitive effect observed in the test was the number of molts completed by the end of the sixteen- week treatment period. The number of molts over the sixteen-week period in the control, 10, 20, 40, 80 mg Se/kg ww dietary treatments were 21, 17, 19, 5, and 0, respectively. The 40 and 80 mg Se/kg ww treatments were significantly reduced relative to the control. The NOEC, LOEC and MATC for this test were determined as 20, 40 and 28.3 mg Se/kg ww dietary selenium, respectively, based on the number of molts endpoint. These dietary concentrations of selenium are more than double those in which egg hatchability effects were observed in mallards.

Groups of twelve flightling male mallards were exposed to 0, 10, 25, and 60 mg Se/kg ww (25% moisture) dietary selenium as seleno-L-methionine by O'Toole and Raisbeck (1997). Birds ate little of the 60 mg Se/kg ww diet and became emaciated. Birds on the 25 mg Se/kg ww diet ate approximately 25% less than birds on the control and 10 mg Se/kg ww diet, but body-weight reductions were statistically significant only intermittently, mostly during the first half of the test. Alopecia (baldness) was observed at 25 mg Se/kg ww but not in the control, 10 mg Se/kg ww, or 60 mg Se/kg ww groups. The dietary NOEC is 10 mg Se/kg ww or 13.3 mg Se/kg dw, and the dietary LOEC is 25 mg Se/kg ww or 33.3 mg Se/kg dw for the food consumption endpoint. However, reduction of risk by avoidance of selenium contaminated food is not thought to occur in real-world situations (U.S. EPA 2016a). If this study's dosing is thought to have produced an unpalatable diet, then it might not be usable for estimating effect thresholds.

DeVink et al. (2008) fed breeding pairs of two-year old lesser scaup (*Aythya affinis*) environmentally relevant doses (at nominal concentrations of <1, 7.5, and 15 mg Se/kg dw) of dietary selenium for thirty days. Seleno-L-methionine was added to commercial feed at measured selenium dry weight concentrations of 0.65 mg Se/kg dw (control), 7.7 mg Se/kg dw, and 14.9 mg Se/kg dw. There were no effects from selenium on adult survival or the number of hens laying eggs. The study had a secondary focus of measuring the decrease of selenium in eggs after the exposure period ended. Egg selenium concentrations decreased from approximately 33 mg Se/kg dw in the high dietary treatment (on the final day of the 30-day exposure) to

approximately 5 mg Se/kg dw in eggs collected 20 days after the selenium supplemented diet ended. A similar rapid decrease in egg selenium occurred in the 7.7 mg Se/kg diet. Eggs collected at the end of the 30-day exposure contained approximately 28 mg Se/kg dw; eggs collected 20 days after the selenium treatment stopped contained approximately 3 mg Se/kg dw. No selenium effect levels for chronic effects analysis were determined for this study.

Brady et al. (2013) exposed lesser scaup (*Aythya affinis*) to background/control (0.8 mg Se/kg dw), moderate (8.1 mg Se/kg dw) and high (20.7 mg Se/kg dw) levels of dietary selenium as seleno-L-methionine. Fifty-four wild-strain, captive ducks (twenty-eight females and twenty-six males) were fed the dietary treatments in pens for twenty-three weeks. The ducks in the high dietary treatment had significantly lower lipids after ten weeks; however, this difference was not observed after twenty-three weeks of exposure. After the twenty-three week exposure, there were no survival effects, selenium-related oxidative stress, or cell-mediated immunity, although immuno-stimulatory effects on antibody production were observed. No selenium effect levels for chronic effects analysis were determined for this study.

Falconiformes (Falcons and Caracaras)

Yamamoto and Santolo (2000) exposed groups of American kestrels to measured dietary selenium concentrations of 0.63, 6.3, and 12 mg Se/kg dw for a period of seventy-seven days. The control group consisted of ten male-female pairs. The treatment groups consisted of fifteen male-female pairs. Observations of the health of the male birds began at the end of exposure and continued for 197 days (after the 77-day exposure). The authors excluded the female birds from the analysis because their weights were too variable. The authors did not report body weights at the beginning of exposure. If it could be assumed that the groups began the exposure with equal weights, then relative to the control slight average reductions in total body weight were observed at the end of the seventy-seven-day exposure period (that is, the beginning of the observation period): 2.9% reduction at 6.3 mg Se/kg dw and 6.6% reduction at 12 mg Se/kg dw. By the end of the 197-day observation period, differences were less; average weights in the 6.3 mg Se/kg dw group were 2.2% greater than controls, and in the 12 mg Se/kg dw group were 3.9% less than controls. Within-group variability yielded considerable overlap between groups. Most of the body weight differences were in fat rather than lean tissue (measured non-invasively by total

body electrical conductivity). Overall, the effect of selenium on total body weight was less than 10% and does not provide a basis for estimation of a threshold.

Part 5 TRANSLATING THE BIRD EGG AND FISH TISSUE CRITERION ELEMENTS TO A WATER-COLUMN ELEMENT USING THE PERFORMANCE-BASED APPROACH

5.1 Purpose

This chapter outlines the details of the mechanistic modeling method that can be used to derive a site-specific chronic water-column selenium criterion element. The mechanistic modeling method is one of two performance based approach methods that can be followed, with the other being the bioaccumulation factor (BAF) method. This chapter also summarizes the translation of the fish tissue criterion element to a national water column element in the 2016 selenium aquatic life criterion (U.S. EPA 2016a) and discusses the translation of the bird tissue criterion element to a water column element that is comparable to EPA's national 2016 selenium aquatic life criterion following a similar approach.

The mechanistic modeling method includes deriving and applying an equation to translate the fish tissue selenium concentration and bird egg selenium concentration to water column selenium concentrations that are protective of aquatic life and aquatic-dependent wildlife, respectively. Part 5.5 discusses the translation of the fish (Part 5.5.1) and bird (Part 5.5.2) tissue criterion elements to both lentic and lotic water column elements on a national basis. The fish tissue to water column translation is a summary of EPA's 2016 national selenium aquatic life criterion. Data used in Part 5.5 were obtained from a nationwide search and were used to derive lentic and lotic chronic water column elements for the national 2016 selenium criterion; therefore these data were not considered site-specific. The water elements derived herein are provided as examples of how the mechanistic modeling method can be used to translate the tissue-based elements to a site-specific water value using the performance-based approach.

5.2 Translation from Tissue Concentration to Water Column Concentration Using the Mechanistic Model

As part of the effort to develop EPA's 2016 national aquatic life criterion for selenium (U.S. EPA 2016a), EPA worked with USGS to derive a translation equation that utilizes a

mechanistic model of bioaccumulation previously published in peer-reviewed scientific literature (Connolly 1985; Luoma and Fisher 1997; Luoma and Rainbow 2005; Luoma et. al. 1992; Presser 2013; Presser and Luoma 2006, 2010; Schlekat et al. 2002; Thomann 1981; Wang 2001; Wang et. al. 1996). This model quantifies bioaccumulation in animal tissues by assuming net bioaccumulation is a balance between assimilation efficiency from diet, ingestion rate, rate of direct uptake in dissolved forms, loss rate, and growth rate. The equation uses species-specific food web models, species-specific bioaccumulation parameters (conversion factor (*CF*) and trophic transfer factor (*TTF*)), and a site-specific enrichment factor (*EF*) to calculate a site-specific water column concentration element from the fish egg-ovary and bird egg criterion elements. For more details on the model and the simplification of the model used here, please see Section 3.2.1 of EPA's 2016 aquatic life criterion (U.S. EPA 2016a). The general model is described by (Equation 5-1).

$$C_{water} = \frac{C_{bird\ egg/fish\ egg}}{TTF^{composite} \times EF \times CF}$$

(Equation 5-1)

Where:

C_{water}	=	Selenium concentration (i.e., criterion) in the water column (µg/L)
$C_{bird\ egg/fish\ egg}$	=	Selenium concentration (i.e., criterion) in the eggs of birds (mg/kg) or the eggs or ovaries of fish (mg/kg)
$TTF^{composite}$	=	Composite trophic transfer factor. The overall *TTF*, or level of selenium bioaccumulation from the base of the food chain (i.e., *TL2*) to the tissues of the target species. *TTFs* are defined as concentration in consumer species divided by concentration in food. Composite *TTFs* take into consideration individual *TTFs* from all levels of the food web.
EF	=	Enrichment factor. The concentration of selenium in particulate matter (algae, detritus, sediment) at the base of the food chain divided by the selenium concentration in water collected at the same time and place (L/g)
CF	=	Conversion factor. Whole body to egg-ovary conversion factor (dimensionless ratio) [Used to convert fish egg-ovary to fish whole body. Not used for birds, which is translated fom bird egg value.]]

(Equation 5-1) describes an ecosystem-dependent relationship between concentration of selenium in the eggs and ovaries of fish or the eggs of birds with the concentration of selenium in the water column. This approach explicitly recognizes the sequential transfer of selenium between environmental compartments (water, particulate material, invertebrate tissue, fish tissue, eggs and/or ovary tissue) by incorporating quantitative expressions of selenium transfer from one compartment to the other. *TTF*s and *CF*s are species specific because they are influenced by the physiology of the animal (Presser and Luoma 2010). *EF*s are site specific because of the influence of the local hydrologic environment (Presser and Luoma 2010). Because this approach uses food web modeling along with species-specific *TTF* and *CF* parameters to quantify most of the transfer between compartments, the only field measurements needed to relate selenium in egg-ovary of fish or egg of birds to water are measurements from the water column and particulate material sufficient to calculate *EFs*.

5.3 Equation Parameters

Empirical or laboratory data related to selenium bioaccumulation in aquatic organisms are needed to derive the equation parameters *EF*, *TTF*, and *CF*. EPA obtained these data by

searching published literature using EPA's public ECOTOX database and other publication databases. The studies used here are the same as those used in U.S. EPA (2016a) with the addition of studies that included data on birds. EPA used this collection of selenium measurements to calculate *EF* values and to develop species-specific *TTF* and *CF* values in an unbiased and systematic manner. A more detailed description of how EPA calculated *EFs* is described in EPA's 2016 Aquatic Life Criterion (U.S. EPA 2016a). How EPA calculated *TTFs* and *CFs* as they related to aquatic life is described in detail in Appendix B of EPA's 2016 Aquatic Life Criterion (U.S. EPA 2016a).

5.3.1 Derivation of Trophic Transfer Factor (TTF) Values

The parameter $TTF^{composite}$ (trophic transfer factor) in (Equation 5-1) quantitatively represents all dietary pathways of selenium exposure for a particular fish or bird species within an aquatic system. The parameter is derived from species-specific *TTF* values representing the food web characteristics of the aquatic system and the proportion of species consumed. It is possible to differentiate bioaccumulative potential for different predator species and food webs by modeling different exposure scenarios. For example, where a fish or bird species of interest is a trophic level 4 predator that primarily consumes trophic level 3 fish, the term $TTF^{composite}$ can be represented as the product of all *TTF* parameters that includes the additional trophic level given as:

$$TTF^{composite} = TTF^{TL4} \times TTF^{TL3} \times TTF^{TL2}$$

(Equation 5-2)

where:

TTF^{TL2}	=	the trophic transfer factor of the trophic level 2 species
TTF^{TL3}	=	the trophic transfer factor of the trophic level 3 species
TTF^{TL4}	=	the trophic transfer factor of the trophic level 4 species
$TTF^{composite}$	=	the product of all the trophic transfer factors

Similarly, the consumption of more than one species of organism at the same trophic level can also be modeled by expressing the *TTF* at a particular trophic level as the weighted average of the *TTF*s of all species consumed given as:

$$\overline{TTF}^{TLx} = \sum_{i} \left(TTF_i^{TLx} \times W_i \right)$$

(Equation 5-3)

where:

TTF_i^{TLx} = the trophic transfer factor of the ith species at a particular trophic level

W_i = the proportion of the ith species consumed

These concepts can be used to formulate an expression of $TTF^{composite}$ to model selenium bioaccumulation in ecosystems with different consumer species and food webs. Figure 5-1 describes four example food web scenarios and the formulation of $TTF^{composite}$ to model selenium bioaccumulation in each of them. The parameter $TTF^{composite}$ quantitatively represents all dietary pathways of selenium exposure for a particular fish or bird species within an aquatic system.

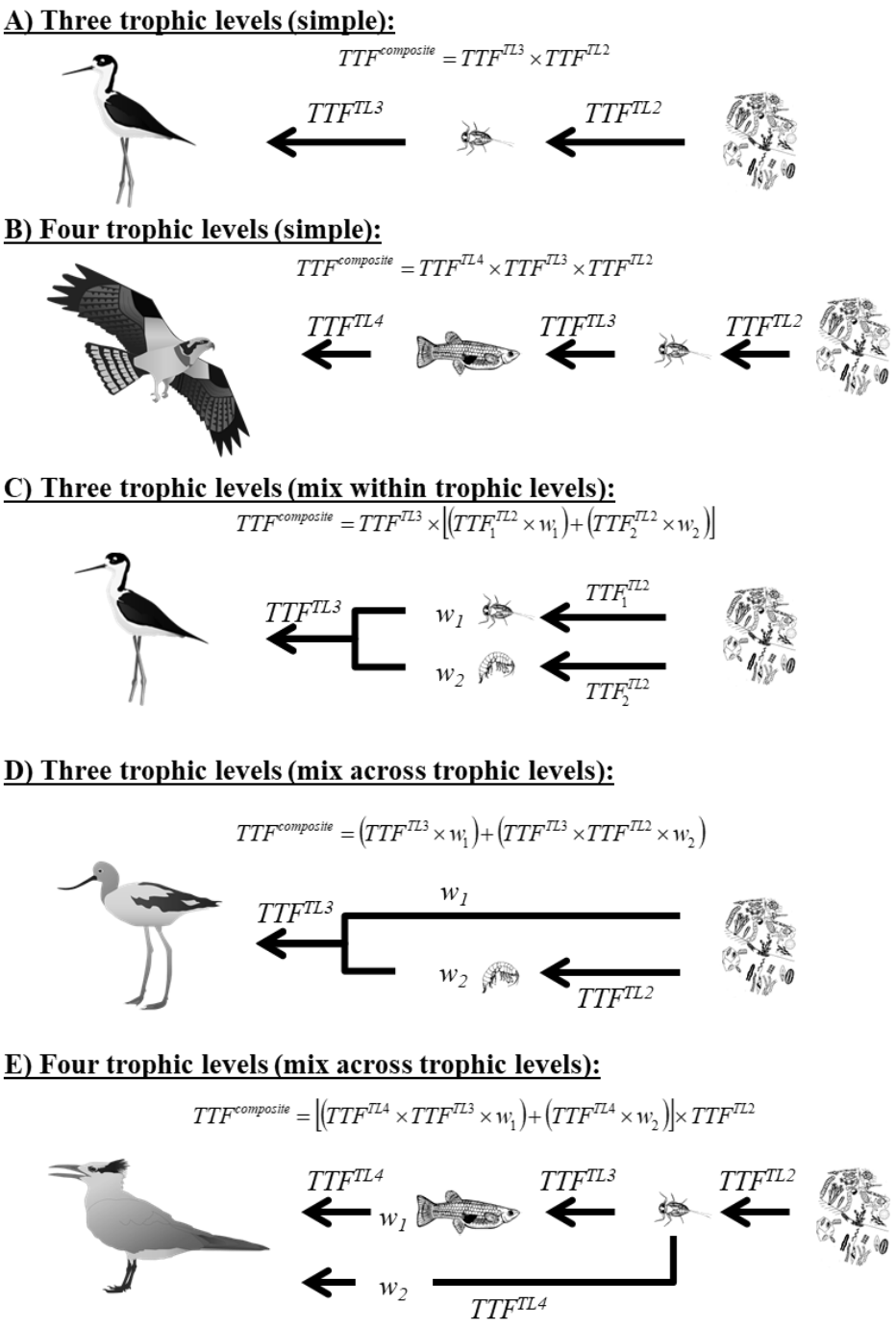


Figure 5-1. Example aquatic system scenarios and the derivation of the equation parameter $TTF^{composite}$.

Example equations shown here are scenario-specific combinations of (Equation 5-2) and (Equation 5-3). [Black-necked stilt and American avocet by Tracey Saxby and Catherine Ward. Osprey by Jane Hawkey. Royal tern by Tracey Saxby. All bird images used with permission from the Integration and Application Network, University of Maryland Center for Environmental Science (http://ian.umces.edu/imagelibrary/)]

The previously derived *TTF*s for invertebrates and fish that are used and summarized in this TSD are described in detail in U.S. EPA (2016a). The following paragraphs generally describe EPA's approach for *TTF*s for all taxonomic groups, including the newly derived *TTF*s for birds.

EPA derived *TTF* values for taxonomic groups of invertebrates, fish, and birds by either using physiological coefficients found in the literature, or by evaluation of the empirical relationship between matched pairs of selenium measurements in organisms and the food they consumed. The latter are empirical measurements of selenium from field studies. For more on physiological coefficients please see Section 3.2.2.1 in EPA's 2016 Aquatic Life Criterion (U.S. EPA 2016a). EPA searched its collection of available selenium measurements and identified measurements taken from aquatic organisms or aquatic-dependent birds. For each measurement from an aquatic organism or bird, EPA searched for additional measurements from other aquatic organisms or particulate material that was collected from the same aquatic site and of a type deemed likely to be ingested as a food source or in conjunction with feeding activity (i.e., lower trophic level). If multiple lower trophic level measurements were matched to an aquatic organism or bird measurement, the median of the lower trophic level measurements was calculated. Each pair of measurements, one taken from a consumer organism and the other representing the diet of the consumer organism, was designated as a matched pair. For every consumer organism-diet organism pair, *TTF*s were calculated using matched measurements from all available sites and studies. EPA limited particulate data used to calculate invertebrate *TTFs* from field data to those aquatic sites with at least two particulate selenium measurements paired with invertebrate selenium measurements, and only used sediment measurements if there was at least one measurement from algae or detritus. If selenium concentrations from more than one category of particulate material (algae, detritus, or sediment) were available, EPA used the median selenium concentration of the available categories as the particulate concentration for that site.

Because selenium is transferred to aquatic animals primarily through aquatic food webs, the observable concentration of selenium in different environmental compartments may vary over time. In Section 3.2.2.1 of U.S. EPA (2016a), an analysis was conducted that suggested that the relationship between selenium concentrations in particulate material and invertebrate tissue and between invertebrate tissue and fish tissue is insensitive to relative collection time within a

one-year period. These results also suggested that selenium becomes relatively persistent in the aquatic ecosystem once dissolved selenium transforms to particulate selenium and becomes bioavailable. Based on these analyses, EPA concluded that selenium measurements from samples collected at the same aquatic site within one year of each other are acceptable to use as matched pairs of measurements from the aquatic sites. For the purposes of matching aquatic-dependent bird egg measurements to lower trophic level measurements, EPA used the same rule established in U.S. EPA (2016a). EPA concluded that use of this rule would be appropriate after conducting an analysis to compare *TTFs* calculated from breeding season data (defined here as April through July) to *TTFs* calculated from all available data for the migratory species. Because many of the bird species analyzed eat invertebrates, and invertebrate sampling collections are typically conducted outside of the breeding season time frames, many of the datasets for the breeding season alone did not produce statistically significant regressions. For those species where enough data were available during the breeding season to produce statistically significant results, the resulting *TTFs* were very close to the *TTFs* calculated from all data for the same bird species. Note that EPA chose a relative collection period of one year based on data taken from many different aquatic sites. Individual aquatic sites may have selenium loads and/or bioaccumulation characteristics that require different relative collection time criteria to accurately characterize selenium relationships.

In Section 3.2.2.1 of EPA's 2016 Selenium Aquatic Life Criterion, EPA evaluated the advantages and disadvantages of using either the median ratio of a distribution of matched pairs of data, or the slopes of linear regression models to derive species-specific *TTF* values for field data and ultimately settled on a hybrid approach (U.S. EPA 2016a). Briefly, the approach includes designating the median of the ratio of matched pairs of selenium measurements as the *TTF* value, but only if ordinary least squares (OLS) linear regression of those data resulted in a significant ($P \leq 0.05$) fit and positive regression coefficient. Requiring a significant positive OLS linear regression coefficient confirms the relationship between selenium in organisms and the food they ingest is adequately represented by the available data. Using the median of the individual ratios provides an estimate of central tendency for that relationship that is less sensitive to potential bias from measurements taken from aquatic systems with very high or very low selenium concentrations. EPA used this same approach for eight new bird *TTF*s derived in this technical support document (TSD).

EPA used previously calculated *TTF* values for 13 invertebrate species and 32 fish species, and newly calculated *TTF* values for eight bird species that live in, or are dependent on, freshwater aquatic environments in North America. The final *TTF* values are listed in Table 5-1, Table 5-2, and Table 5-3, respectively. The invertebrate and fish data used to derive these previously calculated *TTF* values are provided in Appendix B of U.S. EPA (2016a). The presence of physiological coefficients for a taxon in Table 5-1 and Table 5-2 indicates that the *TTF* values were calculated using those parameters based on laboratory studies. The absence of physiological coefficients for a taxon indicates that EPA derived the *TTF* value using field data. If a *TTF* value could be calculated from both physiological coefficients and field data, EPA used the *TTF* value calculated from the substantially larger number of field measurements to minimize statistical uncertainty.

Table 5-1. EPA-Derived Trophic Transfer Factor (*TTF*) Values for Freshwater Aquatic Invertebrates (U.S. EPA 2016a).

Common name	Scientific name	AE^a	IR^a	$k_e{}^a$	*TTF*
Crustaceans					
Amphipod	*Hyalella azteca*	-	-	-	1.22
Copepod	Copepoda	0.520	0.420	0.155	1.41
Crayfish	*Astacidae*	-	-	-	1.46
water flea	*Daphnia magna*	0.406	0.210	0.116	0.74
Insects					
Dragonfly	*Anisoptera*	-	-	-	1.97
Damselfly	*Coenagrionidae*	-	-	-	2.88
Mayfly	*Neocloeon triangulifer*	-	-	-	2.38
Midge	*Chironimidae*	-	-	-	1.90
water boatman	*Corixidae*	-	-	-	1.48
Mollusks					
asian clam[b]	*Corbicula fluminea*	0.550	0.050	0.006	4.58
zebra mussel	*Dreissena polymorpha*	0.260	0.400	0.026	4.00
Annelids					
Blackworm	*Lumbriculus variegatus*	0.165	0.067	0.009	1.29
Other					
Zooplankton		-	-	-	1.89

[a] AE = assimilation efficiency (proportion). IR = ingestion rate (g/g-day). k_e = loss rate (/day).
[b] Not to be confused with *Potamocorbula amurensis*

Table 5-2. EPA-Derived Trophic Transfer Factor (*TTF*) Values for Freshwater Fish (U.S. EPA 2016a).

Common name	Scientific name	AE^a	IR^a	$k_e{}^a$	*TTF*
Cypriniformes					
blacknose dace	*Rhinichthys atratulus*	-	-	-	0.71
bluehead sucker	*Catostomus discobolus*	-	-	-	1.04
longnose sucker	*Catostomus catostomus*	-	-	-	0.90
white sucker	*Catostomus commersonii*	-	-	-	1.11
flannelmouth sucker	*Catostomus latipinnis*	-	-	-	0.98
common carp	*Cyprinus carpio*	-	-	-	1.20
creek chub	*Semotilus atromaculatus*	-	-	-	1.06
fathead minnow	*Pimephales promelas*	-	-	-	1.57
red shiner	*Cyprinella lutrensis*	-	-	-	1.31
redside shiner	*Richardsonius balteatus*	-	-	-	1.08
sand shiner	*Notropis stramineus*	-	-	-	1.56
Cyprinodontiformes					
western mosquitofish	*Gambusia affinis*	-	-	-	1.21
northern plains killifish	*Fundulus kansae*	-	-	-	1.27

Common name	Scientific name	AE^a	IR^a	$k_e{}^a$	*TTF*
Esociformes					
northern pike	*Esox lucius*	-	-	-	1.78
Gasterosteiformes					
brook stickleback	*Culaea inconstans*	-	-	-	1.79
Perciformes					
black crappie	*Pomoxis nigromaculatus*	-	-	-	2.67
Bluegill	*Lepomis macrochirus*	-	-	-	1.03
green sunfish	*Lepomis cyanellus*	-	-	-	1.12
largemouth bass	*Micropterus salmoides*	-	-	-	1.39
smallmouth bass	*Micropterus dolomieu*	-	-	-	0.86
striped bass	*Morone saxatilis*	0.375	0.335	0.085	1.48
Walleye	*Sander vitreus*	-	-	-	1.60
yellow perch	*Perca flavescens*	-	-	-	1.42
Salmoniformes					
brook trout	*Salvelinus fontinalis*	-	-	-	0.88
brown trout	*Salmo trutta*	-	-	-	1.38
mountain whitefish	*Prosopium williamsoni*	-	-	-	1.38
cutthroat trout	*Oncorhynchus clarkii*	-	-	-	1.12
rainbow trout	*Oncorhynchus mykiss*	-	-	-	1.07
Scorpaeniformes					
mottled sculpin	*Cottus bairdi*	-	-	-	1.38
Sculpin	*Cottus sp.*	-	-	-	1.29
Siluriformes					
black bullhead	*Ameiurus melas*	-	-	-	0.85
channel catfish	*Ictalurus punctatus*	-	-	-	0.68

[a] AE = assimilation efficiency (proportion). IR = ingestion rate (g/g-day). k_e = loss rate (/day).

Table 5-3. EPA-Derived Trophic Transfer Factor (*TTF*) Values for Aquatic-Dependent Birds.

Common name	Scientific name	*TTF*
Non-Migratory		
American coot	*Fulica americana*	1.89
red winged blackbird	*Agelaius phoeniceus*	0.86
Migratory		
American avocet	*Recurvirostra americana*	1.44
cinnamon teal	*Anas cyanoptera*	1.79
eared grebe	*Podiceps nigricollis*	2.00
Gadwall	*Anas strepera*	1.78
pied billed grebe	*Podilymbus podiceps*	0.78
yellow headed blackbird	*Xanthocephalus xanthocephalus*	1.04

For fish species without sufficient data to directly calculate a *TTF* value, EPA estimated the *TTF* value by sequentially considering higher taxonomic classifications until one or more taxa for which a calculated *TTF* value was available matched the taxon being considered. If the lowest matching taxon was common to more than one species with a *TTF* value available, EPA used the median *TTF* from the matching species. For example, although data to directly calculate *TTF* for northern redbelly dace (*Chrosomus eos*) were not available, this species is in the family Cyprinidae, which also includes blacknose dace (*Rhinichthys atratulus*), common carp (*Cyprinus carpio*), creek chub (*Semotilus atromaculatus*), fathead minnow (*Pimephales promelas*), red shiner (*Cyprinella lutrensis*), redside shiner (*Richardsonius balteatus)*, and sand shiner (*Notropis stramineus*). Because Cyprinidae is the lowest taxonomic classification where *Chrosomus eos* matches a species with an available *TTF* value, the median of the blacknose dace, common carp, creek chub, fathead minnow, red shiner, redside shiner, and sand shiner *TTF* values was used as the *TTF* value for northern redbelly dace. The data and analyses used to calculate all *TTF* values including those estimated by taxonomic classification are provided in Table B-8 of Appendix B in U.S. EPA (2016a).

Empirical data for egg and diet pairs were not available for eight bird species that were identified as species of concern for California by U.S. FWS (American dipper, brown pelican, bald eagle, Ridgway's rail, Light-footed Ridgway's rail, Yuma Ridgway's rail, black rail, and least tern) (U.S. FWS 2017). Therefore, composite *TTFs* (see Part 5.4.2) were estimated for these species of concern from their species-specific dietary compositions and the application of empirically-derived egg to diet *TTFs* from surrogate species with similar diets. When possible these trophic level *TTFs* were applied from closely related surrogate species (in most cases within the same order). For more details on *TTF* derivation for each bird species, see Appendix B.

5.3.2 Derivation of Egg-Ovary to Whole Body Conversion Factor (CF) Values for Aquatic Life

The parameter *CF* (conversion factor) listed in (Equation 5-1 represents the species-specific partitioning of selenium as measured in the whole body and in egg-ovary tissue of fish. EPA derived species-specific *CF* (Table 5-4) values by applying the same method used to derive species-specific *TTF* values, using empirical measurements of selenium concentrations in different tissues of the same fish. To derive egg-ovary to whole body *CF* values, EPA defined matched pairs of selenium measurements from the whole body and from the eggs or ovaries

measured from the same individual fish or from matched composite samples. Egg-ovary concentration was defined as a measurement from either the eggs or the ovaries. If multiple measurements from both eggs and ovaries of the same individual or matched composite sample were available, the average value was used. As was the case with *TTF*s, *CF*s were calculated using matched tissue measurements from all available sites and studies for a given species.

EPA had sufficient egg-ovary and whole body selenium measurements to directly derive egg-ovary to whole body *CF* values for 13 species of fish. However, matched pairs of selenium measurements in eggs and/or ovaries and muscle (M) tissue, and matched pairs of selenium measurements in muscle and whole body were also available. To derive *CF* values for additional fish species, EPA used either the additional data or a taxonomic classification approach to estimate *CF*. EPA derived 13 *CF* values directly from matched pairs of egg-ovary and whole body selenium measurements and an additional seven *CF* values by multiplying EO/M and M/WB conversion factors. For more details on *CFs* for fish see Section 3.2.2.2 and Appendix B in U.S. EPA (2016a). For the process of translating the bird egg criterion to a water column concentration, *CFs* were not necessary.

Table 5-4. EPA-Derived Egg-Ovary to Whole Body Conversion Factor (*CF*) Values (U.S. EPA 2016a).

Common name	Scientific name	*CF*	Std. Dev.[a]
Acipenseriformes			
white sturgeon	*Acipenser transmontanus*	1.69	
Cypriniformes			
bluehead sucker	*Catostomus discobolus*	1.82	0.19
flannelmouth sucker	*Catostomus latipinnis*	1.41	0.20
white sucker	*Catostomus commersonii*	1.38	0.36
desert pupfish	*Cyprinodon macularius*	1.20	0.10
common carp	*Cyprinus carpio*	1.92	0.49
roundtail chub	*Gila robusta*	2.07	0.29
fathead minnow	*Pimephales promelas*	1.40	0.75
creek chub	*Semotilus atromaculatus*	1.99	1.00
razorback sucker	*Xyrauchen texanus*	3.11	
Esociformes			
northern pike	*Esox lucius*	2.39	
Perciformes			
Bluegill	*Lepomis macrochirus*	2.13	0.68
green sunfish	*Lepomis cyanellus*	1.45	0.23

Common name	Scientific name	*CF*	Std. Dev.[a]
smallmouth bass	*Micropterus dolomieu*	1.42	0.19
Salmoniformes			
brook trout	*Salvelinus fontinalis*	1.38	
Dolly Varden	*Salvelinus malma*	1.61	
brown trout	*Salmo trutta*	1.45	1.81[b]
rainbow trout	*Oncorhynchus mykiss*	2.44	
cutthroat trout	*Oncorhynchus clarkii*	1.96	2.03[b]
mountain whitefish	*Prosopium williamsoni*	7.39	

[a] Standard deviation for *CF* values for those species that had egg-ovary and whole body selenium concentrations.

[b] The brown trout and cutthroat trout standard deviations for *CF* values of 1.81 and 2.03 are considerably higher than the other standard deviations in this table. The brown trout data were taken from two studies, Formation Environmental (2011) and Osmundson et al. (2007). *CF* values for three of the four fish samples from Osmundson et al. were four to six times greater than the median. Also, the Formation Environmental data consisted of samples collected from natural streams and samples collected from a fish hatchery. The *CF* values for the fish hatchery samples were four to seven times lower than the median value. Although collectively, the data set meets the criteria for including the brown trout *CF*, the *CF* values for Osmundson et al. and Formation Environmental hatchery samples may be anomalously high and low, respectively. Excluding these potentially anomalous data reduces the brown trout standard deviation to 0.47. The cutthroat trout *CF* values are from two sources (Formation Environmental 2012 and Hardy 2005). The reason for the higher variability in the cutthroat trout *CF* values is due to the relatively higher *CF* values in the hatchery fish from the Formation study. The standard deviation for cutthroat trout drops to 0.62 if the hatchery fish are excluded. See Appendix B of (U.S. EPA 2016a) for a presentation of the data for both species.

5.3.3 Calculation of Site-Specific Enrichment Factor (EF) Values

The most influential step in selenium bioaccumulation occurs at the base of aquatic food webs (Chapman et al. 2010). The parameter *EF* characterizes this step by quantifying the partitioning of selenium between the dissolved and particulate (algae, detritus, and sediment). *EF* can vary by at least two orders of magnitude across aquatic systems (Presser and Luoma 2010). The greatest reduction in uncertainty when translating a fish tissue or bird tissue concentration of selenium to a water column concentration using (Equation 5-1) is achieved when spatially and temporally coincident site-specific empirical observations of dissolved and particulate selenium of sufficient quality and quantity are used to accurately characterize the *EF*. Thus when deriving the 2016 national selenium aquatic life criterion, EPA only used aquatic sites with sufficient data to calculate a reasonably reliable *EF* value. To calculate site-specific *EF*s when translating the fish and bird tissue criterion elements to a water column element, the State will follow the methods under the performance-based approach (*Translation of Selenium Tissue Criterion Elements to Site-Specific Water Column Criterion Elements for California*).

To calculate the *EF* of aquatic systems for the 2016 national selenium aquatic life criterion, EPA searched its collection of selenium concentration measurements from field studies (see Section 2.7.8 of U.S. EPA 2016a for a description of data sources and acceptability criteria) and identified aquatic sites with measurements from both particulate material and the water column. EPA first identified all selenium measurements from algae, detritus, or sediment, and then searched for corresponding water column measurements from samples collected at the same aquatic site within one year of the particulate sample. If more than one water measurement was available for any given particulate measurement, the median was used. For each of these matched pairs of particulate and water measurements, EPA calculated the ratio of particulate concentration to water concentration. If more than one ratio for any given category of particulate material (algae, detritus, or sediment) was calculated at an aquatic site, EPA used the median ratio to characterize the relationship of that category of particulate material. The geometric mean of the algae, detritus, and sediment ratios was then used as the *EF*. Because there were at most only three possible values (one for algae, one for detritus, and one for sediment), EPA used the geometric mean to reduce the potential for one of the values to have excessive influence on the final site *EF* value.

The availability of selenium measurements from particulate material was limited. In addition, a majority of particulate measurements were from sediment samples with a significantly lower correlation to selenium in water ($r = 0.34$) compared to algae ($r = 0.68$; Fisher r-to-z transformation, $P<0.001$) and detritus ($r = 0.94$; Fisher r-to-z transformation, $P<0.001$). Therefore, to reduce uncertainty in estimating site-specific *EF* values, EPA limited its analysis to those aquatic sites with at least two particulate selenium measurements paired with corresponding water column measurements, and only used sediment measurements if there was at least one other measurement from either algae or detritus. Based on these requirements, *EF* values were calculated for 96 individual aquatic sites, and these calculated *EF* values were used to derive the 2016 national selenium aquatic life criterion.

5.4 Food Web Models

5.4.1 Aquatic Life

For the 65 aquatic sites where an *EF* value was calculated and where fish were sampled, EPA modeled the food webs for the fish species the studies indicated were present. Some of those studies provided information about the species and proportions of organisms ingested by

fish, either through direct analysis of stomach contents, or examination of the presence and prevalence of invertebrate species. For those studies, that site-specific information in the food web models was used. Most studies, however, did not provide site-specific food web information. In those cases, the food webs of fish species present were modeled using information about their typical diet and/or eating habits obtained from the NatureServe database (http://www.natureserve.org).

After EPA developed food web models, EPA identified the appropriate species-specific *TTF* values for each model and calculated $TTF^{composite}$. Although individual *TTF* values were derived for several different taxa of invertebrates and fish (Table 5-1 and Table 5-2), some of the food web models included one or more taxa for which no *TTF* value was available. EPA estimated *TTF* values for these taxa using the same taxonomic approach used to estimate egg-ovary to whole body, egg-ovary to muscle, and muscle to whole body conversion factors for taxa without sufficient data. In brief, for taxa with insufficient data to calculate a *TTF* value, EPA sequentially considered higher taxonomic classifications until one or more taxa for which a *TTF* value was available matched the taxon being considered. If the lowest matching taxon was common to more than one species with a *TTF* value available, EPA used the median *TTF* from the matching species. EPA parameterized food web models with *TTF*s and *EF*s to translate the egg-ovary criterion element to a set of water column concentrations in order to derive the water column concentration element of the selenium criterion. Details of these food web models are shown in Table B-8 of Appendix B in U.S. EPA (2016a).

5.4.2 Aquatic-Dependent Wildlife

EPA modeled the food webs for 16 bird species using species-specific dietary information to calculate composite *TTFs*. Eight of the bird species' composite *TTFs* were derived based on species-specific dietary compositions and an empirically-derived egg to diet *TTF*. The remaining 8 bird species' composite *TTFs* were estimated using their species-specific dietary compositions and the application of empirically-derived egg to diet *TTFs* from surrogate species that had similar diets and, when possible, were taxonomically related (within same order). Details regarding bird $TTF^{composite}$ calculations are included in Appendix B.

5.4.2.1 Species-Specific Composite *TTFs*

Composite *TTFs* were calculated for eight bird species in order to relate selenium concentrations in the bird eggs of those species to selenium in particulate matter at the base of

the food web. Particulate matter is defined here as algae, detritus, and sediment (U.S. EPA 2016a). Bird dietary compositions were modeled using information from species-specific descriptions within the Cornell Lab of Ornithology Birds of North America web site: (http://www.birds.cornell.edu/Page.aspx?pid=1478). The eight bird species included two non-migratory (or resident) species: American coot and red-winged blackbird; and four migratory species: cinnamon teal, eared grebe, pied-billed grebe, and yellow headed blackbird.

EPA first calculated bird egg to diet *TTFs* following the general procedure described for the calculation of *TTFs* in Part 5.3.1 above. Because six of the eight bird species consumed an omnivorous diet, the calculation procedure followed for fish was modified as follows. For bird species whose diet consisted of both plants and animals, information regarding species-specific dietary descriptions was used to calculate the relative proportions of the bird diet consisting of plants and animals. For every egg selenium measurement paired with additional selenium measurements from both aquatic invertebrates and aquatic algae and vascular plants, a weighted dietary selenium concentration was calculated. As with fish, paired data were required to be collected at the same site within a one-year period (see Part 5.3.1 for additional details). Also, following the approach used for fish, all paired invertebrate or primary producer species were included, and considered as surrogates for dietary species from that trophic level. When more than one paired potential diet item from the same trophic level was available, the median selenium concentration was used.

The relationship between egg selenium concentrations and selenium concentrations in modeled diet organisms were natural log transformed and evaluated using linear regression after removing outliers. If the slope of a set of matched pairs of selenium measurements was both positive and statistically significant ($P \leq 0.05$), then the relationship between selenium in the target bird species and the food it consumes is considered adequately represented by the available data. Paired data and regression results, as well as a more detailed description of the procedure used to determine outliers, can be found in Appendix B. For each set of paired data meeting the regression criteria, the ratio of each egg selenium measurement was divided by its corresponding paired dietary selenium measurement, and the species-specific *TTF* for that trophic level was calculated as the median ratio of all pairs of data.

Next, food webs were constructed by estimating the diet of each target bird species from the species-specific descriptions, and a final species-specific $TTF^{composite}$ was calculated using

(Equation 5-2) and (Equation 5-3). The TTF^{TL2} linking the invertebrates consumed by that bird species to the base of the food web is calculated by applying *TTFs* for invertebrate species or groups of species obtained from U.S. EPA (2016a) (Table 5-1 here) to the corresponding invertebrate taxa in the modeled bird species' diet. Table 5-5 lists $TTF^{composite}$ for the eight bird species for which *TTF*s could be calculated. Dietary information and calculations performed to calculate $TTF^{composite}$ for these species are listed in Appendix B.

Table 5-5. EPA-Derived Composite Selenium Trophic Transfer Factors ($TTF^{composite}$) for Aquatic-Dependent Birds.

Non-Migratory Species	$TTF^{composite}$
American coot	2.48
red-winged blackbird	1.67
Migratory Species	$TTF^{composite}$
American avocet	2.61
cinnamon teal	3.04
eared grebe	3.15
gadwall	2.24
pied-billed grebe	1.52
yellow-headed blackbird	1.93

5.4.2.2 Threatened and Endangered Species of Concern Composite *TTFs*

Empirical data for egg and diet pairs were not available for the following species of concern in California: American dipper, brown pelican, bald eagle, Ridgway's rail, light-footed Ridgway's rail, Yuma Ridgway's rail, black rail, and least tern. These species were identified as species of concern by U.S. FWS in the following report: "*Species at Risk from Selenium Exposure in California Inland Surface Waters, Enclosed Bays and Estuaries*" (U.S. FWS 2017). Species-specific dietary descriptions for these Threatened and Endangered (T&E) species were used to model the relative proportions of the bird diet consisting of plants and animals, and then paired selenium data from an appropriate surrogate species were weighted accordingly to calculate a species-specific (egg to diet) *TTF*. Composite *TTFs* were then calculated for these species of concern using their species-specific dietary composition and species-specific *TTF* derived from a surrogate species. The surrogate species selected was based on similarity in dietary composition and if possible taxonomic relatedness (within same order). For bird species

that consumed fish, the pied-billed grebe *TTF* was used as a surrogate, as pied-billed grebe is the only piscivore with sufficient data to calculate a *TTF*. Table 5-6 lists $TTF^{composite}$ values for the eight T&E species in California, with the surrogate species in parentheses. Specific calculations used to generate these $TTF^{composite}$ values are included in Appendix B.

Table 5-6. Composite Selenium Trophic Transfer Factors ($TTF^{composite}$) for Aquatic-Dependent Bird Species of Concern in California.

Surrogate species (from Table 5-3) used for *TTF*s in parentheses.

California Bird Species of Concern (surrogate species used)	Scientific name	$TTF^{composite}$
American dipper (average of red-winged blackbird and yellow-headed blackbird)	*Cinclus mexicanus*	2.11
brown pelican (pied-billed grebe)	*Pelecanus occidentalis*	1.83
bald eagle (pied-billed grebe)	*Haliaeetus leucocephalus*	1.69
Ridgway's rail (American coot[a])	*Rallus obsoletus*	3.16
light-footed Ridgway's rail (American coot[a])	*Rallus obsoletus levipes*	1.70
Yuma Ridgway's rail (American coot[a])	*Rallus obsoletus yumanensis*	1.33
black rail (American coot[a])	*Laterallus jamaicensis*	1.69
least tern (pied-billed grebe)	*Sternula antillarum*	1.83

[a] Species-specific *TTFs* calculated using American coot paired data weighted to account for species-specific plant vs. animal proportions (See Appendix B for details).

5.5 Deriving National Protective Water Column Concentrations for Lentic and Lotic Systems

5.5.1 Aquatic Life

To derive the water column element for the 2016 national selenium aquatic life criterion, EPA translated the egg-ovary criterion element to a distribution of water column concentration values for lentic and lotic aquatic systems. EPA used the *EF* values calculated for 96 aquatic sites, available information about the fish species present at those sites, and food web models of those fish species. Because translation of the egg-ovary criterion element is site- and species-specific, several studies identifying more than one species of fish could potentially provide more than one translated water column concentration (one translated water value for each species). EPA considered using all water column values for all species present to generate distributions of translated water column values from lentic and lotic aquatic sites. However, the number of reported fish species at aquatic sites with an *EF* value varied from one to six fish species. Furthermore, the studies providing data for 31 of the 96 sites with *EF* values do not provide

information on the species of fish that may have been present at the aquatic site. Because the number of fish species at an aquatic site was not consistently reported, and because the number of reported fish species does not necessarily indicate the number of species present at a site, EPA calculated one translated egg-ovary criterion element to water column value for each aquatic site with both an *EF* value and at least one reported fish species. When more than one species was reported at a site, EPA used the lowest translated water value for that site. Using this methodology, EPA translated the egg-ovary FCV into water column concentrations at 26 lentic and 39 lotic aquatic sites. EPA used these distributions of water concentration values translated from the egg-ovary criterion element to derive chronic water column criterion element values for lentic and lotic aquatic systems. Table 5-7 shows the model parameter values used to translate the egg-ovary criterion element to individual water concentrations for each site used in the 2016 national selenium aquatic life criterion, and Figure 5-2 shows the distribution of the translated values. For more information on how EPA classifies lotic (flowing waters) and lentic (standing waters) waters see Section *3.2.4 Classifying Categories of Aquatic Systems* in U.S. EPA (2016a). The translated water column values for each individual site in Table 5-7 were intended to be used as part of a distribution to derive a protective water column element value on a national basis for the 2016 national selenium aquatic life criterion. The State could consider these water column values or follow the methodology described in the performance-based approach to translate the fish tissue criterion elements into a protective water column criterion element value for a specific site under consideration.

Table 5-7. Data for the 65 Site Minimum Translations of the Fish Egg-Ovary Criterion Concentration Element to a Water Column Concentration (U.S. EPA 2016a).

Identification				Model Parameters			Translation
Reference	**Site**	**Species**	**Type**	***EF*[a]**	***CF*[b]**	***TTF*[composite-c]**	**C_{water}[d] (µg/L)**
Birkner 1978	East Allen Reservoir, Medicine Bow WY	Iowa darter	Lentic	2.31	1.45	2.87	1.57
Birkner 1978	Galett Lake, Laramie WY	Iowa darter	Lentic	0.88	1.45	2.87	4.15
Birkner 1978	Larimer Highway 9 Pond, Fort Collins CO	northern plains killifish	Lentic	1.70	1.20	2.44	3.04
Birkner 1978	Meeboer Lake, Laramie WY	northern plains killifish	Lentic	0.58	1.20	2.44	8.96
Birkner 1978	Miller's Lake, Wellington CO	Iowa darter	Lentic	2.37	1.45	2.87	1.53
Birkner 1978	Sweitzer Lake, Delta CO	fathead minnow	Lentic	0.87	1.40	2.78	4.45
Birkner 1978	Twin Buttes Reservoir, Laramie WY	Iowa darter	Lentic	1.21	1.45	2.87	3.01
Bowie et al. 1996	Hyco Reservoir	Bluegill	Lentic	2.35	2.13	2.00	1.51
Butler et al. 1993	Navajo Reservoir, Piedra River Arm, near La Boca	brown trout	Lentic	1.26	1.45	2.78	2.98
Butler et al. 1997	Large pond south of G Road, southern Mancos Valley	fathead minnow	Lentic	2.00	1.40	2.78	1.94
Butler et al. 1997	Pond downstream from site MNP2, southern Mancos Valley	smallmouth bass	Lentic	5.15	1.42	1.93	1.07
Butler et al. 1997	Pond on Woods Canyon at 15 Road	fathead minnow	Lentic	0.90	1.40	2.78	4.29
Grasso et al. 1995	Arapahoe Wetlands Pond	fathead minnow	Lentic	0.86	1.40	2.78	4.49
Lemly 1985	Badin Lake	red shiner	Lentic	12.48	1.95	2.27	0.27
Lemly 1985	Belews Lake	red shiner	Lentic	1.75	1.95	2.27	1.94
Lemly 1985	High Rock Lake	red shiner	Lentic	4.99	1.95	2.27	0.68
Muscatello and Janz 2009	Vulture Lake	northern pike	Lentic	1.01	2.39	4.02	1.56
Orr et al. 2012	Clode Pond 11	cutthroat trout	Lentic	0.71	1.96	2.29	4.70
Orr et al. 2012	Elk Lakes 14	cutthroat trout	Lentic	1.64	1.96	2.29	2.05
Orr et al. 2012	Fording River Oxbow 10	cutthroat trout	Lentic	1.34	1.96	2.29	2.50
Orr et al. 2012	Henretta Lake 27	cutthroat trout	Lentic	0.50	1.96	2.29	6.72
Saiki and Lowe 1987	Kesterson Pond 11	western mosquitofish	Lentic	0.51	1.20	2.37	10.52
Saiki and Lowe 1987	Kesterson Pond 2	western mosquitofish	Lentic	0.32	1.20	2.37	16.83

Identification				Model Parameters			Translation
Reference	**Site**	**Species**	**Type**	***EF*[a]**	***CF*[b]**	***TTF*[composite-c]**	**C_{water}[d] (µg/L)**
Saiki and Lowe 1987	Kesterson Pond 8	western mosquitofish	Lentic	0.60	1.20	2.37	8.84
Saiki and Lowe 1987	Volta Pond 26	western mosquitofish	Lentic	0.93	1.20	2.37	5.69
Stephens et al. 1988	Marsh 4720	common carp	Lentic	0.10	1.92	1.58	52.02
Butler et al. 1991	Uncompahgre River at Colona	rainbow trout	Lotic	0.63	2.44	2.33	4.21
Butler et al. 1993	Spring Cr. at La Boca	brown trout	Lotic	0.18	1.45	2.78	20.97
Butler et al. 1995	Hartman Draw near mouth, at Cortez	fathead minnow	Lotic	0.15	1.40	2.78	26.04
Butler et al. 1995	McElmo Cr. at Hwy. 160, near Cortez	fathead minnow	Lotic	0.90	1.40	2.78	4.32
Butler et al. 1995	McElmo Cr. downstream from Alkali Cyn.	fathead minnow	Lotic	0.37	1.40	2.78	10.57
Butler et al. 1995	McElmo Cr. downstream from Yellow Jacket Cyn.	red shiner	Lotic	0.12	1.95	2.27	28.34
Butler et al. 1995	McElmo Cr. upstream from Yellow Jacket Cyn.	red shiner	Lotic	0.10	1.95	2.27	35.60
Butler et al. 1995	Navajo Wash near Towaoc	speckled dace	Lotic	0.20	1.95	1.36	29.07
Butler et al. 1995	San Juan River at Four Comers	red shiner	Lotic	0.26	1.95	2.27	12.97
Butler et al. 1995	San Juan River at Mexican Hat Utah	common carp	Lotic	0.29	1.92	1.58	17.24
Butler et al. 1995	Woods Cyn. Near Yellow Jacket	fathead minnow	Lotic	0.40	1.40	2.78	9.60
Butler et al. 1997	Cahone Canyon at Highway 666	green sunfish	Lotic	0.20	1.45	2.29	23.22
Butler et al. 1997	Mud Creek at Highway 32, near Cortez	fathead minnow	Lotic	0.07	1.40	2.78	55.27
Casey 2005	Deerlick Creek	rainbow trout	Lotic	2.24	2.44	2.33	1.18
Casey 2005	Luscar Creek	rainbow trout	Lotic	0.33	2.44	2.33	8.14
Formation Environmental 2012	Crow Creek - 1A	brown trout	Lotic	0.80	1.45	2.96	4.42
Formation Environmental 2012	Crow Creek - 3A	brown trout	Lotic	0.81	1.45	2.97	4.37
Formation Environmental 2012	Crow Creek - CC150	brown trout	Lotic	1.04	1.45	2.91	3.44
Formation Environmental 2012	Crow Creek - CC350	brown trout	Lotic	1.16	1.45	2.97	3.02

Identification				Model Parameters			Translation
Reference	**Site**	**Species**	**Type**	***EF*[a]**	***CF*[b]**	***TTF*[composite-c]**	**C_{water}[d] (µg/L)**
Formation Environmental 2012	Crow Creek - CC75	brown trout	Lotic	1.19	1.45	2.87	3.07
Formation Environmental 2012	Deer Creek	brown trout	Lotic	1.55	1.45	3.00	2.25
Formation Environmental 2012	Hoopes Spring – HS	brown trout	Lotic	0.24	1.45	3.86	11.06
Formation Environmental 2012	Hoopes Spring - HS3	brown trout	Lotic	0.54	1.45	2.63	7.40
Formation Environmental 2012	Sage Creek - LSV2C	brown trout	Lotic	0.45	1.45	3.01	7.76
Formation Environmental 2012	Sage Creek - LSV4	brown trout	Lotic	0.69	1.45	2.88	5.22
Formation Environmental 2012	South Fork Tincup Cr.	brown trout	Lotic	1.32	1.45	3.05	2.58
Hamilton and Buhl 2004	Lower East Mill Creek	cutthroat trout	Lotic	1.32	1.96	2.29	2.55
McDonald and Strosher 1998	Elk R. above Cadorna Cr. (745)	mountain whitefish	Lotic	6.30	7.39	2.97	0.11
McDonald and Strosher 1998	Fording R. above Swift Cr. (746)	cutthroat trout	Lotic	0.23	1.96	2.29	14.91
Orr et al. 2012	Elk River 1	cutthroat trout	Lotic	0.55	1.96	2.29	6.14
Orr et al. 2012	Elk River 12	cutthroat trout	Lotic	2.67	1.96	2.29	1.26
Orr et al. 2012	Fording River 23	cutthroat trout	Lotic	0.21	1.96	2.29	16.20
Orr et al. 2012	Michel Creek 2	cutthroat trout	Lotic	0.28	1.96	2.29	11.85
Saiki and Lowe 1987	San Luis Drain	western mosquitofish	Lotic	0.36	1.20	2.37	14.81
Saiki and Lowe 1987	Volta Wasteway	western mosquitofish	Lotic	1.03	1.20	2.37	5.17
Saiki et al. 1993	Mud Slough at Gun Club Road	Bluegill	Lotic	1.37	2.13	1.47	3.53
Saiki et al. 1993	Salt Slough at the San Luis National Wildlife Refuge	Bluegill	Lotic	0.43	2.13	1.47	11.29
Saiki et al. 1993	San Joaquin R. above Hills Ferry Road	Bluegill	Lotic	0.36	2.13	1.47	13.50

Identification				Model Parameters			Translation
Reference	**Site**	**Species**	**Type**	***EF*[a]**	***CF*[b]**	***TTF*[composite-c]**	**C_{water}[d] (µg/L)**
Saiki et al. 1993	San Joaquin R. at Durham Ferry State Recreation Area	Bluegill	Lotic	0.75	2.13	1.47	6.46

a - Geometric mean of the median enrichments factors (*EF*) for all available food types (algae, detritus, and sediment). *EF* (L/g) = C_{food}/C_{water}.
b - Taxa-specific conversion whole body to egg-ovary conversion factor (*CF*; dimensionless ratio).
c - Composite trophic transfer factor ($TTF^{composite}$). Product of *TTF* values for all trophic levels.
d - Translated water selenium concentration corresponding to an egg-ovary criterion element of 15.1 mg Se/kg dw, calculated by (Equation 5-1).

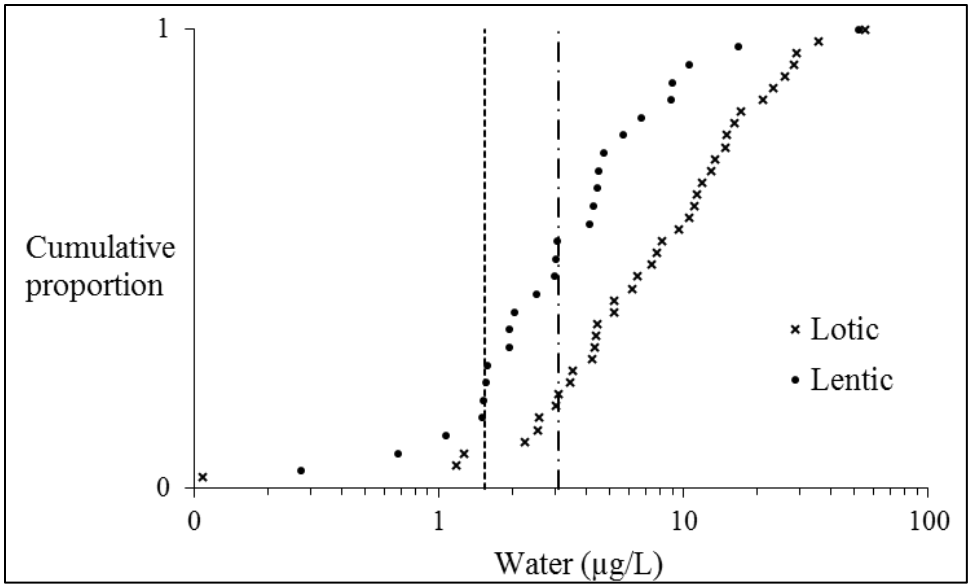


Figure 5-2. Probability distribution of the water column concentrations translated from the fish egg-ovary criterion element at 26 lentic and 39 lotic aquatic sites (U.S. EPA 2016a).
Dashed and dash-dot lines show the 20th percentiles of the lentic and lotic distributions, respectively.

In the 2016 national selenium aquatic life criterion, EPA selected the 20th percentile from the distribution of translated water column values of each category as the final national water column criterion element concentrations (3.1 µg/L for lotic waters and 1.5 µg/L for lentic waters) because the 20th percentile is consistent with past practice as it provides a high probability of protection for most aquatic systems in both lentic and lotic categories. Table 5-8 provides the 20th percentile of the water concentration values translated from the fish egg-ovary criterion element value. These values were calculated by applying the mechanistic modeling method on a national scale, and may be appropriate for California.

Table 5-8. Water Column Criterion Element Concentration Values Translated from the Fish Egg-Ovary Criterion Element in the 2016 National Selenium Aquatic Life Criterion (U.S. EPA 2016a).

	Lentic	Lotic
20th percentile (final 2016 EPA recommended water column criterion element protective of aquatic life)	**1.5 µg/L**	**3.1 µg/L**

As discussed in Section 2.2.2 of U.S. EPA (2016a), selenium bioaccumulation potential depends on several biogeochemical factors that characterize a particular aquatic system. Uncertainty in the translation of the egg-ovary criterion element to the water column element can be reduced by deriving a site-specific water column criterion element that uses site-specific selenium data and information on food web dynamics from a biological assessment of the aquatic system. The general considerations are provided in the performance-based approach (*Translation of Selenium Tissue Criterion Elements to Site-Specific Water Column Criterion Elements for California*) and in Appendix K of U.S. EPA (2016a).

5.5.2 Aquatic-Dependent Wildlife

To translate the bird tissue criterion elements into a water column concentration that is comparable to the 2016 national aquatic life criterion and to determine whether the U.S. EPA (2016a) national water column criterion element (Table 5-8) is also protective of aquatic-dependent wildlife, EPA translated the bird egg tissue element to a distribution of water column concentration values for the same lentic and lotic aquatic systems. To translate the bird egg tissue element, EPA utilized information from the same 65 aquatic sites shown in Table 5-7 and in U.S. EPA (2016a) in addition to the food web models of 16 bird species (see Parts 5.4.2.1 and 5.4.2.2) with a variety of diets, from plants and insects to fish. Using a similar methodology to the one described in Part 5.5.1, EPA translated the bird egg FCV into water column concentrations at 26 lentic and 39 lotic aquatic sites. At each site, EPA used (Equation 5-1) to translate from the bird egg criterion element of 11.2 mg Se/kg dw to a water column concentration using the *EF* for that site and the maximum $TTF^{composite}$ for the 16 modeled bird species. EPA chose to translate the bird egg element using the maximum $TTF^{composite}$ because it generates the most protective water column concentration that would sufficiently protect sensitive species in bioaccumulative food webs. This is consistent with the approach of selecting the most bioaccumlative food web for the fish species analysis in the 2016 final Aquatic Life Selenium Criterion for Freshwater (U.S. EPA 2016a). Table 5-9 shows the model parameter values used to translate the bird egg criterion element value to individual water concentrations using data for each site used in the 2016 national selenium aquatic life criterion, and Figure 5-3 shows the distribution of the translated water column values. The translated water column values for each individual site in Table 5-9 could be used as part of a distribution to calculate a protective water column element on a large geographic scale, paralleling the approach used in the 2016 national selenium aquatic life

criterion. The State could consider these values or follow the methodology described in the performance-based approach to translate the bird tissue criterion element into a protective water column criterion element value for a specific site under consideration.

Table 5-9. Data for the 65 Site Minimum Translations of the Bird Egg Criterion Concentration Element to a Water Column Concentration.

Sites and enrichment factors (*EF*) are those used to translate the fish egg-ovary criterion concentration element to water column concentrations (U.S. EPA 2016a). The $TTF^{composite}$ for Ridgway's rail was used for all sites, as it is the largest among the 16 bird species described in this document, resulting in the most protective water column concentrations.

Identification				Model Parameters		Translation
Reference	**Site**	**Species**	**Type**	EF^{a}	$TTF^{composite-b}$	$C_{water}{}^{c}$ (µg/L)
Birkner 1978	East Allen Reservoir, Medicine Bow WY	Ridgway's rail	Lentic	2.31	3.16	1.53
Birkner 1978	Galett Lake, Laramie WY	Ridgway's rail	Lentic	0.88	3.16	4.05
Birkner 1978	Larimer Highway 9 Pond, Fort Collins CO	Ridgway's rail	Lentic	1.70	3.16	2.08
Birkner 1978	Meeboer Lake, Laramie WY	Ridgway's rail	Lentic	0.58	3.16	6.14
Birkner 1978	Miller's Lake, Wellington CO	Ridgway's rail	Lentic	2.37	3.16	1.49
Birkner 1978	Sweitzer Lake, Delta CO	Ridgway's rail	Lentic	0.87	3.16	4.06
Birkner 1978	Twin Buttes Reservoir, Laramie WY	Ridgway's rail	Lentic	1.21	3.16	2.93
Bowie et al. 1996	Hyco Reservoir	Ridgway's rail	Lentic	2.35	3.16	1.51
Butler et al. 1993	Navajo Reservoir, Piedra River Arm, near La Boca	Ridgway's rail	Lentic	1.26	3.16	2.81
Butler et al. 1997	Large pond south of G Road, southern Mancos Valley	Ridgway's rail	Lentic	2.00	3.16	1.77
Butler et al. 1997	Pond downstream from site MNP2, southern Mancos Valley	Ridgway's rail	Lentic	5.15	3.16	0.69
Butler et al. 1997	Pond on Woods Canyon at 15 Road	Ridgway's rail	Lentic	0.90	3.16	3.92
Grasso et al. 1995	Arapahoe Wetlands Pond	Ridgway's rail	Lentic	0.86	3.16	4.10
Lemly 1985	Badin Lake	Ridgway's rail	Lentic	12.48	3.16	0.28
Lemly 1985	Belews Lake	Ridgway's rail	Lentic	1.75	3.16	2.02
Lemly 1985	High Rock Lake	Ridgway's rail	Lentic	4.99	3.16	0.71
Muscatello and Janz 2009	Vulture Lake	Ridgway's rail	Lentic	1.01	3.16	3.51
Orr et al. 2012	Clode Pond 11	Ridgway's rail	Lentic	0.71	3.16	4.96

Identification				Model Parameters		Translation
Reference	**Site**	**Species**	**Type**	***EF*[a]**	***TTF*[composite-b]**	**C_{water}[c] (µg/L)**
Orr et al. 2012	Elk Lakes 14	Ridgway's rail	Lentic	1.64	3.16	2.16
Orr et al. 2012	Fording River Oxbow 10	Ridgway's rail	Lentic	1.34	3.16	2.64
Orr et al. 2012	Henretta Lake 27	Ridgway's rail	Lentic	0.50	3.16	7.08
Saiki and Lowe 1987	Kesterson Pond 11	Ridgway's rail	Lentic	0.51	3.16	7.00
Saiki and Lowe 1987	Kesterson Pond 2	Ridgway's rail	Lentic	0.32	3.16	11.21
Saiki and Lowe 1987	Kesterson Pond 8	Ridgway's rail	Lentic	0.60	3.16	5.89
Saiki and Lowe 1987	Volta Pond 26	Ridgway's rail	Lentic	0.93	3.16	3.79
Stephens et al. 1988	Marsh 4720	Ridgway's rail	Lentic	0.10	3.16	37.00
Butler et al. 1991	Uncompahgre River at Colona	Ridgway's rail	Lotic	0.63	3.16	5.63
Butler et al. 1993	Spring Cr. at La Boca	Ridgway's rail	Lotic	0.18	3.16	19.81
Butler et al. 1995	Hartman Draw near mouth, at Cortez	Ridgway's rail	Lotic	0.15	3.16	23.76
Butler et al. 1995	McElmo Cr. at Hwy. 160, near Cortez	Ridgway's rail	Lotic	0.90	3.16	3.94
Butler et al. 1995	McElmo Cr. downstream from Alkali Cyn.	Ridgway's rail	Lotic	0.37	3.16	9.64
Butler et al. 1995	McElmo Cr. downstream from Yellow Jacket Cyn.	Ridgway's rail	Lotic	0.12	3.16	29.56
Butler et al. 1995	McElmo Cr. upstream from Yellow Jacket Cyn.	Ridgway's rail	Lotic	0.10	3.16	37.13
Butler et al. 1995	Navajo Wash near Towaoc	Ridgway's rail	Lotic	0.20	3.16	18.10
Butler et al. 1995	San Juan River at Four Comers	Ridgway's rail	Lotic	0.26	3.16	13.53
Butler et al. 1995	San Juan River at Mexican Hat Utah	Ridgway's rail	Lotic	0.29	3.16	12.26
Butler et al. 1995	Woods Cyn. Near Yellow Jacket	Ridgway's rail	Lotic	0.40	3.16	8.76
Butler et al. 1997	Cahone Canyon at Highway 666	Ridgway's rail	Lotic	0.20	3.16	18.15
Butler et al. 1997	Mud Creek at Highway 32, near Cortez	Ridgway's rail	Lotic	0.07	3.16	50.44
Casey 2005	Deerlick Creek	Ridgway's rail	Lotic	2.24	3.16	1.59
Casey 2005	Luscar Creek	Ridgway's rail	Lotic	0.33	3.16	10.89
Formation Environmental 2012	Crow Creek - 1A	Ridgway's rail	Lotic	0.80	3.16	4.44

Identification				Model Parameters		Translation
Reference	**Site**	**Species**	**Type**	***EF*[a]**	***TTF*[composite-b]**	**C_{water}[c] (µg/L)**
Formation Environmental 2012	Crow Creek - 3A	Ridgway's rail	Lotic	0.81	3.16	4.40
Formation Environmental 2012	Crow Creek - CC150	Ridgway's rail	Lotic	1.04	3.16	3.40
Formation Environmental 2012	Crow Creek - CC350	Ridgway's rail	Lotic	1.16	3.16	3.05
Formation Environmental 2012	Crow Creek - CC75	Ridgway's rail	Lotic	1.19	3.16	2.98
Formation Environmental 2012	Deer Creek	Ridgway's rail	Lotic	1.55	3.16	2.29
Formation Environmental 2012	Hoopes Spring – HS	Ridgway's rail	Lotic	0.24	3.16	14.50
Formation Environmental 2012	Hoopes Spring - HS3	Ridgway's rail	Lotic	0.54	3.16	6.62
Formation Environmental 2012	Sage Creek - LSV2C	Ridgway's rail	Lotic	0.45	3.16	7.93
Formation Environmental 2012	Sage Creek - LSV4	Ridgway's rail	Lotic	0.69	3.16	5.11
Formation Environmental 2012	South Fork Tincup Cr.	Ridgway's rail	Lotic	1.32	3.16	2.68
Hamilton and Buhl 2004	Lower East Mill Creek	Ridgway's rail	Lotic	1.32	3.16	2.69
McDonald and Strosher 1998	Elk R. above Cadorna Cr. (745)	Ridgway's rail	Lotic	6.30	3.16	0.56
McDonald and Strosher 1998	Fording R. above Swift Cr. (746)	Ridgway's rail	Lotic	0.23	3.16	15.72
Orr et al. 2012	Elk River 1	Ridgway's rail	Lotic	0.55	3.16	6.47
Orr et al. 2012	Elk River 12	Ridgway's rail	Lotic	2.67	3.16	1.33
Orr et al. 2012	Fording River 23	Ridgway's rail	Lotic	0.21	3.16	17.08
Orr et al. 2012	Michel Creek 2	Ridgway's rail	Lotic	0.28	3.16	12.49

Identification				Model Parameters		Translation
Reference	**Site**	**Species**	**Type**	***EF*[a]**	***TTF*[composite-b]**	**C_{water}[c] (µg/L)**
Saiki and Lowe 1987	San Luis Drain	Ridgway's rail	Lotic	0.36	3.16	9.86
Saiki and Lowe 1987	Volta Wasteway	Ridgway's rail	Lotic	1.03	3.16	3.44
Saiki et al. 1993	Mud Slough at Gun Club Road	Ridgway's rail	Lotic	1.37	3.16	2.59
Saiki et al. 1993	Salt Slough at the San Luis National Wildlife Refuge	Ridgway's rail	Lotic	0.43	3.16	8.30
Saiki et al. 1993	San Joaquin R. above Hills Ferry Road	Ridgway's rail	Lotic	0.36	3.16	9.92
Saiki et al. 1993	San Joaquin R. at Durham Ferry State Recreation Area	Ridgway's rail	Lotic	0.75	3.16	4.75

a - Geometric mean of the median enrichment factors (*EF*) for all available food types (algae, detritus, and sediment). *EF* (L/g) = C_{food}/C_{water}.

b - Composite trophic transfer factor ($TTF^{composite}$). Product of *TTF* values for all trophic levels.

c - Translated water selenium concentration corresponding to a bird egg criterion element of 11.2 mg Se/kg dw, calculated by (Equation 5-1).

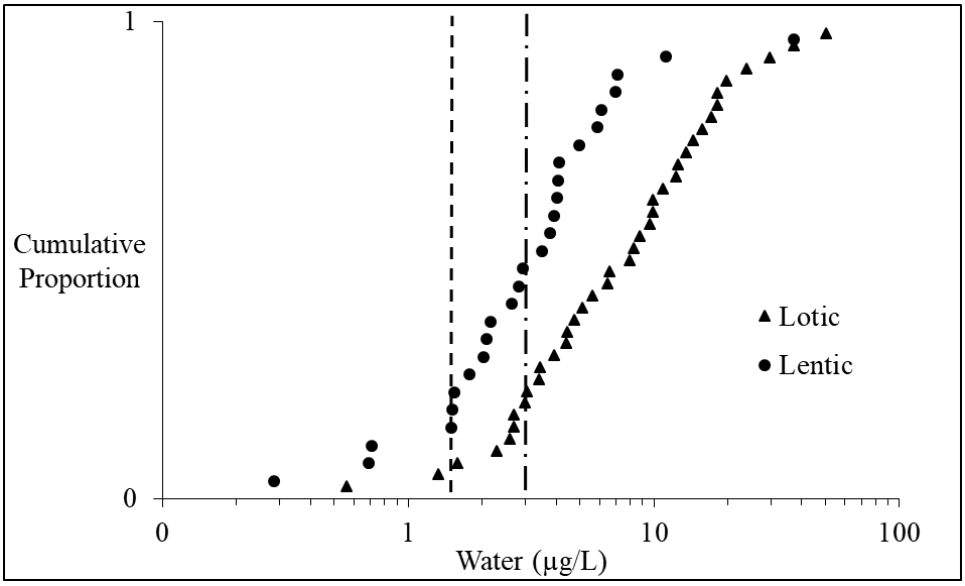


Figure 5-3. Probability distribution of the water column concentrations translated from the bird egg criterion element at the 26 lentic and 39 lotic aquatic sites from U.S. EPA (2016a). Dashed and dash-dot lines show the 20th percentiles of the lentic and lotic distributions, respectively.

As in the 2016 national aquatic life criterion for selenium, EPA presented the 20th percentile from the distribution of translated water column values of each category as the water column concentrations (3.0 µg/L for lotic waters and 1.5 µg/L for lentic waters) so that a direct comparison can be made to the aquatic life water-column concentrations, and because the 20th percentile is consistent with past practice as it provides a high probability of protection for most aquatic systems in both lentic and lotic categories. Table 5-10 provides the 20th percentile of the water concentration values translated from the bird egg criterion element value. Since the EPA translated water column concentration values for aquatic-dependent wildlife for both lentic and lotic systems are equal to or extremely close (1.5 µg/L for lentic waters and 3.0 µg/L for lotic waters) to the translated water column concentration values for aquatic life (1.5 µg/L for lentic waters and 3.1 µg/L for lotic waters), it appears that EPA's 2016 national selenium aquatic life water column criterion elements for lentic and lotic waters would be protective of aquatic-dependent wildlife as well. The differences in the translated water column concentration value for lotic waters between the aquatic life and aquatic-dependent wildlife are within the range of uncertainty of the 2016 national selenium water column criterion elements.

As discussed in Section 2.2.2 of U.S. EPA (2016a), selenium bioaccumulation potential depends on several biogeochemical factors that characterize a particular aquatic system. Uncertainty in the translation of the egg-ovary criterion element to the water column element can be reduced by deriving a site-specific water column criterion element that uses site-specific selenium data and information on food web dynamics from a biological assessment of the aquatic system. The general considerations are provided in the performance-based approach (*Translation of Selenium Tissue Criterion Elements to Site-Specific Water Column Criterion Elements for California*) and in Appendix K of U.S. EPA (2016a).

EPA conducted a separate analysis to run the model with five additional sites where *EF*s could be calculated for California waters. The sites are located within two selenium impacted areas, and when added to the dataset, the translated water column concentrations for birds changed from 1.5 µg/L to 1.6 µg/L for lentic systems, and remained at 3.0 µg/L for lotic systems. This analysis was also conducted with the water column criterion elements translated from fish egg-ovary criterion. After the five California sites were added, the translated lentic water column concentration increased from 1.5 µg/L to 1.6 µg/L, and remained unchanged at 3.1 µg/L for lotic systems. A comparison is shown in Table 5-11.

Table 5-10. Water Column Concentration Values Translated from the Bird Egg Criterion Element Using the 26 Lentic and 39 Lotic Sites in the National Selenium Aquatic Life Criterion (U.S. EPA 2016a).

	Lentic	**Lotic**
20th percentile (protective aquatic-dependent wildlife water column value)	**1.5 µg/L**	**3.0 µg/L**

Table 5-11. Comparison of 20th Percentile Water Column Concentration Values (µg/L) Translated from the Fish Egg-Ovary Criterion Element and the Bird Egg Criterion Element for the 26 Lentic and 39 Lotic Sites from the 2016 Aquatic Life Criteria (ALC) Dataset and the 65 Sites from the ALC Dataset + 5 Additional California Sites (4 Lentic and 1 Lotic).

Translation Site Dataset	**Translated from Fish Egg-Ovary**		**Translated from Bird Egg**	
	Lentic	**Lotic**	**Lentic**	**Lotic**
26 Lentic and 39 Lotic (2016 ALC Sites)	1.5 µg/L	3.1 µg/L	1.5 µg/L	3.0 µg/L
65 ALC Sites + 5 CA Sites (4 Lentic and 1 Lotic)	1.6 µg/L	3.1 µg/L	1.6 µg/L	3.0 µg/L

5.6 Derivation of Averaging Period for Chronic Water Criterion Element and Intermittent-Exposure Water Criterion Element

A previous analysis done in U.S. EPA 2016a (see Section 3.2.6 and Appendix J in U.S. EPA 2016a) demonstrated that a 30-day averaging period for the chronic water criterion element affords protection under all conditions for fish. EPA is proposing the same averaging period for the water column elements of California's selenium criterion.

Chapman et al. (2009) noted that selenium acute toxicity has been reported rarely in the aquatic environment and that traditional methods for predicting effects based on direct exposure to dissolved concentrations do not work well for selenium. As demonstrated in Appendix J of U.S. EPA (2016a), the kinetics of selenium accumulation and depuration are sufficiently slow that attainment of the water criterion element concentration by ambient 30-day averages will protect sensitive aquatic life species even where concentrations exhibit a high degree of variability.

To address situations where pulsed exposures of selenium could result in bioaccumulation in the ecosystem and potential chronic effects in aquatic life and aquatic-dependent wildlife, EPA is providing an intermittent-exposure water criterion element concentration intended to limit cumulative exposure to selenium, derived from the chronic 30-day water criterion element magnitude and from its duration, which was obtained from the kinetic analysis of Appendix J in U.S. EPA (2016a). That is, the intermittent criterion element is based on the same kinetic analysis used to derive the water chronic averaging period (30 days).

The 30-day average concentration, $C_{30\text{-}day}$, is given by **(**Equation 5-4):

$$C_{30-day} = C_{int} f_{int} + C_{bkgrnd}(1 - f_{int})$$

(Equation 5-4)

Where:

C_{int} = the intermittent spike concentration (µg/L)

f_{int} = the fraction of any 30-day period during which elevated selenium concentrations occur

C_{bkgrnd} = the average daily background concentration occurring during the remaining time, integrated over 30 days.

$C_{30\text{-}day}$ is not to exceed the chronic criterion element, $WQC_{30\text{-}day}$. If the intent is to apply a criterion element, WQC_{int}, to the intermittent spike concentrations, then replacing C_{int} with WQC_{int} and $C_{30\text{-}day}$ with $WQC_{30\text{-}day}$ in the above equation, and then solving for WQC_{int} yields (Equation 5-5**)**:

$$WQC_{int} = \frac{WQC_{30-day} - C_{bkgrnd}(1 - f_{int})}{f_{int}}$$

(Equation 5-5)

The equation expresses the intermittent-exposure water criterion element in terms of the 30-day average chronic water criterion element, for a lentic or lotic system, as appropriate, while accounting for the fraction (in days) of any 30-day period the intermittent spikes occur and for the concentration background occurring during the remaining time. The reasonable worst-case assumption inherent in this approach is that selenium bioaccumulation is linear over a very wide range of concentrations, that is, *EF* and *TTF* values do not decrease significantly as concentrations increase. For more information and examples on the intermittent-exposure water criterion element, please see Section 3.3 of U.S. EPA (2016a).

Part 6 AQUATIC AND AQUATIC-DEPENDENT WILDLIFE CRITERIA FOR SELENIUM IN CALIFORNIA'S FRESH WATERS

The available data indicate that aquatic life and aquatic-dependent wildlife would be protected from the toxic effects of selenium by applying the following criteria, recognizing that fish tissue elements and bird egg elements supersede the translated site-specific water elements (except in special situations, see footnote 4 in Table 6-1) and that the fish egg-ovary elements supersede all other fish tissue elements:

1. The concentration of selenium in bird eggs does not exceed 11.2 mg/kg, dry weight;
2. The concentration of selenium in the eggs or ovaries of fish does not exceed 15.1 mg/kg, dry weight;
3. The concentration of selenium (a) in whole body of fish does not exceed 8.5 mg/kg dry weight, or (b) in muscle tissue of fish (skinless, boneless fillet) does not exceed 11.3 mg/kg dry weight;
4. The 30-day average concentration of selenium in water does not exceed more than once in three years on average the value derived on a site-specific basis using the methodology described in *Translation of Selenium Tissue Criterion Elements to Site-Specific Water Column Criterion Elements for California*.
5. The intermittent concentration of selenium in either a lentic or lotic water, as appropriate, does not exceed $WQC_{int} = \frac{WQC_{30-day} - C_{bkgrnd}(1-f_{int})}{f_{int}}$ more than once in three years on average.

Table 6-1. Summary of the Proposed California Selenium Ambient Chronic Water Quality Criteria for Protection of Aquatic Life and Aquatic-Dependent Wildlife.

Media Type	**Bird Tissue**	**Fish Tissue[1]**		**Water Column[4]**	
Criterion Element	**Bird Egg[2]**	**Egg-Ovary[2]**	**Fish Whole Body or Muscle[3]**	**Monthly Average Exposure**	**Intermittent Exposure[5]**
Magnitude	11.2 mg/kg dw	15.1 mg/kg dw	8.5 mg/kg dw whole body or 11.3 mg/kg dw muscle (skinless, boneless filet)	Derived on a site-specific basis using the methodology described in *Translation of Selenium Tissue Criterion Elements to Site-Specific Water Column Criterion Elements for California*	$WQC_{int} = \frac{WQC_{30-day} - C_{bkgrnd}(1 - f_{int})}{f_{int}}$
Duration	Instan-taneous measure-ment[6]	Instan-taneous measure-ment[6]	Instantaneous measurement[6]	30 days	Number of days/month with an elevated concentration
Frequency	Not to be exceeded	Not to be exceeded	Not to be exceeded	Not more than once in three years on average	Not more than once in three years on average

1. Fish tissue elements are expressed as steady-state.
2. Fish Egg-Ovary supersedes any whole-body, muscle, or translated water column element for that taxon when fish egg-ovary are measured. Bird Egg supersedes translated water column elements for that taxon when both are measured.
3. Fish whole-body or muscle tissue supersedes the translated water column element when both fish tissue and water concentrations are measured.
4. Translated water column values ($WQC_{30\text{-day}}$) will be based on dissolved total selenium in water and will be derived using the methodology described in *Translation of Selenium Tissue Criterion Elements to Site-Specific Water Column Criterion Elements for California*. Water column values are the applicable criterion element in the absence of bird tissue data and steady-state condition fish tissue data.
5. Where $WQC_{30\text{-day}}$ is the water column monthly element derived using the methodology described in *Translation of Selenium Tissue Criterion Elements to Site-Specific Water Column Criterion Elements for California*, C_{bkgrnd} is the average background selenium concentration, and f_{int} is the fraction of any 30-day period during which elevated selenium concentrations occur, with f_{int} assigned a value ≥ 0.033 (corresponding to 1 day).
6. Fish tissue and bird tissue data provide instantaneous point measurements that reflect integrative accumulation of selenium over time and space in bird or fish population(s) at a given site.

The chronic selenium criterion is derived to be protective of the entire aquatic community, including fish, amphibians, invertebrates, and aquatic-dependent wildlife. Based on this analysis and EPA's previous work in U.S. EPA (2016a), fish and birds are the most sensitive taxa to selenium effects. When both endpoints are translated to protective lentic and lotic water column concentration, the results are equal or nearly equal. Selenium water quality criterion elements based on fish tissue (egg-ovary, whole body, and/or muscle) or bird egg sample data override the criterion elements based on water column selenium data due to the fact, noted above, that fish and bird tissue concentrations provide the most robust and direct information on potential selenium effects in fish and birds. However, because selenium concentrations in fish and bird tissue are a result of selenium bioaccumulation via dietary exposure, there are two specific circumstances where the fish concentrations do not fully represent potential effects on fish and the aquatic ecosystem: 1) in "fishless" waters for the fish tissue elements, and 2) in areas with new selenium inputs for both taxa.

Fishless waters are defined as waters with insufficient instream habitat and/or flow to support a population of any fish species on a continuing basis, or waters that once supported populations of one or more fish species but no longer support fish (i.e., extirpation) due to temporary or permanent changes in water quality (e.g., due to selenium pollution), flow or instream habitat. Because of the inability to collect sufficient fish tissue to measure selenium concentrations in fish in such waters, water column concentrations will best represent selenium levels required to protect aquatic communities and downstream waters in such areas. It is possible that birds will still represent potential effects of selenium in these fishless waters. As shown in the Part 5 of this TSD, some birds that consume invertebrates bioaccumulate more selenium than birds that eat primarily fish and may therefore more susceptible to selenium effects.

Footnote 1 in Table 6-1 indicates that the fish tissue concentrations of the criterion are expressed as steady-state. Since avian taxa are more mobile across aquatic habitats, the bird tissue concentrations of the criterion are not expressed in terms of steady-state. An organism is in steady-state when the rates of chemical uptake and depuration are equal and tissue concentrations remain constant over time (U.S. EPA 2003). For the purposes of EPA's 2016 recommended aquatic life selenium criterion, steady-state refers to conditions where sufficient time has passed after the introduction of a new or increased discharge of selenium into a water

body so that fish tissue concentrations of selenium are no longer increasing (U.S. EPA 1991). For a fish tissue measurement to be meaningful, the system from which the sample is taken should not be experiencing recent new inputs of selenium. In the EPA's Aquatic Life Ambient Water Quality Criterion for Selenium–Freshwater 2016, new inputs are defined as new anthropogenic activities resulting in the release of selenium into a lentic or lotic aquatic system. New inputs do not refer to seasonal variability of selenium that occurs naturally within a system (e.g. spring run-off events or precipitation-driven pulses). New inputs will likely result in a greater concentration of selenium in the food web and a relatively slow increase in the selenium concentration in fish. Fish tissue data should not be utilized for implementation of the criterion until after selenium concentrations in the fish have stopped increasing. EPA estimates that the concentration of selenium in fish tissue will not reach steady-state for several months in lotic systems and for longer time periods (e.g., 2–3 years) in lentic systems. Achievement of steady-state in an aquatic system depends on the hydrodynamics of the aquatic system (particularly reservoirs with multiple riverine inputs and controlled releases of water into downstream water bodies), the location of the selenium input, and the particular food web. EPA expects the time needed to achieve steady-state with new or increased selenium inputs to be site-specific. Thus, the EPA recommends that fish tissue criterion elements not take precedence over the water column criterion elements until the aquatic system achieves steady-state. In the interim, the EPA recommends sampling and using site-specific data to gain a better understanding of the selenium bioaccumulation dynamics in a receiving water and to determine when steady-state conditions have been reached.

6.1 Protection of Downstream Waters

EPA regulations at 40 CFR § 131.10(b) provide that "[i]n designating uses of a waterbody and the appropriate criteria for those uses, the state shall take into consideration the water quality standards of downstream waters and ensure that its water quality standards provide for the attainment and maintenance of the water quality standards of downstream waters." Especially in cases where downstream waters are lentic waterbody types (e.g., lakes, impoundments), or harbor more sensitive species, a selenium criterion more stringent than that required to protect in-stream uses may be necessary to ensure that water quality standards provide for the attainment and maintenance of the water quality standards of downstream waters.

6.2 Site-specific Criteria

All elements of the proposed California selenium criterion may be modified to reflect site-specific conditions where the scientific evidence indicates that different values will be protective of aquatic life and aquatic-dependent wildlife and provide for the attainment of designated uses.

Since the fish egg-ovary criterion element is based on a robust set of toxicity data, California may modify that element by applying the Recalculation Procedure (U.S. EPA 2013) to modify the species toxicity database to reflect taxonomic relatedness to the site assemblage, while including tested surrogates for untested resident species. If the Recalculation Procedure is used, the State will follow the process to develop a site-specific criterion instead of the performance-based approach. For aquatic-dependent wildlife, the Recalculation Procedure would not be appropriate as the bird tissue criterion element was derived for the most sensitive bird species in the literature and is considered a surrogate for all birds. However under the performance-based approach, California could translate the bird EC_{10} to a site-specific water column criterion with the use of a species-specific *TTF* if site-specific data indicated this was needed to ensure protection of aquatic-dependent wildlife.

It is important to note that species in the data set presented here that are not present at a site should not be deleted from the data set if those species serve as surrogate(s) for other species known or expected to be present at a site. To further improve confidence in the applied tissue criterion element, further testing of fish species or bird species that are residents at the site can be conducted. The most relevant testing would measure survival and occurrence of deformities in offspring of wild-caught female fish, or hatching success of wild breeding bird pairs to determine an EC_{10} for selenium in the eggs or ovaries (e.g., following Janz and Muscatello 2008).

Using either the proposed bird egg, fish egg-ovary, fish whole body, or fish muscle criterion concentration element or a site-specific bird egg, fish egg-ovary, fish whole body, or fish muscle criterion element, translation of a tissue criterion to a protective water concentration should be performed in a manner that accounts for site-specific conditions and is consistent with the performance-based approach (*Translation of Selenium Tissue Criterion Elements to Site-Specific Water Column Criterion Elements for California*). Both the performance-based approach (*Translation of Selenium Tissue Criterion Elements to Site-Specific Water Column Criterion Elements for California*) and Appendix K in U.S. EPA (2016a) provides information

on the data necessary to derive a site-specifc water column criterion element translated from the fish and bird tissue criterion elements and a site-specific criterion, as well as scientifically defensible methods, including the use of traditional Bioaccumulation Factors (BAFs), in addition to the more comprehensive mechanistic modeling used in the criteria derivation in this TSD.

Part 7 SUMMARY OF U.S. FISH AND WILDLIFE SERVICE AQUATIC-DEPENDENT WILDLIFE SELENIUM REPORT

Section 7(a)(2) of the Endangered Species Act (ESA) requires federal agencies, in consultation with the U.S. Fish and Wildlife Service (U.S. FWS 2017) and/or the U.S. National Marine Fisheries Service (NMFS), to ensure that actions they authorize/approve, fund, or carry out are not likely to jeopardize the continued existence of federally listed endangered or threatened species or result in the destruction or adverse modification of designated critical habitat of such species as cited in 16 U.S.C. § 1536 (a)(2). Therefore, EPA coordinated with the U.S. FWS to determine the aquatic and aquatic-dependent wildlife species most at-risk for selenium toxicity in California inland surface waters, and enclosed bays and estuaries, excluding the San Francisco Bay and Delta. U.S FWS (2017) includes a preliminary list of aquatic and aquatic-dependent wildlife species (including Threatened and Endangered (T&E) species) in California (excluding resident species found in the San Francisco Bay and Delta) that can potentially be adversely affected by elevated selenium levels in the aquatic environment and aquatic food web.

The U.S. FWS's report included the identification of potentially sensitive California aquatic life and aquatic-dependent wildlife species including T&E species. U.S. FWS further analyzed the list of potentially sensitive species to determine each species' potential exposure to selenium, in order to narrow the list to a manageable size of approximately 4 to 12 species per geographic ecoregion of California (to approximately 20 to 41 species total). The seven geographic ecoregions included the Klamath Basin, North/Central Coast and Inland Ranges, California Great Valley (San Joaquin Valley), Sierra Nevada Mountains and Foothills, Great Basin, Deserts, and Southern Coast and Inland Ranges (coastal areas included enclosed bays and estuaries).

U.S. FWS analyzed each species on the smaller list to determine the final list of most at-risk species in California. The detailed analysis first looked at each species occurrence data and life history information. The analysis included detailed information on each species exposure potential using parameters or factors such as aquatic dependency, habitat, population status (including Federal listing status and history, if applicable), body size/weights, dietary composition (and associated volumes), and percentages of components that make up the diet (to assist in determining its food web), and any other factors necessary to make an informed determination as to the species' exposure potential to selenium toxicity.

In addition, the analysis specifically looked at whether the aquatic and aquatic-dependent wildlife species are migratory (part-time residents) and whether exposure to selenium is prior to or during breeding cycles. U.S. FWS further provided a final list of most at-risk species as Table A1 in their 2017 report for all geographic ecoregions combined, ranked in order of those expected to be most affected from elevated selenium levels in the aquatic environment, based on U.S. FWS's analysis and expertise. EPA will consider the information in the U.S. FWS's report to prepare its biological evaluation of potential effects to federally listed threatened and endangered species from our selenium criteria promulgation for California waters.

REFERENCES

Ackerman, J.T. and C.A. Eagles-Smith. 2009. Selenium bioaccumulation and body condition in shorebirds and terns breeding in San Francisco Bay, California, USA. Environ. Toxicol. Chem. 28(10): 2134-2141.

Ackerman, J. T., C. A. Hartman, M. P. Herzog, J. Y. Takekawa, J. A. Robinson, L. W. Oring, J. P. Skorupa, and R. Boettcher. 2013. American Avocet (*Recurvirostra americana*), version 2.0. Retrieved from The Birds of North America (A. F. Poole, Editor). Cornell Lab of Ornithology, Ithaca, NY, USA. https://doi.org/10.2173/bna.275

Ackerman, J.T., C.A. Eagles-Smith, M.P. Herzog and C.A. Hartman. 2016. Maternal transfer of contaminants in birds: Mercury and selenium concentrations in parents and their eggs. Environ. Pollut. 210: 145-154.

Adams, W.J., K.V. Brix, K.A. Cothern, L.A. Tear, R.D. Cardwell, A. Fairbrother and J. Toll. 1998. Assessment of selenium food chain transfer and critical exposure factors for avian wildlife species: need for site-specific data. In: Environmental Toxicology and Risk Assessment: Seventh Volume. ASTM STP 1333. Little, E. E., DeLonafy, A. J., and Greenberg, B. M. (Eds). American Society for Testing and Materials, Philadelphia, Pennsylvania.

Adams, W.J., K.V. Brix, M. Edwards, L.M. Tear, D.K. DeForest, and A. Fairbrother. 2003. Analysis of field and laboratory data to derive selenium toxicity thresholds for birds. Environmental Toxicology and Chemistry 22(9): 2020-2029.

Albers, P.H., D.E. Green and C.J. Sanderson. 1996. Diagnostic criteria for selenium toxicosis in aquatic birds: dietary exposure, tissue concentrations and macroscopic effects. J. Wildlife Diseases 32(3): 468-485.

Allan, C.B., G.M. Lacourciere and T.C. Stadtman. 1999. Responsiveness of selenoproteins to dietary selenium. Annu. Rev. Nutr. 19: 1-16.

ATSDR (Agency for Toxic Substances and Disease Registry). 2003. Toxicological Profile for Selenium. Atlanta, GA: U.S. Department of Health and Human Services, Public Health Service Agency for Toxic Substances and Disease Registry.

Audas, A., G.R. Hogan and H. Razniak. 1995. Incubation temperature as a modifying factor on survival of *Tenebrio molitor* reared in selenium-containing media. J. Toxicol. Environ. Health 44(1): 115-122.

Baines, S.B., N.S. Fisher, M.A. Doblin and G.A. Cutter. 2001. Uptake of dissolved organic selenides by marine phytoplankton. Limnol. and Oceanog. 46: 1936-1944.

Beckon, W.N., C. Parkins, A. Maximovich and A.V. Beckon. 2008. A general approach to modeling biphasic relationships. Environ. Sci. Tech. 42: 1308-1314.

Besser, J.M., W.G. Brumbaugh, D.M. Papoulias, C.D. Ivey, J.L. Kunz, M. Annis and C.G. Ingersoll. 2012. Bioaccumulation and toxicity of selenium during a life-cycle exposure with desert pupfish (*Cyprinodon macularius*). U.S. Geological Survey Scientific Investigations Report 2012–5033, 30 pp. with appendices.

Birkner, J.H. 1978. Selenium in aquatic organisms from seleniferous habitats. Ph.D. Thesis. Colorado State University, Fort Collins, CO. Available from: University Microfilms, Ann Arbor, MI. Order No. 78-20841.

Bowie, G.L., J.G. Sanders, G.F. Riedel, C.C. Gilmour, D.L. Breitburg, G.A. Cutter and D.B. Porcella. 1996. Assessing selenium cycling and accumulation in aquatic ecosystems. Water Air Soil Pollut. 90(1/2): 93-104.

Brady, C., S. Petrie, M. Schummer, S. Badzinski, N. Belzile andY-W. Chen. 2013. Effects of dietary selenium on the health and survival of captive wintering lesser scaup. Environ. Pollut. 175: 8-15.

Brain, P. and R. Cousens. 1989. An equation to describe dose-responses where there is stimulation of growth at low doses. Weed Res. 29: 93-96.

Brisbin Jr., I. Lehr and Thomas B. Mowbray. 2002. American Coot (*Fulica americana*), Retrieved from The Birds of North America. P.G. Rodewald (Ed). Ithaca: Cornell Lab of Ornithology. Retrieved from The Birds of North America: https://birdsna.org/Species-Account/bna/species/y00475

Breynaert, E., C. Bruggerman and A. Maes. 2008. XANES-EXAFS analysis of Se solid-phase reaction products formed upon contacting Se (IV) with FeS_2 and FeS. Environ. Sci. Technol. 42: 3595-3601.

Buehler, D. A. 2000. Bald Eagle (*Haliaeetus leucocephalus*), version 2.0. Retrieved from The Birds of North America (A. F. Poole and F. B. Gill, Editors). Cornell Lab of Ornithology, Ithaca, NY, USA. https://doi.org/10.2173/bna.506

Burau, R.G. 1985. Environmental chemistry of selenium. California Agriculture. July-August: 16-18.

Butler, D.L., R.P. Krueger, B.C. Osmundson, A.L. Thompson and S.K. McCall. 1991. Reconnaissance investigation of water quality, bottom sediment, and biota associated with irrigation drainage in the Gunnison and Uncompahgre river basins and at Sweitzer Lake, west-central Colorado, 1988-89. U.S. Geological Survey Water-Resources Investigations Report No. 91-4103. Denver, CO.

Butler, D.L., R.P. Krueger, B.C. Osmundson and A.L. Thompson. 1993. Reconnaissance investigation of water quality, bottom sediment, and biota associated with irrigation drainage in the pine river project area, Southern Ute Indian Reservation, Southwestern Colorado and

Northwestern New Mexico, 1988-9. U.S. Geological Survey Water-Resources Investigations Report No. 92-4188. Denver, CO.

Butler, D.L., W.G. Wright, D.A. Hahn, R.P. Krueger and B.C. Osmundson. 1994. Physical, chemical, and biological data for detailed study of irrigation drainage in the Uncompahgre project area and in the Grand Valley, west-central Colorado, 1991-92. U.S. Geological Survey Open File Report No. 94-110. Denver, CO.

Butler, D.L., R.P. Krueger, B.C. Osmundson and E.G. Jensen. 1995. Reconnaissance investigation of water quality, bottom sediment, and biota associated with irrigation drainage in the Dolores project area, Southwestern Colorado and Southeastern Utah, 1990-91. U.S. Geological Survey. Water-Resources Investigations Report 94-4041. Denver, CO 1995.

Butler, D.L., B.C. Osmundson, and R.P. Krueger. 1997. Field screening of water, soil, bottom sediment, and biota associated with irrigation drainage in the Dolores Project and the Mancos River Basin, Southwestern Colorado, 1994. U.S. Geological Survey. Water-Resources Investigations Report 97-4008. Denver, Colorado 1997.

Byron, E. and G. Santolo. 2010. 2010 Selenium Monitoring Results: Fish and Bird Egg Tissue Chemistry, Newport Bay Watershed. Prepared by CH2M Hill for Orange County. August 12, 2010.

Byron, E., G. Santolo, P. Sexton, K. Ashby and D. Apt. 2012. 2011 Selenium Monitoring Results: Fish and Bird Egg Tissue Chemistry, Newport Bay Watershed. Prepared for Orange County. January 18, 2012.

Byron, E. and G. Santolo. 2014. Selenium and Organochlorine Compounds in Newport Bay Watershed Fish and Bird Eggs, 2013. Prepared by CH2M Hill for Orange County. March 20, 2014.

Caffrey, P.B. 1989. The effects of zinc deprivation on selenium requirements in daphnids (Crustacea). Available from Univ. Microfilms Int., Order No. DA9008884, From: Diss. Abstr. Int. B 50(11): 4959.

Carolina Power & Light. 1997. Largemouth Bass Selenium Bioassay- Report. Carolina Power & Light Company, Environmental Services Section, NC. December 1997.

Casey, R. 2005. Results of aquatic studies in the McLeod and Upper Smoky River systems. Alberta Environment. 62 pp.

CEDEN. 2015. California Environmental Data Exchange Network (CEDEN: http://www.ceden.org/) database. Last accessed on February 4, 2015.

CH2M Hill. 2008. Development of a selenium standard for the open waters of the Great Salt Lake. Prepared for: State of Utah. Department of Environmental Quality. Division of Water Quality. May 2008.

Chapman, P.M., W.J. Adams, M.L. Brooks, C.G. Delos, S.N. Luoma, W.A Maher, H.M. Ohlendorf, T.S. Presser and D.P. Shaw. 2009. Ecological Assessment of Selenium in the Aquatic Environment: Summary of a SETAC Pellston Workshop. Pensacola FL (USA): Society of Environmental Toxicology and Chemistry (SETAC).

Chapman, P.M., W.J. Adams, M.L. Brooks, C.G. Delos, S.N. Luoma, W.A Maher, H.M. Ohlendorf, T.S. Presser and D.P. Shaw (Eds). 2010. Ecological Assessment of Selenium in the Aquatic Environment. SETAC Press, Pensacola, FL, USA.

Chen, Y.W., H.Y.T. Truong and N. Belzile. 2008. Abiotic formation of elemental selenium and role of iron oxide surfaces. Chemosphere 74: 1079-1084.

Cleveland, L., E.E. Little, D.R. Buckler and R.H. Wiedmeyer. 1993. Toxicity and bioaccumulation of waterborne and dietary selenium in juvenile bluegill (*Lepomis macrochirus*). Aquat. Toxicol. (Amsterdam) 27(3-4): 265-279.

Connolly, J. P. 1985. Predicting single-species toxicity in natural water systems. Environ. Toxicol. Chem. 4(4): 573-582.

Cooney, J.D., G.M. DeGraeve, E.L. Moore, W.D. Palmer and T.L. Pollock. 1992. Effects of food and water quality on culturing of *Ceriodaphnia dubia*. Environ. Toxicol. Chem. 11(6): 823-837.

Cowgill, U.M. 1987. Critical analysis of factors affecting the sensitivity of zooplankton and the reproducibility of toxicity test results. Water Res. 21(12): 1453-1462.

Cowgill, U.M. and D.P. Milazzo. 1989. The culturing and testing of two species of duckweed. ASTM Spec. Tech. Publ. 1027. Aquat. Toxicol. Hazard Assess. 12: 379-391.

Coyle, J.J., D.R. Buckler, C.G. Ingesoll, J.F. Fairchild and T.W. May. 1993. Effect of dietary selenium on the reproductive success of bluegills *Lepomis macrochirus*. Environ. Toxicol. Chem. 12(3): 551-565.

Cullen, S. A., J. R. Jehl Jr., and G. L. Nuechterlein. 1999. Eared Grebe (*Podiceps nigricollis*), version 2.0. Retrieved from The Birds of North America (A. F. Poole and F. B. Gill, Editors). Cornell Lab of Ornithology, Ithaca, NY, USA. https://doi.org/10.2173/bna.433

Cutter, G.A. 1989. The estuarine behavior of selenium in San Francisco Bay. Estuar. Coast. Shelf Sci. 28: 13-34.

Cutter, G.A. and K.W. Bruland. 1984. The marine biogeochemistry of selenium: a re-evaluation Limnol. Oceanogr. 29: 1179-1192.

Cutter, G.A. and L.S. Cutter. 2004. Selenium biogeochemistry in the San Francisco Bay estuary: changes in water column behavior. Estuar. Coast. Shelf S. 61: 463-476.

Cutter, G.A. and M.L.C. San Diego-McGlone. 1990. Temporal variability of selenium fluxes in the San Francisco Bay. Sci. Total Environ. 97: 235-250.

DeVink, J-M.A., R.G. Clark, S.M Slattery and T.M. Scheuhammer. 2008. Effects of dietary selenium on reproduction and body mass of captive lesser scaup. Environ. Toxicol. Chem. 27(2): 471-477.

Diaz, X., W.P. Johnson, W.A. Oliver and D.L. Naftz. 2009. Volatile selenium flux from the Great Salt Lake. Environ. Sci. Technol. 43: 53-59.

Diplock, A.T. and W.G. Hoekstra. 1976. Metabolic aspects of selenium action and toxicity. CRC Cr Rev. Toxicol. 5: 271-329.

Doblin, M.A., S.B. Baines, L.S. Cutter and G.A. Cutter. 2006. Sources and biogeochemical cycling of particulate selenium in the San Francisco Bay estuary. Estuar. Coast. Shelf Sci. 67: 681-694.

Doroshov, S., J. Van Eenennaam, C. Alexander, E. Hallen, H. Bailey, K. Kroll and C. Restrepo. 1992. Development of Water Quality Criteria for Resident Aquatic Species of the San Joaquin River; Part II, Bioaccumulation of Dietary Selenium and its Effects on Growth and Reproduction in Bluegill (*Lepomis macrochirus*). Final Report to State Water Resources Control Board, State of California. Contract Number 7-197-250-0.

Doucette, G.J., N.M. Price and P.J. Harrison. 1987. Effects of selenium deficiency on the morphology and ultrastructure of the coastal marine diatom *Thalassiosira pseudonana* (Bacillariophyceae). J. Phycol. 23(1): 9-17.

Dumont, E., F. Vanhaecke, and R. Cornelis. 2006. Selenium speciation from food source to metabolites: a critical review. Anal. Bioanal. Chem. 385, 1304–1323.

Eddleman, W. R., R. E. Flores, and M. Legare. 1994. Black Rail (*Laterallus jamaicensis*), version 2.0. Retrieved from The Birds of North America (A. F. Poole and F. B. Gill, Editors). Cornell Lab of Ornithology, Ithaca, NY, USA. https://doi.org/10.2173/bna.123

Egerer-Sieber, C., V. Herl, F. Muller-Uri, W. Kreisb and Y.A. Muller. 2006. Crystallization and preliminary crystallographic analysis of selenomethionine-labelled progesterone 5*b*-reductase from *Digitalis lanata* Ehrh. Acta Crystall Sec F. 62: 186–188.

Elendt, B.P. 1990. Selenium deficiency in crustacea, an ultrastructural approach to antennal damage in *Daphnia magna* Straus. Protoplasma 154(1): 25-33.

Elendt, B.P. and W.R. Bias. 1990. Trace nutrient deficiency in *Daphnia magna* cultured in standard medium for toxicity testing effects of the optimization of culture conditions on life history parameters of *Daphnia magna*. Water Res. 24(9): 1157-1168.

EPRI (Electric Power Research Institute). 2006. Fate and Effects of Selenium in Lentic and Lotic Systems. Technical Paper, 104 pp. Available at website: http://www.epri.com/abstracts/Pages/ProductAbstract.aspx?ProductId=000000000001005315.

Emsley, J. 2011. Nature's Building Blocks: An A-Z Guide to the Elements. Oxford University Press.

Fairbrother, A., K.V. Brix, J.E. Toll, S. McKay and W.J. Adams. 1999. Egg selenium concentrations as predictors of avian toxicity. Human and Ecological Risk Assessment 5: 1229-1253.

Fan, T.W.M., S.J. Teh, D.E. Hinton and R.M. Higashi. 2002. Selenium biotransformations into proteinaceous forms by foodweb organisms of selenium-laden drainage waters in California. Aquatic Toxicol. 57: 65-84.

Feck, J.M. 2002. Assessment of the American dipper (*Cinclus mexicanus*) as a biological indicator of water quality. M.S. Thesis, University of Wyoming, Laramie. 41 p.

Fernández-Turiel, J.L., W. Carvalho, M. Cabañas, X. Querol and A. López-Soler. 1994. Mobility of heavy metals from coal fly ash. Environ. Geol. 23: 264-270.

Formation Environmental. 2011. Brown Trout Laboratory Reproduction Studies Conducted in Support of Development of a Site-Specific Selenium Criterion. Prepared for J.R. Simplot Company by Formation Environmental. Revised October 2011.

Formation Environmental. 2012. Appendix A – Summary of Exposure Conditions: Surface Water, Sediment, Tissue. Technical Support Document: Proposed Site-Specific Selenium Criterion, Sage and Crow Creeks, Idaho. Prepared for J.R. Simplot Company. January 2012.

Freeman, J.L., L.H. Zhang, M.A. Marcus, S. Fakra, S.P. McGrath and E.A.P. Pilon-Smits. 2006. Spatial imaging, speciation, and quantification of selenium in the hyperaccumulator plants *Astragalus bisulcatus* and *Stanleya pinnata*. Plant Physiol. 142: 124-134.

Gammonley, J. H. 2012. Cinnamon Teal (*Spatula cyanoptera*), version 2.0. Retrieved from The Birds of North America (A. F. Poole, Editor). Cornell Lab of Ornithology, Ithaca, NY, USA. https://doi.org/10.2173/bna.209

Gatlin III, D.M.I and R.P. Wilson. 1984. Dietary selenium requirement of fingerling channel catfish. J. Nutr. 114: 627-633.

Gillespie, R.B. and P.C. Baumann. 1986. Effects of high tissue concentrations of selenium on reproduction by bluegills (*Lepomis macrochirus*). Trans. Am. Fish. Soc. 115(2): 208-213.

Gissel-Nielsen, G. and M. Gissel-Nielsen. 1978. Sensitivity of trout to chronic and acute exposure to selenium. Agric. Environ. 4: 85-91.

Golder Associates. 2009. Development of a site-specific selenium toxicity threshold for Dolly Varden Char. Report Number 04-1421-101/2000. Report to Northgate Minerals Corporation, Smithers, British Columbia.

Grasso, D.N., M.E. Jennings and W.J. Sadler. 1995. Field Screening of Water Quality, Bottom Sediment, and Biota Associated with Irrigation Drainage, Wind River Indian Reservation, Wyoming, 1992-1993. Open-File Report 95-121.

Hamilton, S.J. and K.J. Buhl. 2004. Selenium in water, sediment, plants, invertebrates, and fish in the Blackfoot River drainage. Water Air Soil Pollut. 159: 3-34.

Harding, L.E. 2008. Non-linear uptake and hormesis effects of selenium in red-winged blackbirds (*Agelaius phoeniceus*). Sci. Total Environ. 389: 350-366.

Harding, L.E, M. Graham and D. Paton. 2005. Accumulation of selenium and lack of severe effects on productivity of American dippers (*Cinclus mexicanus*) and spotted sandpipers (*Actitis macularia*). Arch. Environ. Contam. Toxicol. 48: 414-423.

Hardy, R.W. 2005. Effects of dietary selenium on cutthroat trout (*Oncorhynchus clarkii*) growth and reproductive performance. Report for Montgomery Watson Harza. December 14.

Harrison, P.J, P.W. Yu, P.A. Thompson, N.M. Price and D.J. Phillips. 1988. Survey of selenium requirements in marine phytoplankton. Mar. Ecol. Prog. Ser. 47(1): 89-96.

Haygarth, P.M. 1994. Global Importance and Global Cycling of Selenium. In: Selenium in the Environment. Frankenberger, W.T. and S. Benson (Eds). Marcel Dekker, Inc. New York.

Heinz, G.H. 1993. Re-exposure of mallards to selenium after chronic exposure. Environ. Toxicol. Chem. 12: 1691-1694.

Heinz, G.H. and M.A. Fitzgerald. 1993a. Reproduction of mallards following overwinter exposure to selenium. Env. Poll. 81(2): 117-122.

Heinz, G.H. and M.A. Fitzgerald. 1993b. Overwinter survival of mallards fed selenium. Arch. Environ. Contam. Toxicol. 25: 90-94.

Heinz, G.H. and D.J. Hoffman. 1996. Comparison of the effects of seleno-L-methionine, seleno-DL-methionine, and selenized yeast on reproduction of mallards. Environ. Pollut. 91: 169-175.

Heinz, G.H. and D.J. Hoffman. 1998. Methylmercury chloride and selenomethionine interactions on the health and reproduction in mallards. Environ. Toxicol. Chem. 17: 139-145.

Heinz, G.H., D.J. Hoffman, A.J. Krynitsky and D.M.G. Weller. 1987. Reproduction in mallards fed selenium. Environ. Toxicol. Chem. 6: 423-433.

Heinz, G.H., D.J. Hoffman and L.G. Gold. 1988. Toxicity of organic and inorganic selenium to mallard ducklings. Arch. Environ. Contam. Toxicol. 17: 561-568.

Heinz, G.H., D.J. Hoffman and L.G. Gold. 1989. Impaired reproduction of mallards fed an organic form of selenium. J. Wildl. Manage. 53: 418-428.

Heinz, G.H., G.W. Pendelton, A.J. Krynitsky, and L.G. Gold. 1990. Selenium accumulation and elimination in mallards. Arch. Environ. Contam. Toxicol. 19:375-379.

Hermanutz, R.O., K.N. Allen, T.H. Roush and S.F. Hedtke. 1992. Effects of elevated selenium concentrations on bluegills *Lepomis macrochirus* in outdoor experimental streams. Environ. Toxicol. Chem. 11(2): 217-224.

Hermanutz, R.O., K.N. Allen, N.E. Detenbeck and C.E. Stephan. 1996. Exposure to bluegill (*Lepomis macrochirus*) to selenium in outdoor experimental streams. U.S. EPA Report. Mid-Continent Ecology Division. Duluth, MN.

Hilton, J.W. and P.V. Hodson. 1983. Effect of increased dietary carbohydrate on selenium metabolism and toxicity in rainbow trout (*Salmo gairdneri*). J. Nutr. 113: 1241-1248.

Hodson, P.V., D.J. Spry and B.R. Blunt. 1980. Effects on rainbow trout (*Salmo gairdneri*) of a chronic exposure to waterborne selenium. Can. J. Fish. Aquat. Sci. 37: 233-240.

Hoffman, D.J. 2002. Role of selenium toxicity and oxidative stress in aquatic birds. Aquat. Toxicol. 57: 11-26.

Hoffman, D.J. and G.H. Heinz. 1988. Embryotoxic and teratogenic effects of selenium in the diet of mallards. J. Toxicol. Environ. Health. 24: 477-490.

Hoffman, D.J., H.M. Ohlendorf and T.W. Aldrich. 1988. Selenium teratogenesis in natural populations of aquatic birds in central California. Arch. Environ. Contam. Toxicol. 17: 519-525.

Hoffman, D.J., C.J. Sanderson, L.J. LeCaptain, E. Cromartie and G.W. Pendleton. 1992. Interactive effects of selenium, methionine, and dietary protein on survival, growth, and physiology in mallard ducklings. Arch. Environ. Contam. Toxicol. 23: 163-171.

Hoffman, D.J., C.M. Marn, K.C. Marois, E. Sproul, M. Dunne and J.P. Skorupa. 2002. Sublethal effects in avocet and stilt hatchlings from selenium-contaminated sites. Environ. Toxicol. Chem. 21(3): 561–566.

Holm, J. 2002. Sublethal effects of selenium on rainbow trout (*Oncorhynchus mykiss*) and brook trout (*Salvelinus fontinalis*). Masters Thesis. Department of Zoology, University of Manitoba, Winnipeg, MB.

Holm, J., V.P. Palace, K. Wautier, R.E. Evans, C.L. Baron, C. Podemski, P. Siwik and G. Sterling. 2003. An assessment of the development and survival of rainbow trout (*Oncorhynchus*

mykiss) and brook trout (*Salvelinus fontinalis*) exposed to elevated selenium in an area of active coal mining. Proceedings of the 26th Annual Larval Fish Conference 2003, Bergen, Norway.

Holm, J., V.P. Palace, P. Siwik, G. Sterling, R. Evans, C. Baron, J. Werner and K. Wautier. 2005. Developmental effects of bioaccumulated selenium in eggs and larvae of two salmonid species. Environ. Toxicol. Chem. 24: 2373-2381.

Hopkins, W.A., J. Congdon and J.K. Ray. 2000. Incidence and impact of axial malformations in larval bullfrogs (*Rana catesbeiana*) developing in sites polluted by a coal-burning power plant. Environ. Toxicol. Chem. 19: 862-868.

Hothem, R.L. and H.M. Ohlendorf. 1989. Contaminants in foods of aquatic birds at Kesterson Reservoir, California, USA, 1985. Arch. Environ. Contam. Toxicol. 18(6): 773-786.

Huggins, F.E., C.L. Senior, P. Chu, K. Ladwig and G.P. Huffman. 2007. Selenium and arsenic speciation in fly ash from full-scale coal-burning utility plants. Environ. Sci. Technol. 41: 3284-3289.

Hunt, W.G., J.M. Jenkins, R.E. Jackman, C.G. Thelander and A.T. Gerstell. 1992. Foraging ecology of bald eagles on a regulated river. J. Raptor Res. 26(4): 243-256.

Hyne, R.V., A. Padovan, D.L. Parry and S.M. Renaud. 1993. Increased fecundity of the cladoceran *Moinodaphnia macleayi* on a diet supplemented with a green alga, and its use in uranium toxicity tests. Aust. J. Mar. Freshwater Res. 44(3): 389-399.

Jackman, R.E., W.G. Hunt, J.M. Jenkins and P.J. Detrich. 1999. Prey of nesting bald eagles in northern California. J. Raptor Res. 33(2): 87-96.

Janz, D.M. 2011. Selenium. In: Fish Physiology Homeostatis and Toxicology of Essential Metals. Wood, C.M., A.P. Farrell, and C.J. Brauner (Eds). Volume 31 Part A. pages 327-374. Elservier Press

Janz, D. and J. Muscatello. 2008. Standard operating procedure for evaluating selenium-induced deformities in early life stages of freshwater fish. Appendix A in: Selenium tissue thresholds: Tissue selection criteria, threshold development endpoints, and potential to predict population or community effects in the field. North America Metals Council – Selenium Working Group, Washington, DC.

Janz, D.M., D.K. DeForest, M.L. Brooks, P.M. Chapman, G. Gilron, D. Hoff, W.A. Hopkins, D.O. McIntyre, C.A. Mebane, V.P. Palace, J.P. Skorupa and M. Wayland. 2010. Selenium Toxicity to Aquatic Organisms. In: Ecological Assessment of Selenium in the Aquatic Environment. Chapman P.M., Adams W.J., Brooks M.L., Delos C.G., Luoma S.N., Maher W.A., Ohlendorf H.M., Presser T.S., and Shaw D.P. (Eds). SETAC Press, Pensacola, FL, USA.

Jaramillo, F. 2006. Selenium Nutrition of *Morone* Hybrids Including Dietary Requirements, Bioavailability, Toxicity and Effects on Immune Response and Disease Resistance. Ph.D. Dissertation. Texas A&M University. May 2006.

Keating, K.I. and P.B. Caffrey. 1989. Selenium deficiency induced by zinc deprivation in a crustacean. Proc. Nat. Acad. Sci. U.S.A. 86(16): 6436-6440.

Keller, M.D., R.C. Selvin, W. Claus and R.R.L. Guillard. 1987. Media for the culture of oceanic ultraphytoplankton. J. Phycol. 23(4): 633-638.

King, K.A., C.L.H. Marr, A.L. Velasco and M. Schotborgh. 2003. Contaminants in waterbirds, grackles, and swallows nesting on the Lower Colorado River, Arizona, 2000-2001. U.S. Fish and Wildlife Service. Arizona Ecological Services Field Office. Phoenix, AZ. April 2003.

Kupsco, A. and D. Schlenk. 2014. Mechanisms of Selenomethionine Developmental Toxicity and the Impacts of Combined Hypersaline Conditions on Japanese Medaka (*Oryzias lat*ipes). Environ. Sci. Technol. 48: 7062−7068.

Lambing, J.H., W.E. Jones, and J.W. Sutphin. 1988. Reconnaissance investigation of water quality, bottom sediment, and biota associated with irrigation drainage in Bowdoin National Wildlife Refuge and adjacent areas of the Milk River Basin, Northeastern Montana, 1986-87. U.S. Geological Survey. Open-File Report 87-4243. Helena, Montana 1988.

Lambing, J.H., D.A. Nimick and J.R. Knapton. 1994. Physical, chemical, and biological data for detailed study of the Sun River irrigation project, Freezout Lake wildlife management area, and Benton Lake national wildlife refuge, West-Central Montana, 1990-92, with selected data for 1987-89. U.S. Geological Survey. Open-File Report 94-120. Helena, Montana 1994.

Larsen, L.F. and P. Bjerregaard. 1995. The effect of selenium on the handling of mercury in the shore crab *Carcinus maenas*. Mar. Pollut. Bull. 31(1-3): 78-83.

LeBlanc, K. L. and D. Wallschlager. 2016. Production and Release of Selenomethionine and Related Organic Selenium Species by Microorganisms in Natural and Industrial Waters. Environ. Sci. Technol. **50**(12): 6164-6171.

Leschack, C. R., S. K. McKnight, and G. R. Hepp. 1997. Gadwall (*Mareca strepera*), version 2.0. Retrieved from The Birds of North America (A. F. Poole and F. B. Gill, Editors). Cornell Lab of Ornithology, Ithaca, NY, USA. https://doi.org/10.2173/bna.283

Lemly, A.D. 1985. Toxicology of selenium in a freshwater reservoir: Implications for environmental hazard evalutaion and safety. Ecotoxicol. Environ. Safe. 10: 314-338.

Lemly, A.D. 1993a. Teratogenic effects of selenium in natural populations of freshwater fish. Ecotoxicol. Environ. Safe. 26(2): 181-204.

Lemly, A.D. 1993b. Guidelines for evaluating selenium data from aquatic monitoring and assessment studies. Environ. Monitor. Assess. 28(1): 83-100.

Lemly, A.D. 1997. Ecosystem recovery following selenium contamination in a freshwater reservoir. Ecotoxicol. Environ. Saf. 36(3): 275-281.

Lemly, A.D. 2002. *Selenium Assessment in Aquatic Ecosystems*. Springer-Verlag, NY.

Lim, C. and D.M. Akiyama. 1995. Nutrient requirements of penaeid shrimp. In: Nutr. Util. Technol. Aquacult. Lim, C.E. and D.J. Sessa (Eds). AOCS Press: Champaign, IL. pp. 60-73.

Lindstrom, K. 1991. Nutrient requirements of the dinoflagellate *Peridinium gatunense*. J. Phycol. 27(2): 207-219.

Linville, R.G. 2006. Effects of selenium on health and reproduction of white sturgeon *Acinpenser transmontanus*: Implications for San Francisco Bay-Delta. PhD thesis. University of California, Davis, CA, USA.

Low, W.H. and W.H. Mullins. 1990. Reconnaissance investigation of water quality, bottom sediment, and biota associated with irrigation drainage in the American Falls Reservoir area, Idaho, 1988-89. U.S. Geological Survey. Open-File Report 90-4120. Boise, Idaho 1990.

Luckey, T.D. and Venugopal, B. 1977. Metal Toxicity in Mammals. Vol 1. New York, Plenum Press.

Luoma, S.N. and N.S. Fisher. 1997. Uncertainties in assessing contaminant exposure from sediment. In: Ecological risk assessment of contaminated sediment. Ingersol, C.G., T. Dillon and G.R. Biddinger (Eds). SETAC special publication series, Pensacola, FL, pp. 211-237.

Luoma, S.N. and P.S. Rainbow. 2005. Why is metal bioaccumulation so variable? Biodynamics as a unifying concept. Environ. Sci. Technol. 39: 1921-1931.

Luoma, S.N. and P.S. Rainbow. 2008. Metal contamination in aquatic environments. Cambridge (UK): Cambridge Univ. Pr.

Luoma, S.N. and T.S. Presser. 2009. Emerging opportunities in management of selenium contamination: Environ. Sci. Technol. 43: 8483-8487.

Luoma, S.N., C. Johns, N.S. Fisher, N.A. Steinberg, R.S. Oremland and J.R. Reinfelder. 1992. Determination of selenium bioavailability to a benthic bivalve from particulate and solute pathways. Environ. Sci. Technol. 26(3): 485-491.

Lusk, J.D. 1993. Selenium in aquatic habitats at Imperial National Wildlife Refuge. M.S. Thesis. University of Arizona. Tuscon, AZ. 151 pp.

Maher, W., A. Roach, M. Doblin, T. Fan, S. Foster, R. Garrett and D. Wallschläger. 2010. Environmental sources, speciation, and partitioning of selenium. Ecological Assessment of Selenium in the Aquatic Environment, 47-92.

Maier, K.J. and A.W. Knight. 1994. Ecotoxicology of selenium in freshwater systems. Rev Environ. Contam. Toxicol. 134: 31-48.

Martinez, C.T. 1994. Selenium levels in selected species of aquatic birds on Imperial National Wildlife Refuge. M.S. Thesis. University of Arizona. Tuscon, AZ. 74 pp.

Massé, A.J., J.R. Muscatella and D.M. Janz. 2016. Effects of elevated ovo selenium exposure on late stage development of *Xenopus laevis* tadpoles. Bull. Environ. Contam, Toxicol. 97: 463-468.

Massé, A.J., J.R. Muscatello and D.M. Janz. 2015. Dose-dependent early life stage toxicities in *Xenopus laevis* exposed *in ovo* to selenium. Environ. Sci. Technol. 49: 13658-13666.

Mayland, H.F., L.F. James, K.E. Panter and J.L. Sonderegger. 1989. Selenium in Seleniferous Environments. In: Selenium in Agriculture and the Environment, Soil Science Society of America and American Society of Agronomy Special Publication 23: 15-50.

McDonald, L.E. and M.M. Strosher. 1998. Selenium Mobilization from Surface Coal Mining in the Elk River Basin, British Columbia: A Survey of Water, Sediment and Biota. Pollution Prevention Ministry of Environment, Lands and Parks, Cranbrook, British Columbia, Canada.

McLean, J.E. and B.E. Bledsoe. 1992. Ground Water Issue: Behavior of Metals in Soils. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response and Office of Research and Development. EPA/540/S-92/018. October 1992.

Mechaly, A., A. Teplitsky, V. Belakhov, T. Baasov, G. Shoham and Y. Shoham. 2000. Overproduction and characterization of seleno-methionine xylanase T-6. J Biotech. 78: 83-86.

Meseck, S.C. and G.A. Cutter. 2006. Evaluating the biogeochemical cycle of selenium in San Francisco Bay through modeling. Limnol. Oceanog. 51:2018-2032.

Moffitt, J. 1941. Notes on the food of the California Clapper Rail. Condor no. 43: 270-273.

Morrissey C.A., L.I. Bendell-Young and J.E. Elliott. 2004. Linking contaminant profiles to the diet and breeding location of American dippers using stable isotopes. J. Appl. Ecol. 41:502–512.

Morrissey, C.A., J.E. Elliott and S.J. Ormerod. 2010. Diet shifts during egg laying: Implications for measuring contaminants in bird eggs. Environ. Pollut. 158(2): 447-454.

Morrissey, C. A., I.L. Pollet, S.J. Ormerod and J.E. Elliott. 2012. American Dippers Indicate Contaminant Biotransport by Pacific Salmon. Environ. Sci. Technol. 46, 1153−1162.

Moxon, A. L. and O.E. Olson. 1974. Selenium in agriculture. In Selenium (R. A. Zingaro & W. C. Cooper, eds), pp. 675-707. Van Nostrand Reinhold Company, NY.

Muller, M. J. and R. W. Storer. 1999. Pied-billed Grebe (*Podilymbus podiceps*), version 2.0. Retrieved from The Birds of North America (A. F. Poole and F. B. Gill, Editors). Cornell Lab of Ornithology, Ithaca, NY, USA. https://doi.org/10.2173/bna.410

Muscatello, J.R. and D.M. Janz. 2009. Selenium accumulation in aquatic biota downstream of a uranium mining and milling operation. Sci. Total Environ. 407: 1318-1325.

Muscatello, J., P.M. Bennett, K.T. Himbeault, A.M. Belknap and D.M. Janz. 2006. Larval deformities associated with selenium accumulation in northern pike (*Esox lucius*) exposed to metal mining effluent. Environ. Sci. Technol. 40: 6506-6512.

NAS. 1976. National Academy of Sciences (NAS), Medical and biologic effects of environmental pollutants. In: Selenium, Division of Medical Sciences, National Research Council, NAS, Washington, DC (1976), p. 203.

Nautilus Environmental. 2011. Evaluation of the Effects of Selenium on Early Life Stage Development of Westslope Cutthroat Trout from the Elk Valley, BC. Report to Elk Valley Selenium Task Force, November 24, 2011.

Newman, M.C. and S.V. Mitz. 1988. Size dependence of zinc elimination and uptake from water by mosquitofish *Gambusia affinis* (Baird and Girard). Aquat. Toxicol. 12: 17-32.

OECD. 2006. Current approaches in the statistical analysis of ecotoxicology data: A guidance to application. Series on Testing and Assessment, No. 54. Environmental Health and Safety Publications. ENV/JM/MONO(2006)18. Organisation for Economic Co-operation and Development (OECD), Paris, France.

Ogle, R.S. and A.W. Knight. 1996. Selenium bioaccumulation in aquatic ecosystems: 1. Effects of sulfate on the uptake and toxicity of selenate in *Daphnia magna*. Arch. Environ. Contam. Toxicol. 30(2): 274-279.

Ohlendorf, H.M. 2002. The birds of Kesterson Reservoir: A historical perspective. Aquatic Toxicol. 57:1–10.

Ohlendorf, H.M. 2003. Ecotoxicology of selenium. Pages 465-500 in: Handbook of Ecotoxicology. D.J. Hoffman, B.A. Rattner, G.A. Burton Jr. and J. Cairns Jr. (Eds). Lewis Publishers, Boca Raton, FL.

Ohlendorf, H.M and G.H. Heinz 2011. Selenium in birds. In W.N. Beyer & J.P. Meador (Eds), Environmental contaminants in biota: interpreting tissue concentrations (2nd ed.). Boca Raton: CRC Press.

Ohlendorf, H.M. and R.L. Hothem. 1995. Agricultural drainwater effects on wildlife in central California. In Hoffman, D.J. A.B. Rattner, G.A. Burton Jr., and J. Cairns Jr. editors. Handbook of Ecotoxicology. Boca Raton (FL USA): Lewis Publishers. P 577-595.

Ohlendorf, H.M., D.J. Hoffman, M.K. Saiki and T.W. Aldrich. 1986. Embryonic mortality and abnormalities of aquatic birds: Apparent impacts of selenium from irrigation drainwater. Sci. Total Environ. 52: 49-63.

Ohmart, R.D. and R.E. Tomlinson. 1977. Foods of Western Clapper Rails. Wilson Bulletin 89:332–336.

Oliver, M.N., G. Ros-McGauran, D.A. Jessup, B.B. Norman and C.E. Franti. 1990. Selenium concentrations in blood of free-ranging mule deer in California. Trans. West Sec. Wildl. Soc. 26: 80-86.

Oremland, R.S., N.A. Steinberg, A.S. Maest, L.G. Miller and J.T. Hollibaugh. 1990. Measurement of in situ rates of selenate removal by dissimilatory bacterial reduction in sediments. Environ. Sci. Technol. 24: 1157-1164.

Orr, P.L., C.I.E. Wiramanaden, M.D. Paine, W. Franklin and C. Fraser. 2012. Food chain model based on field data to predict westslope cutthroat trout (*Oncorhynchus clarkii lewisi*) ovary selenium concentrations from water selenium concentrations in the Elk Valley, British Columbia. Environ. Toxicol. Chem. 31: 672-680.

Ort, J.F. and J.D. Latshaw. 1977. The effect of intermediate levels of selenium in the diets of S.C.W.L. laying hens. Poultry Sci. 56: 1744.

Ort, J.F. and J.D. Latshaw. 1978. The toxic level of sodium selenite in the diet of laying chickens. J. Nutr. 108: 1114-1120.

Osmundson, B.C., T. May, J. Skorupa and R. Krueger. 2007. Selenium in fish tissue: Prediction equations for conversion between whole body, muscle, and eggs. Poster presentation at the 2007 Annual Meeting of the Society of Environmental Toxicology and Chemistry, and three supporting unpublished spreadsheets. Milwaukee, WI, USA.

O'Toole, D. and M.F. Raisbeck. 1997. Experimentally induced selenosis of adult mallard ducks: clinical signs, lesions, and toxicology. Vet. Pathol. 34: 330-340.

Ouerdane, L., F. Aureli, P. Flis, K. Bierla, H. Preud'homme, F. Cubadda and J. Szpunar. 2013. Comprehensive speciation of low-molecular weight selenium metabolites in mustard seeds using HPLC – electrospray linear trap/orbiting tandem mass spectrometry. Metallomics 5: 1294-1304.

Palace, V.P., J.E. Spallholz, J. Holm, K. Wautier, R.E. Evans, and C.L. Baron. 2004. Metabolism of selenomethionine by rainbow trout (*Oncorhynchus mykiss*) embryos can generate oxidative stress. Ecotoxicol. Environ. Safe. 58: 17-21.

Peterson, D.A, T.F. Harms, P. Ramirez, Jr., G.T. Allen, and A.H. Christenson. 1991. Reconnaissance investigation of water quality, bottom sediment, and biota associated with irrigation drainage in the Riverton Reclamation Project, Wyoming, 1988-89. U.S. Geological Survey. Open-File Report 90-4187. Cheyenne, Wyoming 1991.

Petrov, P.K., J.W. Charters and D. Wallschlager. 2012. Identification and determination of selenosulfate and selenocyanate in flue gas desulfurization waters. Environ. Sci. Technol. 46: 1716-1723.

Presser, T.S. 2013. Selenium in Ecosystems within the Mountaintop Coal Mining and Valley-Fill Region of Southern West Virginia - Assessment and Ecosystem-Scale Modeling. U.S. Department of the Interior, U.S. Geological Survey, Professional Paper 1803.

Presser, T.S. and S.N. Luoma. 2006. Forecasting selenium discharges to the San Francisco Bay-Delta estuary: ecological effects of a proposed San Luis drain extension. Professional Paper 1646, U.S. Department of Interior, U.S. Geological Survey, Reston, Virginia.

Presser, T.S. and S.N. Luoma. 2010. A Methodology for Ecosystem-Scale Modeling of Selenium. Integr. Environ. Asses. 6: 685-710.

Presser, T.S. and H.M. Ohlendorf. 1987. Biogeochemical cycling of selenium in the San Joaquin Valley, California, USA. Environ. Manage. 11(6): 804-822.

Presser, T.S., M.A. Sylvester and W.H. Low. 1994. Bioaccumulation of selenium from natural geologic sources in western states and its potential consequences. Environ. Manage. 18(3): 423-436.

Presser, T.S., D.Z. Piper, K.J. Bird, J.P. Skorupa, S.J. Hamilton, S.J. Detwiler and M.A. Huebner. 2004. The Phosphoria Formation: A Model for Forecasting Global Selenium Sources in the Environment, Handbook of Exploration and Environmental Chemistry, Vol. 8 (Hein J.R., editor) pp 300-303.

Price, N.M. 1987. Urea and selenium nutrition of marine phytoplankton: a physiological and biochemical study. Avail. NLC, From: Diss. Abstr. Int. B 49(5): 1498-1499.

Price, N.M., P.A. Thompson and P.J. Harrison. 1987. Selenium: An essential element for growth of the coastal marine diatom *Thalassiosira pseudonana* (Bacillariophyceae). J. Phycol. 23(1): 1-9.

Ranjard, L., S. Nazaret and B. Cournoyer. 2003. Freshwater bacteria can methylate selenium through the thiopurine methyltransferase pathway. Appl. Environ. Micorbiol. 69(7): 3784-3790.

Ratti, J.T., A.M. Moser, E.O. Garton and R. Miller. 2006. Selenium levels in bird eggs and effects on avian reproduction. J. Wildlife Manage. 70(2): 572-578.

Reddy, C.C. and E.J. Massaro. 1983. Biochemistry of selenium: an overview. Fundam. Appl. Toxicol. 3: 431-436.

Riedel, G.F. and J.G. Sanders. 1996. The influence of pH and media composition on the uptake of inorganic selenium by *Chlamydomonas reinhardtii*. Environ. Toxicol. Chem. 15(9): 1577-1583.

Rinella, F.A., and C.A. Schuler. 1992. Reconnaissance investigation of water quality, bottom sediment, and biota associated with irrigation drainage in the Malheur National Wildlife Refuge, Harney County, Oregon, 1988-89. Open-File Report 91-4085. Portland, Oregon 1992.

Rinella, F.A., W.H. Mullins, C.A. Schuler. 1994. Reconnaissance investigation of water quality, bottom sediment, and biota associated with irrigation drainage in the Owyhee and Vale projects, Oregon and Idaho, 1990-91. U.S. Geological Survey. Water-Resources Investigations Report 93-4156. Portland, Oregon 1994.

Rudolph, B.-L., I. Andreller and C.J. Kennedy. 2008. Reproductive success, early life stage development, and survival of Westslope cutthroat trout (*Oncorhynchus clarkii lewisi*) exposed to elevated selenium in an area of active coal mining. Environ. Sci. Technol. 42: 3109-3114.

Rush, S.A., K.F. Gaines, W.R. Eddleman and C.J. Conway. 2012. Clapper Rail (*Rallus longirostris*), Retrieved from the Birds of North America Online (A. Poole, Ed.). Cornell Lab of Ornithology, Ithaca, NY, USA.
http://bna.birds.cornell.edu.bnaproxy.birds.cornell.edu/bna/species/340

Saiki, M.K. and T.P. Lowe. 1987. Selenium in aquatic organisms from subsurface agricultural drainage water, San Joaquin Valley, California (USA). Arch. Environ. Contam. Toxicol. 16(6): 657-670.

Saiki, M.K., M.R. Jennings and W.G. Brumbaugh. 1993. Boron, molybdenum and selenium in aquatic food chains from the Lower San Joaquin River and its tributaries California. Arch. Environ. Contam. Toxicol. 24(3): 307-319.

Schlekat, C.E., B.G. Lee and S.N. Luoma. 2002. Assimilation of selenium from phytoplankton by three benthic invertebrates: effect of phytoplankton species. Mar. Ecol. Prog. Ser. 237: 79-85.

Schultz, R. and R. Hermanutz. 1990. Transfer of toxic concentrations of selenium from parent to progeny in the fathead minnow (*Pimephales promelas*). Bull. Environ. Contam. Toxicol. 45: 568-573.

Seiler, R., J.P. Skorupa and L.A. Peltz. 1999. Areas Susceptible to Irrigation-Induced Selenium Contamination of Water and Biota in the Western United States. U.S. Geological Survey Circular, Volumes 1180.

Seiler, R., J. Skorupa, D. Naftz and B. Nolan. 2003. Irrigation-induced contamination of water, sediment, and biota in the Western United States: synthesis of data from the National Irrigation Water Quality Program. Denver, CO: U.S. Geological Survey Professional Paper.

Simmons, D.B. and D. Wallschläger. 2005. A critical review of the biogeochemistry and ecotoxicology of selenium in lotic and lentic environments. Environ. Toxicol. Chem. 24: 1331-1343.

Simmons, D. B. and D. Wallschlager. 2011. Release of reduced inorganic selenium species into waters by the green fresh water algae Chlorella vulgaris. Environ. Sci. Tech. 45: 2165-2171.

Skorupa, J.P. and H.M. Ohlendorf. 1991. Contaminants in drainage water and avian risk thresholds. In: The Economics and Management of Water and Drainage in Agriculture. Dinar, A. and D. Zilberman (Eds). Kluwer Academic Publishers, Boston, pp. 345-368.

Skorupa, J.P., S.P. Mormon and J.S. Sefchick-Edwards. 1996. Guidelines for interpreting selenium exposures of biota associated with nonmarine aquatic habitats. Report to the National Irrigation Water Quality Program (NIWQP). U.S. Fish and Wildlife Service, Sacramento, CA.

Skorupa 1998a. Chapter 18. Selenium poisoning in fish and wildlife in nature: Lessons from 12 real-world examples. In Frankenberger W.T. and R.A. Engberg, eds, Environmental Chemistry of Selenium. Marcel Dekker, Inc. New York, NY, pp. 315–354.

Skorupa 1998b. Risk assessment for the biota database of the National Irrigation Water Quality Program. U.S. Fish and Wildlife Service, Sacramento, CA.

Smith, G.J., G.H. Heinz, D.J. Hoffman, J.W. Spann and A.J. Krynitsky. 1988. Reproduction in black-crowned night-herons fed selenium. Lake Reservoir Mgt. 4(2): 175-180.

Sotherland, P.R. and H. Rahn. 1987. On the composition of bird eggs. The Condor 89: 48-65.

Stadtman, T.C. 1996. Selenocysteine. Ann. Rev. Biochem. 65: 83-100.

Stanley Jr., T.R., J.W. Spann, G.J. Smith and R. Rosscoe. 1994. Main and interactive effects of arsenic and selenium on mallard reproduction and duckling growth and survival. Arch. Environ. Contam. Toxicol. 26:444-451.

Stanley Jr., T.R., G.J. Smith, D.J. Hoffman, G.H. Heinz and R. Rosscoe. 1996. Effects of boron and selenium on mallard reproduction and duckling growth and survival. Environ. Toxicol. Chem. 15:1124-1132.

Stephan, C.E., D.I. Mount, D.J. Hansen, J.H. Gentile, G.A. Chapman and W.A. Brungs. 1985. Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and their Uses. PB85-227049. National Technical Information Service, Springfield, VA.

Stephens, D.W., B.W. Waddell and J.B. Miller. 1988. Reconnaissance investigation of water quality, bottom sediment and biota associated with irrigation drainage in the Middle Green River basin, Utah, 1986-87. U.S.G.S. Water-Resources Investigations Report 88-4011.

Stewart, A.R., S.N. Luoma, C.E. Schlekat, M.A. Doblin and K.A. Hieb. 2004. Food web pathway determines how selenium affects aquatic ecosystems: a San Francisco Bay case study. Environ. Sci. Technol. 38:4519-4526.

Stewart, R., M. Grosell, D. Buchwalter, N.S. Fisher, S.N. Luoma, T. Mathews, P. Orr and W.X. Wang. 2010. Bioaccumulation and trophic transfer of selenium. Ecological Assessment of Selenium in the Aquatic Environment, Edited by Peter M. Chapman, Pensacola, FL, p. 91-137.

Stewart, A. R., S. N. Luoma, K. A. Elrick, J. L. Carter and M. van der Wegen. 2013. Influence of estuarine processes on spatiotemporal variation in bioavailable selenium. Mar. Ecol. Prog. Ser. 492: 41–56.

Sunde, R.A. 1984. The biochemistry of selenoproteins. J. Am. Oil Chem. Soc. 61: 1891-1900.

Thomann, R. V. 1981. Equilibrium model of fate of microcontaminants in diverse aquatic food chains. Can. J. Fish. Aquat. Sci. 38: 280-296.

Thompson, P.A. and W. Hosja. 1996. Nutrient limitation of phytoplankton in the Upper Swan River Estuary, Western Australia. Mar. Freshwater Res. 47(4): 659-667.

Thorarinsson, R., M.L. Landolt, D.G. Elliott, R.J. Pascho and R.W. Hardy. 1994. Effect of dietary vitamin E and selenium on growth, survival and the prevalence of *Renibacterium salmoninarum* infection in Chinook salmon (*Oncorhynchus tshawytscha*). Aquacult. 121(4): 343-358.

Tokunaga, T.K., G.E. Brown Jr., I.J. Pickering, S.R. Sutton and S. Bajt. 1997. Selenium redox reactions and transport between ponded waters and sediments. Environ. Sci. Technol. 31: 1419-1425.

Twedt, D. J. and R. D. Crawford. 1995. Yellow-headed Blackbird (*Xanthocephalus xanthocephalus*), version 2.0. Retrieved from The Birds of North America (A. F. Poole and F. B. Gill, Editors). Cornell Lab of Ornithology, Ithaca, NY, USA. https://doi.org/10.2173/bna.192

Unrine, J.M., W.A. Hopkins, C.S. Romanek and B.P. Jackson. 2007. Bioaccumulation of trace elements in omnivorous amphibian larvae: Implications for amphibian health and contaminant transport. Environ. Pollut. 149: 182-192.

U.S. DOI (United States Department of the Interior, Bureau of Reclamation/Fish and Wildlife Service/Geological Survey/Bureau of Indian Affairs). 1998. Guidelines for Interpretation of the Biological Effects of Selected Constituents in Biota, Water, and Sediment. National Irrigation Water Quality Program Information Report No. 3. Bureau of Reclamation, Denver, CO. 198 p.

U.S. EPA (U.S. Environmental Protection Agency). 1987. Ambient water quality criteria for selenium. EPA-440/5-87-006. National Technical Information Service.

U.S. EPA (U.S. Environmental Protection Agency). 1991. Technical Support Document For Water Quality-based Toxics Control. Office of Water. EPA/505/2-90-001.

U.S. EPA (U.S. Environmental Protection Agency). 1995. Great Lakes Water Quality Initiative Criteria Documents for the Protection of Wildlife: DDT, Mercury, 2,3,7,8-TCDD, PCBs. Office of Water. EPA/820/B-95/008.

U.S. EPA (U.S. Environmental Protection Agency). 1997. Mercury Study Report to Congress, Volume VII: Characterization of Human Health and Wildlife Risks from Mercury Exposure in the United States. Office of Research and Development. EPA-452/R-97-009.

U.S. EPA (U.S. Environmental Protection Agency). 1998. Report on the Peer Consultation Workshop on Selenium Aquatic Toxicity and Bioaccumulation. Office of Water. EPA-822-R-98-007.

U.S. EPA (U.S. Environmental Protection Agency). 2003. Methodology for Deriving Ambient Water Quality Criteria for the Protection of Human Health (2000) Volume 2: Development of National Bioaccumulation Factors. Office of Water. EPA-882-R-03-030. https://nepis.epa.gov/Exe/ZyPDF.cgi/P1005EZQ.PDF?Dockey=P1005EZQ.PDF

U.S. EPA (U.S. Environmental Protection Agency). 2004. Draft aquatic life water quality criteria for selenium - 2004. Office of Science and Technology, Office of Water. USEPA-822-D-04-001.

U.S. EPA (U.S. Environmental Protection Agency). 2011a. Re: EPA Action on the Gilbert Bay Selenium Criterion and Footnote (14). U.S. EPA Region 8. Denver, CO 80202. December 12, 2011.

U.S. EPA (U.S. Environmental Protection Agency). 2011b. TRAP. Mid-Continent Ecology Division. Accessed 28 April 2011. Available online at: http://www.epa.gov/med/Prods_Pubs/trap.htm.

U.S. EPA (U.S. Environmental Protection Agency). 2012. Ecological Effects Test Guidelines. OCSPP 850.2300: Avian Reproduction Test. EPA 712-C-02.

U.S. EPA (U.S. Environmental Protection Agency). 2013. The revised deletion process for the site-specific recalculation procedure for aquatic life criteria. EPA 822-R-13-001. National Technical Information Service, Springfield, VA.

U.S. EPA (U.S. Environmental Protection Agency). 2014. External Peer Review Draft Aquatic Life Ambient Water Quality Criteria for Selenium – Freshwater. EPA 822-P-14-001. Office of Water.

U.S. EPA (U.S. Environmental Protection Agency). 2016a. Aquatic Life Ambient Water Quality Criteria for Selenium – Freshwater 2016. EPA 822-R-16-006. Office of Water, Office of Science and Technology, Washington, DC.

U.S. EPA (U.S. Environmental Protection Agency). 2016b. Technical support document for the proposed aquatic life and aquatic-dependent wildlife selenium water quality criteria for the San Francisco Bay and delta. U.S. EPA Region 9. Water Division. San Francisco, CA. June, 2016.

U.S. FWS (U.S. Fish and Wildlife Service). 1985. Recovery plan for the California least tern, *Sterna antillarum brownii*. U.S. Fish and Wildlife Service, Portland, Oregon 112. pp.

U.S. FWS (U.S. Fish and Wildlife Service). 2003. Evaluation of the Clean Water Act Section 304(a) human health criterion for methylmercury: protectiveness for threatened and endangered wildlife in California. U.S. Fish and Wildlife Service, Sacramento Fish and Wildlife Office, Environmental Contaminants Division. Sacramento, California. 96 pp

U.S. FWS (U.S. Fish and Wildlife Service). 2010. Yuma Clapper Rail Recovery Plan (*Rallus longirostris yumanensis*). Draft First Revision. February 10, 2010. Albuquerque, New Mexico: U.S. Fish and Wildlife Service, Southwest Region.

U.S. FWS (U.S. Fish and Wildlife Service). 2017. Species at risk from selenium exposure in California inland surface waters, enclosed bays and estuaries. Final report to the U.S. EPA. Inter-Agency Agreement No. DW-14-95825001-0. Sacramento, CA. October 2017.

Walls, S.J., C.B. Meyer, J. Iannuzzi and T.H. Schlekat. 2015. Effects of coal fly ash on tree swallow reproduction in Watts Bar Reservoir, Tennessee. Integrat. Env. Assess. 11(1): 56-66.

Wang, C. and R.T. Lovell. 1997. Organic selenium sources, selenomethionine and selenoyeast, have higher bioavailability than an inorganic selenium source, sodium selenite, in diets for channel catfish (*Ictalurus punctatus*). Aquacult. 152(1-4): 223-234.

Wang, W.X. 2001. Comparison of metal uptake rate and absorption efficiency in marine bivalves. Environ. Toxicol. Chem. 20(6): 1367-1373.

Wang, W.X., N.S. Fisher and S.N. Luoma. 1996. Kinetic determinations of trace element bioaccumulation in the mussel *Mytilus edulis*. Mar. Ecol. Prog. Ser. 140(1-3): 91-113.

Watanabe, T., V. Kiron and S. Satoh. 1997. Trace minerals in fish nutrition. Aquaculture 151: 185-207.

Weech, S.A., A.M. Scheuhammer and M.E. Wayland. 2012. Selenium accumulation and reproduction in birds breeding downstream of a uranium mill in northern Saskachewan, Canada. Ecotoxicol. 21: 280-288.

Wehr, J.D. and L.M. Brown. 1985. Selenium requirement of a bloom-forming planktonic alga from softwater and acidified lakes. Can. J. Fish. Aquat. Sci. 42: 1783-1788.

Wiemeyer, S.N. and D.J. Hoffman. 1996. Reproduction in Eastern screech-owls fed selenium. J. Wildlife Manage. 60(2): 332-341.

Willson, M. F. and H. E. Kingery. 2011. American Dipper (*Cinclus mexicanus*), version 2.0. Retrieved from The Birds of North America (A. F. Poole, Editor). Cornell Lab of Ornithology, Ithaca, NY, USA. https://doi.org/10.2173/bna.229

Wilson, D.S., P. Zhang, R. He, R. Ota and S.T. Omaye. 1997. Kinetics of selenium incorporation into tissues of female mallard ducks. Toxicol. 122: 51-60.

Winner, R.W. 1989. Multigeneration life-span tests of the nutritional adequacy of several diets and culture waters for *Ceriodaphnia dubia*. Environ. Toxicol. Chem. 8(6): 513-520.

Winner, R.W. and T.C. Whitford. 1987. The interactive effects of a cadmium stress, a selenium deficiency and water temperature on the survival and reproduction of *Daphnia magna* Straus. Aquat. Toxicol. (Amsterdam) 10(4): 217-224.

Yamamoto, J.T. and G.M. Santolo. 2000. Body condition effects in American kestrels fed selenomethionine. J. Wildlife Diseas. 36(4): 646-652.

Yasukawa, K. and W. A. Searcy. 1995. Red-winged Blackbird (*Agelaius phoeniceus*), version 2.0. Retrieved from The Birds of North America (A. F. Poole and F. B. Gill, Editors). Cornell Lab of Ornithology, Ithaca, NY, USA. https://doi.org/10.2173/bna.184

Yuan, T., A.M. Weljie and H.J. Vogel. 1998. Tryptophan fluorescence quenching by methionine and selenomethionine residues of calmodulin: orientation of peptide and protein binding. Biochemistry 37: 3187-3195.

Zhang, Y. and J.N. Moore. 1996. Selenium fractionation and speciation in a wetland system. Environ. Sci. Technol. 30: 2613-2619.

Appendix A SUMMARY INFORMATION FOR QUANTITATIVE AND QUALITATIVE BIRD STUDIES

Summary

The following three tables include summary information from the studies considered for the derivation of the aquatic-dependent wildlife egg criterion described in Part 4. Appendix Table A-1 summarizes the three mallard studies that were combined to calculate the mallard egg EC10 in Part 4.2.1. Appendix Table A-2 summarizes bird studies with reproductive endpoints that provided qualitative support for the final egg tissue criteria described in Part 4.6.1. Appendix Table A-3 summarizes bird studies with non-reproductive endpoints that provided qualitative support for the final egg tissue criteria described in Section 4.6.2.

Appendix Table A-1. Quantitative aquatic-dependent wildlife toxicity data considered and used for criterion development.

Data from these studies were combined into a meta-analysis with a resulting EC_{10} for egg hatchability of 11.2 mg egg/kg dw.

Species, Life Stage	Animal Origin/Site	Maternal Age	Chemical Form	Exposure Media Type and Duration	Selected Sensitive Endpoint or Effect	Toxicity Value		Reference
						Diet mg/kg ww	Egg[a] mg/kg dw	
Anseriformes (Ducks, Geese, and Swans)								
Mallard, *Anas platyrhynchos*, Adult	Game farm	2 years old	Seleno- DL-methionine	Dietary (3 week pre- breeding exposure through end of egg laying)	Hatchability	NOEC: Control[b] LOEC: 10[b]	NOEC: 0.17 LOEC: 15.3	Heinz et al. 1987
Mallard, *Anas platyrhynchos*, Adult	Game farm	2 years old	Seleno- DL-methionine	Dietary (Duration not specified, but >100 days.)	Hatchability	NOEC: 8[b] LOEC: 16[b]	NOEC: 36.7 LOEC: 60	Heinz et al. 1989
Mallard, *Anas platyrhynchos*, Adult	Game farm (Outdoorsman Hunting Club, Webb, IA)	1 year old	Seleno- DL-methionine	Dietary 120-122 days	Hatchability	NOEC: 3.5[b] LOEC: 7[b]	NOEC: 12.07 LOEC: 24.48	Stanley et al. 1996

[a] All egg concentrations are measured from a subset of eggs
[b] Nominal
[c] Measured

Appendix Table A-2. Qualitative aquatic-dependent wildlife toxicity reproductive data considered for criterion development.

Species, Life Stage	Animal Origin/Site	Maternal Age	Chemical Form	Exposure Media Type and Duration	Selected Sensitive Endpoint or Effect	Toxicity Value		Reference
						Diet mg/kg ww	Egg[a] mg/kg dw	
Anseriformes (Ducks, Geese, and Swans)								
Mallard, *Anas platyrhynchos*, Adult	Game farm	Not specified	Seleno- DL-methionine	Dietary (4 week pre-breeding exposure through end of egg laying)	Hatchability	NOEC: control[b] LOEC: 10[b]	NOEC: 1.35 LOEC: 30.4	Heinz and Hoffman 1996
Mallard, *Anas platyrhynchos*, Adult	Game farm	18 month old (Whistling Wings, Hanover , Il)	Seleno- DL-methionine	Dietary (4 week pre-breeding exposure through end of egg laying)	Hatchability	NOEC: control[b] LOEC: 10[b]	NOEC: 1.16 LOEC: 25.1	Heinz and Hoffman 1998
Mallard, *Anas platyrhynchos*, Adult	Game farm (Frost Waterfowl Trust, Coloma, WI)	1 year old	Seleno- DL-methionine	Dietary 21 weeks	onset of egg-laying	NOEC: control[b] LOEC: 15[b]		Heinz and Fitzgerald 1993a
Mallard, *Anas platyrhynchos*, Adult	Game farm	2 years old	selenite	Dietary Duration not specified, but >100 days	percentage of abnormal embryos[c]	NOEC: 5[b] LOEC: 10[b]	NOEC: 0.6 LOEC: 1.77	Hoffman and Heinz 1988
Mallard, *Anas platyrhynchos*, Adult	Game farm	2 years old	Seleno- DL-methionine	Dietary Duration not specified, but >100 days	percentage of malformed embryos[c]	NOEC: 4[b] LOEC: 8[b]	NOEC: 11.3 LOEC: 36.7	

Species,		Maternal				Toxicity Value		Reference
Mallard, *Anas platyrhynchos*, Adult	Game farm (Frost Waterfowl Trust, Coloma, WI)	1 year old	Seleno- DL-methionine	Dietary 115-124 days	Hatchability	NOEC: 0.37[c] (dw) LOEC: 6.5[c] (dw)	NOEC: 1.6 LOEC: 42	Stanley et al. 1994
Pelecaniformes (Pelicans, Herons, Ibises, and Allies)								
Black-crowned night heron, *Nycticorax nycticorax* Adult	Obtained from captive breeding colony. Patuxent Wildlife Center. Laurel, MD	Not specified	Seleno- DL-methionine		Hatchability, Malformations	NOEC: >10[b]	NOEC: >3.3 (ww)	Smith et al 1988
Strigiformes (Owls)								
Eastern screech owl, *Megascops asio* Adult	Captive birds	3-4 years old	Seleno- DL-methionine (measured in egg and diet)	Dietary Duration not specified. Through clutch completion.	Femur length Nestling survival to 5 d Adult weight	NOEC: 8.81[d] (dw) LOEC: 30[d] (dw)	NOEC: 2.57 (ww) LOEC: 7.44 (ww)	Wiemeyer 1996
Charadriiformes (Plovers, Sandpipers, and Allies)								
American avocet, *Recurvirostra americana* Eggs and nestlings of field-exposed adults	Collected eggs from north and south Tulare Lake drainage district, CA; and Westfarmers, CA	n/a (field collected eggs)	Naturally occurring Se	Field exposure. Lifetime maternal exposure.	Chick weight	n/a	NOEC: 6.7 LOEC: 31.4 (weights were 7% lower at high Se site vs. low Se site (3.3 mg/kg dw egg)	Hoffman et al. (2002)

Species,		Maternal				Toxicity Value		Reference
Black necked stilt; *Himantopus mexicanus* Eggs and nestlings of field-exposed adults	Collected eggs from north and south Tulare Lake drainage district, CA; and Westfarmers, CA	n/a (field collected eggs)	Naturally occurring Se	Field exposure. Lifetime maternal exposure.	Chick weight	n/a	NOEC: >20.5 (no differences across sites)	Hoffman et al. (2002)
Spotted sandpiper, *Actitis macularia* Eggs and nestlings of field-exposed adults	5 reference and 3 Se exposed areas in S. Alberta, CA	n/a (nest observations)	Naturally occurring Se	Field exposure. Lifetime maternal exposure.	Fledglings per nest.	n/a	NOEC: >7.3	Harding et al. (2005)
Passeriformes (Perching Birds)								
Tree swallows, *Tachycineta bicolor* Eggs and nestlings of field-exposed adults	1 reference site and four Se impacted sites near Key Lake, northern Saskatchewan	n/a (nest observations)	Naturally occurring Se	Field exposure. Lifetime maternal exposure.	Mean clutch size Hatchability Nestling growth	n/a	NOEC: >9 (average egg concentration at site with highest Se.	Weech et al. (2012)
Tree swallows, *Tachycineta bicolor* Eggs and nestlings of field-exposed adults	Watts Bar Reservoir, TN (6 impacted sites and one reference site)	n/a (nest observations)	Naturally occurring Se	Field exposure. Lifetime maternal exposure.	Hatching success	n/a	NOEC:>3.15-4.75[e]	Walls et al. (2015)

Species,		Maternal				Toxicity Value		Reference
Red-winged blackbirds, *Agelaius phoeniceus* Eggs and nestlings of field-exposed adults	Elk River Valley, SE British Columbia	n/a (nest observations)	Naturally occurring Se	Field exposure. Lifetime maternal exposure.	Hatchability	n/a	Point of downward inflection of quadratic curve[f] :22	Harding 2008
Red-winged blackbirds, *Agelaius phoeniceus* Eggs and nestlings of field-exposed adults	SE Idaho in the vicinity of Soda Springs	n/a (nest observations)	Naturally occurring Se	Field exposure. Lifetime maternal exposure.	All measured endpoints (nest success, clutch size, chicks hatched/fledged, egg/hatchling weight)	n/a	NOEC:>7.18	Ratti et al. 2006
American robin, *Turdus migratorius* Eggs and nestlings of field-exposed adults	SE Idaho in the vicinity of Soda Springs	n/a (nest observations)	Naturally occurring Se	Field exposure. Lifetime maternal exposure.	All measured endpoints (nest success, clutch size, chicks hatched/fledged, egg/hatchling weight)	n/a	NOEC:>4.48	Ratti et al. 2006

[a] All egg concentrations are measured from a subset of eggs
[b] Nominal
[c] Abnormal embryos includes those with malformations, edema, or stunted growth. Malformed embryos do not include those with edema.
[d] Measured
[e] Hatching success was significantly lower at two of the six impacted sites compared to reference. However, differences could not be attributable to Se because of multiple potential co-contaminant. No combinations of potential contaminants could explain the differences in hatching success.
[f] Not statistically significant.

Appendix Table A-3. Qualitative aquatic-dependent wildlife toxicity non-reproductive data considered for criterion development.

Species, Life Stage	Animal Origin/Site	Chemical Form	Exposure Media Type and Duration	Selected Sensitive Endpoint or Effect	Toxicity Value of Diet mg/kg ww	Reference
Anseriformes (Ducks, Geese, and Swans)						
Mallard, *Anas platyrhynchos*, Adult males	Game farm	Seleno- DL-methionine	Dietary 16 weeks	Number of molts completed by 16 weeks	NOEC: 20[a] LOEC: 40[a]	Albers 1996
Lesser Scaup, *Aythya affinis* Adult	Captive-reared	Seleno- DL-methionine	Dietary 4 months	Survival Weight	NOEC: >20.66[b] (dw)	Brady et al. 2013
Lesser Scaup, *Aythya affinis* Adult	Captive-reared	Seleno- DL-methionine	Dietary 30 days	Survival	NOEC: >14.9[b] (dw)	DeVink et al. 2008
Mallard, *Anas platyrhynchos*, 1-day old hatchlings	Game farm (Spring Farm, Sag Harbor, NY)	Selenite	Dietary 6 weeks	Food consumption	NOEC: 10[a] LOEC: 20[a]	Heinz et al. 1988
Mallard, *Anas platyrhynchos*, 1-day old hatchlings	Game farm (Spring Farm, Sag Harbor, NY)	Seleno- DL-methionine	Dietary 6 weeks	Food consumption	NOEC: 10[a] LOEC: 20[a]	
Mallard, *Anas platyrhynchos*, Adult males	Game farm (Frost Waterfowl Trust, Coloma, WI)	Seleno- DL-methionine	Dietary 21 weeks+12 weeks all control+5 weeks[c]	Survival Weight	NOEC:>15[a,d]	Heinz 1993
Mallard, *Anas platyrhynchos*, Adult males	Game farm (Frost Waterfowl Trust, Coloma, WI)	Seleno- DL-methionine	Dietary 16 weeks	Survival Weight	NOEC: 10[a] LOEC: 20[a]	Heinz and Fitzgerald 1993b

Species, Life Stage	Animal Origin/Site	Chemical Form	Exposure Media Type and Duration	Selected Sensitive Endpoint or Effect	Toxicity Value of Diet mg/kg ww	Reference
Mallard, *Anas platyrhynchos*, 1-day old hatchlings	Game farm (Oak Ridge Game Farm, Gravette, AR)	Seleno- DL-methionine	Dietary 4 weeks	Weight (standard-22%-protein diet)	NOEC: 15[a] LOEC: 60[a]	Hoffman et al. 1992
Mallard, *Anas platyrhynchos*, 1-day old hatchlings	Game farm (Oak Ridge Game Farm, Gravette, AR)	Seleno- DL-methionine	Dietary 4 weeks	Survival (low-11%-protein diet)	NOEC: 15[a] LOEC: 60[a]	
Mallard, *Anas platyrhynchos*, 1-day old hatchlings	Game farm (Oak Ridge Game Farm, Gravette, AR)	Seleno- DL-methionine	Dietary 4 weeks	Weight (high-44%-protein diet)	NOEC: control[a] LOEC: 15[a]	
Mallard, *Anas platyrhynchos*, Flightling males	Game farm (Whistling Wings, Inc., Hanover, Il)	Seleno- DL-methionine	Dietary 150 days	Alopecia	NOEC: 10[a] LOEC: 25[a]	O'Toole and Raisbeck 1997
Falconiformes (Falcons and Caracaras)						
American kestrel, *Falco sparverius* Adult	Captive-reared (McGill University, Montreal, Quebec)	Seleno- DL-methionine	Dietary 77 days	Lean mass 49 d after end of exposure[e]	NOEC: 6.3[b] (dw) LOEC: 12[b] (dw)	Yamamoto and Santolo 2000

[a] nominal
[b] measured
[c] treatment group fed 15 mg/kg Se for 21 weeks, then control diet for 12 weeks, then both groups fed 100 mg/kg Se for 5 weeks.
[d] No effects at 15 mg/kg diet. Effects for all endpoints at 100 mg/kg diet regardless of pre-exposure.
[e] Weights were not measured prior to Se exposure. No statistically significant differences in mass on final day of exposure.

Appendix B CALCULATION OF TROPHIC TRANSFER FACTORS

Paired Data Used to Calculate Bird Trophic Transfer Factors (*TTF*)

The following tables (Appendix Table B-1 through Appendix Table B-11) list paired data used to calculate bird trophic transfer factors (*TTF*) listed in Table 5-3, which were then used to calculate the bird composite *TTF*s listed in Table 5-5 and Table 5-6. As described in Part 5.4.2.1, for each species, quantitative food web information was used to determine the proportion of the diet consisting of plants (including algae) and animals. Using these proportions, a weighted diet (plant+animal) selenium concentration was calculated for every site where an egg selenium measurement was paired with both a plant and animal selenium measurement, as follows.

$$Diet\ Se = (Plant\ Se \times Plant\ Diet\ Proportion) + (Invertebrate\ Se \times Invertebrate\ Se\ Proportion)$$

In order to be considered, paired data were required to be collected at the same site within a one-year period. The one-year period for matched data is based on an analysis described in U.S. EPA (2016a) suggesting the relationship between selenium in paired tissue is insensitive to collection time within one year.

The relationship between paired egg and weighted diet selenium concentrations was evaluated using linear regression following natural log transformation after removing outliers. For each regression model, outliers were identified by examining four residual plots: residual vs. fitted values; standardized residuals vs. theoretical quantiles (Q-Q plot); square root of standardized residuals vs. fitted values; and standardized residuals vs. leverage (Cook's distance). An observation was identified as an outlier or overly influential if the observation was greater than the 50th percentile in the Cook's distance plot, or if it was identified as an outlier in three of the four plots listed above. Up to three passes of this outlier analysis was performed for each regrerssion model, after removing outliers from previous passes. If the slope of a set of matched pairs of selenium measurements was both positive and statistically significant ($P \leq 0.05$), then the relationship between selenium in the target bird species and the food it consumes is considered adequately represented by the available data.

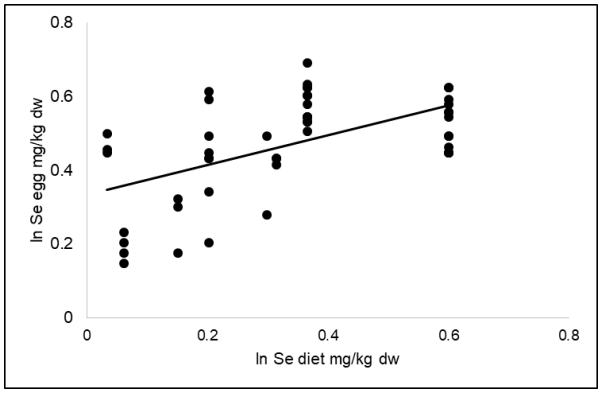
Appendix Table B-1. American Avocet. Bird Egg to Diet Trophic Transfer Factor (*TTF*).

Rows with data pairs that were removed during outlier analysis are identified with bold and italics.

Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
Lambing et al. 1994	B-23	1.80	0.13	2.40	0.87	2.32	4.2	1.81
Lambing et al. 1994	B-23	1.80	0.13	2.40	0.87	2.32	4	1.72
Lambing et al. 1994	B-23	1.80	0.13	2.40	0.87	2.32	3.5	1.51
Lambing et al. 1994	B-23	1.80	0.13	2.40	0.87	2.32	4.2	1.81
Lambing et al. 1994	B-23	1.80	0.13	2.40	0.87	2.32	3.2	1.38
Lambing et al. 1994	B-23	1.80	0.13	2.40	0.87	2.32	3.5	1.51
Lambing et al. 1994	B-23	1.80	0.13	2.40	0.87	2.32	3.8	1.64
Lambing et al. 1994	B-23	1.80	0.13	2.40	0.87	2.32	3.5	1.51
Lambing et al. 1994	B-23	1.80	0.13	2.40	0.87	2.32	3.5	1.51
Lambing et al. 1994	B-23	1.80	0.13	2.40	0.87	2.32	4	1.72
Lambing et al. 1994	B-23	1.80	0.13	2.40	0.87	2.32	3.4	1.46
Lambing et al. 1994	B-23	1.80	0.13	2.40	0.87	2.32	4.3	1.85
Lambing et al. 1994	B-23	1.80	0.13	2.40	0.87	2.32	4.9	2.11
Lambing et al. 1994	B-23	1.80	0.13	2.40	0.87	2.32	3.4	1.46
Lambing et al. 1994	B-26	8.90	0.13	3.25	0.87	3.98	3.1	0.78
Lambing et al. 1994	B-26	8.90	0.13	3.25	0.87	3.98	2.8	0.70
Lambing et al. 1994	B-26	8.90	0.13	3.25	0.87	3.98	3.9	0.98
Lambing et al. 1994	B-26	8.90	0.13	3.25	0.87	3.98	3.8	0.95
Lambing et al. 1994	B-26	8.90	0.13	3.25	0.87	3.98	3.1	0.78
Lambing et al. 1994	B-26	8.90	0.13	3.25	0.87	3.98	3.6	0.90
Lambing et al. 1994	***B-26***	***8.90***	***0.13***	***3.25***	***0.87***	***3.98***	***11***	***2.76***
Lambing et al. 1994	B-26	8.90	0.13	3.25	0.87	3.98	2.8	0.70
Lambing et al. 1994	B-26	8.90	0.13	3.25	0.87	3.98	3.5	0.88

Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
Lambing et al. 1994	B-26	8.90	0.13	3.25	0.87	3.98	3.6	0.90
Lambing et al. 1994	B-26	8.90	0.13	3.25	0.87	3.98	4.2	1.05
Lambing et al. 1994	B-26	8.90	0.13	3.25	0.87	3.98	4.2	1.05
Lambing et al. 1994	B-26	8.90	0.13	3.25	0.87	3.98	2.8	0.70
Lambing et al. 1994	B-26	8.90	0.13	3.25	0.87	3.98	2.9	0.73
Lambing et al. 1994	B-27	1.80	0.13	2.10	0.87	2.06	2.7	1.31
Lambing et al. 1994	B-27	1.80	0.13	2.10	0.87	2.06	2.7	1.31
Lambing et al. 1994	B-27	1.80	0.13	2.10	0.87	2.06	2.6	1.26
Lambing et al. 1994	B-29	1.20	0.13	1.65	0.87	1.59	3.1	1.95
Lambing et al. 1994	B-29	1.20	0.13	1.65	0.87	1.59	2.8	1.76
Lambing et al. 1994	B-29	1.20	0.13	1.65	0.87	1.59	2.2	1.38
Lambing et al. 1994	B-29	1.20	0.13	1.65	0.87	1.59	2.7	1.70
Lambing et al. 1994	B-29	1.20	0.13	1.65	0.87	1.59	2.7	1.70
Lambing et al. 1994	B-29	1.20	0.13	1.65	0.87	1.59	4.1	2.58
Lambing et al. 1994	B-29	1.20	0.13	1.65	0.87	1.59	1.6	1.01
Lambing et al. 1994	B-29	1.20	0.13	1.65	0.87	1.59	3.9	2.45
Rinella and Schuler 1992	***Harney***	***0.51***	***0.13***	***1.55***	***0.87***	***1.42***	***1.3***	***0.92***
Rinella and Schuler 1992	Harney	0.51	0.13	1.55	0.87	1.42	2	1.41
Rinella and Schuler 1992	Harney	0.51	0.13	1.55	0.87	1.42	2.1	1.48
Rinella and Schuler 1992	Harney	0.51	0.13	1.55	0.87	1.42	1.5	1.06
Rinella and Schuler 1992	***N Malheur***	***0.56***	***0.13***	***2.20***	***0.87***	***1.99***	***0.87***	***0.44***

Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
Rinella and Schuler 1992	N Malheur	0.56	0.13	2.20	0.87	1.99	1.9	0.96
Rinella and Schuler 1992	N Malheur	0.56	0.13	2.20	0.87	1.99	3.1	1.56
Rinella and Schuler 1992	S Malheur	0.82	0.13	1.20	0.87	1.15	1.5	1.30
Rinella and Schuler 1992	S Malheur	0.82	0.13	1.20	0.87	1.15	1.7	1.48
Rinella and Schuler 1992	S Malheur	0.82	0.13	1.20	0.87	1.15	1.4	1.22
Rinella and Schuler 1992	S Malheur	0.82	0.13	1.20	0.87	1.15	1.6	1.39
Rinella et al. 1994	Ft. Boise WMA	0.78	0.13	1.13	0.87	1.08	3.15	2.92
Rinella et al. 1994	Ft. Boise WMA	0.78	0.13	1.13	0.87	1.08	2.8	2.59
Rinella et al. 1994	Ft. Boise WMA	0.78	0.13	1.13	0.87	1.08	2.86	2.65

Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
							**Median *TTF***	1.44
							Adjusted r^2	0.29
							F	21.38
							df	48
							P	<0.001

Appendix Table B-2. American Coot. Bird Egg to Diet Trophic Transfer Factor (*TTF*).

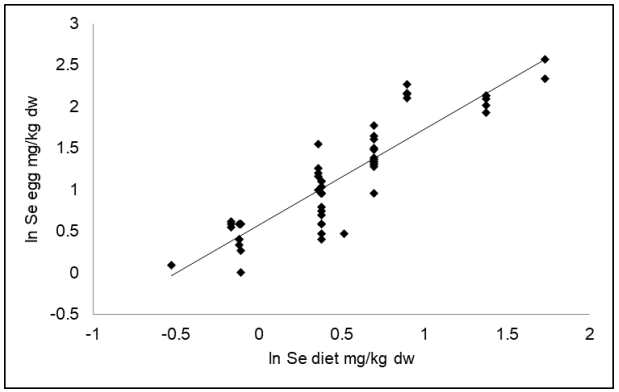
Rows with data pairs that were removed during outlier analysis are identified with bold and italics.

Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
Butler et al. 1991	***7***	***10.72***	***0.8***	***31.75***	***0.2***	***14.93***	***11.1***	***0.74***
Butler et al. 1995	TT	0.41	0.8	1.33	0.2	0.59	1.1	1.86
Butler et al. 1995	***TT***	***0.41***	***0.8***	***1.33***	***0.2***	***0.59***	***2.4***	***4.06***
Butler et al. 1997	DCP1	1.45	0.8	6.45	0.2	2.45	8.2	3.35
Butler et al. 1997	***DCP1***	***1.45***	***0.8***	***6.45***	***0.2***	***2.45***	***18***	***7.35***
Butler et al. 1997	DCP1	1.45	0.8	6.45	0.2	2.45	8.6	3.51
Butler et al. 1997	DCP1	1.45	0.8	6.45	0.2	2.45	9.7	3.96
Butler et al. 1997	DCP1	1.45	0.8	6.45	0.2	2.45	8.7	3.55
Butler et al. 1997	MNP2	3.85	0.8	4.40	0.2	3.96	8.1	2.05
Butler et al. 1997	***MNP2***	***3.85***	***0.8***	***4.40***	***0.2***	***3.96***	***3.6***	***0.91***
Butler et al. 1997	MNP2	3.85	0.8	4.40	0.2	3.96	6.9	1.74
Butler et al. 1997	MNP2	3.85	0.8	4.40	0.2	3.96	8.4	2.12
Butler et al. 1997	MNP2	3.85	0.8	4.40	0.2	3.96	7.5	1.89
Butler et al. 1997	MNP2	3.85	0.8	4.40	0.2	3.96	8.4	2.12
Lambing 1988	***7***	***0.47***	***0.8***	***6.50***	***0.2***	***1.68***	***1.4***	***0.84***
Lambing 1988	7	0.47	0.8	6.50	0.2	1.68	1.6	0.95
Lambing 1988	***7***	***0.47***	***0.8***	***6.50***	***0.2***	***1.68***	***1.1***	***0.66***
Lambing et al. 1994	B-21	1.42	0.8	4.38	0.2	2.01	5.9	2.94
Lambing et al. 1994	B-21	1.42	0.8	4.38	0.2	2.01	2.6	1.29
Lambing et al. 1994	B-21	1.42	0.8	4.38	0.2	2.01	5	2.49

Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
Lambing et al. 1994	B-21	1.42	0.8	4.38	0.2	2.01	3.8	1.89
Lambing et al. 1994	B-21	1.42	0.8	4.38	0.2	2.01	5.2	2.59
Lambing et al. 1994	B-21	1.42	0.8	4.38	0.2	2.01	3.6	1.79
Lambing et al. 1994	B-21	1.42	0.8	4.38	0.2	2.01	3.9	1.94
Lambing et al. 1994	B-21	1.42	0.8	4.38	0.2	2.01	3.8	1.89
Lambing et al. 1994	B-21	1.42	0.8	4.38	0.2	2.01	3.7	1.84
Lambing et al. 1994	B-21	1.42	0.8	4.38	0.2	2.01	4.4	2.19
Lambing et al. 1994	B-21	1.42	0.8	4.38	0.2	2.01	3.7	1.84
Lambing et al. 1994	B-21	1.42	0.8	4.38	0.2	2.01	3.8	1.89
Lambing et al. 1994	B-21	1.42	0.8	4.38	0.2	2.01	3.8	1.89
Lambing et al. 1994	B-21	1.42	0.8	4.38	0.2	2.01	4	1.99
Lambing et al. 1994	B-21	1.42	0.8	4.38	0.2	2.01	4.5	2.24
Lambing et al. 1994	B-22	1.00	0.8	3.15	0.2	1.43	3.5	2.45
Lambing et al. 1994	B-22	1.00	0.8	3.15	0.2	1.43	3.3	2.31
Lambing et al. 1994	B-22	1.00	0.8	3.15	0.2	1.43	2.7	1.89
Lambing et al. 1994	B-22	1.00	0.8	3.15	0.2	1.43	4.7	3.29
Lambing et al. 1994	B-22	1.00	0.8	3.15	0.2	1.43	3.2	2.24
Lambing et al. 1994	B-26	0.77	0.8	4.20	0.2	1.46	1.5	1.03
Lambing et al. 1994	B-26	0.77	0.8	4.20	0.2	1.46	1.6	1.10
Lambing et al. 1994	B-26	0.77	0.8	4.20	0.2	1.46	2	1.37
Lambing et al. 1994	B-26	0.77	0.8	4.20	0.2	1.46	2.6	1.78
Lambing et al. 1994	B-26	0.77	0.8	4.20	0.2	1.46	3	2.05
Lambing et al. 1994	B-26	0.77	0.8	4.20	0.2	1.46	2.8	1.92
Lambing et al. 1994	B-26	0.77	0.8	4.20	0.2	1.46	2.2	1.51
Lambing et al. 1994	B-26	0.77	0.8	4.20	0.2	1.46	2.6	1.78

Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
Lambing et al. 1994	B-26	0.77	0.8	4.20	0.2	1.46	2.6	1.78
Lambing et al. 1994	B-26	0.77	0.8	4.20	0.2	1.46	2.8	1.92
Lambing et al. 1994	B-26	0.77	0.8	4.20	0.2	1.46	2.6	1.78
Lambing et al. 1994	B-26	0.77	0.8	4.20	0.2	1.46	2.6	1.78
Lambing et al. 1994	B-26	0.77	0.8	4.20	0.2	1.46	2.1	1.44
Lambing et al. 1994	B-26	0.77	0.8	4.20	0.2	1.46	3	2.05
Lambing et al. 1994	B-26	0.77	0.8	4.20	0.2	1.46	1.8	1.23
Lambing et al. 1994	B-26	0.77	0.8	4.20	0.2	1.46	1.8	1.23
Peterson et al. 1991	3	4.64	0.8	9.62	0.2	5.64	10.3	1.83
Peterson et al. 1991	3	4.64	0.8	9.62	0.2	5.64	13.1	2.32
Rinella and Schuler 1992	N Malheur	0.56	0.8	2.20	0.2	0.89	1.8	2.03
Rinella and Schuler 1992	N Malheur	0.56	0.8	2.20	0.2	0.89	1.5	1.69
Rinella and Schuler 1992	N Malheur	0.56	0.8	2.20	0.2	0.89	1.4	1.58
Rinella and Schuler 1992	N Malheur	0.56	0.8	2.20	0.2	0.89	1.5	1.69
Rinella and Schuler 1992	S Malheur	0.82	0.8	1.20	0.2	0.90	1.3	1.45
Rinella and Schuler 1992	S Malheur	0.82	0.8	1.20	0.2	0.90	1	1.12
Rinella and Schuler 1992	S Malheur	0.82	0.8	1.20	0.2	0.90	1.8	2.01
Rinella and Schuler 1992	S Malheur	0.82	0.8	1.20	0.2	0.90	1.8	2.01
Rinella et al. 1994	Ft. Boise	0.78	0.8	1.13	0.2	0.85	1.8	2.13

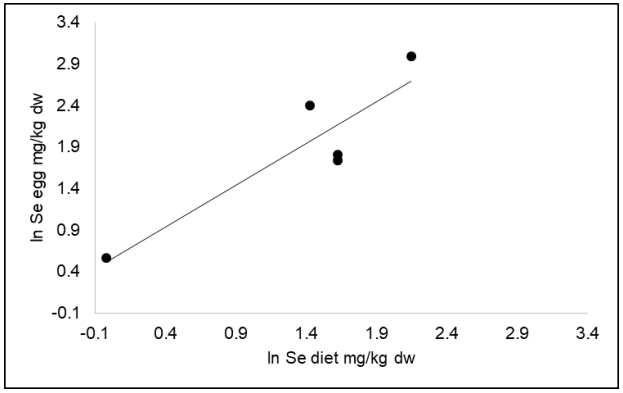
Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
	WMA							
Rinella et al. 1994	Ft. Boise WMA	0.78	0.8	1.13	0.2	0.85	1.73	2.05
Rinella et al. 1994	Ft. Boise WMA	0.78	0.8	1.13	0.2	0.85	1.85	2.19



Median *TTF*	1.89
Adjusted r^2	0.80
F	232.7
df	58
P	<0.001

Appendix Table B-3. Cinnamon Teal. Bird Egg to Diet Trophic Transfer Factor (*TTF*).

Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
Butler et al. 1997	CHP	3.75	0.42	12.00	0.58	8.54	20	2.34
Butler et al. 1997	DCP3	2.10	0.42	7.20	0.58	5.06	5.7	1.13
Butler et al. 1997	DCP3	2.10	0.42	7.20	0.58	5.06	6.1	1.21
Butler et al. 1997	MNP2	3.85	0.42	4.40	0.58	4.17	11	2.64
Rinella et al. 1994	Ft. Boise WMA	0.78	0.42	1.13	0.58	0.98	1.75	1.79



Median *TTF*	1.79
Adjusted r^2	0.76
F	13.81
df	3
P	0.034

Appendix Table B-4. Eared Grebe. Bird Egg to Diet Trophic Transfer Factor (*TTF*).

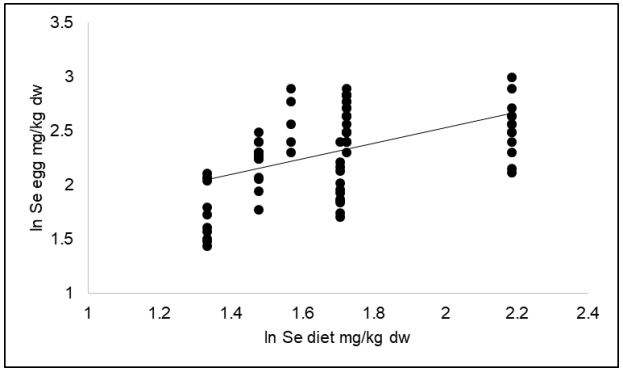
Because eared grebes eat a 100% invertebrate diet, all paired invertebrate-egg measurements were used, regardless of whether a paired plant measurement was available.

Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
Lambing et al. 1994	B-16	n/a	0	4.80	1	4.80	13	2.71
Lambing et al. 1994	B-16	n/a	0	4.80	1	4.80	18	3.75
Lambing et al. 1994	B-16	n/a	0	4.80	1	4.80	16	3.33
Lambing et al. 1994	B-16	n/a	0	4.80	1	4.80	11	2.29
Lambing et al. 1994	B-16	n/a	0	4.80	1	4.80	10	2.08
Lambing et al. 1994	B-19	n/a	0	5.60	1	5.60	17	3.04
Lambing et al. 1994	B-19	n/a	0	5.60	1	5.60	12	2.14
Lambing et al. 1994	B-19	n/a	0	5.60	1	5.60	14	2.50
Lambing et al. 1994	B-19	n/a	0	5.60	1	5.60	12	2.14
Lambing et al. 1994	B-19	n/a	0	5.60	1	5.60	14	2.50
Lambing et al. 1994	B-19	n/a	0	5.60	1	5.60	15	2.68
Lambing et al. 1994	B-19	n/a	0	5.60	1	5.60	10	1.79
Lambing et al. 1994	B-19	n/a	0	5.60	1	5.60	11	1.96
Lambing et al. 1994	B-19	n/a	0	5.60	1	5.60	12	2.14
Lambing et al. 1994	B-19	n/a	0	5.60	1	5.60	13	2.32
Lambing et al. 1994	B-19	n/a	0	5.60	1	5.60	16	2.86
Lambing et al. 1994	B-19	n/a	0	5.60	1	5.60	17	3.04
Lambing et al. 1994	B-19	n/a	0	5.60	1	5.60	15	2.68
Lambing et al. 1994	B-19	n/a	0	5.60	1	5.60	18	3.21
Lambing et al. 1994	B-19	n/a	0	5.60	1	5.60	17	3.04
Lambing et al. 1994	B-19	n/a	0	5.60	1	5.60	12	2.14
Lambing et al. 1994	B-19	n/a	0	5.60	1	5.60	17	3.04

Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
Lambing et al. 1994	B-19	n/a	0	5.60	1	5.60	14	2.50
Lambing et al. 1994	B-19	n/a	0	5.60	1	5.60	11	1.96
Lambing et al. 1994	B-19	n/a	0	5.60	1	5.60	16	2.86
Lambing et al. 1994	B-19	n/a	0	5.60	1	5.60	11	1.96
Lambing et al. 1994	B-21	n/a	0	4.38	1	4.38	9.4	2.14
Lambing et al. 1994	B-21	n/a	0	4.38	1	4.38	5.9	1.35
Lambing et al. 1994	B-21	n/a	0	4.38	1	4.38	7.9	1.80
Lambing et al. 1994	B-21	n/a	0	4.38	1	4.38	10	2.28
Lambing et al. 1994	B-21	n/a	0	4.38	1	4.38	7.8	1.78
Lambing et al. 1994	B-21	n/a	0	4.38	1	4.38	11	2.51
Lambing et al. 1994	B-21	n/a	0	4.38	1	4.38	9.6	2.19
Lambing et al. 1994	B-21	n/a	0	4.38	1	4.38	10	2.28
Lambing et al. 1994	B-21	n/a	0	4.38	1	4.38	7	1.60
Lambing et al. 1994	B-21	n/a	0	4.38	1	4.38	11	2.51
Lambing et al. 1994	B-21	n/a	0	4.38	1	4.38	10	2.28
Lambing et al. 1994	B-21	n/a	0	4.38	1	4.38	9.4	2.14
Lambing et al. 1994	B-21	n/a	0	4.38	1	4.38	9.8	2.24
Lambing et al. 1994	B-21	n/a	0	4.38	1	4.38	12	2.74
Lambing et al. 1994	B-21	n/a	0	8.90	1	8.90	13	1.46
Lambing et al. 1994	B-21	n/a	0	8.90	1	8.90	10	1.12
Lambing et al. 1994	B-21	n/a	0	8.90	1	8.90	8.6	0.97
Lambing et al. 1994	B-21	n/a	0	8.90	1	8.90	14	1.57
Lambing et al. 1994	B-21	n/a	0	8.90	1	8.90	8.3	0.93
Lambing et al. 1994	B-21	n/a	0	8.90	1	8.90	11	1.24
Lambing et al. 1994	B-21	n/a	0	8.90	1	8.90	18	2.02

Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
Lambing et al. 1994	B-21	n/a	0	8.90	1	8.90	20	2.25
Lambing et al. 1994	B-21	n/a	0	8.90	1	8.90	12	1.35
Lambing et al. 1994	B-21	n/a	0	8.90	1	8.90	12	1.35
Lambing et al. 1994	B-21	n/a	0	8.90	1	8.90	14	1.57
Lambing et al. 1994	B-21	n/a	0	8.90	1	8.90	15	1.69
Lambing et al. 1994	B-21	n/a	0	8.90	1	8.90	13	1.46
Lambing et al. 1994	B-23	n/a	0	3.79	1	3.79	8.2	2.16
Lambing et al. 1994	B-23	n/a	0	3.79	1	3.79	7.7	2.03
Lambing et al. 1994	B-23	n/a	0	3.79	1	3.79	4.8	1.26
Lambing et al. 1994	B-23	n/a	0	3.79	1	3.79	4.2	1.11
Lambing et al. 1994	B-23	n/a	0	3.79	1	3.79	4.4	1.16
Lambing et al. 1994	B-23	n/a	0	3.79	1	3.79	5	1.32
Lambing et al. 1994	B-23	n/a	0	3.79	1	3.79	4.5	1.19
Lambing et al. 1994	B-23	n/a	0	3.79	1	3.79	6	1.58
Lambing et al. 1994	B-23	n/a	0	3.79	1	3.79	7.9	2.08
Lambing et al. 1994	B-23	n/a	0	3.79	1	3.79	5.6	1.48
Lambing et al. 1994	B-26	n/a	0	5.50	1	5.50	6.3	1.15
Lambing et al. 1994	B-26	n/a	0	5.50	1	5.50	7.1	1.29
Lambing et al. 1994	B-26	n/a	0	5.50	1	5.50	6.5	1.18
Lambing et al. 1994	B-26	n/a	0	5.50	1	5.50	8.7	1.58
Lambing et al. 1994	B-26	n/a	0	5.50	1	5.50	6.4	1.16
Lambing et al. 1994	B-26	n/a	0	5.50	1	5.50	6.9	1.25
Lambing et al. 1994	B-26	n/a	0	5.50	1	5.50	8.4	1.53
Lambing et al. 1994	B-26	n/a	0	5.50	1	5.50	9.1	1.65
Lambing et al. 1994	B-26	n/a	0	5.50	1	5.50	5.7	1.04

Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
Lambing et al. 1994	B-26	n/a	0	5.50	1	5.50	11	2.00
Lambing et al. 1994	B-26	n/a	0	5.50	1	5.50	7.5	1.36
Lambing et al. 1994	B-26	n/a	0	5.50	1	5.50	5.5	1.00



Median *TTF*	2.00
Adjusted r^2	0.24
F	23.95
df	73
P	<0.01

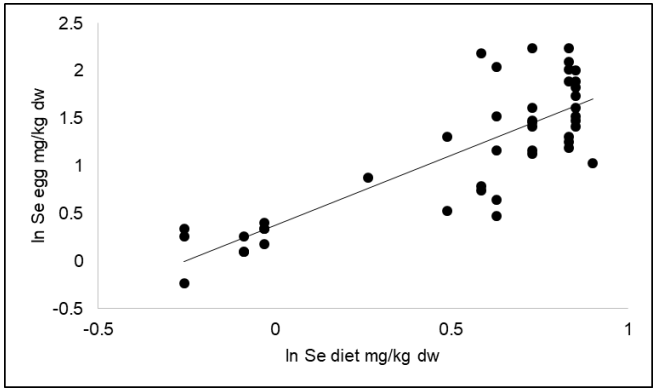
Appendix Table B-5. Gadwall. Bird Egg to Diet Trophic Transfer Factor (*TTF*).

Rows with data pairs that were removed during outlier analysis are identified with bold and italics.

Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
Lambing et al. 1994	B-21	1.69	0.75	4.30	0.25	2.34	4.1	1.75
Lambing et al. 1994	B-21	1.69	0.75	4.30	0.25	2.34	5	2.13
Lambing et al. 1994	B-21	1.69	0.75	4.30	0.25	2.34	6.6	2.82
Lambing et al. 1994	B-21	1.69	0.75	4.30	0.25	2.34	4.4	1.88
Lambing et al. 1994	B-21	1.69	0.75	4.30	0.25	2.34	6.2	2.65
Lambing et al. 1994	B-21	1.69	0.75	4.30	0.25	2.34	7.4	3.16
Lambing et al. 1994	B-21	1.69	0.75	4.30	0.25	2.34	4.6	1.96
Lambing et al. 1994	B-21	1.69	0.75	4.30	0.25	2.34	5.7	2.43
Lambing et al. 1994	B-21	1.85	0.75	4.30	0.25	2.46	2.8	1.14
Lambing et al. 1994	B-22	1.20	0.75	3.57	0.25	1.79	2.1	1.17
Lambing et al. 1994	B-22	1.20	0.75	3.57	0.25	1.79	8.9	4.96
Lambing et al. 1994	***B-22***	***1.20***	***0.75***	***3.57***	***0.25***	***1.79***	***10***	***5.58***
Lambing et al. 1994	B-22	1.20	0.75	3.57	0.25	1.79	2.2	1.23
Lambing et al. 1994	***B-22***	***1.20***	***0.75***	***3.57***	***0.25***	***1.79***	***14***	***7.81***
Lambing et al. 1994	***B-22***	***5.60***	***0.75***	***3.57***	***0.25***	***5.09***	***2.7***	***0.53***
Lambing et al. 1994	***B-22***	***5.60***	***0.75***	***3.57***	***0.25***	***5.09***	***2.6***	***0.51***
Lambing et al. 1994	***B-22***	***5.60***	***0.75***	***3.57***	***0.25***	***5.09***	***4.2***	***0.82***
Lambing et al. 1994	***B-22***	***5.60***	***0.75***	***3.57***	***0.25***	***5.09***	***3.3***	***0.65***
Lambing et al. 1994	B-23	1.80	0.75	3.79	0.25	2.30	3.3	1.44
Lambing et al. 1994	***B-23***	***1.80***	***0.75***	***3.79***	***0.25***	***2.30***	***13***	***5.66***
Lambing et al. 1994	B-23	1.80	0.75	3.79	0.25	2.30	8.1	3.52
Lambing et al. 1994	B-23	1.80	0.75	3.79	0.25	2.30	3.5	1.52
Lambing et al. 1994	B-23	1.80	0.75	3.79	0.25	2.30	3.7	1.61

Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
Lambing et al. 1994	B-23	1.80	0.75	3.79	0.25	2.30	6.6	2.87
Lambing et al. 1994	B-23	1.80	0.75	3.79	0.25	2.30	9.4	4.09
Lambing et al. 1994	B-23	1.80	0.75	3.79	0.25	2.30	7.5	3.26
Lambing et al. 1994	B-26	0.66	0.75	3.23	0.25	1.30	2.4	1.84
Lambing et al. 1994	B-26	0.77	0.75	4.20	0.25	1.63	3.7	2.27
Lambing et al. 1994	B-26	0.77	0.75	4.20	0.25	1.63	1.7	1.04
Lambing et al. 1994	B-26	1.36	0.75	4.20	0.25	2.07	4.3	2.08
Lambing et al. 1994	B-26	1.36	0.75	4.20	0.25	2.07	4.1	1.98
Lambing et al. 1994	B-26	1.36	0.75	4.20	0.25	2.07	5	2.41
Lambing et al. 1994	B-26	1.36	0.75	4.20	0.25	2.07	3.1	1.50
Lambing et al. 1994	B-26	1.36	0.75	4.20	0.25	2.07	3.2	1.54
Lambing et al. 1994	B-26	1.36	0.75	4.20	0.25	2.07	4.4	2.12
Lambing et al. 1994	B-26	1.36	0.75	4.20	0.25	2.07	9.4	4.54
Lambing et al. 1994	B-26	1.10	0.75	4.20	0.25	1.88	3.2	1.71
Lambing et al. 1994	B-26	1.10	0.75	4.20	0.25	1.88	4.6	2.45
Lambing et al. 1994	B-26	1.10	0.75	4.20	0.25	1.88	7.7	4.11
Lambing et al. 1994	B-26	1.10	0.75	4.20	0.25	1.88	1.6	0.85
Lambing et al. 1994	B-26	1.10	0.75	4.20	0.25	1.88	1.9	1.01
Rinella and Schuler 1992	Harney	0.51	0.75	1.55	0.25	0.77	1.4	1.81
Rinella and Schuler 1992	Harney	0.51	0.75	1.55	0.25	0.77	0.79	1.02
Rinella and Schuler 1992	Harney	0.51	0.75	1.55	0.25	0.77	1.3	1.68
Rinella and Schuler 1992	N Malheur	0.56	0.75	2.20	0.25	0.97	1.2	1.24

Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
Rinella and Schuler 1992	N Malheur	0.56	0.75	2.20	0.25	0.97	1.4	1.44
Rinella and Schuler 1992	N Malheur	0.56	0.75	2.20	0.25	0.97	1.4	1.44
Rinella and Schuler 1992	N Malheur	0.56	0.75	2.20	0.25	0.97	1.5	1.55
Rinella and Schuler 1992	S Malheur	0.82	0.75	1.20	0.25	0.92	1.1	1.20
Rinella and Schuler 1992	S Malheur	0.82	0.75	1.20	0.25	0.92	1.3	1.42
Rinella and Schuler 1992	S Malheur	0.82	0.75	1.20	0.25	0.92	1.1	1.20

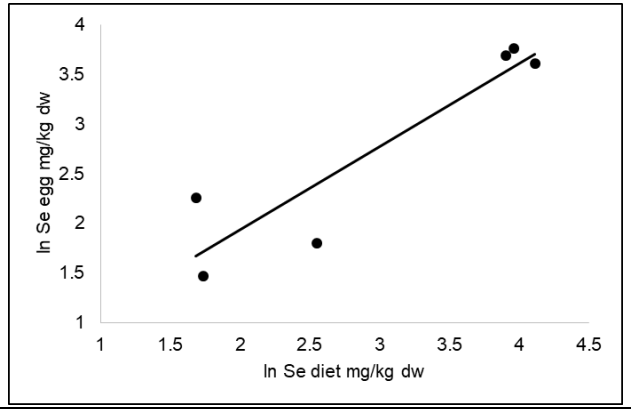


Median *TTF*	1.78
Adjusted r^2	0.66
F	86.17
df	42
P	<0.001

Appendix Table B-6. Pied-Billed Grebe. Bird Egg to Fish (*TTF*).

The *TTF* for this species was calculated using all available paired egg-animal Se measurements.

Study	Site	Fish Se (mg/kg)	Egg Se (mg/kg)	*TTF*
Byron and Santolo 2010	BCW	61.30	36.78	0.60
Byron and Santolo 2010	UCI	5.4	9.55	1.77
Byron et al. 2012	BCW	49.74	40	0.80
Byron et al. 2012	UCI	12.82	6.06	0.47
Byron and Santolo 2014	BCW	52.49	43	0.82
Byron and Santolo 2014	UCI	5.7	4.35	0.76



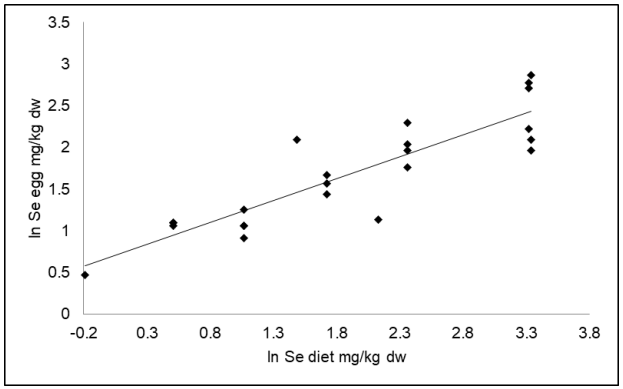
Median *TTF*	0.78
Adjusted r^2	0.81
F	22.25
Df	4
P	0.009

Appendix Table B-7. Red-Winged Blackbird. Bird Egg to Diet (*TTF*).

Rows with data pairs that were removed during outlier analysis are identified with bold and italics.

Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
Butler et al. 1991	7	10.72	0.17	31.75	0.83	28.18	17.6	0.62
Butler et al. 1993	LP4	1.50	0.17	3.20	0.83	2.91	2.9	1.00
Butler et al. 1993	LP4	1.50	0.17	3.20	0.83	2.91	2.9	1.00
Butler et al. 1993	LP4	1.50	0.17	3.20	0.83	2.91	2.5	0.86
Butler et al. 1993	LP4	1.50	0.17	3.20	0.83	2.91	3.5	1.20
Butler et al. 1994	MKP	9.90	0.17	32.00	0.83	28.24	8.1	0.29
Butler et al. 1994	MKP	9.90	0.17	32.00	0.83	28.24	7.1	0.25
Butler et al. 1994	MKP	6.45	0.17	32.00	0.83	27.66	16	0.58
Butler et al. 1994	MKP	6.45	0.17	32.00	0.83	27.66	9.2	0.33
Butler et al. 1994	MKP	6.45	0.17	32.00	0.83	27.66	15	0.54
Butler et al. 1995	DD	0.83	0.17	0.83	0.83	0.83	1.6	1.93
Butler et al. 1991	10	2.55	0.17	4.80	0.83	4.42	8.1	1.83
Butler et al. 1991	***10***	***2.55***	***0.17***	***4.80***	***0.83***	***4.42***	***8.6***	***1.95***
Butler et al. 1997	CHP	3.75	0.17	12.00	0.83	10.60	7.7	0.73
Butler et al. 1997	CHP	3.75	0.17	12.00	0.83	10.60	5.8	0.55
Butler et al. 1997	CHP	3.75	0.17	12.00	0.83	10.60	9.9	0.93
Butler et al. 1997	CHP	3.75	0.17	12.00	0.83	10.60	7.1	0.67
Butler et al. 1997	DCP1	1.45	0.17	6.45	0.83	5.60	4.2	0.75
Butler et al. 1997	DCP1	1.45	0.17	6.45	0.83	5.60	5.3	0.95
Butler et al. 1997	DCP1	1.45	0.17	6.45	0.83	5.60	4.8	0.86
Butler et al. 1997	LCHP1	0.44	0.17	1.91	0.83	1.66	2.9	1.75
Butler et al. 1997	LCHP1	0.44	0.17	1.91	0.83	1.66	3	1.81
Butler et al. 1997	LCHP1	0.44	0.17	1.91	0.83	1.66	3	1.81

Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
Butler et al. 1997	***WCP***	***2.30***	***0.17***	***9.70***	***0.83***	***8.44***	***2.1***	***0.25***
Butler et al. 1997	***WCP***	***2.30***	***0.17***	***9.70***	***0.83***	***8.44***	***2.8***	***0.33***
Butler et al. 1997	WCP	2.30	0.17	9.70	0.83	8.44	3.1	0.37

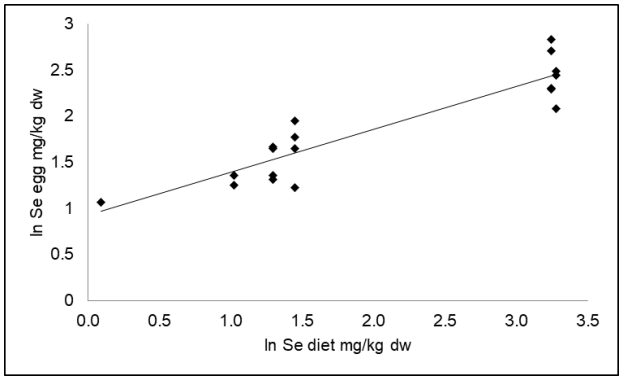


Median *TTF*	0.86
Adjusted r^2	0.77
F	73.92
df	21
P	<0.001

Appendix Table B-8. Yellow-Headed Blackbird. Bird Egg to Diet Trophic Transfer Factor (*TTF*).

Rows with data pairs that were removed during outlier analysis are identified with bold and italics.

Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
Butler et al. 1991	7	10.72	0.25	31.75	0.75	26.50	8	0.30
Butler et al. 1991	7	10.72	0.25	31.75	0.75	26.50	11.5	0.43
Butler et al. 1994	MKP	9.90	0.25	32.00	0.75	26.48	12	0.45
Butler et al. 1994	MKP	6.45	0.25	32.00	0.75	25.61	9.9	0.39
Butler et al. 1994	MKP	6.45	0.25	32.00	0.75	25.61	10	0.39
Butler et al. 1994	MKP	6.45	0.25	32.00	0.75	25.61	15	0.59
Butler et al. 1994	MKP	6.45	0.25	32.00	0.75	25.61	17	0.66
Butler et al. 1997	MNP2	3.85	0.25	4.40	0.75	4.26	7	1.64
Butler et al. 1997	MNP2	3.85	0.25	4.40	0.75	4.26	5.2	1.22
Butler et al. 1997	MNP2	3.85	0.25	4.40	0.75	4.26	3.4	0.80
Butler et al. 1997	MNP2	3.85	0.25	4.40	0.75	4.26	5.9	1.38
Butler et al. 1993	LP4	1.50	0.25	3.20	0.75	2.78	3.9	1.41
Butler et al. 1993	LP4	1.50	0.25	3.20	0.75	2.78	3.5	1.26
Butler et al. 1993	R1	4.64	0.25	3.33	0.75	3.66	3.9	1.07
Butler et al. 1993	R1	4.64	0.25	3.33	0.75	3.66	5.3	1.45
Butler et al. 1993	R1	4.64	0.25	3.33	0.75	3.66	5.2	1.42
Butler et al. 1993	R1	4.64	0.25	3.33	0.75	3.66	3.7	1.01
Butler et al. 1995	TT	0.41	0.25	1.33	0.75	1.10	2.9	2.64
Butler et al. 1995	***TT***	***0.41***	***0.25***	***1.33***	***0.75***	***1.10***	***4.8***	***4.38***

Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
							**Median *TTF***	1.04
							Adjusted r^2	0.83
							F	81.74
							df	16
							P	<0.001

Calculation of $TTF^{composite}$ for Species with measured *TTF*

This section describes the calculation of $TTF^{composite}$ for the eight bird species with measured TTF using data listed in the preceding tables. $TTF^{composite}$ were calculated from food webs modeled using information from the Cornell Lab of Ornithology Birds of North America web site: (http://www.birds.cornell.edu/Page.aspx?pid=1478), and following the methods describes in Part 5.4.2.1. Calculations were made using different combinations of (Equation 5-2) and (Equation 5-3) depending on the specific modeled food web.

$$\boldsymbol{TTF^{composite} = TTF^{TL4} \times TTF^{TL3} \times TTF^{TL2}}$$

(Appendix Equation B-1)

where:

TTF^{TL2}	=	the trophic transfer factor of the trophic level 2 species
TTF^{TL3}	=	the trophic transfer factor of the trophic level 3 species
TTF^{TL4}	=	the trophic transfer factor of the trophic level 4 species
$TTF^{composite}$	=	the product of all the trophic transfer factors

Similarly, the consumption of more than one species of organism at the same trophic level can also be modeled by expressing the *TTF* at a particular trophic level as the weighted average of the *TTF*s of all species consumed given as:

$$\overline{TTF}^{TLx} = \sum_i \left(TTF_i^{TLx} \times W_i\right)$$

(Appendix Equation B-2)

where:

TTF_i^{TLx}	=	the trophic transfer factor of the ith species at a particular trophic level
W_i	=	the proportion of the ith species consumed

$TTF^{composite}$ for these species are listed in Table 5-5. Invertebrate *TTF*s used in these calculations are from Table 5-1.

Non-Migratory Species

American Coot

The diet of American coot is described as consisting of predominantly plant matter, including pond weeds, sedges, algae, and wild and domestic grasses, as well as species such as eelgrass, wild celery, duckweeds, cattail, watermilfoil, and numerous other plants. Animal matter is relatively uncommon, but can be important during the breeding season, especially for growing young. Important animal food items, from greatest to least importance, include insects, mollusks, small crustaceans, and crawfish, as well as some small vertebrates, such as salamander larvae, tadpoles, and small fish (Brisbin et al. 2002).

Based on this information, the American coot diet was modeled as consisting of 80% aquatic plants and 20% aquatic invertebrates. American coot data were available for 66 bird egg-diet selenium pairs (Appendix Table B-2). Data were natural log transformed, and six egg-diet pairs were removed following outlier analysis. The slope of the egg-diet regression model was positive and statistically significant (P<0.001), with an adjusted r^2 of 0.80. The TTF^{TL3} for American coot based on a diet of 80% plants and 20% animals was 1.89.

The American coot diet was modeled as consisting of 80% aquatic plants, 8% insects, 6% mollusks, 4% small crustaceans, and 2% crayfish. The $TTF^{composite}$ is calculated as follows.

$$TTF^{composite} = [1.89 \times 0.8] + [1.89 \times ((2.14\ x\ 0.08) + (4.29\ x\ 0.06) + (1.32\ x\ 0.04) + (1.46\ x\ 0.02))] = \mathbf{2.48}$$

Red-Winged Blackbird

The diet of red-winged blackbird during the breeding season is described as consisting primarily of animal matter, although this can vary with date, sex, and access to agricultural habitats (Yasukawa and Searcy 1995). For example, within agricultural habitats in Ontario, stomach contents were 51% insects and 42% agricultural waste grain. Within marshes in Manitoba, however, diet was 100% animal matter. Dietary animal matter consists almost entirely of insects.

Based on this information the red-winged blackbird diet was modeled as consisting of 17% aquatic plants and 83% aquatic insects, using the average ratio of insects to seeds in stomach contents of four studies conducted during the breeding season. Red-winged blackbird

data were available for 26 bird egg-diet pairs (Appendix Table B-7). Data were natural log transformed, and three egg-diet pairs were removed following outlier analysis. The slope of the egg-diet regression model was positive and statistically significant (P<0.001), with an adjusted r^2 of 0.77. The TTF^{TL3} for red-winged blackbird based on a diet of 17% plants and 83% animals was 0.80.

The red-winged blackbird diet was modeled as consisting of 17% aquatic plants and 83% aquatic insects. The $TTF^{composite}$ is calculated as follows.

$$TTF^{composite} = [0.86 \times 0.17] + [0.86 \times 2.14 \times 0.83] = \mathbf{1.67}$$

Migratory Species

EPA conducted an analysis to compare breeding season data (defined here as April through July) vs. all available data for the migratory species. Because many of the bird species analyzed eat invertebrates, and invertebrate sampling collections are typically conducted outside of the breeding season time frames, many of the data for the breeding season only did not produce a statistically significant regression. For those birds where enough data were available during the breeding season to produce statistically significant results, the resulting *TTFs* were similar to the all data scenarios of the same bird species. For these reasons, EPA decided to derive all of the migratory *TTFs* using all data available in each study.

American avocet

American avocets are generalist tactile feeders, and their diet varies by habitat (Ackerman et al. 2013). Stomach content results from six inland studies across Western North America reveal that avocets consume a range of plant and animal species. Plant matter, primarily seeds, range from 1-35%. Animal matter consists primarily of dipterans, predominantly chironomids, followed by corixidae, beetles, mayflies, annelids, gastropods, crustaceans, other invertebrates, and very rarely small fish and amphibians.

Based on the above information American avocet diet was modeled as consisting of 13% plants and 87% animals. American avocet data were available for 53 bird egg-diet selenium pairs (Appendix Table B-1). Data were natural log transformed, and three egg-diet pairs were removed following outlier analysis. The slope of the egg-plant regression model was positive and

statistically significant (P<0.001), with an r^2 of 0.29. The TTF^{TL3} based on a diet of 13% plants and 87% animals was 1.44. Based on the dietary information described above, the American avocet diet was modeled as consisting of 13% plants, 55% chironomids, 10% corixids, 1% mayflies, 10% other insects (mainly beetles), 3% annelids, 3% mollusks, 2% crustaceans, and 3% other invertebrates. The $TTF^{composite}$ is calculated as follows.

$$TTF^{composite} = [1.49 \times 0.13] + \left[(1.49 \times \left((1.90 \times 0.55) + (1.48 \times 0.10) + (2.38 \times 0.01) + (2.14 \times 0.10) + (1.29 \times 0.03) + (4.29 \times 0.03) + (1.41 \times 0.02) + (1.89 \times 0.03)\right)\right] = \mathbf{2.61}$$

Cinnamon Teal

The diet of cinnamon teal varies with location and season. The average diets according to percent dry weight of esophageal contents from six studies from the Western United States during the spring and summer were approximately 36% dipterans (primarily chironomids), 9.5% gastropods, 4% corixidae, 3.5% cladocerans, 2% beetles, 1% odonates, 2% other invertebrates, and 42% plant matter, primarily seeds (Gammonley 2012).

Based on the information above, cinnamon teal diet was modeled as consisting of 42% aquatic plants and 58% aquatic invertebrates. Cinnamon teal data were available for 5 bird egg-diet selenium pairs (Appendix Table B-3). Data were natural log transformed, and no egg-diet pairs were removed following outlier analysis. The slope of the egg-plant regression model was positive and statistically significant (P=0.03), with an adjusted r^2 of 0.76. The TTF^{TL3} based on a diet of 42% plants and 58% animals was 1.79.

The cinnamon teal diet was modeled as consisting of 42% aquatic plants, 36% chironomids, 9.5% mollusks (gastropods), 4% corixidae, 3.5% cladocerans, 2% other insects (beetles), 1% odonates, and 2% other invertebrates. The $TTF^{composite}$ is calculated as follows.

$$TTF^{composite} = [1.79 \times 0.42] + \left[1.79 \times \left((1.90 \times 0.36) + (4.29 \times 0.095) + (1.48 \times 0.04) + (0.74 \times 0.035) + (2.14 \times 0.02) + (2.425 \times 0.01) + (1.89 \times 0.02)\right)\right] = \mathbf{3.04}$$

Eared Grebe

The diet of eared grebes consists of animals, principally invertebrates but also occasionally small fish (Cullen et al. 1999). In saline lakes, diet consists of predominantly of brine shrimp (60-93%) and brine flies (5-40%) depending on their relative availability. Eared grebes have also been found to feed on pile worms, amphipods and small fish. In breeding grounds and in migration in Western States, eared grebes feed primarily on insects, particularly on water boatmen, as well as diving beetles, caddisflies, mayflies, chironomids, and odonates.

Based on the information above, eared grebe diet was modeled as consisting of 100% aquatic invertebrates. Eared grebe data were available for 75 bird egg-invertebrate selenium pairs (Appendix Table B-4). Because eared grebes do not consume plants, all available bird-egg invertebrate pairs were used, regardless of whether or not they were also paired with a plant measurement. Data were natural log transformed, and no egg-diet pairs were removed following outlier analysis. The slope of the regression model was positive and statistically significant ($P<0.001$), with an adjusted r^2 of 0.24. The TTF^{TL3} based on a 100% animal diet was 2.00.

The eared grebe diet was modeled as consisting of 45% crustaceans (brine shrimp), 25% dipterans (brine flies and chironomids), 20% corixidae, 5% other insects, and 5% annelids. The $TTF^{composite}$ for eared grebe is calculated as follows.

$$TTF^{composite} = [2.00 \times ((1.41 \times 0.45) + (1.90 \times 0.25) + (1.48 \times 0.2) + (2.14 \times 0.05) + (1.29 \times 0.05))] = \mathbf{3.15}$$

Gadwall

The diet of gadwall varies seasonally, with a diet consisting almost entirely of plant matter in the fall and winter, and between 23-46% animal and 42-54% plant matter during the summer (Leschack et al. 1997). Plants eaten include filamentous algae, water milfoil, widgeon grass, duckweed, and pondweed, depending on availability (Leschack et al. 1997). Animal food items consist of midge larvae, aphids, snails, and beetle larvae (Leschack et al. 1997).

Based on the information above, the gadwall diet was modeled as consisting of 75% aquatic plants and 25% aquatic invertebrates. Cinnamon teal data were available for 51 bird egg-diet selenium pairs (Appendix Table B-5). Data were natural log transformed, and seven egg-diet pairs were removed following outlier analysis. The slope of the egg-plant regression model was positive and statistically significant ($P<0.001$), with an adjusted r^2 of 0.66. The TTF^{TL3} based on a diet of 75% plants and 25% animals was 1.78.

The gadwall diet was modeled as consisting of 75% plants and 25% animals, with the modeled animal portion consisting of 11% chironomids, 7% insects (beetles), 5% small crustaceans, and 2% snails. The $TTF^{composite}$ for gadwall is calculated as follows.

$$TTF^{composite} = [1.78 \times 0.75] + \left[1.78 \times \left((1.90 \times 0.11) + (2.14 \times 0.07) + (1.32 \times 0.05) + (4.29 \times 0.02)\right)\right] = \mathbf{2.24}$$

Pied-Billed Grebe

The diet of pied-billed grebes includes decapod crustaceans, especially crayfish, aquatic insects, and fishes. In some areas, also leeches, gizzard shad, or frogs and tadpoles. Pied-billed grebes in the fishless wetlands of Manitoba kill and eat tiger salamanders. Stomach contents of 174 individuals from the Eastern United States contained 376 food items: 62 decapods (crayfish, crabs, shrimps, etc.), 13 dragonfly larvae, 77 bugs, 124 beetles, 76 fishes, 5 mollusks, and 19 other invertebrates (Muller and Storer 1999). Based on the dietary information listed above, and after applying a general weighting factor of 5 to fish and crayfish to account for their larger size, the pied-billed grebe diet was modeled as 41% fish, 33% crayfish, 13% beetles, 8% corixids, 2% other invertebrates, 1% mollusks, and 1% dragonflies.

Pied-billed grebe data were available for six bird egg-fish pairs (Appendix Table B-6). These data were reported in Byron and Santolo (2010, 2014); Byron et al. (2012) for two sites in the Newport Bay, CA watershed. Data were natural log transformed, and no egg-diet pairs were removed following outlier analysis. The slope of the egg-fish regression model was positive and statistically significant (P=0.009), with an adjusted r^2 of 0.81. Based on these data, the TTF^{TL4} for pied-billed grebe diet is 0.78. The $TTF^{composite}$ for pied-billed grebe is calculated as follows using the modeled diet described above. The piscivorous portion of their diet is modeled using a fish $TTF^{composite}$ of 2.34, which is the average fish $TTF^{composite}$ at the 65 sites used in the translation dataset (Appendix Table B-12).

$$TTF^{composite} = [0.78 \times 2.34 \times 0.41] + \left[0.78 \times \left((1.46 \times 0.33) + (2.14 \times 0.13) + (1.48 \times 0.08) + (1.89 \times 0.02) + (4.29 \times 0.01) + (1.97 \times 0.01)\right)\right] = \mathbf{1.52}$$

Data on *TTF*s for piscivorous bird species are limited, and the pied-billed grebe was the only predominantly piscivorous species with sufficient data to calculate a *TTF* following the approach used in the 2016 aquatic life criteria document (U.S. EPA 2016a). Limited paired data exist for two additional species that are largely piscivorous, but insufficient data were available for regression analysis. King et al. (2003) measured selenium in double-crested cormorant eggs during 1999-2000 and in three fish species (largemouth bass, red shiner, threadfin shad) during 2000 from Topock Marsh, Arizona. The double-crested cormorant *TTF* was calculated as 0.84. Martinez (1994) measured selenium in green heron eggs and egg masses (consisting of the ovary and the cluster of developing eggs surrounding the ovary) from two lakes in the lower Colorado River in southwest Arizona during the breeding season of 1993. Lusk (1993) measured selenium in fish and invertebrate prey species from the same two sites in 1991 and 1992. Based on these data, the green heron *TTF* was 1.35 based on diet paired with egg and 2.37 based on diet paired with egg masses. Because the similarity of the pied-billed grebe *TTF* to the *TTF* of the piscivorous double-crested cormorant, and because it is the only species for which a *TTF* could be calculated from paired data, the pied-billed grebe *TTF* was considered to be an acceptable *TTF*, and an acceptable surrogate *TTF* for piscivorous birds.

Yellow-Headed Blackbird

The diet of yellow-headed blackbird consists of a variety of insects and seeds. In a study of 15 birds in Utah, the diet consisted of seven orthoptera, seven odonata, 96 coleoptera, 40 lepidoptera, 13 diptera, 10 hymenoptera, and 109 seeds (Twedt and Crawford 1995). Based on the dietary information listed above, the yellow-headed blackbird diet was modeled as consisting of 25% plants and 75% animals. Because they appear to consume a wide variety of insects, the animal proportion of their diet was modeled as consisting of all insects.

Yellow-headed blackbird data were available for 19 bird egg-diet selenium pairs (Appendix Table B-8). Data were natural log transformed, and one egg-diet pairs were removed following outlier analysis. The slope of the egg-diet regression model was positive and statistically significant (P<0.001), with an adjusted r^2 of 0.83. The TTF^{TL3} for yellow-headed blackbird was 1.04. The $TTF^{composite}$ for yellow-headed blackbird is calculated as follows.

$$TTF^{composite} = [1.04 \times 0.25] + [1.04 \times 2.14 \times 0.75] = \mathbf{1.93}$$

Summary

Composite *TTFs* could be calculated from species-specific measured data for two non-migratory species: American coot and red-winged blackbird, and six migratory species: American avocet, cinnamon teal, eared grebe, gadwall, pied-billed grebe, and yellow-headed blackbird. Available dietary information describes the pied-billed grebe diet as a 100% animal diet consisting of fish and invertebrates; however, the *TTF* for pied-billed grebe was calculated from available paired data, which included only bird egg-fish selenium data. Species level *TTFs* for these species are listed in Table 5-3, and composite *TTFs* for these species are listed in Table 5-5.

Paired Surrogate Data Used to Calculate Bird Trophic Transfer Factors (*TTF*) for Threatened and Endangered (T&E) Species

The following tables (Appendix Table B-9 through Appendix Table B-11) list paired data from the surrogate species with measured trophic transfer factors (*TTF*) described above used to calculate bird *TTF* for T&E species, which were then used to calculate the bird composite *TTF*s for the T&E species listed in Table 5-6 and described below. Methods and data requirements are the same here as previously described for species with measured *TTFs*.

Food web data were used to first determine the proportion of plants and animals in a bird's diet, and then measured data from an appropriate surrogate species were weighted accordingly to calculate a surrogate *TTF*. Next, a composite *TTF* was calculated using specific dietary information following the methods described in Part 5.4.2.1.

Appendix Table B-9. Ridgway's Rail. Bird Egg to Diet Trophic Transfer Factor (*TTF*) after Reweighting Surrogate Species American coot Diet to a 15% Plant and 85% Animal Diet.

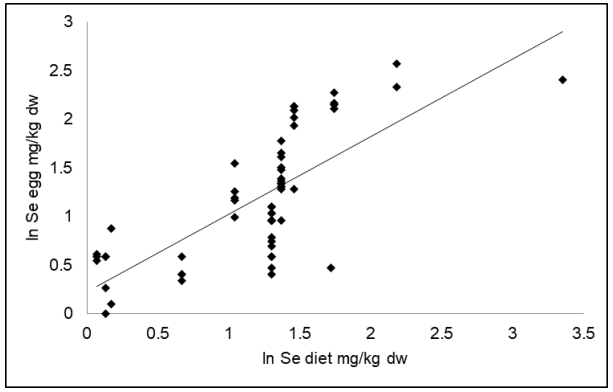
Rows with data pairs that were removed during outlier analysis are identified with bold and italics.

Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
Butler et al. 1991	7	10.72	0.15	31.75	0.85	28.60	11.10	0.39
Butler et al. 1995	TT	0.41	0.15	1.33	0.85	1.19	1.10	0.93
Butler et al. 1995	TT	0.41	0.15	1.33	0.85	1.19	2.40	2.02
Butler et al. 1997	DCP1	1.45	0.15	6.45	0.85	5.70	8.20	1.44
Butler et al. 1997	***DCP1***	***1.45***	***0.15***	***6.45***	***0.85***	***5.70***	***18.00***	***3.16***
Butler et al. 1997	DCP1	1.45	0.15	6.45	0.85	5.70	8.60	1.51
Butler et al. 1997	DCP1	1.45	0.15	6.45	0.85	5.70	9.70	1.70
Butler et al. 1997	DCP1	1.45	0.15	6.45	0.85	5.70	8.70	1.53
Butler et al. 1997	MNP2	3.85	0.15	4.40	0.85	4.32	8.10	1.88
Butler et al. 1997	MNP2	3.85	0.15	4.40	0.85	4.32	3.60	0.83
Butler et al. 1997	MNP2	3.85	0.15	4.40	0.85	4.32	6.90	1.60
Butler et al. 1997	MNP2	3.85	0.15	4.40	0.85	4.32	8.40	1.95
Butler et al. 1997	MNP2	3.85	0.15	4.40	0.85	4.32	7.50	1.74
Butler et al. 1997	MNP2	3.85	0.15	4.40	0.85	4.32	8.40	1.95
Lambing 1988	***7***	***0.47***	***0.15***	***6.50***	***0.85***	***5.60***	***1.40***	***0.25***
Lambing 1988	7	0.47	0.15	6.50	0.85	5.60	1.60	0.29
Lambing 1988	***7***	***0.47***	***0.15***	***6.50***	***0.85***	***5.60***	***1.10***	***0.20***
Lambing et al. 1994	B-21	1.42	0.15	4.38	0.85	3.94	5.90	1.50
Lambing et al. 1994	B-21	1.42	0.15	4.38	0.85	3.94	2.60	0.66
Lambing et al. 1994	B-21	1.42	0.15	4.38	0.85	3.94	5.00	1.27

Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
Lambing et al. 1994	B-21	1.42	0.15	4.38	0.85	3.94	3.80	0.96
Lambing et al. 1994	B-21	1.42	0.15	4.38	0.85	3.94	5.20	1.32
Lambing et al. 1994	B-21	1.42	0.15	4.38	0.85	3.94	3.60	0.91
Lambing et al. 1994	B-21	1.42	0.15	4.38	0.85	3.94	3.90	0.99
Lambing et al. 1994	B-21	1.42	0.15	4.38	0.85	3.94	3.80	0.96
Lambing et al. 1994	B-21	1.42	0.15	4.38	0.85	3.94	3.70	0.94
Lambing et al. 1994	B-21	1.42	0.15	4.38	0.85	3.94	4.40	1.12
Lambing et al. 1994	B-21	1.42	0.15	4.38	0.85	3.94	3.70	0.94
Lambing et al. 1994	B-21	1.42	0.15	4.38	0.85	3.94	3.80	0.96
Lambing et al. 1994	B-21	1.42	0.15	4.38	0.85	3.94	3.80	0.96
Lambing et al. 1994	B-21	1.42	0.15	4.38	0.85	3.94	4.00	1.02
Lambing et al. 1994	B-21	1.42	0.15	4.38	0.85	3.94	4.50	1.14
Lambing et al. 1994	B-22	1.00	0.15	3.15	0.85	2.83	3.50	1.24
Lambing et al. 1994	B-22	1.00	0.15	3.15	0.85	2.83	3.30	1.17
Lambing et al. 1994	B-22	1.00	0.15	3.15	0.85	2.83	2.70	0.95
Lambing et al. 1994	B-22	1.00	0.15	3.15	0.85	2.83	4.70	1.66
Lambing et al. 1994	B-22	1.00	0.15	3.15	0.85	2.83	3.20	1.13
Lambing et al. 1994	B-26	0.77	0.15	4.20	0.85	3.69	1.50	0.41
Lambing et al. 1994	B-26	0.77	0.15	4.20	0.85	3.69	1.60	0.43
Lambing et al. 1994	B-26	0.77	0.15	4.20	0.85	3.69	2.00	0.54
Lambing et al. 1994	B-26	0.77	0.15	4.20	0.85	3.69	2.60	0.71
Lambing et al. 1994	B-26	0.77	0.15	4.20	0.85	3.69	3.00	0.81
Lambing et al. 1994	B-26	0.77	0.15	4.20	0.85	3.69	2.80	0.76

Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
Lambing et al. 1994	B-26	0.77	0.15	4.20	0.85	3.69	2.20	0.60
Lambing et al. 1994	B-26	0.77	0.15	4.20	0.85	3.69	2.60	0.71
Lambing et al. 1994	B-26	0.77	0.15	4.20	0.85	3.69	2.60	0.71
Lambing et al. 1994	B-26	0.77	0.15	4.20	0.85	3.69	2.80	0.76
Lambing et al. 1994	B-26	0.77	0.15	4.20	0.85	3.69	2.60	0.71
Lambing et al. 1994	B-26	0.77	0.15	4.20	0.85	3.69	2.60	0.71
Lambing et al. 1994	B-26	0.77	0.15	4.20	0.85	3.69	2.10	0.57
Lambing et al. 1994	B-26	0.77	0.15	4.20	0.85	3.69	3.00	0.81
Lambing et al. 1994	B-26	0.77	0.15	4.20	0.85	3.69	1.80	0.49
Lambing et al. 1994	B-26	0.77	0.15	4.20	0.85	3.69	1.80	0.49
Peterson et al. 1991	3	4.64	0.15	9.62	0.85	8.87	10.30	1.16
Peterson et al. 1991	3	4.64	0.15	9.62	0.85	8.87	13.10	1.48
Rinella and Schuler 1992	N Malheur	0.56	0.15	2.20	0.85	1.95	1.80	0.92
Rinella and Schuler 1992	N Malheur	0.56	0.15	2.20	0.85	1.95	1.50	0.77
Rinella and Schuler 1992	N Malheur	0.56	0.15	2.20	0.85	1.95	1.40	0.72
Rinella and Schuler 1992	N Malheur	0.56	0.15	2.20	0.85	1.95	1.50	0.77
Rinella and Schuler 1992	S Malheur	0.82	0.15	1.20	0.85	1.14	1.30	1.14
Rinella and Schuler 1992	S Malheur	0.82	0.15	1.20	0.85	1.14	1.00	0.87
Rinella and Schuler 1992	S Malheur	0.82	0.15	1.20	0.85	1.14	1.80	1.57

Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
Rinella and Schuler 1992	S Malheur	0.82	0.15	1.20	0.85	1.14	1.80	1.57
Rinella et al. 1994	Ft. Boise WMA	0.78	0.15	1.13	0.85	1.07	1.80	1.68
Rinella et al. 1994	Ft. Boise WMA	0.78	0.15	1.13	0.85	1.07	1.73	1.61
Rinella et al. 1994	Ft. Boise WMA	0.78	0.15	1.13	0.85	1.07	1.85	1.72



Median *TTF*	0.96
Adjusted r^2	0.53
F	69.67
df	61
P	<0.001

Appendix Table B-10. Black Rail. Bird Egg to Diet (*TTF*) after Reweighting Surrogate Species American coot Diet to a 13% Plant and 87% Animal Diet.

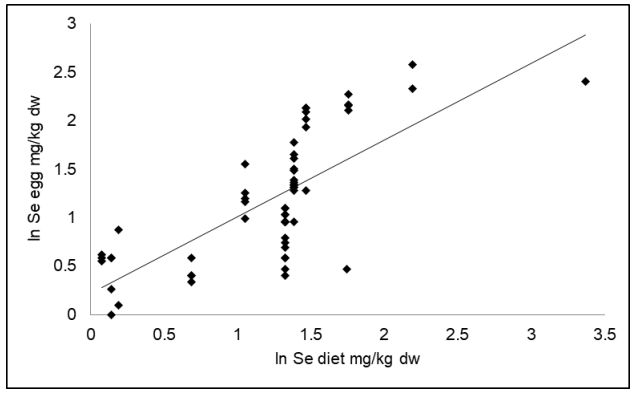
Rows with data pairs that were removed during outlier analysis are identified with bold and italics.

Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
Butler et al. 1991	7	10.72	0.15	31.75	0.85	29.02	11.10	0.38
Butler et al. 1995	TT	0.41	0.15	1.33	0.85	1.21	1.10	0.91
Butler et al. 1995	TT	0.41	0.15	1.33	0.85	1.21	2.40	1.99
Butler et al. 1997	DCP1	1.45	0.15	6.45	0.85	5.80	8.20	1.41
Butler et al. 1997	***DCP1***	***1.45***	***0.15***	***6.45***	***0.85***	***5.80***	***18.00***	***3.10***
Butler et al. 1997	DCP1	1.45	0.15	6.45	0.85	5.80	8.60	1.48
Butler et al. 1997	DCP1	1.45	0.15	6.45	0.85	5.80	9.70	1.67
Butler et al. 1997	DCP1	1.45	0.15	6.45	0.85	5.80	8.70	1.50
Butler et al. 1997	MNP2	3.85	0.15	4.40	0.85	4.33	8.10	1.87
Butler et al. 1997	MNP2	3.85	0.15	4.40	0.85	4.33	3.60	0.83
Butler et al. 1997	MNP2	3.85	0.15	4.40	0.85	4.33	6.90	1.59
Butler et al. 1997	MNP2	3.85	0.15	4.40	0.85	4.33	8.40	1.94
Butler et al. 1997	MNP2	3.85	0.15	4.40	0.85	4.33	7.50	1.73
Butler et al. 1997	MNP2	3.85	0.15	4.40	0.85	4.33	8.40	1.94
Lambing 1988	***7***	***0.47***	***0.15***	***6.50***	***0.85***	***5.72***	***1.40***	***0.24***
Lambing 1988	7	0.47	0.15	6.50	0.85	5.72	1.60	0.28
Lambing 1988	***7***	***0.47***	***0.15***	***6.50***	***0.85***	***5.72***	***1.10***	***0.19***
Lambing et al. 1994	B-21	1.42	0.15	4.38	0.85	4.00	5.90	1.48
Lambing et al. 1994	B-21	1.42	0.15	4.38	0.85	4.00	2.60	0.65
Lambing et al. 1994	B-21	1.42	0.15	4.38	0.85	4.00	5.00	1.25
Lambing et al. 1994	B-21	1.42	0.15	4.38	0.85	4.00	3.80	0.95
Lambing et al. 1994	B-21	1.42	0.15	4.38	0.85	4.00	5.20	1.30

Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
Lambing et al. 1994	B-21	1.42	0.15	4.38	0.85	4.00	3.60	0.90
Lambing et al. 1994	B-21	1.42	0.15	4.38	0.85	4.00	3.90	0.98
Lambing et al. 1994	B-21	1.42	0.15	4.38	0.85	4.00	3.80	0.95
Lambing et al. 1994	B-21	1.42	0.15	4.38	0.85	4.00	3.70	0.93
Lambing et al. 1994	B-21	1.42	0.15	4.38	0.85	4.00	4.40	1.10
Lambing et al. 1994	B-21	1.42	0.15	4.38	0.85	4.00	3.70	0.93
Lambing et al. 1994	B-21	1.42	0.15	4.38	0.85	4.00	3.80	0.95
Lambing et al. 1994	B-21	1.42	0.15	4.38	0.85	4.00	3.80	0.95
Lambing et al. 1994	B-21	1.42	0.15	4.38	0.85	4.00	4.00	1.00
Lambing et al. 1994	B-21	1.42	0.15	4.38	0.85	4.00	4.50	1.13
Lambing et al. 1994	B-22	1.00	0.15	3.15	0.85	2.87	3.50	1.22
Lambing et al. 1994	B-22	1.00	0.15	3.15	0.85	2.87	3.30	1.15
Lambing et al. 1994	B-22	1.00	0.15	3.15	0.85	2.87	2.70	0.94
Lambing et al. 1994	B-22	1.00	0.15	3.15	0.85	2.87	4.70	1.64
Lambing et al. 1994	B-22	1.00	0.15	3.15	0.85	2.87	3.20	1.11
Lambing et al. 1994	B-26	0.77	0.15	4.20	0.85	3.75	1.50	0.40
Lambing et al. 1994	B-26	0.77	0.15	4.20	0.85	3.75	1.60	0.43
Lambing et al. 1994	B-26	0.77	0.15	4.20	0.85	3.75	2.00	0.53
Lambing et al. 1994	B-26	0.77	0.15	4.20	0.85	3.75	2.60	0.69
Lambing et al. 1994	B-26	0.77	0.15	4.20	0.85	3.75	3.00	0.80
Lambing et al. 1994	B-26	0.77	0.15	4.20	0.85	3.75	2.80	0.75
Lambing et al. 1994	B-26	0.77	0.15	4.20	0.85	3.75	2.20	0.59
Lambing et al. 1994	B-26	0.77	0.15	4.20	0.85	3.75	2.60	0.69
Lambing et al. 1994	B-26	0.77	0.15	4.20	0.85	3.75	2.60	0.69
Lambing et al. 1994	B-26	0.77	0.15	4.20	0.85	3.75	2.80	0.75

Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
Lambing et al. 1994	B-26	0.77	0.15	4.20	0.85	3.75	2.60	0.69
Lambing et al. 1994	B-26	0.77	0.15	4.20	0.85	3.75	2.60	0.69
Lambing et al. 1994	B-26	0.77	0.15	4.20	0.85	3.75	2.10	0.56
Lambing et al. 1994	B-26	0.77	0.15	4.20	0.85	3.75	3.00	0.80
Lambing et al. 1994	B-26	0.77	0.15	4.20	0.85	3.75	1.80	0.48
Lambing et al. 1994	B-26	0.77	0.15	4.20	0.85	3.75	1.80	0.48
Peterson et al. 1991	3	4.64	0.15	9.62	0.85	8.97	10.30	1.15
Peterson et al. 1991	3	4.64	0.15	9.62	0.85	8.97	13.10	1.46
Rinella and Schuler 1992	N Malheur	0.56	0.15	2.20	0.85	1.99	1.80	0.91
Rinella and Schuler 1992	N Malheur	0.56	0.15	2.20	0.85	1.99	1.50	0.75
Rinella and Schuler 1992	N Malheur	0.56	0.15	2.20	0.85	1.99	1.40	0.70
Rinella and Schuler 1992	N Malheur	0.56	0.15	2.20	0.85	1.99	1.50	0.75
Rinella and Schuler 1992	S Malheur	0.82	0.15	1.20	0.85	1.15	1.30	1.13
Rinella and Schuler 1992	S Malheur	0.82	0.15	1.20	0.85	1.15	1.00	0.87
Rinella and Schuler 1992	S Malheur	0.82	0.15	1.20	0.85	1.15	1.80	1.56
Rinella and Schuler 1992	S Malheur	0.82	0.15	1.20	0.85	1.15	1.80	1.56
Rinella et al. 1994	Ft. Boise WMA	0.78	0.15	1.13	0.85	1.08	1.80	1.67
Rinella et al. 1994	Ft. Boise	0.78	0.15	1.13	0.85	1.08	1.73	1.60

Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
	WMA							
Rinella et al. 1994	Ft. Boise WMA	0.78	0.15	1.13	0.85	1.08	1.85	1.71



Median *TTF*	0.95
Adjusted r^2	0.52
F	68.24
df	61
P	<0.001

Appendix Table B-11. Light-Footed Ridgeway's rail and Yuma rail. Bird Egg to Diet (*TTF*) after Reweighting Surrogate Species American coot Diet to a 100% Animal Diet.

Because these species eat a 100% animal diet, all paired animal-egg measurements were used, regardless of whether a paired plant measurement was available. Rows with data pairs that were removed during outlier analysis are identified with bold and italics.

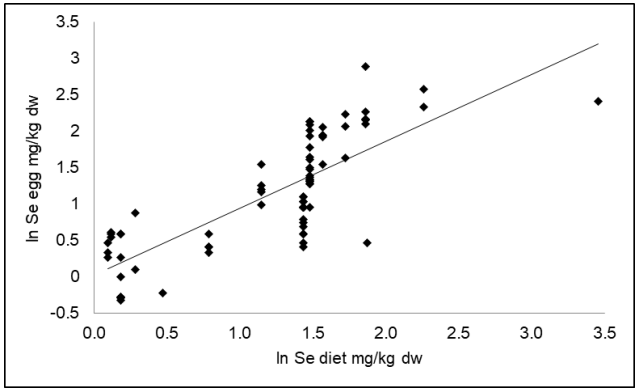
Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
Butler et al. 1991	7	n/a	0.0	31.75	1.0	31.75	11.1	0.35
Butler et al. 1995	TT	n/a	0.0	1.33	1.0	1.33	1.1	0.83
Butler et al. 1995	TT	n/a	0.0	1.33	1.0	1.33	2.4	1.81
Butler et al. 1997	DCP1	n/a	0.0	6.45	1.0	6.45	8.2	1.27
Butler et al. 1997	DCP1	n/a	0.0	6.45	1.0	6.45	18	2.79
Butler et al. 1997	DCP1	n/a	0.0	6.45	1.0	6.45	8.6	1.33
Butler et al. 1997	DCP1	n/a	0.0	6.45	1.0	6.45	9.7	1.50
Butler et al. 1997	DCP1	n/a	0.0	6.45	1.0	6.45	8.7	1.35
Butler et al. 1997	MNP2	n/a	0.0	4.40	1.0	4.40	8.1	1.84
Butler et al. 1997	MNP2	n/a	0.0	4.40	1.0	4.40	3.6	0.82
Butler et al. 1997	MNP2	n/a	0.0	4.40	1.0	4.40	6.9	1.57
Butler et al. 1997	MNP2	n/a	0.0	4.40	1.0	4.40	8.4	1.91
Butler et al. 1997	MNP2	n/a	0.0	4.40	1.0	4.40	7.5	1.70
Butler et al. 1997	MNP2	n/a	0.0	4.40	1.0	4.40	8.4	1.91
Lambing 1988	***7***	**n/a**	**0.0**	***6.50***	**1.0**	***6.50***	***1.4***	***0.22***
Lambing 1988	7	n/a	0.0	6.50	1.0	6.50	1.6	0.25
Lambing 1988	***7***	**n/a**	**0.0**	***6.50***	**1.0**	***6.50***	***1.1***	***0.17***
Lambing 1988	10	n/a	0.0	1.10	1.0	1.10	1.6	1.45
Lambing 1988	10	n/a	0.0	1.10	1.0	1.10	1.4	1.27
Lambing 1988	10	n/a	0.0	1.10	1.0	1.10	1.3	1.18

Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
Lambing et al. 1994	B-16	n/a	0.0	4.80	1.0	4.80	6.8	1.42
Lambing et al. 1994	B-16	n/a	0.0	4.80	1.0	4.80	4.7	0.98
Lambing et al. 1994	B-16	n/a	0.0	4.80	1.0	4.80	7	1.46
Lambing et al. 1994	B-16	n/a	0.0	4.80	1.0	4.80	7.8	1.63
Lambing et al. 1994	B-16	n/a	0.0	4.80	1.0	4.80	7	1.46
Lambing et al. 1994	B-19	n/a	0.0	5.60	1.0	5.60	7.9	1.41
Lambing et al. 1994	B-19	n/a	0.0	5.60	1.0	5.60	9.3	1.66
Lambing et al. 1994	B-19	n/a	0.0	5.60	1.0	5.60	5.1	0.91
Lambing et al. 1994	B-21	n/a	0.0	4.38	1.0	4.38	5.9	1.35
Lambing et al. 1994	B-21	n/a	0.0	4.38	1.0	4.38	2.6	0.59
Lambing et al. 1994	B-21	n/a	0.0	4.38	1.0	4.38	5	1.14
Lambing et al. 1994	B-21	n/a	0.0	4.38	1.0	4.38	3.8	0.87
Lambing et al. 1994	B-21	n/a	0.0	4.38	1.0	4.38	5.2	1.19
Lambing et al. 1994	B-21	n/a	0.0	4.38	1.0	4.38	3.6	0.82
Lambing et al. 1994	B-21	n/a	0.0	4.38	1.0	4.38	3.9	0.89
Lambing et al. 1994	B-21	n/a	0.0	4.38	1.0	4.38	3.8	0.87
Lambing et al. 1994	B-21	n/a	0.0	4.38	1.0	4.38	3.7	0.84
Lambing et al. 1994	B-21	n/a	0.0	4.38	1.0	4.38	4.4	1.00
Lambing et al. 1994	B-21	n/a	0.0	4.38	1.0	4.38	3.7	0.84
Lambing et al. 1994	B-21	n/a	0.0	4.38	1.0	4.38	3.8	0.87
Lambing et al. 1994	B-21	n/a	0.0	4.38	1.0	4.38	3.8	0.87
Lambing et al. 1994	B-21	n/a	0.0	4.38	1.0	4.38	4	0.91
Lambing et al. 1994	B-21	n/a	0.0	4.38	1.0	4.38	4.5	1.03
Lambing et al. 1994	B-22	n/a	0.0	3.15	1.0	3.15	3.5	1.11

Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
Lambing et al. 1994	B-22	n/a	0.0	3.15	1.0	3.15	3.3	1.05
Lambing et al. 1994	B-22	n/a	0.0	3.15	1.0	3.15	2.7	0.86
Lambing et al. 1994	B-22	n/a	0.0	3.15	1.0	3.15	4.7	1.49
Lambing et al. 1994	B-22	n/a	0.0	3.15	1.0	3.15	3.2	1.02
Lambing et al. 1994	B-26	n/a	0.0	4.20	1.0	4.20	1.5	0.36
Lambing et al. 1994	B-26	n/a	0.0	4.20	1.0	4.20	1.6	0.38
Lambing et al. 1994	B-26	n/a	0.0	4.20	1.0	4.20	2	0.48
Lambing et al. 1994	B-26	n/a	0.0	4.20	1.0	4.20	2.6	0.62
Lambing et al. 1994	B-26	n/a	0.0	4.20	1.0	4.20	3	0.71
Lambing et al. 1994	B-26	n/a	0.0	4.20	1.0	4.20	2.8	0.67
Lambing et al. 1994	B-26	n/a	0.0	4.20	1.0	4.20	2.2	0.52
Lambing et al. 1994	B-26	n/a	0.0	4.20	1.0	4.20	2.6	0.62
Lambing et al. 1994	B-26	n/a	0.0	4.20	1.0	4.20	2.6	0.62
Lambing et al. 1994	B-26	n/a	0.0	4.20	1.0	4.20	2.8	0.67
Lambing et al. 1994	B-26	n/a	0.0	4.20	1.0	4.20	2.6	0.62
Lambing et al. 1994	B-26	n/a	0.0	4.20	1.0	4.20	2.6	0.62
Lambing et al. 1994	B-26	n/a	0.0	4.20	1.0	4.20	2.1	0.50
Lambing et al. 1994	B-26	n/a	0.0	4.20	1.0	4.20	3	0.71
Lambing et al. 1994	B-26	n/a	0.0	4.20	1.0	4.20	1.8	0.43
Lambing et al. 1994	B-26	n/a	0.0	4.20	1.0	4.20	1.8	0.43
Low and Mullins 1990	**Spring Creek**	**n/a**	**0.0**	**1.60**	**1.0**	**1.60**	**0.4**	**0.25**
Low and Mullins 1990	Spring Creek	n/a	0.0	1.60	1.0	1.60	0.8	0.50

Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
Ong et al. 1991	24C	n/a	0.0	1.20	1.0	1.20	0.72	0.60
Ong et al. 1991	24C	n/a	0.0	1.20	1.0	1.20	0.75	0.63
Ong et al. 1991	24C	n/a	0.0	1.20	1.0	1.20	0.76	0.63
Ong et al. 1991	24C	n/a	0.0	1.20	1.0	1.20	0.76	0.63
Peterson et al. 1991	3	n/a	0.0	9.62	1.0	9.62	10.3	1.07
Peterson et al. 1991	3	n/a	0.0	9.62	1.0	9.62	13.1	1.36
Rinella and Schuler 1992	N Malheur	n/a	0.0	2.20	1.0	2.20	1.8	0.82
Rinella and Schuler 1992	N Malheur	n/a	0.0	2.20	1.0	2.20	1.5	0.68
Rinella and Schuler 1992	N Malheur	n/a	0.0	2.20	1.0	2.20	1.4	0.64
Rinella and Schuler 1992	N Malheur	n/a	0.0	2.20	1.0	2.20	1.5	0.68
Rinella and Schuler 1992	S Malheur	n/a	0.0	1.20	1.0	1.20	1.3	1.08
Rinella and Schuler 1992	S Malheur	n/a	0.0	1.20	1.0	1.20	1	0.83
Rinella and Schuler 1992	S Malheur	n/a	0.0	1.20	1.0	1.20	1.8	1.50
Rinella and Schuler 1992	S Malheur	n/a	0.0	1.20	1.0	1.20	1.8	1.50
Rinella et al. 1994	Ft. Boise WMA	n/a	0.0	1.13	1.0	1.13	1.8	1.60
Rinella et al. 1994	Ft. Boise WMA	n/a	0.0	1.13	1.0	1.13	1.73	1.54

Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
Rinella et al. 1994	Ft. Boise WMA	n/a	0.0	1.13	1.0	1.13	1.85	1.64



Median *TTF*	0.90
Adjusted r^2	0.60
F	120.8
df	78
P	<0.001

Calculation of $TTF^{composite}$ for T&E Species

This section describes the calculation of $TTF^{composite}$ for the eight T&E bird species with measured TTF of surrogate species using data listed in the preceding tables. $TTF^{composite}$ were calculated from food webs modeled using information from the Cornell Lab of Ornithology Birds of North America web site: (http://www.birds.cornell.edu/Page.aspx?pid=1478), and following the methods described in Part 5.4.2.1. $TTF^{composite}$ for these species are listed in Table 5-5.

Composite *TTF* Results

American Dipper

According to the U.S. FWS (2017), aquatic insects, primarily benthic macroinvertebrate larvae such as mayflies (Ephemeroptera), caddisflies (Trichoptera), and stoneflies (Plecoptera), make up the majority of the American dipper's diet. This species will also consume other aquatic organisms, including small fish and fish eggs. The abundance of prey items determines the presence of dippers within a watershed (Feck 2002). The diet of American dippers varies based time of the year (i.e., breeding season or non-breeding season) and habitat (Wilson and Kingery 2011). Morrissey et al. (2010, 2012) found that female American dippers switched to feeding at a higher trophic level (such as fish and predatory invertebrates) during egg-laying. Additionally, using isotopic signatures, Morrissey et al. (2004) determined that non-migratory dippers ate a higher percentage of fish (42% ± 7) than migrant dippers (22% ± 6).

Based on this information, the American dipper diet was modeled under two scenarios: a low fish diet (diet consisting of 22% small fish and 78% aquatic insect) and a high fish diet (diet consisting of 42% small fish and 58% aquatic insect). Additionally, to calculate a composite *TTF* for American dipper the mean composite fish *TTF* of 2.34 based on the average fish composite TTFs for all translation sites in U.S. EPA (2016a) (Appendix Table B-12) and an aquatic insect *TTF* of 2.14 was used (Table 5-1).

As empirical bird *TTFs* were not available for American dipper, a composite *TTF* for this species of concern was calculated from a closely related surrogate species with empirical bird *TTFs*. To calculate a composite *TTF* for American dipper the passerine *TTF* of 0.95 (based on the average *TTF* for red-winged blackbird and yellow-headed blackbird) was applied.

Low fish diet scenario:

$$TTF^{composite} = [0.95 \times 2.34 \times 0.22] + [0.95 \times 2.14 \times 0.78] = \mathbf{2.07}$$

High fish diet scenario:

$$TTF^{composite} = [0.95 \times 2.34 \times 0.42] + [0.95 \times 2.14 \times 0.58] = \mathbf{2.11}$$

Brown Pelican

U.S. FWS (2017) reported that along the California coast, brown pelicans are dependent on small, surface schooling fish, such as anchovy (*Engraulis mordax*) and Pacific sardines (*Sardinops sagax*). Brown pelican's diet consists of 90% northern anchovy during the breeding season.

Based on this information relating to the dietary composition of brown pelican, the diet of this species was modeled to consist of 100% fish. As the empirically measured fish *TTFs* were not available for the fish species identified by U.S. FWS (2017), a mean composite fish *TTF* of 2.34 was used (Appendix Table B-12).

As empirical bird *TTFs* were not available for brown pelican, a composite *TTF* for this species was calculated using a surrogate species (pied-billed grebe) with a similar diet for which there is an empirically derived *TTF*. Therefore, the *TTF* of 0.78 for pied billed grebe was applied.

$$TTF^{composite} = [0.78 \times 2.34] = \mathbf{1.83}$$

Bald Eagle

As bald eagles are opportunistic foragers, the dietary composition of this species is widely variable and is based on the availability of prey species (Buehler 2000). Generally known as a piscivore, bald eagles also often consume various other prey and carrion. Diets of bald eagles inhabiting northern California commonly consisted of Sacramento sucker (*Catostomus occidentalis*), hardhead (*Mylopharodon conocephalsu)*, Sacramento pikeminnow (*Ptechocheilus grandis*), brown bullhead (*Ameiurus nebulosus*), common carp (*Cyprinus carpio*), tui chub (*Gila bicolor*), rainbow trout (*Onchorhyncus mykiss*), largemouth bass (*Micropterus salmoides*), Sacramento perch (*Archoptlites interruptus*), American coot (*Fulica americana*), mallard (*Anas platyrhynchos*), western grebe (*Aechmophorus occidentalis*), gulls (*Larus spp.*), pied-billed grebe (*Podilymbus podiceps*), common merganser (*Mergus merganser*), and other diving ducks (U.S.

FWS 2017; Hunt et al. 1992; Jackman et al. 1999). In the U.S. FWS (2017) report, a generic dietary composition for northern California bald eagles was estimated to be 71.2% fish, 22.8% bird, and 6% mammal.

As noted above, since the dietary composition for this species can be highly variable (Buehler 2000) and the selenium from the mammalian part of the diet likely may not be related to aquatic exposures, the mammal component of the diet was not included in the calculation of the composite *TTF* below. Therefore, the U.S. FWS (2017) estimated generic dietary composition of 71.2% fish and 22.8% bird was used to calculate a bald eagle composite *TTF* for selenium. Additionally, a mean composite fish *TTF* of 2.34 (Appendix Table B-12) and a mean composite bird *TTF* of 2.15 (the average $TTF^{composite}$ for all bird species excluding bald eagle) was used.

As empirical bird *TTFs* were not available for bald eagle, a composite *TTF* for this species of concern was calculated using an empirically-derived bird to fish *TTF* of 0.78 from a surrogate species of pied-billed grebe. This surrogate species is a piscivore and therefore has a diet consisting of a similar trophic position (trophic level 4). As this is the only empirically-derived *TTF* for a bird species with a largely piscivorous diet, and because there are no known empirically-derived bird *TTFs* for bird eating birds, the pied billed grebe *TTF* was considered the best surrogate *TTF* for bald eagle.

$$TTF^{composite} = [0.78 \times 2.34 \times 0.712] + [0.78 \times 2.15 \times 0.228] = \mathbf{1.69}$$

Ridgway's Rail

Ridgway's rails are omnivorous species with a highly variable diet (Rush et al. 2012). As reported by U.S. FWS (2017) on average, animal matter accounted for roughly 85% of Ridgway's rails diet with the remainder being composed of seed and hull fragments of marsh cordgrass. Moffitt (1941) identified the stomach contents of eighteen Ridgway's rails and found that the animal matter portion of their overall diet consisted of approximately 56.5% plaited horse mussels (*Modiolus demissus*), 15% spiders (Lycosidae), 7.6% macoma clams (*Macoma balthica*), 3.2% yellow shore crabs (*Hemigrapsis oregonesis*), 2% worn-out nassa snails (*Ilyanassa obsoletus*), and 1.1% worms, insects, and carrion (combined). The remaining 15% of their diet consisted of plant matter.

Based on this information, the Ridgway's rail's diet is modeled as consisting of 15% plant matter and 85% invertebrates (64% mussels, 3% crabs, and 18% other invertebrates). A mollusk *TTF* of 4.29, a crustacean *TTF* of 1.41, and an invertebrate *TTF* of 1.89 were used (Table 5-1).

Because there is no empirically-derived bird to plant or bird to invertebrate *TTFs* for Ridgway's rail, the Ridgway's rail *TTFs* was calculated using paired data from the closely related American coot (also from the order Gruiformes), based on a diet consisting of 15% plants and 85% animals (Appendix Table B-9). Data were natural log transformed, and three egg-diet pairs were removed following outlier analysis. The resulting regression for the Ridgway's rail weighted diet was positive and statistically significant, resulting in a TTF^{TL3} of 0.96. The $TTF^{composite}$ for Ridgway's rail is calculated as follows using the modeled diet described above.

$$TTF^{composite} = [0.96 \times 0.15] + \left[0.96 \times \left((4.29 \times 0.64) + (1.41 \times 0.03) + (1.89 \times 0.18)\right)\right] = \mathbf{3.16}$$

Light-Footed Ridgway's Rail

Like the Ridgway's rail, the light-footed Ridgway's rail is an opportunistic forager and omnivore with a highly variable diet (U.S. FWS 2003). In their 2017 report, U.S. FWS reported that light-footed Ridgway's rail relies on salt marsh invertebrates, such as mussels, snails, fiddler and hermit crabs, fish, crayfish, isopods, and beetles. In 2003, U.S. FWS assumed that light-footed Ridgway's rail diet was 10% crayfish and 10% fish, leaving the remaining 80% to be aquatic invertebrates.

From this information, the light-footed Ridgway's rail dietary composition was assumed to be 80% invertebrates, 10% crayfish, and 10% fish. Additionally, an invertebrate *TTF* of 1.89, a crayfish *TTF* of 1.46, and a composite fish *TTF* of 2.34 was used to calculate a composite *TTF* for light-footed Ridgway's rail.

Because there is no empirically-derived bird to plant or bird to invertebrate *TTFs* for light-footed Ridgway's rail, its *TTFs* were calculated using paired data from the closely related American coot (also from the order Gruiformes), based on a diet consisting of 100% animals (Appendix Table B-11). Because light-footed Ridgway's rails do not consume plants, all available bird egg-animal pairs were used, regardless of whether or not they were also paired

with a plant measurement. Data were natural log transformed, and three egg-diet pairs were removed following outlier analysis. The resulting regression for the light-footed Ridegway's rail weighted diet was positive and statistically significant, resulting in a TTF^{TL4} of 0.90. The $TTF^{composite}$ for light footed Ridgway's rail is calculated as follows using the modeled diet described above.

$$TTF^{composite} = [0.90 \times 2.34 \times 0.1] + \left[0.90 \times \left((1.89 \times 0.8) + (1.46 \times 0.1)\right)\right] = \mathbf{1.85}$$

Yuma Ridgway's Rail

As reported by U.S. FWS (2017), the dietary composition of Yuma Ridgway's rail is dominated by two species of crayfish. Ohmart and Tomlinson (1977) found that approximately 95% of the stomach contents of two Yuma Ridgway's rails consisted of crayfish. Other prey items consumed by Yuma Ridgway's rails include small fish, insects, amphibian larvae, clams, and other aquatic invertebrates (U.S. FWS 2010, 2017).

Therefore, the dietary composition of Yuma Ridgway's rail was assumed to be 95% crayfish and 5% other aquatic invertebrates, since in the 2003 report U.S. FWS indicated that fish do not appear to be an important dietary item for Yuma Ridgway's rail residing outside of the Colorado River Delta in Mexico. A crayfish *TTF* of 1.46 and an all invertebrate *TTF* of 1.89 was used to calculate a composite *TTF* for Yuma Ridgway's rail (U.S. EPA 2016a).

Because there is no empirically-derived bird to plant or bird to invertebrate *TTFs* for Yuma Ridgway's rail, its *TTFs* was calculated using paired data from the closely related American coot (also from the order Gruiformes), based on a diet consisting of 100% animals (Appendix Table B-11). Because Yuma Ridgway's rail do not consume plants, all available bird egg-animal pairs were used, regardless of whether or not they were also paired with a plant measurement. Data were natural log transformed, and three egg-diet pairs were removed following outlier analysis. The resulting regression for the Yuma Ridgway's rail weighted diet was positive and statistically significant, resulting in a TTF^{TL3} of 0.90. The $TTF^{composite}$ for Yuma Ridgway's rail is calculated as follows using the modeled diet described above.

$$TTF^{composite} = \left[0.90 \times \left((1.46 \times 0.95) + (1.89 \times 0.05)\right)\right] = \mathbf{1.33}$$

Black Rail

U.S. FWS (2017) reports that dietary information for black rail is limited and notes that this species is likely an opportunistic forager with a variable diet dependent on food availability. This species consumes invertebrates and seeds. Eddleman et al. (1994) indicated that the dietary composition of nesting black rails consisted of 73% predaceous diving, ground and other beetles, 14% earwigs, 13% bulrush seeds, and trace amounts of cattail.

Due to the limited information about and the variability of the black rail's diet, an assumed dietary composition of 13% plant matter and 87% aquatic invertebrates was used to calculate a composite *TTF* for this species. An all invertebrate *TTF* of 1.89 was used (Table 5-1).

Because there is no empirically-derived bird to plant or bird to invertebrate *TTFs* for black rail, its *TTFs* was calculated using paired data from the closely related American coot (also from the order Gruiformes), based on a diet consisting of 13% plants and 87% animals (Appendix Table B-10). Data were natural log transformed, and three egg-diet pairs were removed following outlier analysis. The resulting regression for the black rail weighted diet was positive and statistically significant, resulting in a TTF^{TL3} of 0.95. The $TTF^{composite}$ for black rail is calculated as follows using the modeled diet described above.

$$TTF^{composite} = [0.95 \times 0.13] + [0.95 \times 1.89 \times 0.87] = \mathbf{1.69}$$

Least Tern

As reported by U.S. FWS (2017), least terns primarily consume small fish (<8 cm in length). Fish species commonly consumed include northern anchovy (*Engraulis mordax*), top smelt (*Atherinops affinis*), silversides (*Atherinopsidae*), herring (*Culpeidae*), and yellowfin goby (*Acanthogobius flavimanus*). However, least terns have been documented to consume up 50 different species of fish (U.S. FWS 1985).

Based on this information in U.S. FWS (2017) report, the dietary composition of the least tern was assumed to be 100% fish. As the empirically measured fish *TTFs* were not available for all of the fish species identified by U.S FWS (2017) and highly variable diet of least tern, a mean composite fish *TTF* of 2.34 was used (Appendix Table B-12).

As empirical bird *TTFs* were not available for least tern, a composite *TTF* for this species was calculated using a surrogate species (pied-billed grebe) with a similar diet for which there is

an empirically-derived bird to fish *TTF*. Therefore, a pied-billed grebe bird to fish *TTF* of 0.78 was applied, and composite *TTF* calculations are provided below.

Pied-Billed Grebe Surrogate:

$$TTF^{composite} = [0.78 \times 2.34] = \mathbf{1.83}$$

Summary

Composite *TTFs* could be calculated for eight T&E bird species using paired dietary information from surrogate species to calculate surrogate *TTFs*. $TTF^{composite}$ for these species are listed in Table 5-6.

Calculation of Fish $TTF^{composite}$ for Bird Food Web Modeling

The fish $TTF^{composite}$ used to calculate avian $TTF^{composite}$ for the six bird species that consume fish as part of their diet (pied billed grebe, American dipper, brown pelican, bald eagle, light-footed Ridgway's rail, and least tern) was determined as follows. For the 65 sites in the translation dataset, where an *EF* was calculated and fish were sampled, the $TTF^{composite}$ for each species of fish was recorded. Next, the average $TTF^{composite}$ for all fish species at that site calculated. Finally, the 65 fish $TTF^{composite}$ site averages were averaged into a single overall average fish $TTF^{composite}$ of 2.34 (Appendix Table B-12). Fish $TTF^{composite}$ were obtained from Appendix B of U.S. EPA (2016a).

Appendix Table B-12. Fish $TTF^{composite}$ from the 65 Sites Used in the Tissue to Water Translation Dataset.

The fish $TTF^{composite}$ used to model avian $TTF^{composite}$ was the overall average of the 65 site averages.

Reference	Site Description	Fish Species	Waterbody Type	Fish $TTF^{composite\text{-}a}$	Average Fish $TTF^{composite}$ at a Site
Birkner 1978	East Allen Reservoir, Medicine Bow WY	Iowa darter	Lentic	2.87	2.87
Birkner 1978	Galett Lake, Laramie WY	Iowa darter	Lentic	2.87	2.87
Birkner 1978	Larimer Highway 9 Pond, Fort Collins CO	northern plains killifish	Lentic	2.44	2.44
Birkner 1978	Meeboer Lake, Laramie WY	northern plains killifish	Lentic	2.44	2.44
Birkner 1978	Miller's Lake, Wellington CO	fathead minnow	Lentic	2.78	2.82
Birkner 1978	Miller's Lake, Wellington CO	Iowa darter	Lentic	2.87	
Birkner 1978	Sweitzer Lake, Delta CO	fathead minnow	Lentic	2.78	2.61
Birkner 1978	Sweitzer Lake, Delta CO	northern plains killifish	Lentic	2.44	
Birkner 1978	Twin Buttes Reservoir, Laramie WY	fathead minnow	Lentic	2.78	2.70
Birkner 1978	Twin Buttes Reservoir, Laramie WY	Iowa darter	Lentic	2.87	
Birkner 1978	Twin Buttes Reservoir, Laramie WY	northern plains killifish	Lentic	2.44	
Bowie et al. 1996	Hyco Reservoir	bluegill	Lentic	2.00	2.00
Butler et al. 1993	Navajo Reservoir, Piedra River Arm, near La Boca	brown trout	Lentic	2.78	1.83
Butler et al. 1993	Navajo Reservoir, Piedra River Arm, near La Boca	bullhead	Lentic	1.62	
Butler et al. 1993	Navajo Reservoir, Piedra River Arm, near La Boca	channel catfish	Lentic	1.35	
Butler et al. 1993	Navajo Reservoir, Piedra River Arm, near La Boca	common carp	Lentic	1.58	

Reference	Site Description	Fish Species	Waterbody Type	Fish $TTF^{composite-a}$	Average Fish $TTF^{composite}$ at a Site
Butler et al. 1997	Large pond south of G Road, southern Mancos Valley	fathead minnow	Lentic	2.78	2.78
Butler et al. 1997	Pond downstream from site MNP2, southern Mancos Valley	smallmouth bass	Lentic	1.93	1.93
Butler et al. 1997	Pond on Woods Canyon at 15 Road	fathead minnow	Lentic	2.78	2.78
Grasso et al. 1995	Arapahoe Wetlands Pond	fathead minnow	Lentic	2.78	2.18
Grasso et al. 1995	Arapahoe Wetlands Pond	white sucker	Lentic	1.58	
Lemly 1985	Badin Lake	black bullhead	Lentic	1.72	2.17
Lemly 1985	Badin Lake	common carp	Lentic	1.58	
Lemly 1985	Badin Lake	fathead minnow	Lentic	2.78	
Lemly 1985	Badin Lake	green sunfish	Lentic	2.29	
Lemly 1985	Badin Lake	western mosquitofish	Lentic	2.37	
Lemly 1985	Badin Lake	red shiner	Lentic	2.27	
Lemly 1985	Belews Lake	black bullhead	Lentic	1.72	2.17
Lemly 1985	Belews Lake	common carp	Lentic	1.58	
Lemly 1985	Belews Lake	fathead minnow	Lentic	2.78	
Lemly 1985	Belews Lake	green sunfish	Lentic	2.29	
Lemly 1985	Belews Lake	western mosquitofish	Lentic	2.37	
Lemly 1985	Belews Lake	red shiner	Lentic	2.27	
Lemly 1985	High Rock Lake	black bullhead	Lentic	1.72	2.17
Lemly 1985	High Rock Lake	common carp	Lentic	1.58	
Lemly 1985	High Rock Lake	fathead minnow	Lentic	2.78	
Lemly 1985	High Rock Lake	green sunfish	Lentic	2.29	

Reference	Site Description	Fish Species	Waterbody Type	Fish $TTF^{composite-a}$	Average Fish $TTF^{composite}$ at a Site
Lemly 1985	High Rock Lake	western mosquitofish	Lentic	2.37	
Lemly 1985	High Rock Lake	red shiner	Lentic	2.27	
Muscatello and Janz 2009	Vulture Lake	burbot	Lentic	2.45	2.82
Muscatello and Janz 2009	Vulture Lake	ninespine stickleback	Lentic	3.22	
Muscatello and Janz 2009	Vulture Lake	northern pike	Lentic	4.02	
Muscatello and Janz 2009	Vulture Lake	white sucker	Lentic	1.58	
Orr et al. 2012	Clode Pond 11	cutthroat trout	Lentic	2.29	2.29
Orr et al. 2012	Elk Lakes 14	cutthroat trout	Lentic	2.29	
Orr et al. 2012	Fording River Oxbow 10	cutthroat trout	Lentic	2.29	
Orr et al. 2012	Henretta Lake 27	cutthroat trout	Lentic	2.29	
Saiki and Lowe 1987	Kesterson Pond 11	western mosquitofish	Lentic	2.37	2.37
Saiki and Lowe 1987	Kesterson Pond 2	western mosquitofish	Lentic	2.37	2.37
Saiki and Lowe 1987	Kesterson Pond 8	western mosquitofish	Lentic	2.37	2.37
Saiki and Lowe 1987	Volta Pond 26	western mosquitofish	Lentic	2.37	2.37
Stephens et al. 1988	Marsh 4720	black bullhead	Lentic	1.72	1.65
Stephens et al. 1988	Marsh 4720	common carp	Lentic	1.58	
Butler et al. 1991	Uncompahgre River at Colona	bluehead sucker	Lotic	1.24	2.03
Butler et al. 1991	Uncompahgre River at Colona	brown trout	Lotic	2.78	

Reference	Site Description	Fish Species	Waterbody Type	Fish $TTF^{composite-a}$	Average Fish $TTF^{composite}$ at a Site
Butler et al. 1991	Uncompahgre River at Colona	flannelmouth sucker	Lotic	1.52	
Butler et al. 1991	Uncompahgre River at Colona	mottled sculpin	Lotic	2.72	
Butler et al. 1991	Uncompahgre River at Colona	rainbow trout	Lotic	2.33	
Butler et al. 1991	Uncompahgre River at Colona	white sucker	Lotic	1.58	
Butler et al. 1993	Spring Cr. at La Boca	bluehead sucker	Lotic	1.24	1.95
Butler et al. 1993	Spring Cr. at La Boca	brown trout	Lotic	2.78	
Butler et al. 1993	Spring Cr. at La Boca	fathead minnow	Lotic	2.78	
Butler et al. 1993	Spring Cr. at La Boca	speckled dace	Lotic	1.36	
Butler et al. 1993	Spring Cr. at La Boca	white sucker	Lotic	1.58	
Butler et al. 1995	Hartman Draw near mouth, at Cortez	fathead minnow	Lotic	2.78	1.85
Butler et al. 1995	Hartman Draw near mouth, at Cortez	flannelmouth sucker	Lotic	1.52	
Butler et al. 1995	Hartman Draw near mouth, at Cortez	sucker	Lotic	1.25	
Butler et al. 1995	McElmo Cr. at Hwy. 160, near Cortez	fathead minnow	Lotic	2.78	2.07
Butler et al. 1995	McElmo Cr. at Hwy. 160, near Cortez	speckled dace	Lotic	1.36	
Butler et al. 1995	McElmo Cr. downstream from Alkali Cyn.	bluehead sucker	Lotic	1.24	1.73
Butler et al. 1995	McElmo Cr. downstream from Alkali Cyn.	fathead minnow	Lotic	2.78	
Butler et al. 1995	McElmo Cr. downstream from Alkali Cyn.	flannelmouth sucker	Lotic	1.52	

Reference	Site Description	Fish Species	Waterbody Type	Fish $TTF^{composite-a}$	Average Fish $TTF^{composite}$ at a Site
Butler et al. 1995	McElmo Cr. downstream from Alkali Cyn.	speckled dace	Lotic	1.36	
Butler et al. 1995	McElmo Cr. downstream from Yellow Jacket Cyn.	common carp	Lotic	1.58	2.04
Butler et al. 1995	McElmo Cr. downstream from Yellow Jacket Cyn.	fathead minnow	Lotic	2.78	
Butler et al. 1995	McElmo Cr. downstream from Yellow Jacket Cyn.	flannelmouth sucker	Lotic	1.52	
Butler et al. 1995	McElmo Cr. downstream from Yellow Jacket Cyn.	red shiner	Lotic	2.27	
Butler et al. 1995	McElmo Cr. upstream from Yellow Jacket Cyn.	bluehead sucker	Lotic	1.24	1.83
Butler et al. 1995	McElmo Cr. upstream from Yellow Jacket Cyn.	bullhead	Lotic	1.62	
Butler et al. 1995	McElmo Cr. upstream from Yellow Jacket Cyn.	common carp	Lotic	1.58	
Butler et al. 1995	McElmo Cr. upstream from Yellow Jacket Cyn.	fathead minnow	Lotic	2.78	
Butler et al. 1995	McElmo Cr. upstream from Yellow Jacket Cyn.	flannelmouth sucker	Lotic	1.52	
Butler et al. 1995	McElmo Cr. upstream from Yellow Jacket Cyn.	green sunfish	Lotic	2.29	
Butler et al. 1995	McElmo Cr. upstream from Yellow Jacket Cyn.	red shiner	Lotic	2.27	
Butler et al. 1995	McElmo Cr. upstream from Yellow Jacket Cyn.	speckled dace	Lotic	1.36	
Butler et al. 1995	Navajo Wash near Towaoc	bluehead sucker	Lotic	1.24	1.30
Butler et al. 1995	Navajo Wash near Towaoc	speckled dace	Lotic	1.36	
Butler et al. 1995	San Juan River at Four Comers	bluehead sucker	Lotic	1.24	1.55

Reference	Site Description	Fish Species	Waterbody Type	Fish $TTF^{composite-a}$	Average Fish $TTF^{composite}$ at a Site
Butler et al. 1995	San Juan River at Four Comers	channel catfish	Lotic	1.35	
Butler et al. 1995	San Juan River at Four Comers	common carp	Lotic	1.58	
Butler et al. 1995	San Juan River at Four Comers	flannelmouth sucker	Lotic	1.52	
Butler et al. 1995	San Juan River at Four Comers	red shiner	Lotic	2.27	
Butler et al. 1995	San Juan River at Four Comers	speckled dace	Lotic	1.36	
Butler et al. 1995	San Juan River at Mexican Hat Utah	bluehead sucker	Lotic	1.24	1.42
Butler et al. 1995	San Juan River at Mexican Hat Utah	channel catfish	Lotic	1.35	
Butler et al. 1995	San Juan River at Mexican Hat Utah	common carp	Lotic	1.58	
Butler et al. 1995	San Juan River at Mexican Hat Utah	flannelmouth sucker	Lotic	1.52	
Butler et al. 1995	Woods Cyn. Near Yellow Jacket	fathead minnow	Lotic	2.78	2.78
Butler et al. 1997	Cahone Canyon at Highway 666	green sunfish	Lotic	2.29	2.29
Butler et al. 1997	Mud Creek at Highway 32, near Cortez	bluehead sucker	Lotic	1.24	2.11
Butler et al. 1997	Mud Creek at Highway 32, near Cortez	fathead minnow	Lotic	2.78	
Butler et al. 1997	Mud Creek at Highway 32, near Cortez	green sunfish	Lotic	2.29	
Casey 2005	Deerlick Creek	rainbow trout	Lotic	2.33	2.33
Casey 2005	Luscar Creek	rainbow trout	Lotic	2.33	2.33

Reference	Site Description	Fish Species	Waterbody Type	Fish $TTF^{composite-a}$	Average Fish $TTF^{composite}$ at a Site
Formation 2012	Crow Creek - 1A	brown trout	Lotic	2.96	2.87
Formation 2012	Crow Creek - 1A	sculpin	Lotic	2.78	
Formation 2012	Crow Creek - 3A	brown trout	Lotic	2.97	2.88
Formation 2012	Crow Creek - 3A	sculpin	Lotic	2.78	
Formation 2012	Crow Creek - CC150	brown trout	Lotic	2.91	2.82
Formation 2012	Crow Creek - CC150	sculpin	Lotic	2.74	
Formation 2012	Crow Creek - CC350	brown trout	Lotic	2.97	2.88
Formation 2012	Crow Creek - CC350	sculpin	Lotic	2.79	
Formation 2012	Crow Creek - CC75	brown trout	Lotic	2.87	2.78
Formation 2012	Crow Creek - CC75	sculpin	Lotic	2.69	
Formation 2012	Deer Creek	brown trout	Lotic	3.00	2.90
Formation 2012	Deer Creek	sculpin	Lotic	2.81	
Formation 2012	Hoopes Spring – HS	brown trout	Lotic	3.86	3.74
Formation 2012	Hoopes Spring – HS	sculpin	Lotic	3.63	
Formation 2012	Hoopes Spring - HS3	brown trout	Lotic	2.63	2.55
Formation 2012	Hoopes Spring - HS3	sculpin	Lotic	2.47	
Formation 2012	Sage Creek - LSV2C	brown trout	Lotic	3.01	2.92
Formation 2012	Sage Creek - LSV2C	sculpin	Lotic	2.83	
Formation 2012	Sage Creek - LSV4	brown trout	Lotic	2.88	2.79
Formation 2012	Sage Creek - LSV4	sculpin	Lotic	2.70	
Formation 2012	South Fork Tincup Cr.	brown trout	Lotic	3.05	2.96
Formation 2012	South Fork Tincup Cr.	sculpin	Lotic	2.86	
Hamilton and Buhl 2004	lower East Mill Creek	cutthroat trout	Lotic	2.29	2.29
McDonald and Strosher 1998	Elk R. above Cadorna Cr. (745)	cutthroat trout	Lotic	2.29	2.63

Reference	Site Description	Fish Species	Waterbody Type	Fish $TTF^{composite-a}$	Average Fish $TTF^{composite}$ at a Site
McDonald and Strosher 1998	Elk R. above Cadorna Cr. (745)	mountain whitefish	Lotic	2.97	
McDonald and Strosher 1998	Fording R. above Swift Cr. (746)	cutthroat trout	Lotic	2.29	2.29
Orr et al. 2012	Elk River 1	cutthroat trout	Lotic	2.29	2.29
Orr et al. 2012	Elk River 12	cutthroat trout	Lotic	2.29	2.29
Orr et al. 2012	Fording River 23	cutthroat trout	Lotic	2.29	2.29
Orr et al. 2012	Michel Creek 2	cutthroat trout	Lotic	2.29	2.29
Saiki and Lowe 1987	San Luis Drain	western mosquitofish	Lotic	2.37	2.29
Saiki and Lowe 1987	Volta Wasteway	western mosquitofish	Lotic	2.37	2.29
Saiki et al. 1993	Mud Slough at Gun Club Road	bluegill	Lotic	1.47	1.87
Saiki et al. 1993	Mud Slough at Gun Club Road	largemouth bass	Lotic	2.04	
Saiki et al. 1993	Mud Slough at Gun Club Road	western mosquitofish	Lotic	2.10	
Saiki et al. 1993	Salt Slough at the San Luis National Wildlife Refuge	bluegill	Lotic	1.47	1.87
Saiki et al. 1993	Salt Slough at the San Luis National Wildlife Refuge	largemouth bass	Lotic	2.04	
Saiki et al. 1993	Salt Slough at the San Luis National Wildlife Refuge	western mosquitofish	Lotic	2.10	
Saiki et al. 1993	San Joaquin R. above Hills Ferry Road	bluegill	Lotic	1.47	1.87
Saiki et al. 1993	San Joaquin R. above Hills Ferry Road	largemouth bass	Lotic	2.04	
Saiki et al. 1993	San Joaquin R. above Hills Ferry Road	western mosquitofish	Lotic	2.10	

Reference	Site Description	Fish Species	Waterbody Type	Fish $TTF^{composite-a}$	Average Fish $TTF^{composite}$ at a Site
Saiki et al. 1993	San Joaquin R. at Durham Ferry State Recreation Area	bluegill	Lotic	1.47	1.87
Saiki et al. 1993	San Joaquin R. at Durham Ferry State Recreation Area	largemouth bass	Lotic	2.04	
Saiki et al. 1993	San Joaquin R. at Durham Ferry State Recreation Area	western mosquitofish	Lotic	2.10	
Average Fish $TTF^{composite}$					**2.34**

[a] Sum of all TTFs relating whole body Se concentrations in fish to the base of the food web. From Appendix B of (U.S. EPA 2016a).

Appendix C TOTAL SELENIUM AND DISSOLVED SELENIUM CONCENTRATIONS IN CALIFORNIA WATER BODIES

Information Summary

Data Source: California Environmental Data Exchange Network (CEDEN) at www.ceden.org/.

Description: The total selenium and dissolved selenium concentrations in California water bodies were collected over a 10-year period from October 5, 2004 to June 3, 2014. The data were downloaded from the CEDEN database, which was last accessed on February 4, 2015. The assigned HUC12 was used as the identifier for the water body name where total selenium and/or dissolved selenium concentrations in water samples were reported. The data summary tables shown below were developed using the Microsoft Excel pivot table function. The level of confidence in the environmental data is high because the CEDEN database from which the data were derived for this analysis is the most comprehensive and largest source of selenium environmental monitoring data collected in California. The sample sites are not, however, randomly selected.

Appendix Table C-1. Total Selenium Concentrations in California Water Bodies.

California Regional Board Sampling Sites (HUC12)	Number of Samples	Minimum Se (µg/L)	Mean Se (µg/L)	Maximum Se (µg/L)
Central Coast Regional Board (6 HUC12 Sites)	8	1.6	**6.9**	**14.6**
Chorro Creek	2	1.6	2.6	**3.7**
Corralitos Canyon	1	**14.6**	**14.6**	**14.6**
Dos Pueblos Canyon-Frontal Santa Barbara Channel	1	**12.4**	**12.4**	**12.4**
Lower Arroyo Grande Creek	1	**10.4**	**10.4**	**10.4**
Lower San Luis Obispo Creek	1	**5.8**	**5.8**	**5.8**
Oso Flaco Creek	2	1.9	**3.3**	**4.6**
Central Valley Regional Board (114 HUC12 Sites)	10637	0.0	**12.8**	**1591.0**
Agua Fria Creek	14	0.6	1.5	**3.9**
Anderson Creek-Sacramento River	2	1.0	2.0	3.0
Ash Slough-Fresno River	1	**5.0**	**5.0**	**5.0**
Bear Creek	17	0.1	0.6	2.0
Bennett Valley-San Joaquin River	53	0.0	2.0	**9.2**

California Regional Board Sampling Sites (HUC12)	Number of Samples	Minimum Se (µg/L)	Mean Se (µg/L)	Maximum Se (µg/L)
Berenda Slough	4	0.1	0.2	0.5
Big Buttonwillow Lake-Salt Slough	18	0.3	0.6	**(1.7)**
Boggs Slough-Fresno Slough	1	0.1	0.1	0.1
Bolinas Bay	8	0.0	0.1	0.2
Boscha Lake (Historical)-Stanislaus River	4	0.2	0.9	**(2.3)**
Brooks Creek-Cache Creek	2	1.0	1.0	1.0
Brush Creek-South Fork American River	1	0.9	0.9	0.9
Caesar Ditch-Cross Creek	3	1.0	1.0	1.0
Chanac Creek	1	**10.0**	**10.0**	**10.0**
Deadmans Slough-Salt Slough	305	0.0	0.6	**4.1**
Deep Slough-Bear Creek	22	0.1	0.4	1.6
Drumheller Slough-Butte Creek	3	0.2	0.2	0.2
East Branch Cross Creek-Cross Creek	1	2.0	2.0	2.0
Escarpado Canyon-Panoche Creek	12	**6.2**	**10.0**	**18.0**
Fancher Creek Canal	6	0.1	0.2	0.4
Fresno Slough	16	0.2	3.0	**6.5**
Gilsizer Slough-Snake River	4	1.0	1.3	2.0
Hog Slough	17	0.1	0.1	0.3
Hospital Creek	25	0.2	0.7	1.6
Ingram Creek	30	0.3	1.2	2.9
Jones Drain-Merced River	23	0.0	0.6	**5.1**
Kern Canyon-San Joaquin River	28	0.1	0.9	2.6
Laguna Seca Creek	353	0.02	**40.2**	**167**
Lake Ramona-San Joaquin River	332	0.01	1.3	**(3.7)**
Lake Success-Tule River	1	1.0	1.0	1.0
Little Creek	4	1.0	1.3	2.0
Lone Willow Slough-San Joaquin River	15	0.1	1.0	**5.3**
Los Banos Creek	31	0.2	1.9	**5.1**
Los Sauces Creek-Frontal Pacific Ocean	1	**7.5**	**7.5**	**7.5**
Lower Bear Creek	21	0.1	0.2	0.9
Lower Cantua Creek	18	0.4	3.0	**7.6**
Lower Cottonwood Creek	14	0.1	0.3	0.8
Lower Del Puerto Creek	24	0.3	1.1	**3.4**
Lower Dry Creek	7	0.1	0.4	0.6
Lower Duck Creek	13	0.1	0.1	0.6
Lower Elk Bayou	1	1.0	1.0	1.0
Lower Freshwater Creek	2	1.0	1.0	1.0
Lower Kellogg Creek	7	0.3	1.1	3.0

California Regional Board Sampling Sites (HUC12)	Number of Samples	Minimum Se (µg/L)	Mean Se (µg/L)	Maximum Se (µg/L)
Lower Laguna	1	**(3.0)**	**(3.0)**	**(3.0)**
Lower Little Panoche Creek	1	**15.0**	**15.0**	**15.0**
Lower Logan Creek	2	1.0	1.5	2.0
Lower Lone Tree Creek	4	0.2	0.4	0.8
Lower Los Gatos Creek	19	0.0	2.4	**6.5**
Lower Mariposa Slough-Deadman Creek	37	0.1	0.6	3.0
Lower Marsh Creek	33	0.6	2.2	**6.0**
Lower Owens Creek	8	0.3	0.5	0.8
Lower Poso Slough-Salt Slough	30	0.1	0.7	1.8
Lower Ulatis Creek	11	0.3	1.3	3.0
Lower Walker Creek	1	0.1	0.1	0.1
Lower West Side Canal	63	0.2	1.2	**6.7**
Lower White Lake-San Joaquin River	249	0.0	0.9	**(4.9)**
Mariposa Creek-Duck Slough	32	0.1	0.4	1.0
Markley Canyon-San Joaquin River	7	0.1	0.1	0.1
McGrath Lake-Frontal Pacific Ocean	2	0.5	0.6	0.6
McLeod Lake-Mormon Slough	6	0.1	0.4	0.8
Middle Elk Bayou	3	1.0	1.0	1.0
Middle Lone Tree Creek	5	0.2	1.0	3.0
Middle River-San Joaquin River	16	0.1	0.2	0.5
Modesto Reservoir-Dry Creek	13	0.1	0.3	1.0
Moreno Gulch	961	0.0	**14.5**	**120**
Mosquito Creek-Cross Creek	4	0.2	0.7	1.0
Mud 1085 Dam-Fresno Slough	2	**4.2**	**4.8**	**5.5**
Mud Slough	3685	0.01	**27.7**	**1591**
Murphy Creek-Mokelumne River	4	0.1	0.7	2.0
Mustang Creek-Los Banos Creek	40	0.2	0.6	2.1
North Branch Tule River-Tule River	6	1.0	1.2	2.0
Old Channel Tule River	5	1.0	1.2	2.0
Oso Creek-Orestimba Creek	70	0.0	2.0	**9.1**
Packer Lake-Sacramento River	1	0.5	0.5	0.5
Pear Slough-San Joaquin River	2294	0.0	1.4	**4.7**
Ping Slough-Coon Creek	1	1.0	1.0	1.0
Pixley Slough	2	0.3	0.4	0.6
Porter Slough	4	1.0	1.3	2.0
Red Bridge Slough-San Joaquin River	215	0.0	0.7	**3.8**
Riley Slough	21	0.1	0.1	0.3
Roberts Island-Trapper Slough	24	0.1	0.5	1.5

California Regional Board Sampling Sites (HUC12)	Number of Samples	Minimum Se (µg/L)	Mean Se (µg/L)	Maximum Se (µg/L)
Rock Creek-Pit River	2	0.1	0.1	0.1
Rodden Creek-Stanislaus River	2	0.1	0.1	0.1
Saint Johns River	4	0.7	1.2	2.0
Salt Creek	6	0.3	0.5	0.8
Shag Slough-San Joaquin River	322	0.0	0.5	**4.1**
Simmons Creek-Littlejohns Creek	7	0.2	0.3	0.4
South Branch Island Canal-Kings River	6	0.1	0.3	0.7
South Slough-Deadman Creek	21	0.1	0.3	1.2
Stockton Diverting Canal-Calaveras River	1	0.8	0.8	0.8
Stone Corral Canyon-Cottonwood Creek	5	0.3	1.1	2.0
Stone Corral Creek	2	0.5	0.6	0.7
Sycamore Slough	35	0.1	0.3	1.2
Telephone Cut-Bishop Cut	3	0.1	0.1	0.1
Threemile Slough-Sacramento River	5	0.1	0.1	0.2
Toe Drain-Cache Slough	6	0.2	1.3	**6.0**
Town of Famoso-Poso Creek	2	0.8	0.9	1.0
Town of Hilmar-San Joaquin River	21	0.1	0.9	2.0
Town of Lemoore-Kings River	1	0.8	0.8	0.8
Town of Riverdale Park-Tuolumne River	5	0.0	0.3	0.9
Town of Terra Bella-Deer Creek	3	1.0	1.3	2.0
Tule Canal-Toe Drain	10	1.0	**3.5**	**7.8**
Turlock Lake	7	0.3	0.5	1.0
Union Island	22	0.3	1.4	3.0
Upper Lone Tree Creek	4	0.1	0.2	0.3
Upper Marsh Creek	3	1.0	1.0	1.0
Upper Poso Slough	15	0.6	**8.4**	**21.0**
Upper Ruth Lake-Mud Slough	353	0.0	1.2	**(5.0)**
Upper West Side Canal	5	0.3	1.0	1.7
Venice Island-Little Connection Slough	6	0.1	0.1	0.2
Walker Slough-French Camp Slough	13	0.1	0.3	1.0
Walthall Slough-San Joaquin River	39	0.1	0.2	0.9
Wildcat Canyon	346	0.0	1.0	**5.7**
Wilson Creek-North Honcut Creek	3	0.1	0.3	0.6
Lahontan Regional Board (2 HUC12 Sites)	18	0.2	0.6	1.9
Mammoth Creek	16	0.2	0.4	0.7
Tecopa Wash-Amargosa River	2	1.4	1.7	1.9

California Regional Board Sampling Sites (HUC12)	Number of Samples	Minimum Se (µg/L)	Mean Se (µg/L)	Maximum Se (µg/L)
Los Angeles Regional Board (45 HUC12 Sites)	116	0.4	**16.4**	**335.0**
Abadi Creek-Sespe Creek	2	1.0	1.9	2.8
Alhambra Wash-Rio Hondo	1	**5.1**	**5.1**	**5.1**
Arroyo Sequit-Frontal Pacific Ocean	5	1.2	2.2	**3.4**
Big Sycamore Canyon	2	2.1	2.6	3.1
Boulder Creek-Sespe Creek	3	1.0	**9.3**	**25.6**
Cedar Creek-Piru Creek	1	1.0	1.0	1.0
Cold Creek-Malibu Creek	11	1.2	**3.7**	**6.8**
Coyote Creek	1	0.9	0.9	0.9
Coyote Creek-San Gabriel River	1	1.3	1.3	1.3
Elizabeth Lake Canyon	2	0.6	1.3	**(2.1)**
Fish Creek-Piru Creek	2	0.9	1.1	1.3
Garapito Creek	2	0.7	1.7	2.7
Harmon Canyon-Santa Clara River	1	**9.2**	**9.2**	**9.2**
Hopper Canyon	1	1.7	1.7	1.7
Hosler Canyon-Piru Creek	3	2.0	**3.4**	**4.7**
Iron Fork-San Gabriel River	1	1.1	1.1	1.1
Las Posas Arroyo	1	**7.6**	**7.6**	**7.6**
Las Virgenes Creek	14	**7.7**	**70.8**	**335**
Lockwood Creek	1	1.5	1.5	1.5
Los Sauces Creek-Frontal Pacific Ocean	7	**7.8**	**24.5**	**42.6**
Lower Conejo Arroyo	2	**4.5**	**4.7**	**5.0**
Lower Ventura River	3	0.8	1.9	3.0
Lower West Fork San Gabriel River	1	0.5	0.5	0.5
Matilija Creek	1	1.8	1.8	1.8
McGrath Lake-Frontal Pacific Ocean	3	0.8	**(2.1)**	**(4.6)**
Medea Creek	12	**3.8**	**10.9**	**36.5**
Mugu Lagoon	1	**55.2**	**55.2**	**55.2**
North Fork San Gabriel River	1	0.8	0.8	0.8
Pole Creek-Santa Clara River	1	**4.0**	**4.0**	**4.0**
Salt Canyon-Santa Clara River	7	1.5	**4.3**	**6.6**
San Antonio Creek	1	**3.4**	**3.4**	**3.4**
San Francisquito Canyon	2	0.6	0.7	0.7
Santa Fe Flood Control Basin-San Gabriel River	2	0.4	**4.4**	**8.4**
Santa Monica Beach-Frontal Santa Monica Bay	1	1.6	1.6	1.6
Santa Paula Creek	1	**298**	**298**	**298**
Snowy Creek-Piru Creek	1	0.7	0.7	0.7
Solstice Canyon-Frontal Santa Monica Bay	2	**3.4**	**4.7**	**6.0**

California Regional Board Sampling Sites (HUC12)	Number of Samples	Minimum Se (µg/L)	Mean Se (µg/L)	Maximum Se (µg/L)
South Fork Santa Clara River	1	**6.4**	**6.4**	**6.4**
Timber Canyon-Santa Clara River	1	2.5	2.5	2.5
Tule Creek-Sespe Creek	5	0.6	1.1	2.5
Upper Bouquet Canyon	1	0.9	0.9	0.9
Upper Conejo Arroyo	1	**6.4**	**6.4**	**6.4**
Upper Simi Arroyo	1	**8.5**	**8.5**	**8.5**
Upper Ventura River	1	1.3	1.3	1.3
Zuma Canyon-Frontal Pacific Ocean	1	2.1	2.1	2.1
North Coast Regional Board (56 HUC12 Sites)	352	0.1	1.0	**126.0**
Alder Creek-Big Sulphur Creek	5	0.1	0.3	0.4
Bear Creek-Eel River	5	0.1	0.2	0.3
Bittenbender Creek-Klamath River	3	0.1	0.2	0.4
Brooks Creek-Russian River	8	0.1	0.3	0.6
Brush Creek-Klamath River	2	0.2	0.4	0.6
Bunton Hollow Creek-Shasta River	8	0.3	0.9	1.9
Burright Creek-East Fork Russian River	5	0.1	0.2	0.3
Butte Creek-South Fork Eel River	3	0.7	0.8	1.0
Cameron Creek-Eel River	11	0.1	0.4	1.0
Canoe Creek-South Fork Eel River	12	0.2	0.4	0.9
Cummings Creek-Van Duzen River	1	0.8	0.8	0.8
Deadwood Creek-Trinity River	3	0.1	0.3	0.7
Deerhorn Creek-Trinity River	8	0.1	0.4	1.0
Division Creek-Eel River	10	0.1	0.4	0.8
Dutch Bill Creek-Russian River	12	0.1	0.5	1.2
East Fork Russian River-Russian River	10	0.1	0.4	0.7
Elder Creek-South Fork Eel River	12	0.1	0.4	0.9
Elk River	2	0.7	0.8	0.9
Empire Creek-Klamath River	3	0.1	0.2	0.4
Estero Americano	1	1.0	1.0	1.0
Freshwater Creek	3	0.7	0.7	0.8
Gill Creek-Russian River	14	0.1	0.3	0.7
Goforth Creek-Middle Fork Eel River	13	0.3	0.6	1.1
Hardscrabble Creek-Smith River	8	0.2	0.4	0.9
Jacoby Creek	1	0.9	0.9	0.9
Kohl Creek-Klamath River	3	0.1	0.3	0.4
Lake Mendocino-East Fork Russian River	4	0.2	0.3	0.5
Little River	1	0.6	0.6	0.6

California Regional Board Sampling Sites (HUC12)	Number of Samples	Minimum Se (µg/L)	Mean Se (µg/L)	Maximum Se (µg/L)
Little Salmon Creek-Salmon Creek	9	0.6	1.2	2.4
Lower Garcia River	3	0.7	1.0	1.6
Lower Indian Creek	2	1.6	**63.8**	**126.0**
Lower Mattole River	2	0.6	0.6	0.7
Lower North Fork Eel River	6	0.2	0.4	0.9
Lower Santa Rosa Creek	19	0.4	2.3	**9.7**
Lower South Fork Smith River	6	0.1	0.2	0.3
McArthur Creek-Redwood Creek	12	0.2	0.4	0.7
Middle Garcia River	1	0.8	0.8	0.8
Mill Creek-Mad River	10	0.1	0.3	0.7
Mingo Creek-South Fork Trinity River	7	0.2	0.4	1.0
Morrison Creek-Russian River	2	0.2	0.2	0.2
North Fork Mattole River	2	0.6	0.7	0.7
Orrs Creek-Russian River	10	0.2	0.4	1.1
Porter Creek-Mark West Creek	21	0.1	1.0	2.6
Russian Gulch-Frontal Pacific Ocean	1	0.8	0.8	0.8
Salmon Creek	4	0.7	0.8	0.9
Sharber Creek-Trinity River	1	0.7	0.7	0.7
Slate Creek-Klamath River	5	0.1	0.3	0.5
Smith River	7	0.1	0.2	0.5
South Fork Gualala River-Gualala River	9	0.1	0.7	2.4
Thomas Creek-Eel River	6	0.1	0.5	1.0
Town of Scott Bar-Scott River	11	0.1	0.4	0.8
Upper Garcia River	1	0.7	0.7	0.7
Upper Indian Creek	1	**7.8**	**7.8**	**7.8**
Ward Creek-Austin Creek	5	0.2	0.5	0.8
West Slough-Dry Creek	11	0.1	0.3	0.6
Yreka Creek	7	0.1	0.5	1.0
San Diego Regional Board (30 HUC12 Sites)	53	0.6	**4.5**	**24.1**
Aliso Creek	3	**10.9**	**13.5**	**18.2**
Arroyo Trabuco	1	**11.6**	**11.6**	**11.6**
Boden Canyon-Santa Ysabel Creek	1	1.8	1.8	1.8
Boulder Creek	1	0.6	0.6	0.6
Buena Vista Creek	1	**6.4**	**6.4**	**6.4**
Cedar Creek	1	1.4	1.4	1.4
Conejos Creek	2	1.3	1.8	2.4
Dan Price Creek-Santa Ysabel Creek	1	1.2	1.2	1.2

California Regional Board Sampling Sites (HUC12)	Number of Samples	Minimum Se (µg/L)	Mean Se (µg/L)	Maximum Se (µg/L)
El Capitan Reservoir-San Diego River	2	0.7	0.8	0.9
Forester Creek	1	**7.7**	**7.7**	**7.7**
Los Penasquitos Creek	1	**7.7**	**7.7**	**7.7**
Lower Escondido Creek	3	**4.7**	**5.9**	**8.2**
Lower Otay Reservoir	2	**(3.2)**	**(3.4)**	**(3.6)**
Lower Pine Valley Creek	1	1.0	1.0	1.0
Lower San Juan Creek	4	2.4	**9.4**	**15.0**
McAlmond Canyon-Cottonwood Creek	1	1.9	1.9	1.9
Middle Pine Valley Creek	3	1.9	2.5	3.0
Middle San Mateo Creek	4	0.8	1.3	1.9
Morena Reservoir-Cottonwood Creek	1	**(6.3)**	**(6.3)**	**(6.3)**
Paradise Creek-San Luis Rey River	1	1.8	1.8	1.8
Prima Deshecha Canada-Frontal Capistrano Bight	1	**24.1**	**24.1**	**24.1**
Rainbow Creek-Santa Margarita River	4	1.7	2.4	**3.4**
Ritchie Creek-San Diego River	2	1.5	1.5	1.6
Salt Creek-Frontal Gulf of Santa Catalina	1	**7.8**	**7.8**	**7.8**
San Marcos Creek	1	**4.3**	**4.3**	**4.3**
San Pasqual Valley-Santa Ysabel Creek	1	**5.2**	**5.2**	**5.2**
Sandia Canyon	1	**4.7**	**4.7**	**4.7**
Upper Pine Valley Creek	2	1.0	2.4	**3.9**
Upper San Juan Creek	3	0.9	1.4	2.1
Upper San Mateo Creek	2	1.0	1.1	1.3
San Francisco Bay Regional Board (7 HUC12 Sites)	99	0.03	1.3	**4.8**
Calabazas Creek-Frontal San Francisco Bay Estuaries	2	0.3	0.4	0.5
Denniston Creek-Frontal Pacific Ocean	47	0.6	1.7	**4.8**
Dry Creek-Arroyo Valle	6	0.2	0.6	2.0
Guadalupe River	3	1.2	1.3	1.6
San Leandro Creek	4	0.2	0.2	0.3
Walnut Creek-Frontal Suisun Bay Estuaries	6	0.5	2.7	**4.4**
Ward Creek-Frontal San Francisco Bay Estuaries	31	0.0	0.6	2.9
Santa Ana Regional Board (10 HUC12 Sites)	12	0.1	1.2	**5.1**
East Twin Creek	1	0.4	0.4	0.4
Fish Creek-Santa Ana River	1	0.3	0.3	0.3
Moreno Valley	1	1.4	1.4	1.4
North Fork San Jacinto River	1	0.6	0.6	0.6
San Antonio Canyon	2	0.5	0.6	0.6

California Regional Board Sampling Sites (HUC12)	Number of Samples	Minimum Se (µg/L)	Mean Se (µg/L)	Maximum Se (µg/L)
San Timoteo Canyon-San Timoteo Wash	1	1.0	1.0	1.0
Santa Anna Wash-Santa Anna River	1	0.3	0.3	0.3
Strawberry Creek-San Jacinto River	2	0.1	0.2	0.3
Upper Chino Creek	1	**5.1**	**5.1**	**5.1**
Upper San Diego Creek	1	**3.9**	**3.9**	**3.9**
Grand Total	**11290**			

Data Source: California Environmental Data Exchange Network (CEDEN) at www.ceden.org/.

Data includes reported Total Se in water samples collected from October 5, 2004 to June 3, 2014.

CEDEN was last accessed on February 4, 2015.

San Francisco Bay Regional Board summary excludes San Francisco Bay selenium data.

Bolded sampling sites indicate a lentic system (lakes and reservoirs).
Bolded numbers in parenthesis indicate that total selenium exceeded 1.5 µg Se/L in lentic systems.

Bolded numbers indicate that total selenium exceeded 3.1 µg Se/L in lotic systems.

HUC12 is Hydrologic Unit Code 12. The HUC12 designation is the name of sampling site.

Appendix Table C-2. Dissolved Selenium Concentrations in California Water Bodies.

California Regional Board Sampling Sites (HUC12)	Number of Samples	Minimum Se (µg/L)	Mean Se (µg/L)	Maximum Se (µg/L)
Central Valley Regional Board (38 HUC12 Sites)	178	0.01	2.7	**106**
Black Butte Dam-Stony Creek	1	1.2	1.2	1.2
Bolinas Bay	8	0.02	0.4	2.7
Boscha Lake (Historical)-Stanislaus River	1	0.1	0.1	0.1
Compton Creek-Los Angeles River	1	1.6	1.6	1.6
Deadmans Slough-Salt Slough	4	0.2	0.7	1.2
Drumheller Slough-Butte Creek	2	0.2	1.5	2.8
Hoag Slough-Sacramento River	2	0.02	0.3	0.5
Hog Slough	2	0.1	0.4	0.7
Ingalsbe Slough-Merced River	2	1.5	2.1	2.7
Jack Slough	2	0.2	0.6	1.1
Jones Drain-Merced River	1	0.1	0.1	0.1
Laguna Seca Creek	4	0.2	1.8	**4.6**
Lake Ramona-San Joaquin River	4	0.1	0.6	1.4
Lower Antelope Creek	2	0.6	0.7	0.7
Lower Poso Slough-Salt Slough	2	0.3	1.1	1.9
Lower White Lake-San Joaquin River	4	0.2	1.0	**(2.0)**

California Regional Board Sampling Sites (HUC12)	Number of Samples	Minimum Se (µg/L)	Mean Se (µg/L)	Maximum Se (µg/L)
McLeod Lake-Mormon Slough	2	0.1	0.6	1.1
Middle Dry Creek	2	1.3	1.8	2.3
Middle Walker Creek	1	0.8	0.8	0.8
Modesto Reservoir-Dry Creek	2	1.2	1.3	1.4
Moreno Gulch	11	0.1	1.3	2.2
Mud Slough	43	0.03	**5.1**	**40.9**
Oso Creek-Orestimba Creek	4	0.1	0.5	0.9
Pear Slough-San Joaquin River	25	0.01	1.0	2.4
Pixley Slough	2	0.1	0.3	0.5
Red Bridge Slough-San Joaquin River	4	0.2	0.5	0.7
Red Spring-Colorado River	12	0.7	**11.7**	**106**
Shag Slough-San Joaquin River	7	0.03	0.6	2.0
South Fork Ditch-Willow Slough	1	0.8	0.8	0.8
Sycamore Slough	1	0.8	0.8	0.8
Town of French Camp-San Joaquin River	1	1.6	1.6	1.6
Town of Hilmar-San Joaquin River	4	0.2	1.2	2.2
Town of Riverdale Park-Tuolumne River	1	0.1	0.1	0.1
Union Island	3	0.7	1.4	1.8
Upper Ruth Lake-Mud Slough	3	0.2	0.5	1.0
Upper Steelhead Creek	1	0.2	0.2	0.2
Wildcat Canyon	4	0.1	1.3	2.6
Yankee Slough	2	0.1	0.2	0.2
Colorado River Regional Board (16 HUC12 Sites)	201	0.03	**6.1**	**46**
Ash Main Canal-Alamo River	23	0.7	**7.7**	**23.7**
Cinnabar Wash-Palo Verde Valley	27	0.9	**3.7**	**10.3**
City of Indio-Whitewater River	7	1.4	2.3	**4.1**
Colorado River-Imperial Reservoir	13	**(1.6)**	**(3.2)**	**(6.4)**
Frontal Salton Sea	2	**5.5**	**5.6**	**5.7**
Gieselmann Lake-Alamo River	6	**3.7**	**6.4**	**9.7**
Guadalupe Creek-Whitewater River	12	0.03	**3.7**	**7.9**
Lower New River	17	**4.2**	**12.5**	**46**
Middle New River	2	**5.4**	**5.5**	**5.6**
Ramer Lake-Alamo River	2	**6.4**	**7.8**	**9.1**
Salton Sea	38	0.7	1.4	**4.3**
Town of Calipatria-Alamo River	23	0.6	**9.4**	**27.1**
Town of El Centro	2	**4.2**	**5.1**	**6.0**
Town of Fuller-Alamo River	4	2.7	**9.4**	**21.0**

California Regional Board Sampling Sites (HUC12)	Number of Samples	Minimum Se (µg/L)	Mean Se (µg/L)	Maximum Se (µg/L)
Town of Niland-Frontal Salton Sea	4	2.0	**5.3**	**11.7**
Upper New River	19	0.1	**12.1**	**38.5**
Lahontan Regional Board (2 HUC12 Sites)	18	0.3	0.5	1.4
Mammoth Creek	16	0.3	0.4	0.4
Tecopa Wash-Amargosa River	2	1.0	1.2	1.4
Los Angeles Regional Board (48 HUC12 Sites)	109	0.3	**9.1**	**129**
Abadi Creek-Sespe Creek	3	0.8	1.5	2.3
Alamitos Bay	2	0.5	0.9	1.4
Alhambra Wash-Rio Hondo	2	**5.3**	**8.1**	**10.9**
Arroyo Seco	1	**4.8**	**4.8**	**4.8**
Arroyo Sequit-Frontal Pacific Ocean	3	1.0	1.7	2.7
Boulder Creek-Sespe Creek	3	1.3	**7.9**	**20.7**
Bull Creek	1	**14.6**	**14.6**	**14.6**
Cedar Creek-Piru Creek	1	0.9	0.9	0.9
Cold Creek-Malibu Creek	11	1.1	**4.0**	**7.2**
Compton Creek-Los Angeles River	1	**6.5**	**6.5**	**6.5**
Coyote Creek	1	0.9	0.9	0.9
Coyote Creek-San Gabriel River	1	0.8	0.8	0.8
Elizabeth Lake Canyon	1	**(1.6)**	**(1.6)**	**(1.6)**
Fish Creek-Piru Creek	2	0.9	0.9	0.9
Garapito Creek	2	0.9	1.9	2.9
Harmon Canyon-Santa Clara River	1	**9.9**	**9.9**	**9.9**
Hopper Canyon	1	2.2	2.2	2.2
Hosler Canyon-Piru Creek	3	1.6	**3.4**	**5.2**
Iron Fork-San Gabriel River	1	0.9	0.9	0.9
Las Posas Arroyo	1	**7.4**	**7.4**	**7.4**
Las Virgenes Creek	14	**9.0**	**37.5**	**129**
Lockwood Creek	1	0.9	0.9	0.9
Lower Conejo Arroyo	2	**4.6**	**4.9**	**5.3**
Lower Ventura River	2	2.0	2.3	2.6
Lower West Fork San Gabriel River	1	0.5	0.5	0.5
Matilija Creek	1	3.0	3.0	3.0
Medea Creek	12	**3.8**	**10.6**	**37.1**
Mugu Lagoon	1	**58.9**	**58.9**	**58.9**
North Fork San Gabriel River	1	0.6	0.6	0.6
Pole Creek-Santa Clara River	1	**4.0**	**4.0**	**4.0**

California Regional Board Sampling Sites (HUC12)	Number of Samples	Minimum Se (µg/L)	Mean Se (µg/L)	Maximum Se (µg/L)
Salt Canyon-Santa Clara River	7	1.5	**4.6**	**7.5**
San Antonio Creek	1	**3.6**	**3.6**	**3.6**
San Francisquito Canyon	2	0.8	0.9	0.9
Santa Fe Flood Control Basin-San Gabriel River	1	0.3	0.3	0.3
Santa Monica Beach-Frontal Santa Monica Bay	1	1.6	1.6	1.6
Santa Paula Creek	1	2.3	2.3	2.3
Snowy Creek-Piru Creek	1	0.9	0.9	0.9
Solstice Canyon-Frontal Santa Monica Bay	2	**4.3**	**4.9**	**5.4**
South Fork Santa Clara River	1	**5.6**	**5.6**	**5.6**
Timber Canyon-Santa Clara River	1	2.3	2.3	2.3
Tujunga Wash-Los Angeles River	2	0.8	**3.9**	**7.0**
Tule Creek-Sespe Creek	5	0.6	1.1	1.8
Upper Bouquet Canyon	1	1.0	1.0	1.0
Upper Conejo Arroyo	1	**5.8**	**5.8**	**5.8**
Upper Simi Arroyo	1	**8.0**	**8.0**	**8.0**
Upper Ventura River	1	1.4	1.4	1.4
Verdugo Wash	1	**5.3**	**5.3**	**5.3**
Zuma Canyon-Frontal Pacific Ocean	1	1.8	1.8	1.8
North Coast Regional Board (1 HUC12 Site)	8	0.8	3.1	**9.3**
Lower Santa Rosa Creek	8	0.8	3.1	**9.3**
San Diego Regional Board (48 HUC12 Sites)	123	0.2	**10.4**	**250**
Aliso Creek	2	**10.4**	**13.9**	**17.3**
Arroyo Trabuco	1	**11.3**	**11.3**	**11.3**
Bee Canyon-Cottonwood Creek	2	**6.3**	**6.9**	**7.5**
Boden Canyon-Santa Ysabel Creek	1	1.7	1.7	1.7
Boulder Creek	3	0.8	1.2	1.8
Buena Vista Creek	1	**5.8**	**5.8**	**5.8**
Cedar Creek	1	1.5	1.5	1.5
Conejos Creek	2	1.2	1.2	1.3
Dan Price Creek-Santa Ysabel Creek	1	0.9	0.9	0.9
El Capitan Reservoir-San Diego River	4	0.7	**(4.2)**	**(12.4)**
Forester Creek	7	**5.0**	**8.4**	**21.3**
Guajome Lake-San Luis Rey River	3	**(2.4)**	**(8.0)**	**(16.0)**
Hellers Bend-San Luis Rey River	3	**3.2**	**6.4**	**11.5**
Keys Creek	3	**3.5**	**9.0**	**18.5**
La Posta Creek	4	0.3	2.2	**3.6**

California Regional Board Sampling Sites (HUC12)	Number of Samples	Minimum Se (µg/L)	Mean Se (µg/L)	Maximum Se (µg/L)
Los Coches Creek-San Diego River	3	2.4	**9.4**	**17.4**
Los Penasquitos Creek	1	**7.4**	**7.4**	**7.4**
Loveland Reservoir-Sweetwater River	2	**(2.3)**	**(5.0)**	**(7.7)**
Lower Escondido Creek	3	**4.5**	**5.4**	**7.4**
Lower Otay Reservoir	2	**(3.2)**	**(3.3)**	**(3.4)**
Lower Pine Valley Creek	1	1.0	1.0	1.0
Lower San Juan Creek	5	1.7	**7.2**	**14.9**
Lower Tecate Creek	4	**5.1**	**10.5**	**14.5**
McAlmond Canyon-Cottonwood Creek	1	1.7	1.7	1.7
Middle Pine Valley Creek	3	1.7	2.2	2.6
Middle San Mateo Creek	4	0.8	1.3	2.4
Mission Valley-San Diego River	3	2.6	**12.0**	**18.5**
Moosa Canyon	3	1.8	**5.8**	**12.3**
Morena Reservoir-Cottonwood Creek	1	**(5.4)**	**(5.4)**	**(5.4)**
Murray Reservoir	3	**(6.1)**	**(14.4)**	**(26.8)**
O'Neill Lake-Santa Margarita River	1	**(2.0)**	**(2.0)**	**(2.0)**
Paradise Creek-San Luis Rey River	4	0.8	2.4	**4.4**
Prima Deshecha Canada-Frontal Capistrano Bight	1	**25.2**	**25.2**	**25.2**
Rainbow Creek-Santa Margarita River	4	1.8	2.6	**3.9**
Rice Canyon-Sweetwater River	4	**12.5**	**34.9**	**43.6**
Ritchie Creek-San Diego River	2	0.9	1.3	1.7
Salt Creek-Frontal Gulf of Santa Catalina	1	**8.4**	**8.4**	**8.4**
San Diego Bay	8	**7.7**	**63.4**	**250**
San Marcos Creek	1	**3.8**	**3.8**	**3.8**
San Pasqual Valley-Santa Ysabel Creek	1	**5.3**	**5.3**	**5.3**
Sandia Canyon	1	**5.1**	**5.1**	**5.1**
Tijuana River-Frontal Pacific Ocean	2	**9.9**	**11.0**	**12.1**
Upper Pine Valley Creek	2	1.3	1.7	2.2
Upper San Juan Creek	3	0.8	1.6	2.4
Upper San Mateo Creek	2	1.2	1.3	1.4
Upper San Vicente Creek	3	2.2	**4.9**	**9.9**
Viejas Creek-Sweetwater River	3	2.6	**8.4**	**19.7**
West Fork San Luis Rey River	3	0.2	1.0	1.6
San Francisco Bay Regional Board (10 HUC12 Sites)	57	0.03	1.2	**5.1**
Bolinas Lagoon	6	0.5	1.3	2.3
Calabazas Creek-Frontal San Francisco Bay Estuaries	2	0.3	0.3	0.3
Cerrito Creek-Frontal San Francisco Bay Estuaries	12	0.9	1.7	2.6

California Regional Board Sampling Sites (HUC12)	Number of Samples	Minimum Se (µg/L)	Mean Se (µg/L)	Maximum Se (µg/L)
Guadalupe River	3	0.8	1.0	1.3
Lobos Creek-Frontal San Francisco Bay Estuaries	3	0.6	1.5	2.7
Lower Arroyo Mocho	3	0.7	1.4	2.1
San Leandro Creek	4	0.1	0.1	0.2
Sausal Creek-Frontal San Francisco Bay Estuaries	12	1.0	2.2	**5.1**
Walnut Creek-Frontal Suisun Bay Estuaries	6	0.1	0.3	0.5
Ward Creek-Frontal San Francisco Bay Estuaries	6	0.03	0.1	0.1
Santa Ana Regional Board (8 HUC12 Sites)	16	0.2	**13.1**	**44.7**
Lower San Diego Creek	8	0.8	**25.2**	**44.7**
Moreno Valley	1	0.5	0.5	0.5
North Fork San Jacinto River	1	0.2	0.2	0.2
San Antonio Canyon	2	0.2	0.3	0.4
San Timoteo Canyon-San Timoteo Wash	1	0.5	0.5	0.5
Strawberry Creek-San Jacinto River	1	0.2	0.2	0.2
Upper Chino Creek	1	3.0	3.0	3.0
Upper San Diego Creek	1	2.8	2.8	2.8
Grand Total (171 HUC Sites)	710			

Data Source: California Environmental Data Exchange Network (CEDEN) at www.ceden.org/.

Data includes reported Dissolved Se in water samples collected from October 5, 2004 to June 3, 2014.

CEDEN was last accessed on February 4, 2015.
San Francisco Bay Regional Board summary excludes San Francisco Bay selenium data, since this is a separate rulemaking effort.

Bolded sampling sites indicate a lentic system (lakes and reservoirs).

Bolded numbers in parenthesis indicate that dissolved selenium exceeded 1.5 µg Se/L in lentic systems.

Bolded numbers indicate that dissolved selenium exceeded 3.1 µg Se/L in lotic systems.

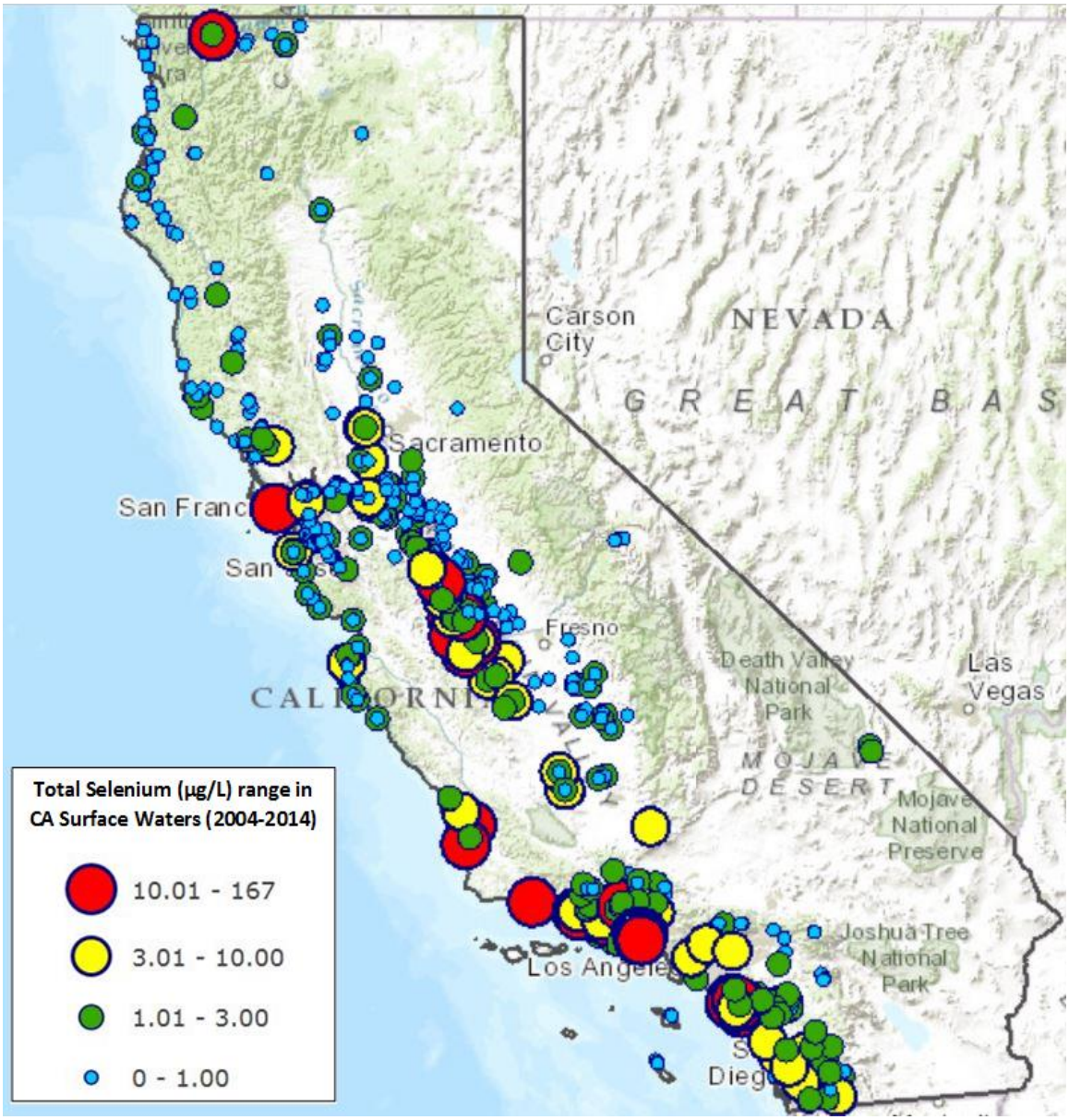
NA is not available.

HUC12 is Hydrologic Unit Code 12. The HUC12 designation is the name of sampling site.

Selenium Concentrations in California Water Bodies for Comparison

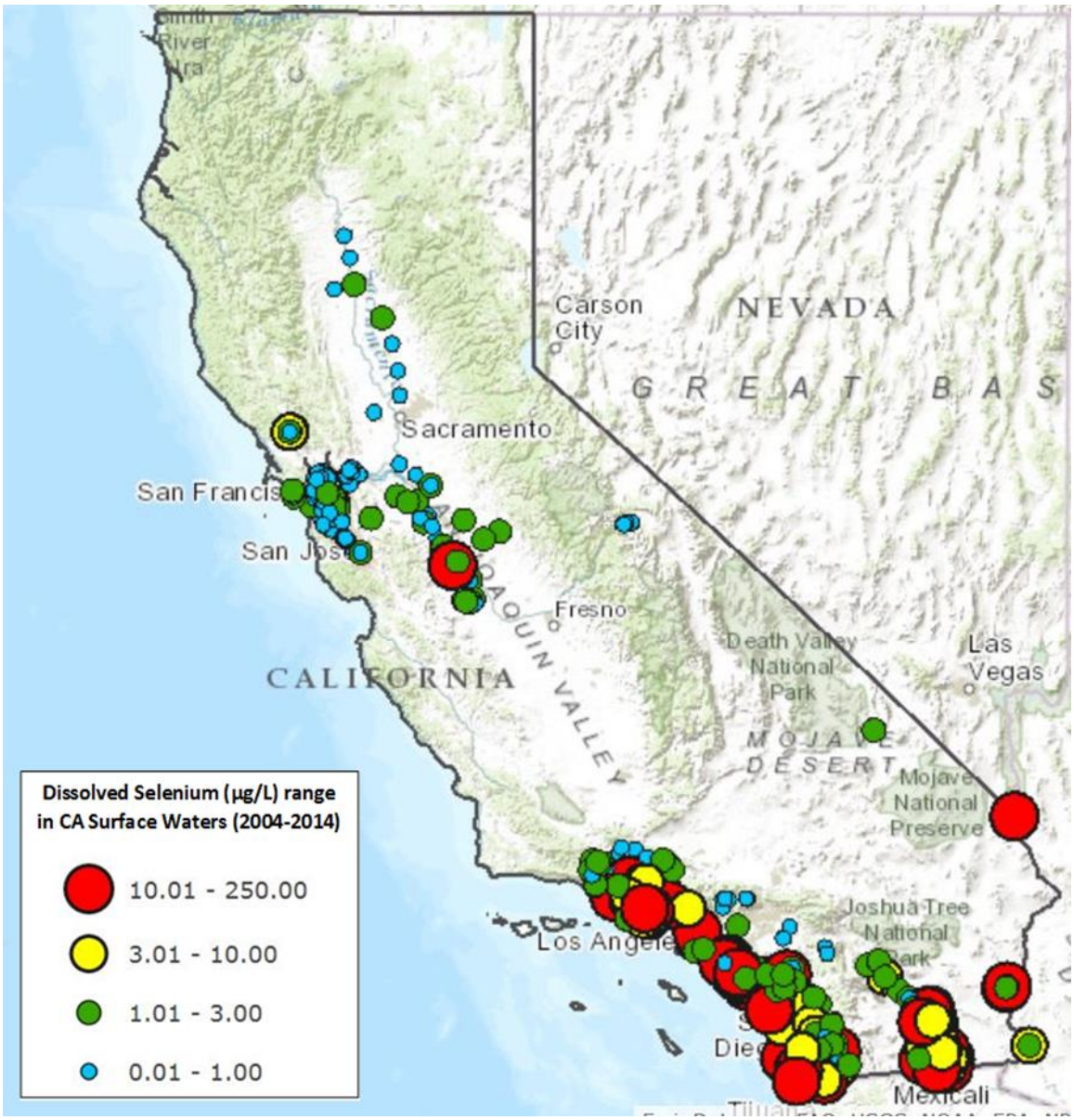
The EPA proposed water column dissolved selenium criterion elements (1.5 µg/L for lentic and 3.1 µg/L for lotic systems) are discussed next as they relate to recently reported selenium concentrations and distributions in California water bodies. EPA's level of confidence in the environmental data is high because the CEDEN database from which the data were derived for this analysis is the most comprehensive and largest source of selenium environmental monitoring data collected in California. A summary report of the selenium concentrations and distributions in California water bodies is provided in Appendix Table C-1 and Appendix Table C-2 above.

Appendix Figure C-1 maps the distributions and abundances of the reported total selenium concentrations (µg/L) in California surface water samples collected from October 5, 2004 through June 3, 2014. In addition to the reported total selenium concentrations, dissolved selenium concentrations in California surface water samples were also reported over the same 10-year period (CEDEN 2015). Appendix Figure C-2 maps the distributions and abundances of the reported dissolved selenium concentrations (µg/L) in surface water samples.



Appendix Figure C-1. Distributions and abundances of total selenium concentrations (µg/L) in surface water samples collected from October 5, 2004 through June 3, 2014.

The environmental data was last accessed through the California Environmental Data Exchange Network website (**CEDEN: http://www.ceden.org/**) on February 4, 2015.



Appendix Figure C-2. Distributions and abundances of dissolved selenium concentrations (µg/L) in surface water samples collected from October 5, 2004 through June 3, 2014.

The environmental data was last accessed through the California Environmental Data Excdhange Network website (**CEDEN: http://www.ceden.org/**) on February 4, 2015.

The total selenium concentration distribution was further characterized by sampling site location in its respective California Regional Water Quality Control Board (Regional Board) area and results are summarized in Appendix Table C-3. The Regional Board areas where the mean total selenium concentration exceeded 3.1 µg/L included Central Valley, Central Coast, Los Angeles, and San Diego. The dissolved selenium concentration distribution was also characterized by Regional Board area and is summarized in Appendix Table C-4. The Regional Board areas where the mean dissolved selenium concentration exceeded 3.1 µg/L included Colorado River, Los Angeles, San Diego, and Santa Ana.

Appendix Table C-3. Total Selenium Concentrations by Regional Board Area.

Regional Board	Number of HUC12 Sites	Number of Samples	Minimum Se (µg/L)	Mean Se (µg/L)	Maximum Se (µg/L)
1 North Coast	56	352	0.1	1.0	126
2 San Francisco Bay	7	99	0.03	1.3	4.8
3 Central Coast	6	8	1.6	6.9	14.6
4 Los Angeles	45	116	0.4	16.4	335
5 Central Valley	114	10,637	0.01	12.8	1591
6 Lahontan	2	18	0.2	0.6	1.9
7 Colorado River	NA	NA	NA	NA	NA
8 Santa Ana	10	12	0.1	1.2	5.1
9 San Diego	30	53	0.6	4.5	24.1
Grand Total	270	11,290			

Data Source: California Environmental Data Exchange Network (CEDEN) at www.ceden.org/.
Data includes reported Total Se concentrations (µg/L) in water samples collected from October 5, 2004 to June 3, 2014.
CEDEN was last accessed on February 4, 2015.
San Francisco Bay Regional Board summary excludes data from within the San Francisco Bay.
NA is not available.
HUC12 is Hydrologic Unit Code 12.

Appendix Table C-4. Dissolved Selenium Concentrations by Regional Board Area.

Regional Board	Number of HUC12 Sites	Number of Samples	Minimum Se (µg/L)	Mean Se (µg/L)	Maximum Se (µg/L)
1 North Coast	1	8	0.8	3.1	9.3
2 San Francisco Bay	10	57	0.03	1.2	5.1
3 Central Coast	NA	NA	NA	NA	NA
4 Los Angeles	48	109	0.3	9.1	129
5 Central Valley	38	178	0.01	2.7	106
6 Lahontan	2	18	0.3	0.5	1.4
7 Colorado River	16	201	0.03	6.1	46
8 Santa Ana	8	16	0.2	13.1	45
9 San Diego	48	123	0.2	10.4	250
Grand Total	171	710			

Data Source: California Environmental Data Exchange Network (CEDEN) at www.ceden.org/.

Data includes reported Dissolved Se concentrations (µg/L) in water samples collected from October 5, 2004 to June 3, 2014.

CEDEN was last accessed on February 4, 2015.

San Francisco Bay Regional Board summary excludes data from within the San Francisco Bay.

NA is not available.

HUC12 is Hydrologic Unit Code 12.