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FINAL

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

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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 the U.S. Environmental Protection Agency's (EPA) basis for and derivation of the final 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 the 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 the 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.

The EPA previously derived freshwater selenium aquatic life chronic tissue-based criterion elements for egg-ovary, whole body and/or muscle concentrations in fish, and translated the tissue-based criterion elements into freshwater selenium water column criterion elements for aquatic life (U.S. EPA 2016a), both of which are summarized in this document. The aquatic-dependent wildlife chronic tissue-based criterion element for bird eggs was not part of the 2016 aquatic life selenium criterion and therefore is derived herein.

The EPA utilized the translation equation from the peer-reviewed and validated Ecosystem Scale Selenium Model (Presser and Luoma 2010) to independently derive protective selenium concentrations in the water column based on bird egg criterion element concentration. This mechanistic model approach is also the basis for the Performance Based Approach (PBA) discussed in *Method for Translating Selenium Tissue Criterion Elements into Site-specific Water Column Criterion Elements for California, Version 2 December 2024* (EPA document number: 820-R-24-008**;** U.S. EPA 2024a), which can be used to derive site-specific protective selenium concentrations in the water column from the fish tissue and bird egg criterion element concentration in California. This mechanistic approach previously used for estimating protective selenium concentrations in the water column from fish egg-ovary criterion concentrations (U.S. EPA 2016a) is summarized in Part 5 of the present document. 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 or bird 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 are appropriate for California. In this analysis, the EPA found that the selenium water column-based criterion elements previously derived by the EPA (U.S. EPA 2016a) to protect aquatic life are also protective of aquatic-dependent wildlife, based on an independent analysis of the aquatic-dependent wildlife data. These tissue-based and default water column criterion element concentrations were developed to protect aquatic life and aquatic-dependent wildlife from reproductive effects associated with dietary exposure and maternal transfer to eggs resulting in mortality, teratogenicity, and decreased hatchability. The available data and modeling results demonstrate that aquatic life and aquatic-dependent wildlife are expected to be protected from the toxic effects of selenium in California by applying the following multi-element criterion:

Table 1-1. Summary of the Final 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[5]**	**Intermittent Exposure[6]**
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)	1.5 µg/L in lentic aquatic systems 3.1 µg/L in lotic aquatic systems	$WQC_{int} = \frac{WQC_{30-day} - C_{bkgrnd}(1 - f_{int})}{f_{int}}$
Duration	Instantaneous measurement[7]	Instantaneous measurement[7]	Instantaneous measurement[7]	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 criterion elements are expressed as steady-state.
2. Fish egg-ovary supersedes any whole-body, muscle, or water column criterion elements for aquatic life when fish egg-ovaries are measured, except as noted in footnote 4. Bird egg supersedes water column criterion elements for aquatic-dependent wildlife when bird eggs are measured, except as noted in footnote 4. The bird tissue criterion element is independently applicable from and equivalent to the fish tissue criterion elements.
3. Fish whole-body or muscle tissue supersedes the water column criterion elements when both fish tissue and water concentrations are measured, except as noted in footnote 4.
4. Water column criterion elements are based on dissolved total selenium in water and are derived from fish tissue and bird tissue criterion elements via bioaccumulation modeling. When selenium inputs are increasing, water column criterion elements are the applicable criterion elements in the absence of steady-state condition fish tissue or bird tissue data.
5. The water column criterion element, which applies independently to the respective aquatic life and aquatic-dependent wildlife uses, is applicable for all CWA purposes and consists of a water column value of 1.5 µg/L in lentic aquatic systems and 3.1 µg/L in lotic aquatic systems unless or until a site-specific water column criterion element is derived for a particular waterbody following the methodology described in *Method for Translating Selenium Tissue Criterion Elements into Site-specific Water Column Criterion Elements for California, Version 2 December 2024*. This publication is incorporated by reference into this section with the approval of the Director of the Federal Register under 5 U.S.C. 552(a) and 1 CFR part 5 1. All approved material is available at EPA, OW Docket, EPA West, Room 3334, 1301 Constitution Ave., NW, Washington, DC, 20004, (202) 566-2426. It is also available for inspection at the National Archives and Records Administration (NARA). For information on the availability of this material at NARA, call 202-741 -6030 or go to www.archives.gov/federal-register/cfr/ibr-locations.html.
6. Where $WQC_{30\text{-}day}$ is the applicable water column monthly criterion element, 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).
7. 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 EPA is finalizing the freshwater selenium criterion described in Table 1-1 for certain waters in California, as provided in Section V - Applicability of the EPA Promulgated Water Quality Standards of the final rule preamble. The EPA is finalizing the 2016 CWA section 304(a) selenium criterion for freshwater with the addition of a bird tissue criterion element, with associated default water column criterion elements, with the addition of the ability to develop a water column criterion element using a performance-based approach (PBA) for translating the tissue elements into a corresponding water-column criterion element on a site-specific basis. This approach allows for flexibility for dischargers and the State to derive site-specific water-column criterion elements based on site-specific data, as appropriate. Available data indicate that applying the criterion in Table 1-1 is expected to be protective of aquatic life and aquatic-dependent wildlife from the toxic effects of selenium, recognizing that the fish tissue elements and the bird egg 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 and one element is based on the concentration of selenium in bird eggs. The final tissue criterion 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 freshwater 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 default water column criterion lentic and lotic elements are based on translation of the tissue criterion elements using the mechanistic Presser and Luoma (2010) model to derive protective selenium concentrations in the water. The EPA is also finalizing intermittent exposure water column elements for lentic and lotic waters. The PBA consists of a methodology to allow translation of the tissue criterion elements into site-specific water column criterion elements. The EPA is finalizing a bird tissue criterion element that is independently applicable from and equivalent to the fish tissue elements. All tissue elements supersede the translated water column criterion elements, either using the default or PBA water column values, for the specific taxon when both are measured. The final selenium criterion, expressed as a single criterion composed of multiple elements, is expected to be protective of

aquatic life and aquatic-dependent wildlife from potential chronic effects of selenium in aquatic ecosystems.

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 final selenium water quality criterion for certain waters in California. 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 (CWA) section 303(c) and the 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. 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 the EPA's best estimate of the maximum concentrations of selenium, with associated frequency and duration specifications, that will support protection of 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 regulations, nor is this document a regulation itself. Thus, this document cannot, and does not, impose any legally binding requirements on the 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)(1) of the CWA. As provided for by the Clean Water Act, the EPA reviews and from time to time revises 304(a) AWQC recommendations (CWA 304(a) recommended criterion/a) 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 communities 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. States and authorized Tribes may adopt this criterion into their water quality standards (WQS) to protect aquatic life and aquatic-dependent wildlife designated uses. States and authorized Tribes may also modify this criterion before adopting these into standards. After adoption, states/authorized Tribes submit new and revised WQS to EPA for review and approval or disapproval. When adopted into state or tribal WQS and approved by the EPA, this criterion can become a basis for establishing National Pollutant Discharge Elimination System (NPDES) program permit limits, listing impaired waters under Section 303(d) and establishing Total Maximum Daily Loads (TMDLs).

In 1980, the EPA first published CWA 304(a) recommended numeric aquatic life criteria for selenium in freshwater (acute criterion 260 µg/L and chronic criterion 35 µg/L, U.S. EPA 1980). 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, the EPA published updated recommended selenium criteria in 1987 (U.S. EPA 1987) to address field-based toxicity observed in aquatic ecosystems at concentrations below the existing criteria values. The 1987 criteria were field-based and were based upon 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, the 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, the EPA held a peer consultation workshop to evaluate new science available for selenium relevant to the selenium aquatic life criterion (U.S. EPA 1998). 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 the 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, the 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).

The EPA used the scientific principles established in a 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 recommended criterion (U.S. EPA 2014) that was reviewed by an expert external peer review panel. In the 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. The EPA's 2016 final recommended 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 EPA national recommended aquatic life water quality criteria for selenium.

In 2016, the EPA recommended a national selenium criterion expressed as four elements (U.E. EPA 2016a). All elements are protective against chronic selenium effects in aquatic life. 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). The 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. The EPA recommended the intermittent values to address short-term exposures, such as noncontinuous discharges containing selenium, that could contribute to chronic exposures and effects through selenium bioaccumulation in either lotic or lentic systems. The 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 the EPA's recommended chronic criterion from potential acute effects of selenium if adopted and applied by states. Chapman et al. (2009) noted that selenium acute toxicity has been reported rarely in the aquatic environment. The most harmful effects of selenium on aquatic life and aquatic-

dependent wildlife come from the bioaccumulation of selenium through the food web (see Part 2.4 of this document below). As such, these chronic effects occur at lower concentrations of selenium than acute exposures; thus an acute criterion element was not derived.

The EPA has not established national selenium criteria recommendations for the protection of aquatic-dependent wildlife. However, the EPA has been involved in two separate efforts dealing with wildlife criteria for selenium. On December 12, 2011, the EPA approved a selenium wildlife criterion for Gilbert Bay of the Great Salt Lake (U.S. EPA 2011a).[1] The EPA 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, the EPA proposed to revise the current federal CWA selenium water quality criteria applicable to the San Francisco Bay and Delta to ensure that the criteria are protective of aquatic life and aquatic-dependent wildlife. Within the analysis that supports the final rule, the EPA reviewed avian toxicity studies and determined that the most "at risk" birds in this system are expected to be protected by the final criteria (U.S. EPA 2016b).[2]

1.2 California Toxics Rule

On May 18, 2000, the 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).[3] The CTR established numeric water quality criteria for priority toxic pollutants for inland surface waters and enclosed bays and estuaries within California. The EPA promulgated the CTR after California rescinded its water quality control plans containing pollutant objectives (criteria). The criteria that the 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.

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 freshwater criteria for selenium from the CTR, based solely on direct water column toxicity, is not considered

1 https://deq.utah.gov/water-quality/great-salt-lake-water-quality-standards
2 https://www.gpo.gov/fdsys/pkg/FR-2016-07-15/pdf/2016-16266.pdf
3 https://www.gpo.gov/fdsys/pkg/FR-2000-05-18/pdf/00-11106.pdf

adequately protective of 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). This technical support document (TSD) provides a scientifically-defensible revised selenium water quality criterion based on dietary exposures to selenium for certain waters in California in accordance with CWA section 303(c) and EPA's implementing regulations. This criterion is based on the best available data and best professional scientific judgments on the toxicological effects of selenium. The criterion herein relies heavily on the documented science supporting the EPA's 2016 final recommended freshwater chronic criterion for selenium (U.S. EPA 2016a), as well as additional toxicity and exposure data specific to aquatic-dependent wildlife in California that was not part of the aquatic life criterion, and 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). The criterion also reflects public comments and feedback the EPA received on the proposed rulemaking in order to ensure broad protection for both aquatic and aquatic-dependent species.

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 Wallschläger 2011; LeBlanc and Wallschläger 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 (Figure 2-1). 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 (Seiler et al. 1999). 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.

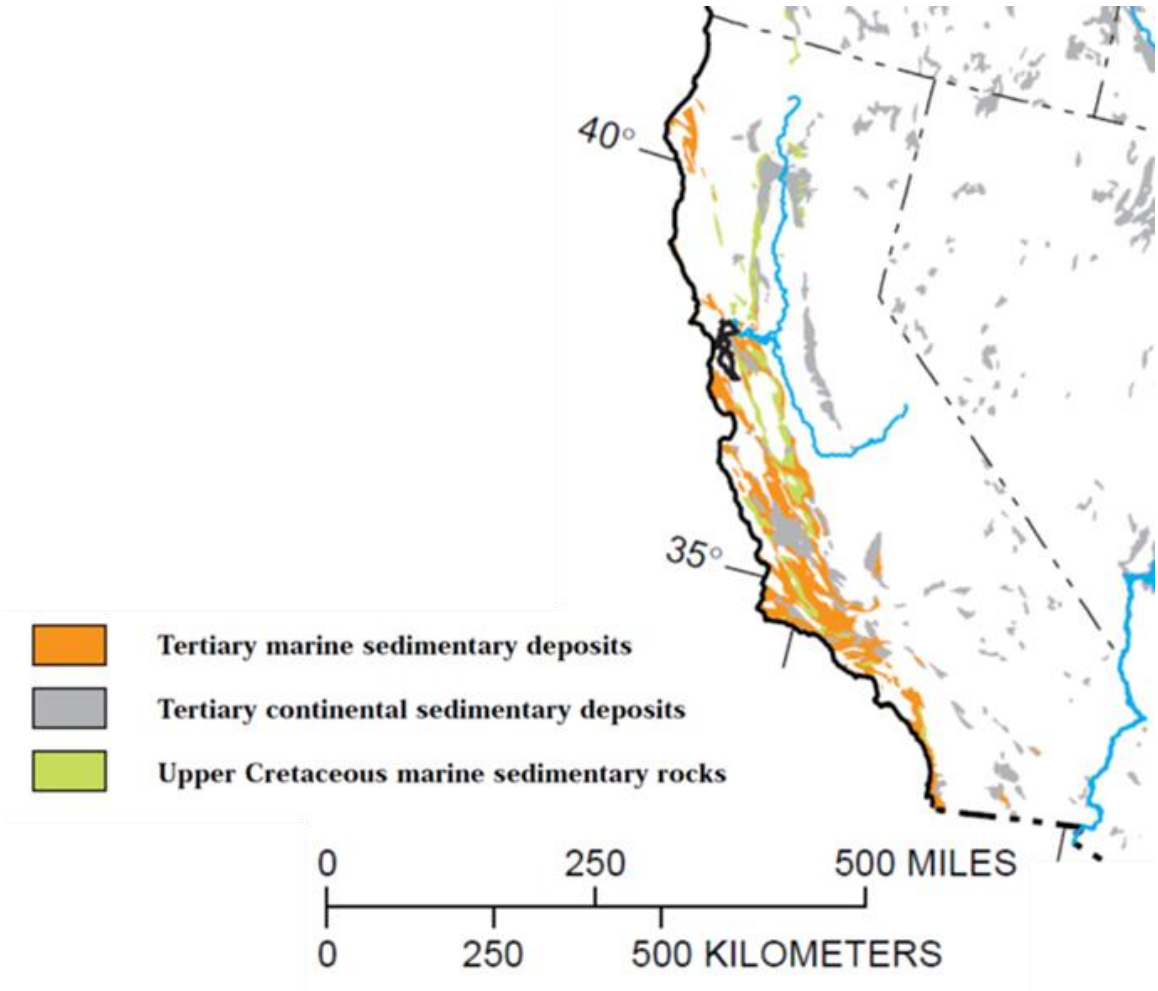

Figure 2-1. Areas of California with seleniferous marine geology.
Modified from: Seiler et al. (1999).

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.

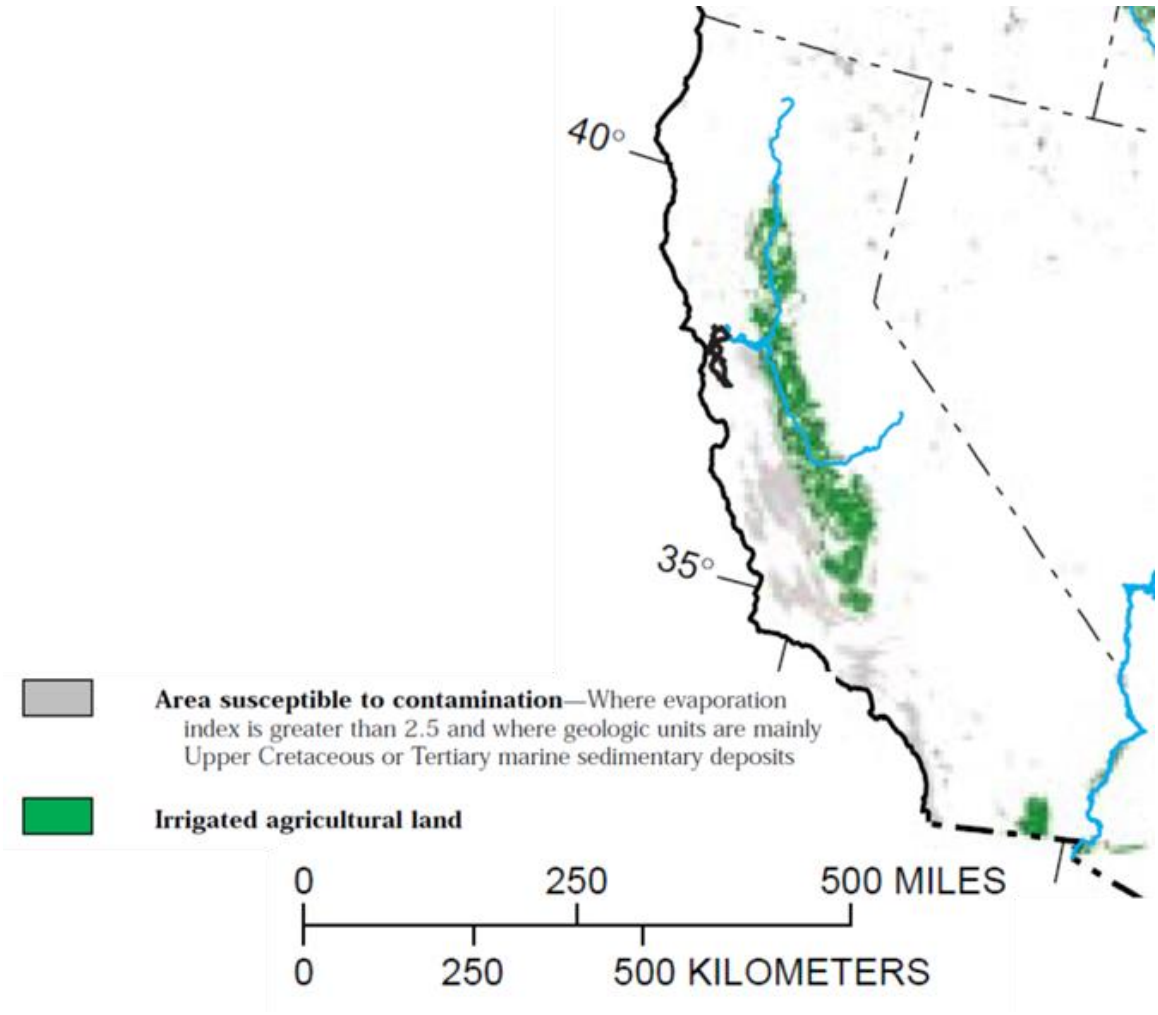

Figure 2-2. Areas of California susceptible to selenium contamination (gray) and where agricultural land is irrigated (green).

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.

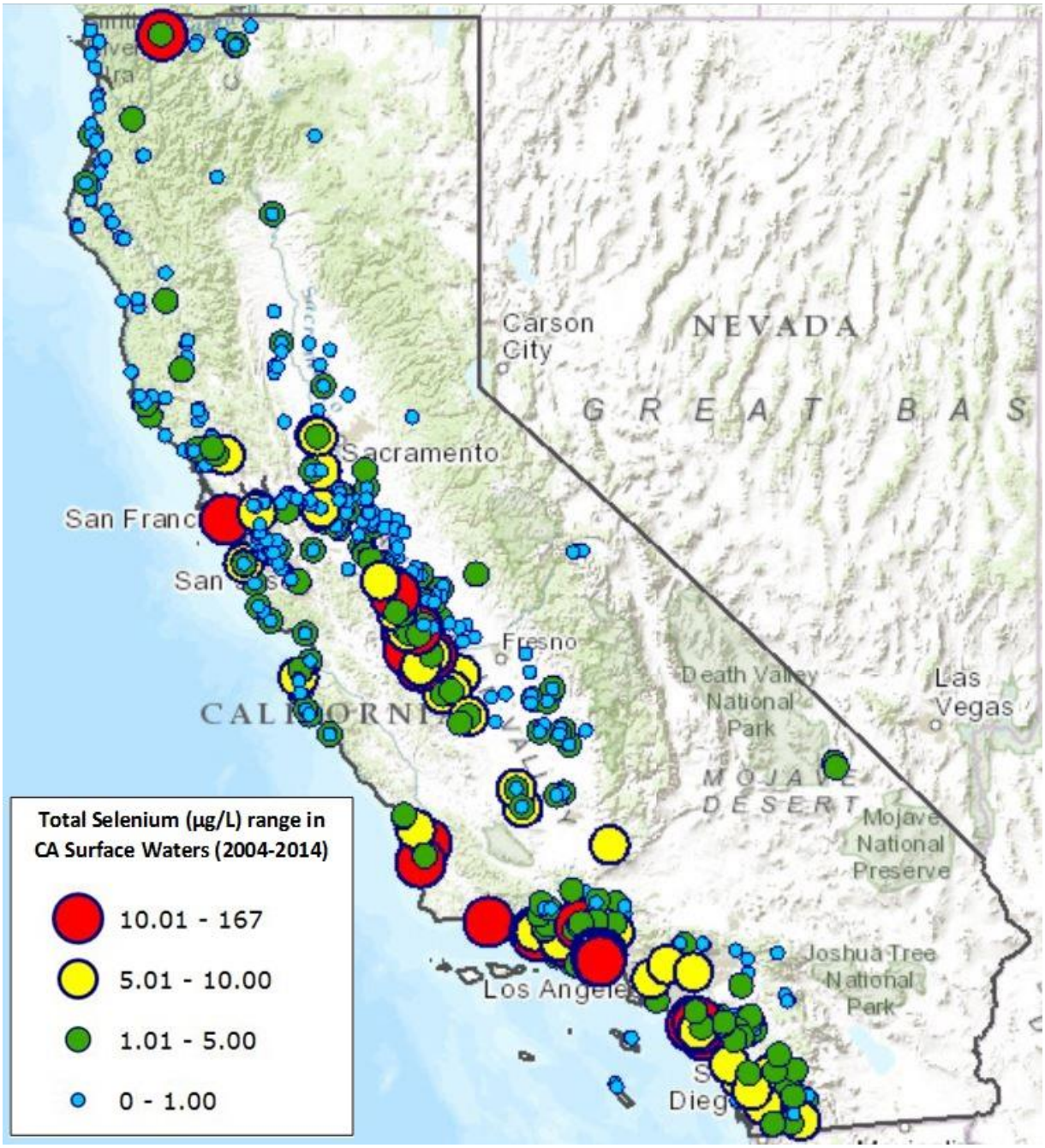

Figure 2-3. Distributions and abundances of total selenium concentrations (µg/L) in surface water samples collected from October 5, 2004 through June 3, 2014.

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 forms present in the food web, which result from transformation of inorganic forms 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 chemical speciation (Cutter and Cutter 2004; Meseck and Cutter 2006; Riedel and Sanders 1996). Selenium exists in four oxidation states (VI, IV, 0, -II) and in a wide range of chemical forms 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 Wallschläger 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, 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 (Ohlendorf and Heinz 2011; Palace et.al. 2004). 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; U.S. EPA 1987; Wang and Lovell 1997; Wilson et al. 1997), 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 transferred 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 cytochrome P-450 or flavin monoxygenase can biotransform organoselenium compounds into selenoxides (Palace et al. 2004).

A variety of lethal and sublethal deformities (terata) can occur in 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 95th 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 the EPA's 2016 "*Aquatic Life Ambient Water Quality Criterion for Selenium – Freshwater, 2016*." This previously published aquatic life criterion was developed using a genus sensitivity distribution (GSD), which represented 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, the 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 the 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 the 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 the 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 the EPA's derivation of effects concentrations for aquatic life, see the EPA's 2016 "*Aquatic Life Ambient Water Quality Criterion for Selenium – Freshwater, 2016*." This approach is consistent with the 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

As described in U.S. EPA (2016a), the 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 the EPA and USGS use the same model to relate the concentration of selenium in fish tissue to water, the 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 the 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 criterion elements that would also be protective of aquatic-dependent wildlife, the 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. The default water column criterion elements for California are the same as those defined in the *Aquatic Life Ambient Water Quality Criterion for Selenium – Freshwater, 2016*. These water criterion elements are subordinate to the bird and fish tissue criterion elements.

The EPA also developed a PBA for translating the fish and bird tissue elements into a corresponding water-column criterion element that could be applied on a site-specific basis, as appropriate. If 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). A PBA-based 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, the 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, and the default water column criterion elements herein. An analysis of the translation from bird egg to water column criterion elements showed that the water column element translated from fish tissue is also protective of aquatic-dependent birds.

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 and 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 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 this one species since the current literature does not include sufficient toxicity data to develop a sensitivity distribution for a range of avian 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).

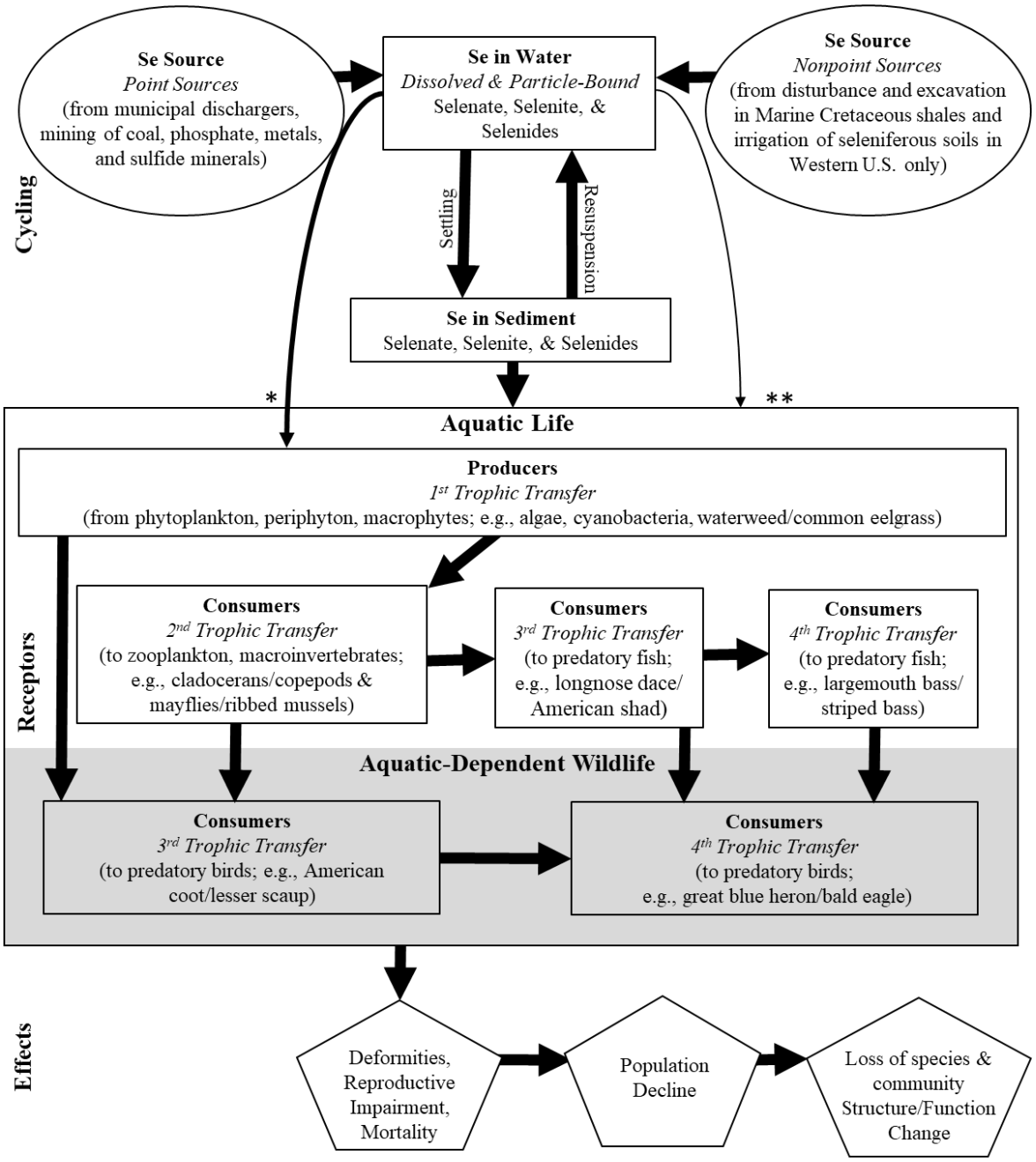

Figure 2-4. Conceptual model diagram of sources, compartmental partitioning, and trophic transfer pathways of selenium in the aquatic environment and bioaccumulation and effects in aquatic-dependent wildlife.

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 the 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).[4] the EPA is finalizing 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 the EPA's 2016 "*Aquatic Life Ambient Water Quality Criterion for Selenium – Freshwater, 2016*," data were obtained primarily by search of published literature using the 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 the 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 from 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

4 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 chronic 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 the 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 5th 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 (Janz et al. 2010; Ohlendorf 2003), 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.

Aquatic-dependent wildlife data considered for inclusion in this California selenium criterion were obtained from published literature reports on chronic exposures of selenium that were associated with effects on mortality, growth, and/or reproduction. This set of published literature was identified by both the 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% or greater 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 because 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 by the EPA OW generally as described by Stephan et al. (1985) in the 1985 Guidelines and the EPA OW's Open Literature Standard Operating Procedure (SOP) (U.S. EPA 2024b). These data quality reviews ensured the studies used to derive the criterion element were scientifically robust. 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. Mallard (*Anas platyrhynchos*) was 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 to selenium toxicity data in the current literature, with Harding (2008) reporting adverse effects on hatchability at selenium egg concentrations of approximately 22.0 mg/kg dw for this species. (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 quantitatively 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 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 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% hatch and 297 g, respectively) compared to the control group (65.7% hatch and 371 g, respectively), these reductions were not statistically significantly different from controls. However, an 18.3% increase in abnormal embryos was observed in the selenium treatment group as were reductions in the percentage 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 that was consistently observed and most sensitive 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, because hatchability was one of the most sensitive

endpoints reported, was consistently observed and significantly different from controls in two of the three studies Heinz et al. 1989 and Stanley et al. 1996), 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 each observation was weighted according to sample size based on the number of eggs. The bird egg EC_{10} 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 the 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 the 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 the 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.

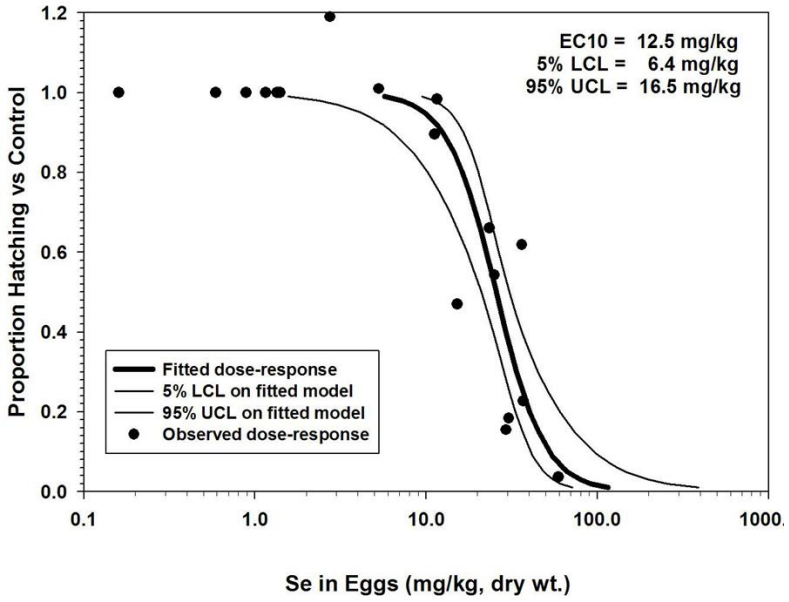

Figure 4-2. Mallard egg hatchability as a function of selenium concentration in eggs.
Source is Figure 17.2 from Ohlendorf (2003).
The data were normalized to their respective control hatchability values. LCL = lower confidence limit. UCL = upper confidence limit.

2011 EPA Reanalysis of the Six Mallard Toxicity Studies

The 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).

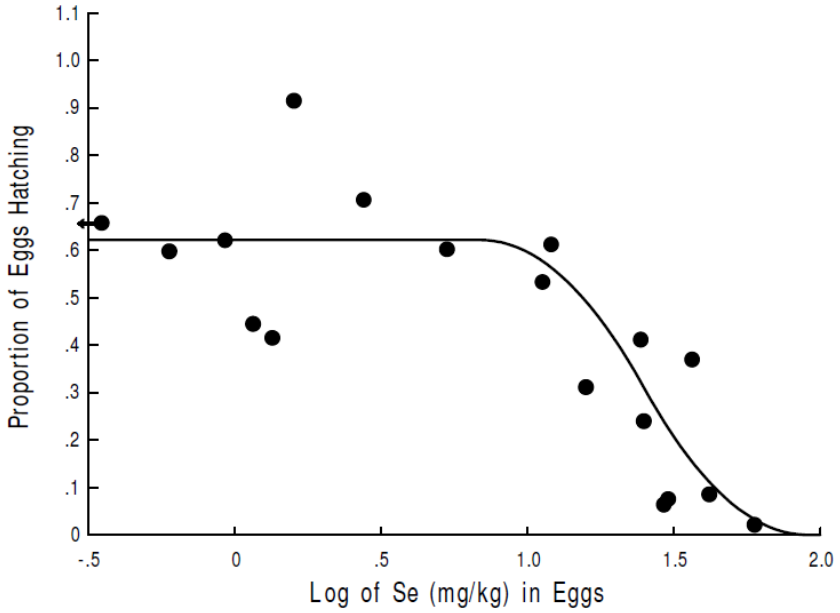

Figure 4-3. Mallard egg hatchability in the six studies, fitted using a tolerance distribution model, without normalization to control values. Source is Figure 2 in U.S. EPA (2011a). Confidence intervals surrounding the EC_{10} were not included in the source document.

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, the EPA derived an EC_{10} for all six mallard studies after first normalizing egg hatchability from each study to their respective controls. The 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), the 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 one approach over the other, the different values derived from the application of these two models to the same data are both scientifically defensible.

The 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.

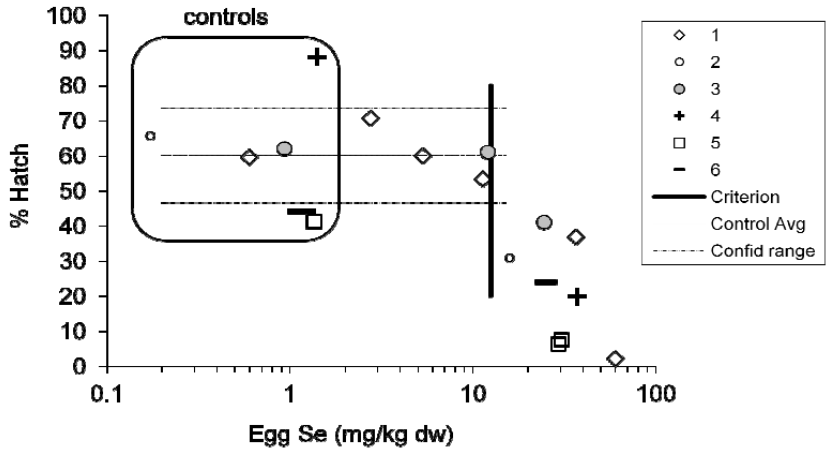

Figure 4-4. Mallard percent hatch v. egg concentration for six studies values.

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, the 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 the 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, the 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 for reduced egg hatchability when applying a biphasic model to the mallard egg hatchability data reported by Heinz et al. (1989).[5] 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), 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). 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.

The 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 toxicity 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 EC_{10} of 11.2 mg Se /kg dw.

5 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).

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 the 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. The EPA is finalizing a mallard egg EC_{10} of 11.2 mg Se/kg dw as an 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, the 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 translation from bird egg to water that is equivalent to the previously derived 2016

national aquatic life selenium criterion. In this analysis, the 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 for 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 the 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 a 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. 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. The seleno-DL-methionine test was found to be to be acceptable during data quality review. Therefore, this test could be considered for quantitative use. However, this endpoint (% malformed embryos) was generally less sensitive (LOEC reported here as 36.7 mg Se/kg dw) than the most sensitive endpoint of hatchabililty (LOECs measured in eggs of 15.3, 60.0 and 24.48 mg Se/kg dw, respectively) that was reported in other papers (Heinz et al. 1987, 1989 and Stanley et al. 1996) and that was used quantitatively in the derivation of the mallard EC_{10} of 11.2 mg Se/kg dw). Therefore, the data presented in Hoffman and Heinz (1988) was not used to estimate an EC_{10} for hatchability (as the endpoints were different and the hatchability endpoint was measured for this study but reported in separate publications) or in the derivation of the criteria.

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. 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. This test was of acceptable data quality and was considered for quantitative use. However, the authors only reported dietary concentrations and did not measure or report the concentration in eggs. Thus, without the egg concentrations this study could not be included to estimate an EC_{10} for mallard egg hatchability. Alternatively, this study indicates that the three studies used to estimate an EC_{10} for hatchability in mallard are protective since the dietary, author-reported LOECs are in line (ranging between 7 and 16 mg Se/kg ww) with the dietary, author reported LOEC of 15 mg Se/kg ww for this study.

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 the 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). This study was not included in the combined mallard dataset since the high control hatchability observed in this study was determined to be statistically different from the other control groups in the meta-analysis (91.4% compared to 57-64.4% in the remaining studies). Thus, the hatchability data from this study provide qualitative support of the wildlife criterion.

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).

This test was determined to be of sufficient data quality. However, the study authors, who are 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 study 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. Therefore, this study was not used quantitatively to derive the selenium criterion for aquatic-dependent wildlife and was instead used to demonstrate the protectiveness of the criterion at 11.2 mg Se/kg dw based on the most sensitive endpoint, hatchability, in the most sensitive species, mallard.

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). 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 and the MATC was 16.26 mg Se/kg dw in diet or 4.37 mg Se/kg ww. The egg concentrations were further converted

to dry weights assuming a moisture content of 82.2% (Sotherland and Rahn 1987) to make them comparable with other egg toxicity values reported in the literature and to the bird egg criterion for selenium. The converted reported egg NOEC was 14.44 mg Se/kg dw, LOEC was 41.80 mg Se/kg dw, and MATC 24.57 mg Se/kg dw. Therefore, this study was not used quantitatively to derive the selenium criterion for aquatic-dependent wildlife and was instead used to demonstrate the protectiveness of the criterion at 11.2 mg Se/kg dw.

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 with an MATC of 14.5 mg Se/kg dw. This test was considered for qualitative use to support the protectiveness of the bird egg criterion of 11.2 mg Se/kg dw 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 (6.7 mg/kg dw in eggs) and high exposure sites (31.4 mg/kg dw in eggs).

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 author-reported egg toxicity value for this study was > NOEC of 7.3 mg Se/kg dw because the differences in the responses (egg failure, hatchability, and nestling survival) between exposed and reference areas were not significantly different. Further, 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. The author-reported NOEC of > 7.3 mg Se/kg dw was used qualitatively to support the protectiveness of the bird egg criterion of 11.2 mg Se/kg dw as spotted sandpipers would likely be protected by the bird egg criterion given the lack of effects on any measured endpoints at the 7.3 mg Se/kg dw. A decision framework for non-definitive values was applied that is consistent with past practice (U.S.EPA 2013), The decision framework was not to use "greater than" values for concentrations of low magnitude or "less than" values for concentrations of high magnitude because they do not provide meaningful toxicity information. Additionally, considering the life history characteristics of spotted sandpipers, which inhabit areas near or along the shoreline of freshwater and consume small invertebrates (e.g., emergent insects, snails and small crustaceans), they are unlikely to be exposed to high selenium concentrations.

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. Therefore, this study was used qualitatively to demonstrate the protectiveness of the bird egg criterion at 11.2 mg Se/kg dw.

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, the NOEC was > 13.3 mg Se/kg dw. This study was considered for qualitative use since it was a field study with other chemicals potentially present. In particular, the study site was downstream of a uranium mill and the study authors note that several metals are present at the site in varying concentrations. However, this study provides insight into the relative sensitivity of tree swallows exposed to selenium and indicates that the bird egg criterion of 11.2 mg Se/kg dw based on mallards would be protective of this species.

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 than 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. The study authors indicated that this was likely due to larger clutch sizes in the impacted colonies that was independent of selenium exposure. 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. The author-reported NOEC was > 4.75 mg Se/kg dw. This study was considered for qualitative use given that it was a field study with known mixtures and given the decision framework for non-definitive values that was applied to be consistent with past practice (U.S.EPA 2013). The decision framework was not to use "greater than" values for concentrations of low magnitude because they do not provide meaningful toxicity information. In particular, the study site was downstream of a coal fly ash spill that occurred in 2008, and the study authors note that several metals are present at the site in varying concentrations. This study provides little insight into the relative sensitivity of tree swallows exposed to selenium. However, when the life history characteristics of tree swallow are considered, since they prefer to inhabit areas near water, but also nest in areas such as fields with an abundance of small insects, the exposure of selenium to tree swallows is not expected to be as high as waterfowl, such as mallard, which are the most sensitive species with selenium toxicity data and are protected by the selenium aquatic-dependent wildlife criterion.

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 reported by the study authors found a significant relationship between hatchability and egg selenium ($P < 0.001$, $n = 116$). The study authors indicated adverse effects on hatchability at approximately 22 mg Se/kg dw egg. This author-reported value was considered a NOEC by the EPA in the development of this selenium aquatic-dependent wildlife criterion. This study was considered for qualitative use given that it was a field study with potential mixtures. In particular, the study site was

downstream of a coal mining site. Additionally, the study use classification was influenced by the amount of scatter in the hatchability data, which made the independent calculation of an EC_{10} value problematic. However, this study provides insight into the relative sensitivity of red-winged blackbirds exposed to selenium and indicates that the bird egg criterion of 11.2 mg Se/kg dw based on mallards would be protective of this species.

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 NOECs 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. This study was considered for qualitative use since it was a field study with known mixtures. In particular, some study sites were in mining areas. However, this study provides insight into the relative sensitivity of red-winged blackbirds and American robins exposed to selenium and indicates that the bird egg criterion of 11.2 mg/kg dw based on mallards would likely be protective of this species. This is especially the case when the life history characteristics of blackbirds and robins are considered. Blackbirds prefer to nest in areas near water, such as wetlands and marshes, but also nest in areas such as meadows with an abundance of small insects. American robins inhabit woodlands and forested areas. Therefore, the exposure of red-winged blackbirds and American robins to selenium are not expected to be as high as waterfowl, such as mallard, that appear to be more sensitive to and are likely to experience higher exposure to selenium.

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. The dietary NOEC and LOECs of the standard and low protein treatments of this study were 15 and 60 mg Se/kg dw and the NOEC of the high protein treatment group was < 15 mg Se/kg dw since effects were observed in the lowest selenium treatment group compared to controls. These toxicity values were 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; Stanley et al. 1996).

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. As with the previously summarized toxicity studies, this dietary NOEC was higher than those observed for the reproductive studies of mallard with dietary LOEC values ranging between 7 to 16 mg Se/kg dw (Heinz et al. 1989; Heinz et al. 1987; Stanley et al. 1996). Therefore, the selenium criterion for aquatic-dependent wildlife based on the most sensitive endpoint of hatchability is expected to be protective of the growth effects observed in this study on adult male mallards.

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, with dietary LOEC values ranging from 7 to 16 mg Se/kg dw. Therefore, the selenium criterion for aquatic-dependent wildlife based on the most sensitive endpoint of hatchability is expected to be protective of the growth effects observed in this study on adult male mallards.

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, and 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. Therefore, the selenium criterion for aquatic-dependent wildlife based on the most sensitive endpoint of hatchability is expected to be protective of the growth effects observed in this study on adult male 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. However, this dietary LOEC was higher than those observed for the reproductive studies of mallard with dietary LOEC values ranging between 7 to 16 mg Se/kg dw (Heinz et al. 1989; Heinz et al. 1987; Stanley et al. 1996). Therefore, the selenium criterion for aquatic-dependent wildlife based on the most sensitive endpoint of hatchability is expected to be protective of the effects observed in this study on adult male mallards.

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. The unbounded dietary NOEC of > 14.9 mg Se/kg dw was higher than the dietary LOECs of 7 to 16 mg Se/kg dw from the reproductive studies in which hatchability was the most sensitive endpoint (Heinz et al. 1989; Heinz et al. 1987; Stanley et al. 1996). Therefore, the selenium criterion for aquatic-dependent wildlife based on the most sensitive endpoint of hatchability is expected to be protective of the effects observed in this study on adult male mallards.

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. Therefore, the dietary NOEC for this study was determined to be > 20.7 mg Se/kg dw. This NOEC was higher than the dietary LOECs (ranging from 7 to 16 mg Se/kg dw) from reproductive mallard studies that were used to derive the selenium criterion for aquatic-dependent wildlife.

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. Therefore, it was determined that the NOEC was 6.3 mg Se/kg dw and the LOEC was 12 mg Se/kg dw for this study. The dietary LOEC from this study was higher than some of the dietary LOECs (ranging from 7 to 16 mg Se/kg dw) and was in line with the genus mean chronic value of 9.125 mg Se/kg dw from reproductive mallard studies that were used to derive the selenium criterion for aquatic-dependent wildlife. Therefore, the selenium criterion for aquatic-dependent wildlife based on the

most sensitive endpoint of hatchability is expected to be protective of the effects observed in this study on adult male mallards.

Part 5 METHOD USED TO TRANSLATE THE BIRD EGG AND FISH TISSUE CRITERION ELEMENTS INTO WATER COLUMN ELEMENTS

5.1 Purpose

This chapter outlines the details of the mechanistic modeling method that was used to calculate protective default water column values and which also can be used to derive a site-specific chronic water-column selenium criterion element. 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 equivalent to the 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. The fish tissue-to-water column translation is a summary of the 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 to demonstrate that the 2016 national selenium aquatic life criterion water column elements are protective of aquatic-dependent wildlife. The approach serves as an example 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 the EPA's 2016 national aquatic life criterion for selenium (U.S. EPA 2016a), the 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, please see Section 3.2.1 of the EPA's 2016 aquatic life criterion (U.S. EPA 2016a). The general model is described by (Equation 5-1).

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

(Equation 5-1)

Where:

C_{water}	=	Selenium concentration in the water column (µg/L) (i.e., water column criterion element)
$C_{bird\ egg\ or\ fish\ egg}$	=	Selenium concentration in the eggs of birds (mg/kg) or the eggs or ovaries of fish (mg/kg) (i.e., egg criterion element)
$TTF^{composite}$	=	Composite trophic transfer factor. The overall *TTF*, or level of selenium bioaccumulation from the base of the food chain 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 uses only the bird egg value.]

(Equation 5-1) describes the translation from the concentration of selenium in the eggs of birds or the eggs and ovaries of fish at the egg or egg/ovary tissue criterion element, respectively, to the concentration of selenium in the water column that would be protective of these tissue criterion elements. This translation 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*. The EPA obtained these data by searching published literature using the 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. The 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 the EPA calculated *EFs* is described in the EPA's 2016 Aquatic Life Criterion (U.S. EPA 2016a). How the EPA calculated *TTFs* and *CFs* as they related to aquatic life is described in detail in Appendix B of the EPA's 2016 Aquatic Life Criterion (U.S. EPA 2016a).

5.3.1 Derivation of Trophic Transfer Factor (TTF) Values

The parameter $TTF^{composite}$ (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:

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

(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 for aquatic-dependent birds 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.

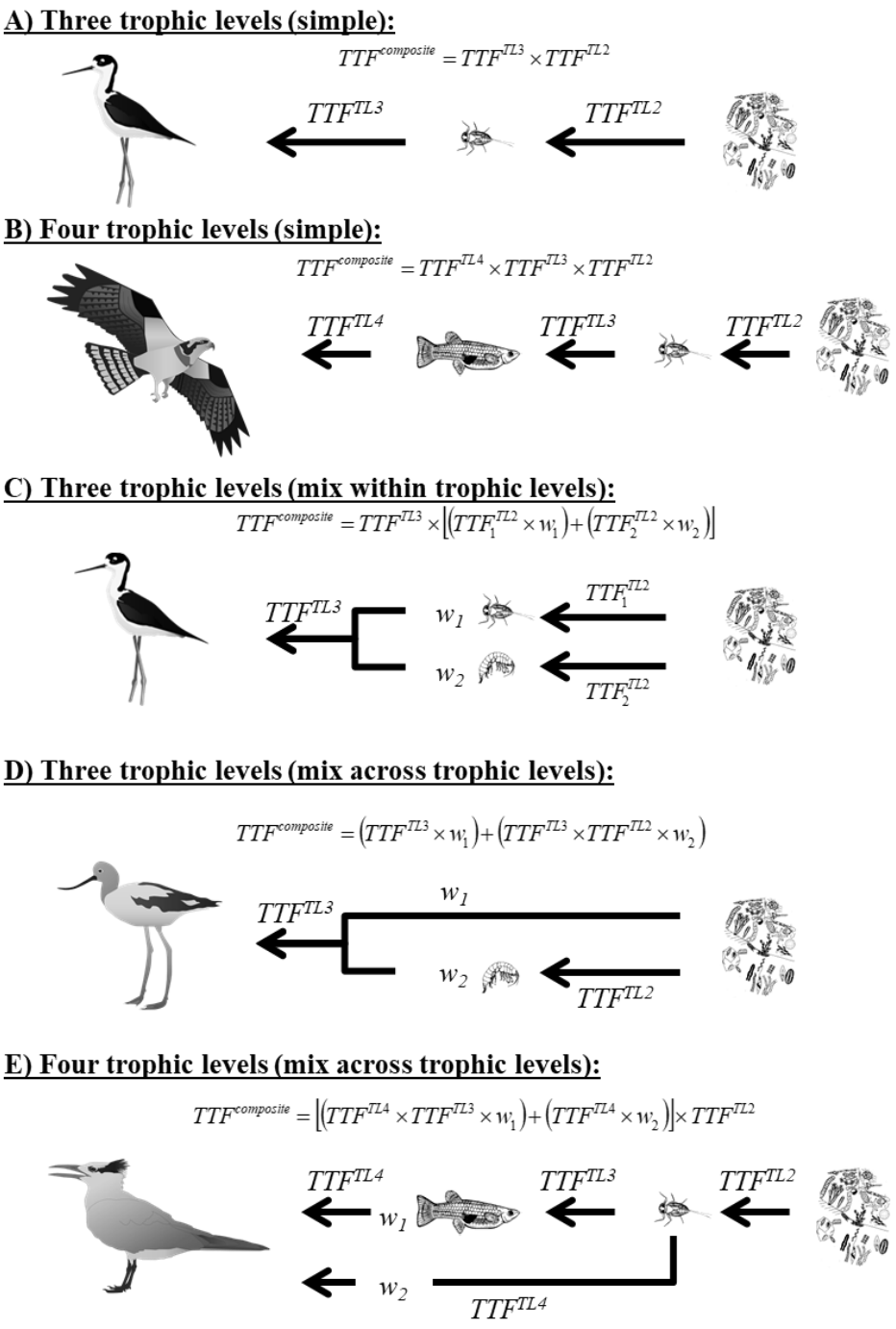

Figure 5-1. Example aquatic system scenarios and the derivation of the equation parameter $TTF^{composite}$.

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 the EPA's approach for *TTF*s for all taxonomic groups, including the newly derived *TTF*s for birds.

The 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 the EPA's 2016 Aquatic Life Criterion (U.S. EPA 2016a). The 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 aquatic-dependent bird, the EPA searched for additional measurements from other aquatic organisms or particulate material that were 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. The 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, the 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 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, the 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, the EPA used the same rule established in U.S. EPA (2016a). The 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 the 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 the EPA's 2016 Selenium Aquatic Life Criterion, the 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. The EPA used this approach for the new bird *TTF*s derived in this technical support document (TSD).

The EPA used the 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 the EPA derived the *TTF* value using field data. If a *TTF* value could be calculated from both physiological coefficients and field data, the 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, the 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, the 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 *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. The 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, the 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.

The 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, the EPA used either the additional data or a taxonomic classification approach to estimate *CF*. The 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, because the only tissue value for birds is for eggs.

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
smallmouth bass	*Micropterus dolomieu*	1.42	0.19

Common name	Scientific name	*CF*	Std. Dev.[a]
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, the EPA only used aquatic sites with sufficient data to calculate a reasonably reliable *EF* value.

To calculate the *EF* of aquatic systems for the 2016 national selenium aquatic life criterion, the 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. The 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, the 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, the 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), the 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, the 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.

In using a site-specific PBA approach 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 PBA approach (*Method for Translating Selenium Tissue Criterion Elements into Site-specific Water Column Criterion Elements for California, Version 2 December 2024;* U.S. EPA 2024a).

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, the 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 the EPA developed food web models, the 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. The 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, the 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, the EPA used the median *TTF* from the matching species. The 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

The 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 *TTF*. The remaining 8 bird species' composite *TTFs* were estimated using their species-specific dietary compositions and the application of empirically-derived *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 for Bird Species*

Composite *TTFs* were calculated for eight bird species with empirically derived *TTFs* 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 the World web site: (https://birdsoftheworld.org/bow/home). The eight bird species with empirically derived *TTFs* included two non-migratory (or resident) species: American coot and red-winged blackbird; and four migratory species: American avocet, cinnamon teal, eared grebe, gadwall, pied-billed grebe, and yellow headed blackbird.

The EPA first calculated bird *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.

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	1.47
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*	2.08
brown pelican (pied-billed grebe)	*Pelecanus occidentalis*	1.79
bald eagle (pied-billed grebe)	*Haliaeetus leucocephalus*	1.75
Ridgway's rail (American coot[a])	*Rallus obsoletus*	3.19
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*	1.84

[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, the EPA translated the egg-ovary criterion element to a distribution of water column concentration values for lentic and lotic aquatic systems. The 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). The 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, the 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, the EPA used the lowest translated water value for that site. Using this methodology, the EPA translated the egg-ovary FCV into water column concentrations at 26 lentic and 39 lotic aquatic sites. The 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 the 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 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 resultant concentration of selenium concentration the water column 1.5 µg/L in lentic aquatic systems and 3.1 µg/L in lotic aquatic systems unless or until a site-specific water column criterion element is derived for a particular waterbody following the methodology does not exceed described in *Method for Translating Selenium Tissue Criterion Elements into Site-specific Water Column Criterion Elements for California, Version 2 December 2024*.

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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).

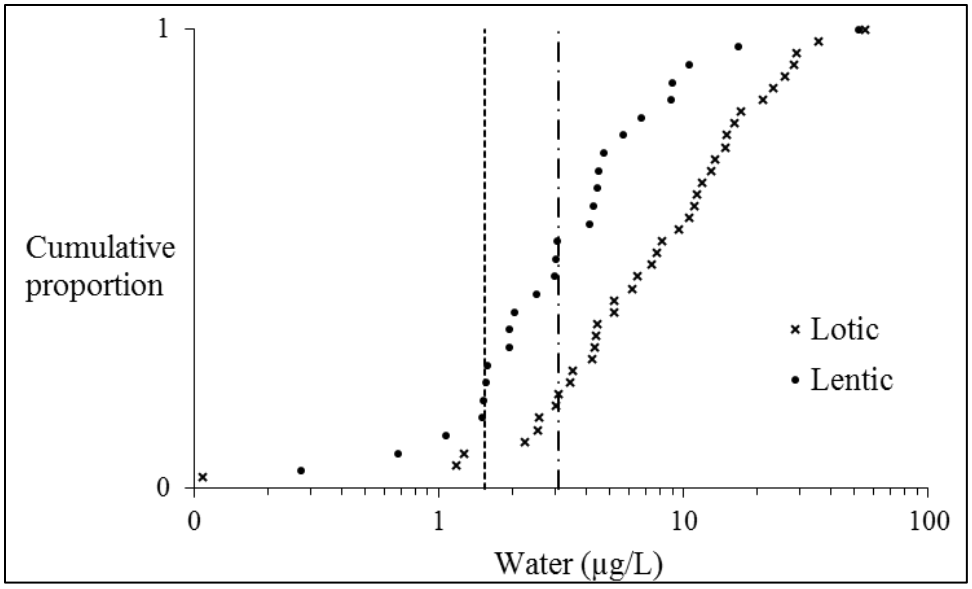

Figure 5-2. Probability distribution of the water column concentrations translated from the fish egg-ovary criterion element at 26 lentic and 39 lotic aquatic sites (U.S. EPA 2016a).
Dashed and dash-dot lines show the 20th percentiles of the lentic and lotic distributions, respectively.

In the 2016 national selenium aquatic life criterion, the 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 are considered 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 (*Method for Translating Selenium Tissue Criterion Elements into Site-specific Water Column Criterion Elements for California, Version 2 December 2024*) 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, the 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, the 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, the EPA translated the bird egg final criteria value into water column concentrations at 26 lentic and 39 lotic aquatic sites. At each site, the 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. The 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 bioaccumulative 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 were used as part of a

distribution to calculate a protective water column element, paralleling the approach used in the 2016 national selenium aquatic life criterion. These values are default values for the State. The State could also consider following 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	3.19	1.52
Birkner 1978	Galett Lake, Laramie WY	Ridgway's rail	Lentic	0.88	3.19	4.01
Birkner 1978	Larimer Highway 9 Pond, Fort Collins CO	Ridgway's rail	Lentic	1.70	3.19	2.06
Birkner 1978	Meeboer Lake, Laramie WY	Ridgway's rail	Lentic	0.58	3.19	6.08
Birkner 1978	Miller's Lake, Wellington CO	Ridgway's rail	Lentic	2.37	3.19	1.48
Birkner 1978	Sweitzer Lake, Delta CO	Ridgway's rail	Lentic	0.87	3.19	4.02
Birkner 1978	Twin Buttes Reservoir, Laramie WY	Ridgway's rail	Lentic	1.21	3.19	2.91
Bowie et al. 1996	Hyco Reservoir	Ridgway's rail	Lentic	2.35	3.19	1.50
Butler et al. 1993	Navajo Reservoir, Piedra River Arm, near La Boca	Ridgway's rail	Lentic	1.26	3.19	2.78
Butler et al. 1997	Large pond south of G Road, southern Mancos Valley	Ridgway's rail	Lentic	2.00	3.19	1.75
Butler et al. 1997	Pond downstream from site MNP2, southern Mancos Valley	Ridgway's rail	Lentic	5.15	3.19	0.68
Butler et al. 1997	Pond on Woods Canyon at 15 Road	Ridgway's rail	Lentic	0.90	3.19	3.88
Grasso et al. 1995	Arapahoe Wetlands Pond	Ridgway's rail	Lentic	0.86	3.19	4.06
Lemly 1985	Badin Lake	Ridgway's rail	Lentic	12.48	3.19	0.28
Lemly 1985	Belews Lake	Ridgway's rail	Lentic	1.75	3.19	2.01
Lemly 1985	High Rock Lake	Ridgway's rail	Lentic	4.99	3.19	0.70
Muscatello and Janz 2009	Vulture Lake	Ridgway's rail	Lentic	1.01	3.19	3.47
Orr et al. 2012	Clode Pond 11	Ridgway's rail	Lentic	0.71	3.19	4.91

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	3.19	2.14
Orr et al. 2012	Fording River Oxbow 10	Ridgway's rail	Lentic	1.34	3.19	2.61
Orr et al. 2012	Henretta Lake 27	Ridgway's rail	Lentic	0.50	3.19	7.02
Saiki and Lowe 1987	Kesterson Pond 11	Ridgway's rail	Lentic	0.51	3.19	6.94
Saiki and Lowe 1987	Kesterson Pond 2	Ridgway's rail	Lentic	0.32	3.19	11.11
Saiki and Lowe 1987	Kesterson Pond 8	Ridgway's rail	Lentic	0.60	3.19	5.83
Saiki and Lowe 1987	Volta Pond 26	Ridgway's rail	Lentic	0.93	3.19	3.76
Stephens et al. 1988	Marsh 4720	Ridgway's rail	Lentic	0.10	3.19	36.65
Butler et al. 1991	Uncompahgre River at Colona	Ridgway's rail	Lotic	0.63	3.19	5.57
Butler et al. 1993	Spring Cr. at La Boca	Ridgway's rail	Lotic	0.18	3.19	19.63
Butler et al. 1995	Hartman Draw near mouth, at Cortez	Ridgway's rail	Lotic	0.15	3.19	23.54
Butler et al. 1995	McElmo Cr. at Hwy. 160, near Cortez	Ridgway's rail	Lotic	0.90	3.19	3.90
Butler et al. 1995	McElmo Cr. downstream from Alkali Cyn.	Ridgway's rail	Lotic	0.37	3.19	9.55
Butler et al. 1995	McElmo Cr. downstream from Yellow Jacket Cyn.	Ridgway's rail	Lotic	0.12	3.19	29.28
Butler et al. 1995	McElmo Cr. upstream from Yellow Jacket Cyn.	Ridgway's rail	Lotic	0.10	3.19	36.78
Butler et al. 1995	Navajo Wash near Towaoc	Ridgway's rail	Lotic	0.20	3.19	17.93
Butler et al. 1995	San Juan River at Four Comers	Ridgway's rail	Lotic	0.26	3.19	13.40
Butler et al. 1995	San Juan River at Mexican Hat Utah	Ridgway's rail	Lotic	0.29	3.19	12.15
Butler et al. 1995	Woods Cyn. Near Yellow Jacket	Ridgway's rail	Lotic	0.40	3.19	8.68
Butler et al. 1997	Cahone Canyon at Highway 666	Ridgway's rail	Lotic	0.20	3.19	17.98
Butler et al. 1997	Mud Creek at Highway 32, near Cortez	Ridgway's rail	Lotic	0.07	3.19	49.96
Casey 2005	Deerlick Creek	Ridgway's rail	Lotic	2.24	3.19	1.57
Casey 2005	Luscar Creek	Ridgway's rail	Lotic	0.33	3.19	10.79
Formation Environmental 2012	Crow Creek - 1A	Ridgway's rail	Lotic	0.80	3.19	4.40

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	3.19	4.36
Formation Environmental 2012	Crow Creek - CC150	Ridgway's rail	Lotic	1.04	3.19	3.37
Formation Environmental 2012	Crow Creek - CC350	Ridgway's rail	Lotic	1.16	3.19	3.02
Formation Environmental 2012	Crow Creek - CC75	Ridgway's rail	Lotic	1.19	3.19	2.96
Formation Environmental 2012	Deer Creek	Ridgway's rail	Lotic	1.55	3.19	2.26
Formation Environmental 2012	Hoopes Spring – HS	Ridgway's rail	Lotic	0.24	3.19	14.36
Formation Environmental 2012	Hoopes Spring - HS3	Ridgway's rail	Lotic	0.54	3.19	6.55
Formation Environmental 2012	Sage Creek - LSV2C	Ridgway's rail	Lotic	0.45	3.19	7.86
Formation Environmental 2012	Sage Creek - LSV4	Ridgway's rail	Lotic	0.69	3.19	5.06
Formation Environmental 2012	South Fork Tincup Cr.	Ridgway's rail	Lotic	1.32	3.19	2.65
Hamilton and Buhl 2004	Lower East Mill Creek	Ridgway's rail	Lotic	1.32	3.19	2.67
McDonald and Strosher 1998	Elk R. above Cadorna Cr. (745)	Ridgway's rail	Lotic	6.30	3.19	0.56
McDonald and Strosher 1998	Fording R. above Swift Cr. (746)	Ridgway's rail	Lotic	0.23	3.19	15.57
Orr et al. 2012	Elk River 1	Ridgway's rail	Lotic	0.55	3.19	6.41
Orr et al. 2012	Elk River 12	Ridgway's rail	Lotic	2.67	3.19	1.32
Orr et al. 2012	Fording River 23	Ridgway's rail	Lotic	0.21	3.19	16.92
Orr et al. 2012	Michel Creek 2	Ridgway's rail	Lotic	0.28	3.19	12.37

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	3.19	9.77
Saiki and Lowe 1987	Volta Wasteway	Ridgway's rail	Lotic	1.03	3.19	3.41
Saiki et al. 1993	Mud Slough at Gun Club Road	Ridgway's rail	Lotic	1.37	3.19	2.57
Saiki et al. 1993	Salt Slough at the San Luis National Wildlife Refuge	Ridgway's rail	Lotic	0.43	3.19	8.22
Saiki et al. 1993	San Joaquin R. above Hills Ferry Road	Ridgway's rail	Lotic	0.36	3.19	9.83
Saiki et al. 1993	San Joaquin R. at Durham Ferry State Recreation Area	Ridgway's rail	Lotic	0.75	3.19	4.71

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).

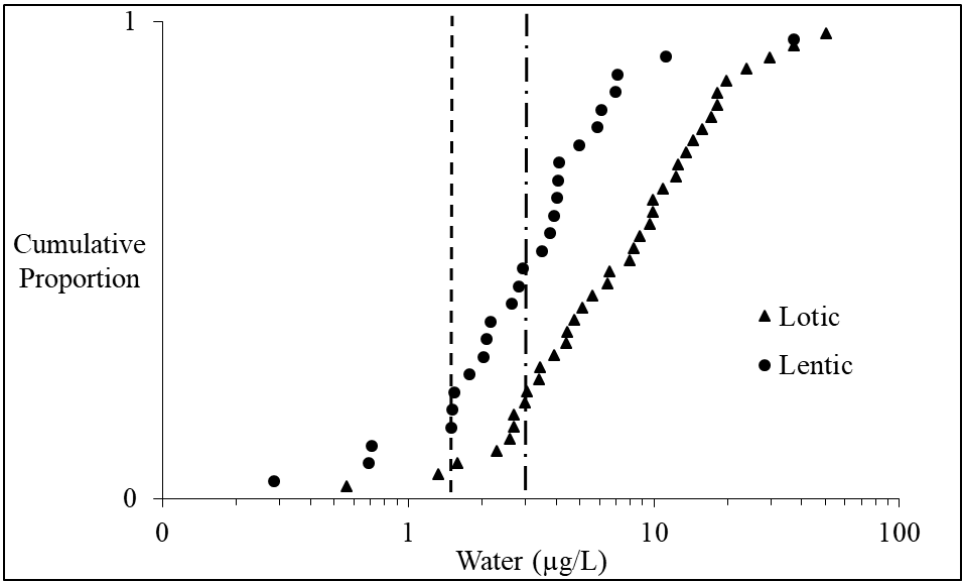

Figure 5-3. Probability distribution of the water column concentrations translated from the bird egg criterion element at the 26 lentic and 39 lotic aquatic sites from U.S. EPA (2016a). Dashed and dash-dot lines show the 20th percentiles of the lentic and lotic distributions, respectively.

As in the 2016 national aquatic life criterion for selenium, the 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), the EPA's 2016 national selenium aquatic life water column criterion elements for lentic and lotic waters is expected to 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 (*Method for Translating Selenium Tissue Criterion Elements into Site-specific Water Column Criterion Elements for California, Version 2 December 2024*) and in Appendix K of U.S. EPA (2016a).

The 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. The EPA is finalizing 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, the 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 1.5 µg/L in lentic aquatic systems and 3.1 µg/L in lotic aquatic systems unless or until a site-specific water column criterion element is derived for a particular waterbody following the methodology does not exceed described in *Method for Translating Selenium Tissue Criterion Elements into Site-specific Water Column Criterion Elements for California, Version 2 December 2024*;
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 Final 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[5]**	**Intermittent Exposure[6]**
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)	1.5 µg/L in lentic aquatic systems 3.1 µg/L in lotic aquatic systems	$WQC_{int} = \frac{WQC_{30-day} - C_{bkgrnd}(1 - f_{int})}{f_{int}}$
Duration	Instantaneous measurement[7]	Instantaneous measurement[7]	Instantaneous measurement[7]	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 criterion elements are expressed as steady-state.
2. Fish egg-ovary supersedes any whole-body, muscle, or water column criterion elements for aquatic life when fish egg-ovaries are measured, except as noted in footnote 4. Bird egg supersedes water column criterion elements for aquatic-dependent wildlife when bird eggs are measured, except as noted in footnote 4. The bird tissue criterion element is independently applicable from and equivalent to the fish tissue criterion elements.
3. Fish whole-body or muscle tissue supersedes the water column criterion elements when both fish tissue and water concentrations are measured, except as noted in footnote 4.
4. Water column criterion elements are based on dissolved total selenium in water and are derived from fish tissue and bird tissue criterion elements via bioaccumulation modeling. When selenium inputs are increasing, water column criterion elements are the applicable criterion elements in the absence of steady-state condition fish tissue or bird tissue data.
5. The water column criterion element, which applies independently to the respective aquatic life and aquatic-dependent wildlife uses, is applicable for all CWA purposes and consists of a water column value of 1.5 µg/L in lentic aquatic systems and 3.1 µg/L in lotic aquatic systems unless or until a site-specific water column criterion element is derived for a particular waterbody following the methodology described in *Method for Translating Selenium Tissue Criterion Elements into Site-specific Water Column Criterion Elements for California, Version 2 December 2024*. This publication is incorporated by reference into this section with the approval of the Director of the Federal Register under 5 U.S.C. 552(a) and 1 CFR part 5 1. All approved material is available at EPA, OW Docket, EPA West, Room 3334, 1301 Constitution Ave., NW, Washington, DC, 20004, (202) 566-2426. It is also available for inspection at the National Archives and Records Administration (NARA). For information on the availability of this material at NARA, call 202-741 -6030 or go to www.archives.gov/federal-register/cfr/ibr-locations.html.
6. Where $WQC_{30\text{-}day}$ is the applicable water column monthly criterion element, 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).
7. 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 the 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 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. The 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. The 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.

Additionally, given that the chronic selenium criterion is derived to be protective of the entire aquatic community, fish tissue and bird tissue elements were independently derived and an analysis was conducted to determine if the fish tissue elements would be protective of aquatic-dependent birds or vice versa (Appendix D). In this analysis, measured bird and fish tissue selenium concentrations were used to indicate if one criterion element would be protective of the other. In the analysis presented in Appendix D, the proportion of sites where both bird and fish attain their respective criterion is higher in lotic sites than lentic sites, and the proportion of sites where birds attain but fish do not is higher in lentic sites than in lotic sites. It is unclear if these lentic-lotic differences represent a general result or are unique to these studies. Despite these differences, the general result that the bird egg criterion will most likely be met so long as the

fish tissue criterion is attaining, but the fish tissue criterion will not necessarily be met if the bird criterion is attaining, applies to both lentic and lotic waterbodies. However, as summarized in Part 1 of the TSD, the EPA derived the bird tissue criterion element in order to protect aquatic-dependent wildlife under California's wildlife designated uses. The national selenium criterion does not incorporate protections for aquatic-dependent wildlife, and the fish tissue criterion was not intended to ensure the protection of aquatic-dependent wildlife. Therefore, to ensure that both aquatic life and aquatic-dependent wildlife in California are protected, the EPA determined that the fish tissue and bird tissue criterion values should be stand-alone elements and that one should not override the other.

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 final 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). For development of a site-specific criteria following a PBA approach, using either the final 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 (*Method for Translating Selenium Tissue Criterion Elements into Site-specific Water Column Criterion Elements for California, Version 2, December 2024;* U.S. EPA 2024a). Both the performance-based approach (*Method for Translating Selenium Tissue Criterion Elements into Site-specific Water Column Criterion Elements for California, Version 2 December 2024;* U.S. EPA 2024a) and Appendix K in U.S. EPA (2016a) provide information on the data necessary to derive a site-specific 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.

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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. Table A-1 summarizes the three mallard studies that were combined to calculate the mallard egg EC10 in Part 4.2.1. 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. 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.

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: 15.9	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: 12.1 LOEC: 24.5	Stanley et al. 1996

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

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: >10[b]	NOEC: >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, 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	
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 (dw) LOEC: 6.5 (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.	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 and Hoffman 1996

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	
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
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
Black necked stilt; *Himantopus mexicanus* Eggs and nestlings of field-exposed adults	Collected eggs from Kesterson Res., Salton Sea, Tulare Basin, and Volta State Wildlife Area, CA.	n/a (field collected eggs)	Naturally occurring Se	Field exposure. Lifetime maternal exposure.	Nest inviability Egg inviability	n/a	EC_{10}: 16.0 (nest inviability) EC_{10}: 20.9-31.0 (egg inviability)	Adams et al. 2003; Skorupa 1998b

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	
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: >13.3 (Maximum 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:> 4.75[e]	Walls et al. 2015
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	NOEC: 22 Point of downward inflection of quadratic curve.[f]	Harding 2008

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	
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 include 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-contaminants. No combinations of potential contaminants could explain the differences in hatching success.
[f] Not statistically significant.

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: >20.7[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	Survival Weight	NOEC: 20[a] LOEC: 40[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	Survival Weight	NOEC: 20[a] LOEC: 40[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 Food consumption	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 from the initial 21-week 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*)

As described in Part 5.4.2.1, the 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, the 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 for the same food type were matched to an aquatic organism or bird measurement, the median measurement for that food type was calculated. If multiple lower trophic level measurements for two or more food types were matched to an aquatic organism or bird measurement, the median measurement of those food types was calculated. 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 (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.

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

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 regression 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.

The following tables (Table B-1 through Table B-8) list the paired data used to calculate bird *TTF* values and the linear regression model results. The "Median *TTF*" reported at the bottom of each of these figures is the species level *TTF* shown in Table 5-3. These species level *TTF* are incorporated into the food web models described below, which were used to calculate the bird composite *TTF*s shown in Table 5-5.

Table B-1. American Avocet 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

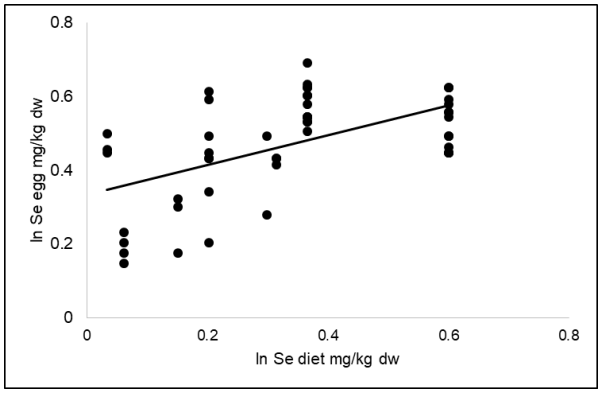
Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
							**Median *TTF***	1.44
							Adjusted r^2	0.29
							F	21.38
							df	48
							P	<0.001

Table B-2. American Coot Trophic Transfer Factor (*TTF*).

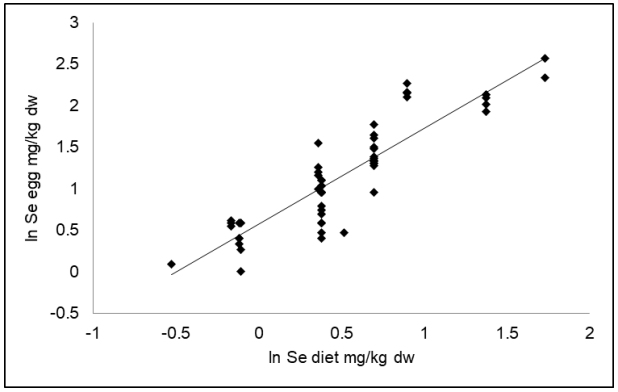
Rows with data pairs that were removed during outlier analysis are identified with bold and italics.

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
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

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	3.9	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
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

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	2.1	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 WMA	0.78	0.8	1.13	0.2	0.85	1.8	2.13
Rinella et al. 1994	Ft. Boise WMA	0.78	0.8	1.13	0.2	0.85	1.73	2.05

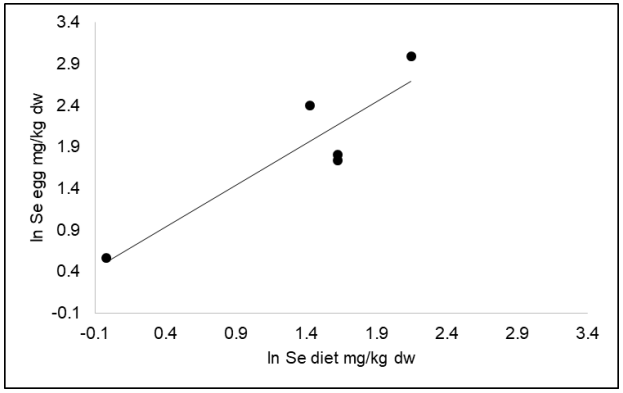
Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
Rinella et al. 1994	Ft. Boise WMA	0.78	0.8	1.13	0.2	0.85	1.85	2.19



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

Table B-3. Cinnamon Teal 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

Table B-4. Eared Grebe Trophic Transfer Factor (*TTF*).

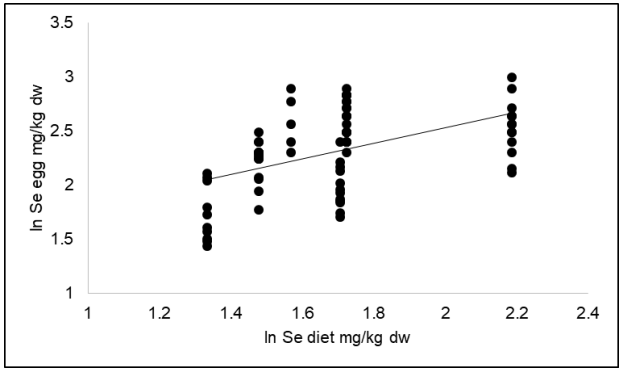
Because eared grebes eat a 100% invertebrate diet, all paired invertebrate-egg measurements were used, regardless of whether a paired plant measurement was available.

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

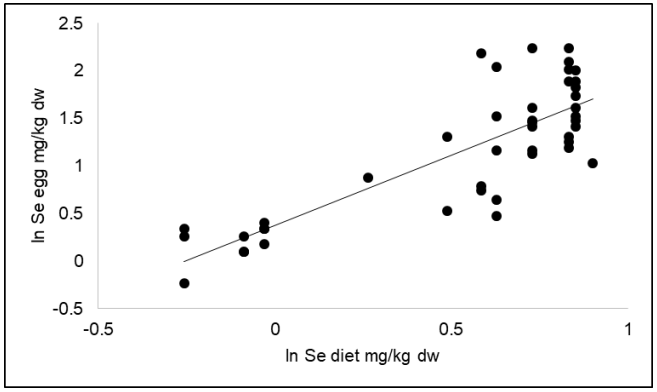
Table B-5. Gadwall Trophic Transfer Factor (*TTF*).

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

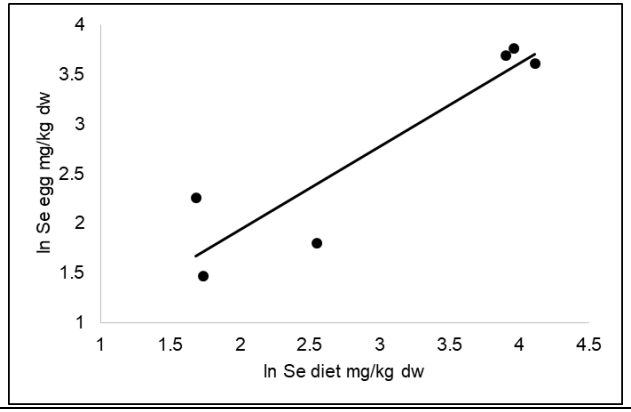


Median *TTF*	1.78
Adjusted r^2	0.66
F	86.17
df	42
P	<0.001

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



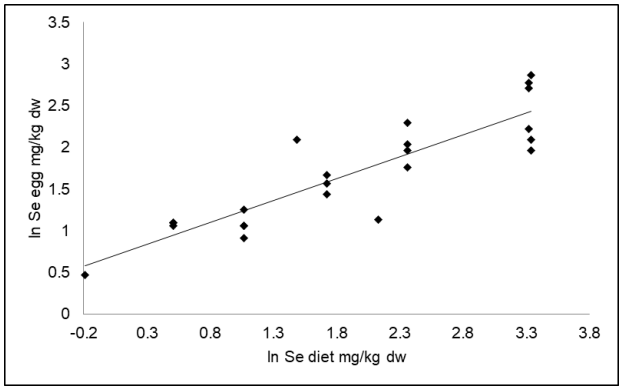
Median *TTF*	0.78
Adjusted r^2	0.81
F	22.25
Df	4
P	0.009

Table B-7. Red-Winged Blackbird 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	0.17	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

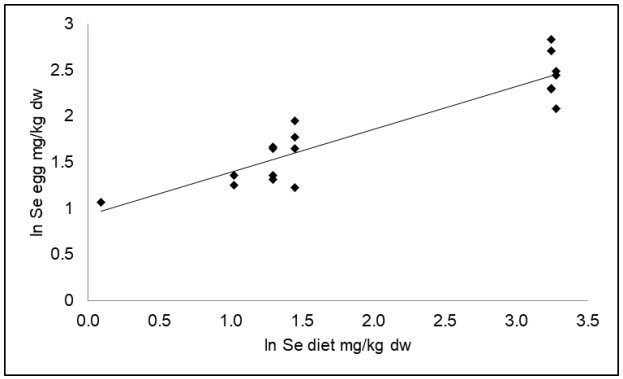


Median *TTF*	0.86
Adjusted r^2	0.77
F	73.92
df	21
P	<0.001

Table B-8. Yellow-Headed Blackbird 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* values 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 the World web site: (https://birdsoftheworld.org/bow/home), and following the methods described in Part 5.4.2.1. Calculations were made using different combinations of (Equation 5-2) and (Equation 5-3), shown below as Appendix Equations B-1 and B-2, 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 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 i^{th} species at a particular trophic level

W_i = the proportion of the i^{th} species consumed

Invertebrate, fish, and bird *TTF*s used in the food web model calculations are from Table 5-1, Table 5-2, and Table 5-3, respectively. $TTF^{composite}$ for these eight bird species are listed in Table 5-5.

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 and Mowbray 2020). Based on this information, the American coot diet was modeled as consisting of 80% aquatic plants, 8% insects, 6% mollusks, 4% small crustaceans, and 2% crayfish.

The American coot species *TTF* value, based on a diet of 80% plants and 20% animals, is 1.89 (Table B-2). The invertebrate *TTF*s are as follows: 2.14 (insects), 4.29 (mollusks), 1.32 (small crustaceans – median amphipod and copepod), and 1.46 (crayfish), respectively (Table 5-1).The American coot $TTF^{composite}$ is 2.48 and 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 red-winged blackbird diet 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 2020). 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.

The red-winged blackbird species *TTF* value, based on a diet of 17% aquatic plants and 83% aquatic insects, is 0.86 (Table B-7). The aquatic insect *TTF*, calculated as the median of all insect orders, is 2.14 (Table 5-1). The red-winged blackbird $TTF^{composite}$ is 1.67 and is calculated as follows.

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

Migratory Species

The 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, the 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. 2020). 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 this information, 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.

The American avocet species *TTF* value, based on a diet of 13% plants and 87% animals, is 1.44 (Table B-1). The invertebrate *TTF*s are as follows: 1.90 (Chironomidae), 1.48 (Corixidae), 4.29 (mollusks), 2.38 (mayflies), 2.14 (insects), 1.29 (annelids), 1.41 (crustaceans), and 1.89 (all invertebrates), respectively (Table 5-1). The American avocet $TTF^{composite}$ is 2.61 and is calculated as follows.

$$TTF^{composite} = [1.44 \times 0.13] + \left[(1.44 \times \left((1.90 \times 0.55) + (1.48 \times 0.10) + (2.38 \times 0.01) + (2.14 \times 0.10) + (1.29 \times 0.03) + (4.29 \times 0.03) + (1.41 \times 0.02) + (1.89 \times 0.03)\right)\right] = \mathbf{2.61}$$

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 2020). Based on this information, 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 cinnamon teal species *TTF* value, based on a diet of 42% plants and 58% animals, is 1.79 (Table B-3). The $TTF^{composite}$ is calculated using the following invertebrate *TTF*s: 1.90 (Chironomidae), 4.29 (mollusks), 1.48 (Corixidae), 0.74 (Cladocera), 2.14 (insects), 2.425 (Odonata), and 1.89 (all invertebrates), respectively (Table 5-1). The cinnamon teal $TTF^{composite}$ is 3.04 and 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. 2020). In saline lakes, their diet consists 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 this information, 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 eared grebe species *TTF* value, based on a 100% animal diet, is 2.00 (Table B-4). The invertebrate *TTF*s are as follows: 1.41 (crustaceans), 1.90 (Diptera), 1.48 (Corixidae), 2.14 (insects), and 1.29 (annelids), respectively (Table 5-1). The $TTF^{composite}$ for eared grebe is 3.15 and is calculated as follows.

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

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. 2020). Plants eaten include filamentous algae, water milfoil, widgeon grass, duckweed, and pondweed, depending on availability. Animal food items consist of midge larvae, aphids, snails, and beetle larvae (Leschack et al. 2020). Based on this information, the gadwall diet was modeled as consisting of 75% plants, 11% chironomids, 7% insects (beetles), 5% small crustaceans, and 2% snails.

The gadwall species *TTF* value, based on a diet of 75% plants and 25% animals, is 1.78 (Table B-5). The invertebrate *TTF*s are as follows: 1.90 (chironomids), 2.14 (insects - beetles), 1.32 (small crustaceans – median amphipod and copepod), and 4.29 (mollusks), respectively (Table 5-1). The $TTF^{composite}$ for gadwall is 2.24 and is calculated as follows.

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

Pied-Billed Grebe

The diet of pied-billed grebes includes decapod crustaceans, especially crayfish, aquatic insects, and fishes. In some areas, prey items also include 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 2020). Based on the dietary information, 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 33% crayfish, 13% beetles, 8% corixids, 2% other invertebrates, 1% mollusks, 1% dragonflies, and 42% fish.

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). The pied-billed grebe species *TTF* value, based on available bird egg-fish paired data reported in Byron and Santolo (2010, 2014); Byron et al. (2012) for two sites in the Newport Bay, CA watershed, is 0.78 (Table B-6). 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.

The invertebrate portion of their diet was modeled using *TTF*s of 1.46 (crayfish), 2.14 (insects - beetles), 1.48 (Corixidae), 1.89 (all invertebrates), 4.29 (mollusks), and 1.97 (dragonflies) from Table 5-1. The piscivorous portion of their diet was modeled using fish $TTF^{composite}$ values for representative fish taxa from Table B-14. For modeling purposes, it was assumed the pied billed grebe consumed equal proportions of carp and minnow species, catfish, sticklebacks, sunfish (Lepomis sp.), sculpin, and killifish (Muller and Storer 2020), with

corresponding $TTF^{composite}$ of 1.58 (Cypriniformes), 1.54 (Siluriformes), 2.47 (Gasterosteiformes), 2.15 (*Lepomis*), 2.69 (*Cottus*), and 2.44 (*Fundulus*).

The $TTF^{composite}$ for pied billed grebe is 1.47 and is calculated as follows.

$$TTF^{composite} = \left[0.78 \times \left((1.46 \times 0.33) + (2.14 \times 0.13) + (1.48 \times 0.08) + (1.89 \times 0.02) + (4.29 \times 0.01) + (1.97 \times 0.01)\right)\right] + \left[0.78 \times \left((1.58 \times 0.07) + (1.54 \times 0.07) + (2.47 \times 0.07) + (2.15 \times 0.07) + (2.69 \times 0.07) + (2.44 \times 0.07)\right)\right] = \mathbf{1.47}$$

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 2020). Based on the dietary information listed above, the yellow-headed blackbird diet was modeled as consisting of 25% plants and 75% insects.

The yellow-headed blackbird species *TTF* value, based on a diet of 25% plants and 75% animals, is 1.04 (Table B-8). The aquatic insect *TTF*, calculated as the median of all insect orders, is 2.14 (Table 5-1). The $TTF^{composite}$ for yellow-headed blackbird is 1.93 and 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

Composite *TTFs* were also calculated for 8 additional T&E bird species that did not have species specific empirical bird *TTF* values: American dipper, brown pelican, bald eagle, Ridgway's rail, light-footed Ridgway's rail, Yuma Ridgway's rail, black rail, and least tern. Surrogate species were selected from the group of eight bird species with sufficient data to calculate species level *TTF* values, based on similarity in dietary composition and if possible taxonomic relatedness (within same order) to account for similarities in dietary composition and physiology/life history, all of which may influence the accumulation of selenium.

After determining an appropriate surrogate species, the surrogate species-level *TTF* value was calculated following the same methodology 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 final surrogate species-level *TTF*. Table B-9 through Table B-13 list paired data from the applicable surrogate bird species, as well as species-level *TTF* values following dietary reweighting for the respective T&E bird species. The original pied-billed grebe *TTF* was used as a surrogate for the T&E bird species whose were largely or entirely piscivorous, because available paired data for pied-billed grebe only included bird egg and fish selenium data. Methods and data requirements for the calculation of species level *TTF*s for re-weighted diets were the same as previously described for species with empirically measured *TTF*s. Finally, species level *TTF*s for T&E species were incorporated into the food web models described below, following methods described in Part 5.4.2.1, which were used to calculate the composite *TTF*s for T&E species listed in Table 5-6.

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

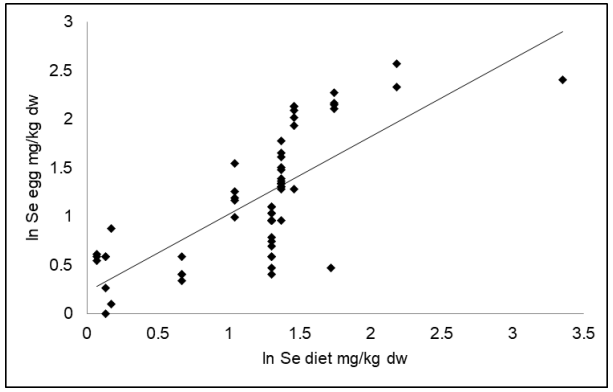
Rows with data pairs that were removed during outlier analysis are identified with bold and italics.

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

Table B-10. Black Rail Trophic Transfer Factor (*TTF*) after Reweighting Surrogate Species American Coot Diet to a 13% Plant and 87% Animal Diet.

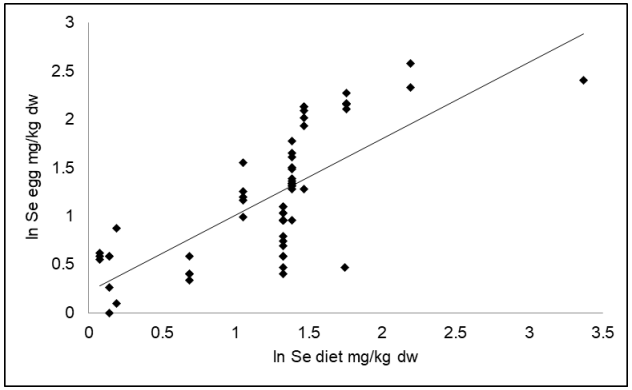
Rows with data pairs that were removed during outlier analysis are identified with bold and italics.

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	0.13	31.75	0.87	29.02	11.10	0.38
Butler et al. 1995	TT	0.41	0.13	1.33	0.87	1.21	1.10	0.91
Butler et al. 1995	TT	0.41	0.13	1.33	0.87	1.21	2.40	1.99
Butler et al. 1997	DCP1	1.45	0.13	6.45	0.87	5.80	8.20	1.41
Butler et al. 1997	***DCP1***	***1.45***	***0.13***	***6.45***	***0.87***	***5.80***	***18.00***	***3.10***
Butler et al. 1997	DCP1	1.45	0.13	6.45	0.87	5.80	8.60	1.48
Butler et al. 1997	DCP1	1.45	0.13	6.45	0.87	5.80	9.70	1.67
Butler et al. 1997	DCP1	1.45	0.13	6.45	0.87	5.80	8.70	1.50
Butler et al. 1997	MNP2	3.85	0.13	4.40	0.87	4.33	8.10	1.87
Butler et al. 1997	MNP2	3.85	0.13	4.40	0.87	4.33	3.60	0.83
Butler et al. 1997	MNP2	3.85	0.13	4.40	0.87	4.33	6.90	1.59
Butler et al. 1997	MNP2	3.85	0.13	4.40	0.87	4.33	8.40	1.94
Butler et al. 1997	MNP2	3.85	0.13	4.40	0.87	4.33	7.50	1.73
Butler et al. 1997	MNP2	3.85	0.13	4.40	0.87	4.33	8.40	1.94
Lambing 1988	***7***	***0.47***	***0.13***	***6.50***	***0.87***	***5.72***	***1.40***	***0.24***
Lambing 1988	7	0.47	0.13	6.50	0.87	5.72	1.60	0.28
Lambing 1988	***7***	***0.47***	***0.13***	***6.50***	***0.87***	***5.72***	***1.10***	***0.19***
Lambing et al. 1994	B-21	1.42	0.13	4.38	0.87	4.00	5.90	1.48
Lambing et al. 1994	B-21	1.42	0.13	4.38	0.87	4.00	2.60	0.65
Lambing et al. 1994	B-21	1.42	0.13	4.38	0.87	4.00	5.00	1.25
Lambing et al. 1994	B-21	1.42	0.13	4.38	0.87	4.00	3.80	0.95
Lambing et al. 1994	B-21	1.42	0.13	4.38	0.87	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	0.13	4.38	0.87	4.00	3.60	0.90
Lambing et al. 1994	B-21	1.42	0.13	4.38	0.87	4.00	3.90	0.98
Lambing et al. 1994	B-21	1.42	0.13	4.38	0.87	4.00	3.80	0.95
Lambing et al. 1994	B-21	1.42	0.13	4.38	0.87	4.00	3.70	0.93
Lambing et al. 1994	B-21	1.42	0.13	4.38	0.87	4.00	4.40	1.10
Lambing et al. 1994	B-21	1.42	0.13	4.38	0.87	4.00	3.70	0.93
Lambing et al. 1994	B-21	1.42	0.13	4.38	0.87	4.00	3.80	0.95
Lambing et al. 1994	B-21	1.42	0.13	4.38	0.87	4.00	3.80	0.95
Lambing et al. 1994	B-21	1.42	0.13	4.38	0.87	4.00	4.00	1.00
Lambing et al. 1994	B-21	1.42	0.13	4.38	0.87	4.00	4.50	1.13
Lambing et al. 1994	B-22	1.00	0.13	3.15	0.87	2.87	3.50	1.22
Lambing et al. 1994	B-22	1.00	0.13	3.15	0.87	2.87	3.30	1.15
Lambing et al. 1994	B-22	1.00	0.13	3.15	0.87	2.87	2.70	0.94
Lambing et al. 1994	B-22	1.00	0.13	3.15	0.87	2.87	4.70	1.64
Lambing et al. 1994	B-22	1.00	0.13	3.15	0.87	2.87	3.20	1.11
Lambing et al. 1994	B-26	0.77	0.13	4.20	0.87	3.75	1.50	0.40
Lambing et al. 1994	B-26	0.77	0.13	4.20	0.87	3.75	1.60	0.43
Lambing et al. 1994	B-26	0.77	0.13	4.20	0.87	3.75	2.00	0.53
Lambing et al. 1994	B-26	0.77	0.13	4.20	0.87	3.75	2.60	0.69
Lambing et al. 1994	B-26	0.77	0.13	4.20	0.87	3.75	3.00	0.80
Lambing et al. 1994	B-26	0.77	0.13	4.20	0.87	3.75	2.80	0.75
Lambing et al. 1994	B-26	0.77	0.13	4.20	0.87	3.75	2.20	0.59
Lambing et al. 1994	B-26	0.77	0.13	4.20	0.87	3.75	2.60	0.69
Lambing et al. 1994	B-26	0.77	0.13	4.20	0.87	3.75	2.60	0.69
Lambing et al. 1994	B-26	0.77	0.13	4.20	0.87	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	0.13	4.20	0.87	3.75	2.60	0.69
Lambing et al. 1994	B-26	0.77	0.13	4.20	0.87	3.75	2.60	0.69
Lambing et al. 1994	B-26	0.77	0.13	4.20	0.87	3.75	2.10	0.56
Lambing et al. 1994	B-26	0.77	0.13	4.20	0.87	3.75	3.00	0.80
Lambing et al. 1994	B-26	0.77	0.13	4.20	0.87	3.75	1.80	0.48
Lambing et al. 1994	B-26	0.77	0.13	4.20	0.87	3.75	1.80	0.48
Peterson et al. 1991	3	4.64	0.13	9.62	0.87	8.97	10.30	1.15
Peterson et al. 1991	3	4.64	0.13	9.62	0.87	8.97	13.10	1.46
Rinella and Schuler 1992	N Malheur	0.56	0.13	2.20	0.87	1.99	1.80	0.91
Rinella and Schuler 1992	N Malheur	0.56	0.13	2.20	0.87	1.99	1.50	0.75
Rinella and Schuler 1992	N Malheur	0.56	0.13	2.20	0.87	1.99	1.40	0.70
Rinella and Schuler 1992	N Malheur	0.56	0.13	2.20	0.87	1.99	1.50	0.75
Rinella and Schuler 1992	S Malheur	0.82	0.13	1.20	0.87	1.15	1.30	1.13
Rinella and Schuler 1992	S Malheur	0.82	0.13	1.20	0.87	1.15	1.00	0.87
Rinella and Schuler 1992	S Malheur	0.82	0.13	1.20	0.87	1.15	1.80	1.56
Rinella and Schuler 1992	S Malheur	0.82	0.13	1.20	0.87	1.15	1.80	1.56
Rinella et al. 1994	Ft. Boise WMA	0.78	0.13	1.13	0.87	1.08	1.80	1.67

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	0.78	0.13	1.13	0.87	1.08	1.73	1.60
Rinella et al. 1994	Ft. Boise WMA	0.78	0.13	1.13	0.87	1.08	1.85	1.71



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

Table B-11. Light-Footed Ridgeway's Rail and Yuma Rail Trophic Transfer Factor (*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.

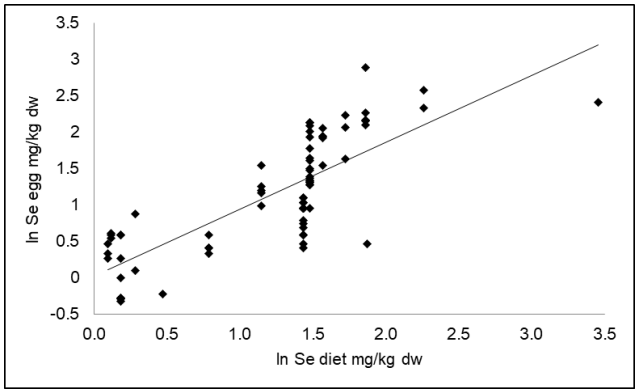
Study	Site	Plant Se (mg/kg)	Plant Diet Prop.	Invert. Se (mg/kg)	Invert Diet Prop.	Diet Se (mg/kg)	Egg Se (mg/kg)	*TTF*
Butler et al. 1991	7	n/a	0.0	31.75	1.0	31.75	11.1	0.35
Butler et al. 1995	TT	n/a	0.0	1.33	1.0	1.33	1.1	0.83
Butler et al. 1995	TT	n/a	0.0	1.33	1.0	1.33	2.4	1.81
Butler et al. 1997	DCP1	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



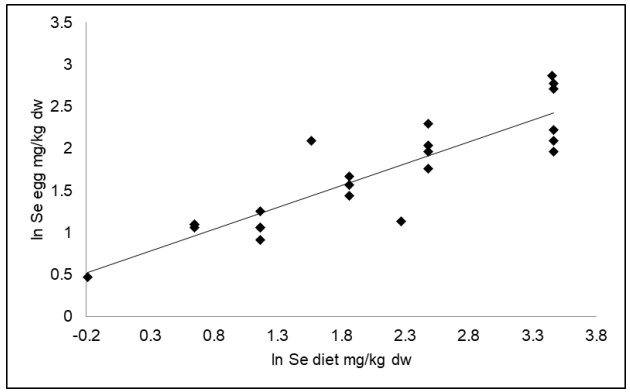
Median *TTF*	0.90
Adjusted r^2	0.60
F	120.8
df	78
P	<0.001

Table B-12. American Dipper. Bird Egg to Diet (*TTF*) after Reweighting Surrogate Species Red-Winged Blackbird to a 100% Animal Diet.

Because this species eats a 100% animal diet, all paired animal-egg measurements were used. 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	17.6	0.55
Butler et al. 1993	LP4	n/a	0.0	3.20	1.0	3.20	2.9	0.91
Butler et al. 1993	LP4	n/a	0.0	3.20	1.0	3.20	2.9	0.91
Butler et al. 1993	LP4	n/a	0.0	3.20	1.0	3.20	2.5	0.78
Butler et al. 1993	LP4	n/a	0.0	3.20	1.0	3.20	3.5	1.09
Butler et al. 1994	MKP	n/a	0.0	32.00	1.0	32.00	8.1	0.25
Butler et al. 1994	MKP	n/a	0.0	32.00	1.0	32.00	7.1	0.22
Butler et al. 1994	MKP	n/a	0.0	32.00	1.0	32.00	16	0.50
Butler et al. 1994	MKP	n/a	0.0	32.00	1.0	32.00	9.2	0.29
Butler et al. 1994	MKP	n/a	0.0	32.00	1.0	32.00	15	0.47
Butler et al. 1995	DD	n/a	0.0	0.83	1.0	0.83	1.6	1.94
Butler et al. 1991	10	n/a	0.0	4.80	1.0	4.80	8.1	1.69
Butler et al. 1991	***10***	n/a	0.0	***4.80***	1.0	***4.80***	***8.6***	***1.79***
Butler et al. 1997	CHP	n/a	0.0	12.00	1.0	12.00	7.7	0.64
Butler et al. 1997	CHP	n/a	0.0	12.00	1.0	12.00	5.8	0.48
Butler et al. 1997	CHP	n/a	0.0	12.00	1.0	12.00	9.9	0.83
Butler et al. 1997	CHP	n/a	0.0	12.00	1.0	12.00	7.1	0.59
Butler et al. 1997	DCP1	n/a	0.0	6.45	1.0	6.45	4.2	0.65
Butler et al. 1997	DCP1	n/a	0.0	6.45	1.0	6.45	5.3	0.82
Butler et al. 1997	DCP1	n/a	0.0	6.45	1.0	6.45	4.8	0.74

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	LCHP1	n/a	0.0	1.91	1.0	1.91	2.9	1.52
Butler et al. 1997	LCHP1	n/a	0.0	1.91	1.0	1.91	3	1.57
Butler et al. 1997	LCHP1	n/a	0.0	1.91	1.0	1.91	3	1.57
Butler et al. 1997	***WCP***	**n/a**	**0.0**	***9.70***	**1.0**	***9.70***	***2.1***	***0.22***
Butler et al. 1997	***WCP***	**n/a**	**0.0**	***9.70***	**1.0**	***9.70***	***2.8***	***0.29***
Butler et al. 1997	WCP	n/a	0.0	9.70	1.0	9.70	3.1	0.32



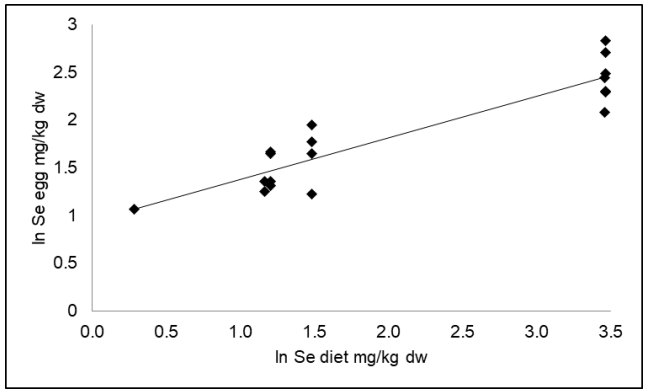
Median *TTF*	0.74
Adjusted r^2	0.77
F	74.15
df	21
P	<0.001

Table B-13. American Dipper. Bird Egg to Diet (*TTF*) after Reweighting Surrogate Species Yellow-Headed Blackbird to a 100% Animal Diet.

Because this species eats a 100% animal diet, all paired animal-egg measurements were used. 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	8	0.25
Butler et al. 1991	7	n/a	0.0	31.75	1.0	31.75	11.5	0.36
Butler et al. 1994	MKP	n/a	0.0	32.00	1.0	32.00	12	0.38
Butler et al. 1994	MKP	n/a	0.0	32.00	1.0	32.00	9.9	0.31
Butler et al. 1994	MKP	n/a	0.0	32.00	1.0	32.00	10	0.31
Butler et al. 1994	MKP	n/a	0.0	32.00	1.0	32.00	15	0.47
Butler et al. 1994	MKP	n/a	0.0	32.00	1.0	32.00	17	0.53
Butler et al. 1997	MNP2	n/a	0.0	4.40	1.0	4.40	7	1.59
Butler et al. 1997	MNP2	n/a	0.0	4.40	1.0	4.40	5.2	1.18
Butler et al. 1997	MNP2	n/a	0.0	4.40	1.0	4.40	3.4	0.77
Butler et al. 1997	MNP2	n/a	0.0	4.40	1.0	4.40	5.9	1.34
Butler et al. 1993	LP4	n/a	0.0	3.20	1.0	3.20	3.9	1.22
Butler et al. 1993	LP4	n/a	0.0	3.20	1.0	3.20	3.5	1.09
Butler et al. 1993	R1	n/a	0.0	3.33	1.0	3.33	3.9	1.17
Butler et al. 1993	R1	n/a	0.0	3.33	1.0	3.33	5.3	1.59
Butler et al. 1993	R1	n/a	0.0	3.33	1.0	3.33	5.2	1.56
Butler et al. 1993	R1	n/a	0.0	3.33	1.0	3.33	3.7	1.11
Butler et al. 1995	TT	n/a	0.0	1.33	1.0	1.33	2.9	2.19
Butler et al. 1995	***TT***	***n/a***	***0.0***	***1.33***	***1.0***	***1.33***	***4.8***	***3.62***

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*	1.10
Adjusted r^2	0.83
F	81.28
df	16
P	<0.001

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

This section describes the calculation of $TTF^{composite}$ values for the eight T&E bird species with measured TTF of surrogate species using data listed in the preceding tables. $TTF^{composite}$ value were calculated from food webs modeled using information from the Cornell Lab of Ornithology Birds of the World web site: (https://birdsoftheworld.org/bow/home), and other published data sources following the methods described in Part 5.4.2.1. $TTF^{composite}$ values for these species are listed in Table 5-6.

Composite *TTF* Results

American Dipper

American dippers consume a wide range of aquatic insects, primarily benthic macroinvertebrate larvae such as mayflies (Ephemeroptera), caddisflies (Trichoptera), stoneflies (Plecoptera), and Dipterans (Kingery and Wilson 2020; U.S. FWS 2017). 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 (Kingery and Wilson 2020). 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). American dippers tend to consume smaller fish, and fish eggs, primarily salmonid species and sculpins (Kingery and Wilson 2020). 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). The piscivorous portion of their diet was modeled as 75% salmonids and 25% sculpins.

An empirical *TTFs* was not available for American dipper, so the *TTF* for this species of concern was calculated from closely related surrogate species with empirical bird *TTFs*. The American dipper *TTF* value, calculated as the average of the two species (red-winged blackbird and yellow-headed blackbird) from the same order (Passeriformes) based on a 100% animal diet, was 0.92 (Table B-12 and Table B-13). The aquatic insect *TTF*, calculated as the median of all insect orders, is 2.14 (Table 5-1). The fish $TTF^{composite}$ values are 2.33 (Salmonidae) and 2.69

(*Cottus*) (Table B-14). The $TTF^{composite}$ for American dipper is 2.03 for the low fish (22%) scenario and 2.08 for the high fish (42%) scenario, and they are calculated as follows.

Low fish diet scenario:

$$TTF^{composite} = [0.92 \times (2.14 \times 0.78)] + \left[0.92 \times \left((2.33 \times 0.165) + (2.69 \times 0.055)\right)\right] = \mathbf{2.03}$$

High fish diet scenario:

$$TTF^{composite} = [0.92 \times 2.14 \times 0.58] + \left[0.92 \times \left((2.33 \times 0.315) + (2.69 \times 0.105)\right)\right] = \mathbf{2.08}$$

Brown Pelican

Along the California coast, brown pelicans are dependent on small, surface schooling fish, such as anchovy (*Engraulis mordax*) and Pacific sardines (*Sardinops sagax*). For example, in the Salton Sea, the Brown pelican diet has been reported to consist of 90% northern anchovy during the breeding season (U.S. FWS 2017). Based on this information, the dietary composition of brown pelicans in California was modeled as 75% northern anchovy and 25% Pacific sardines, based on the assumption that northern anchovies comprise a larger proportion of their diet than Pacific sardines, but slightly less across the state than observed in the Salton Sea.

An empirical bird *TTF* was not available for brown pelican, so the species level *TTF* of 0.78 based on the piscivorous pied billed grebe was used (Table B-6). Composite *TTF*s for the fish species in the brown pelican diet are not included in (Table B-14), so $TTF^{composite}$ values were calculated for these species using dietary information from FishBase. The class-level (Actinopterygii) *TTF* of 1.21 was used in all of the modeled fish $TTF^{composite}$ calculations because no species level fish *TTF* values were available (Table 5-2).

Northern anchovy (https://www.fishbase.se/summary/Engraulis-mordax.html), and Pacific sardines (https://www.fishbase.se/summary/SpeciesSummary.php?ID=1477&AT=Pacific+sardine) primarily consume zooplankton, so their diets were modeled as consisting of 100% zooplankton. The $TTF^{composite}$ for Northern anchovy and Pacific sardines are calculated as follows.

$$TTF^{composite} = [1.21 \times (1.89 \times 1.0)] = \mathbf{2.29}$$

The $TTF^{composite}$ for brown pelican is 1.79 and is calculated as follows.

$$TTF^{composite} = [0.78 \times ((2.29 \times 0.75) + (2.29 \times 0.25))] = \mathbf{1.79}$$

Bald Eagle

Bald eagles are opportunistic foragers, with highly variable diets based on the availability of prey species. Bald eagles are primarily piscivores, showing a preference for fish when they are available, but will also often consume various other prey types, as well as carrion (Buehler 2020). 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. No mammalian $TTF^{composite}$ values were available, so the modeled diet was rescaled as 75.7% fish and 24.3% birds.

An empirical bird *TTF* was not available for bald eagle, so the species level *TTF* of 0.78 based on the piscivorous pied billed grebe was used (Table B-6), as it is the only empirically-derived *TTF* for a bird species with a largely piscivorous diet, and there are no empirically-derived *TTFs* for bird eating birds. Because they consume a wide range of fish species depending on local availability, the fish portion of their diet was modeled using the median $TTF^{composite}$ of 2.35 for all fish species shown in (Table B-14). Because they consume a wide range of bird species depending on local availability, the bird portion of their diet was modeled using the median $TTF^{composite}$ of 1.93 for all bird species (except for bald eagle) shown in (Table 5-5) and (Table 5-6). The $TTF^{composite}$ for the bald eagle is 1.75 and is calculated as follows.

$$TTF^{composite} = [0.78 \times (2.35 \times 0.757)] + [0.78 \times (1.93 \times 0.243)] = \mathbf{1.75}$$

Ridgway's Rail

The Ridgway's rail is an omnivorous species with a highly variable diet (Eddleman and Conway 2020). 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, 66% mollusks, 3% crabs, and 16% other invertebrates.

An empirical *TTF* for Ridgway's rail was not available, so the *TTF* for this species was calculated using paired data from the closely related American coot (also from the order Gruiformes). The Ridgway's rail species *TTF* value, based on a diet consisting of 15% plants and 85% animals is 0.96 (Table B-9). The invertebrate *TTF*s are as follows: 4.29 (mollusks), 1.46 (crabs – crayfish surrogate), and 1.89 (all invertebrates), respectively (Table 5-1). The $TTF^{composite}$ for Ridgway's rail is 3.19 and calculated as follows.

$$TTF^{composite} = [0.96 \times 0.15] + \left[0.96 \times \left((4.29 \times 0.66) + (1.41 \times 0.03) + (1.89 \times 0.16)\right)\right] = \mathbf{3.19}$$

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). U.S. FWS (2017) reported that light-footed Ridgway's rail consume a variety of salt marsh invertebrates, such as mussels, snails, fiddler and hermit crabs, fish, crayfish, isopods, and beetles. Prey fish species include the California killifish and flathead grey mullet (U.S. FWS 1985). U.S. FWS (2003) assumed that light-footed Ridgway's rail diet was 10% crayfish and 10% fish, leaving the remaining 80% to be aquatic invertebrates. Based on this information, the light-footed Ridgway's rail dietary composition was assumed to be 10% crayfish, 10% fish, and 80% other invertebrates.

An empirical-species *TTF* for light-footed Ridgway's rail was not available, so the TTF for this species was calculated using paired data from the closely related American coot (also from the order Gruiformes). The light-footed Ridgway's rail species *TTF* value, based on a diet consisting of 100% animals, is 0.90 (Table B-11). The invertebrate portion of their diet was modeled using *TTF*s of 1.46 (crayfish) and 1.89 (all invertebrates) from Table 5-1. The piscivorous portion of their diet was modeled as 50% California killifish and 50% flathead gray mullet. Composite *TTF*s were not available for either of these fish species in (Table B-14), so $TTF^{composite}$ values were calculated below.

Dietary information for California killifish (*Fundulus parvipinnis*) was obtained from NatureServe (https://explorer.natureserve.org/Taxon/ELEMENT_GLOBAL.2.103892/Fundulus_parvipinnis), which noted the species is an invertivore that feeds primarily on arthropods, as well as annelids, gastropods, and fish eggs. Based on this information, their diet was modeled as consisting of 100% invertebrates. The $TTF^{composite}$ for California killifish is calculated as follows, using a genus-level fish *TTF* of 1.27 (*Fundulus*) from (Table 5-2) and a dietary *TTF* of 1.89 (all invertebrates) from (Table 5-1).

$$TTF^{composite} = [1.27 \times (1.89 \times 1.0)] = \mathbf{2.40}$$

Dietary information for the flathead grey mullet was obtained from FishBase (https://www.fishbase.se/summary/Mugil-cephalus.html) which noted that the species consumes a mix of zooplankton, benthic invertebrates, algae, and detritus. Based on this information, the diet of flathead gray mullet was modeled as 50% zooplankton and 50% benthic invertebrates. The $TTF^{composite}$ for flathead grey mullet is calculated as follows, using a class-level fish *TTF* of 1.21 (Actinopterygii) from (Table 5-2) and dietary *TTF* of 1.89 (zooplankton) and 1.68 (benthic invertebrates – median *TTF* of all benthic crustaceans, benthic insects, and annelids) from (Table 5-1).

$$TTF^{composite} = [1.21 \times (1.89 \times 0.5)] + [1.21 \times (1.68 \times 0.5)] = \mathbf{2.16}$$

The $TTF^{composite}$ for light footed Ridgway's rail is 1.70 and is calculated as follows.

$$TTF^{composite} = \left[0.90 \times \left((1.89 \times 0.8) + (1.46 \times 0.1)\right)\right] + \left[0.90 \times \left((2.40 \times 0.05) + (2.16 \times 0.05)\right)\right] = \mathbf{1.70}$$

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). Based on this information, the Yuma Ridgway's rail diet was modeled as 95% crayfish and 5% other aquatic invertebrates, as U.S. FWS (2003) 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.

An empirical *TTF* for Yuma Ridgway's rail was not available, so the species *TTF* was calculated as 0.90 using paired data from the closely related American coot (also from the order Gruiformes), based on a diet consisting of 100% animals (Table B-11). The invertebrate *TTF*s are 1.46 (crayfish) and 1.89 (all invertebrates) from (Table 5-1). The $TTF^{composite}$ for Yuma Ridgway's rail is 1.33 and is calculated as follows.

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

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. (2020) 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. Based on this information, the black rail diet was modeled as 13% plant matter and 87% aquatic invertebrates.

An empirical *TTF* for black rail was not available, so the *TTF* was calculated using paired data from the closely related American coot (also from the order Gruiformes). The black rail species *TTF* value, based on a diet consisting of 13% plants and 87% animals, is 0.95 (Table

B-10). The invertebrate portion of the black rail diet was modeled using an all-invertebrate *TTF* of 1.89 (Table 5-1). The $TTF^{composite}$ for black rail is 1.69 and is calculated as follows.

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

Least Tern

Least tern consume a variety of shallow-bodied small fish species (<8 cm in length) (Atwood and Kelly 1984, Thompson et al. 2020). In California, the least tern diet is comprised of shallow-bodied small fish species (<8 cm) and smaller young of the year fish from larger species found in shallow waters such as estuaries, nearshore waters and river mouths (U.S. FWS 2017, 2020). Their diet has been known to include up to 50 different fish species (U.S. FWS 1985, 2017). The distributions of prey species based on dropped prey were surveyed at Alameda Point, in the San Francisco Bay (Elliot 2008), and at sites in Southern California (Atwood and Kelly 1984). The most abundant prey species were silversides, particularly topsmelt (*Atherinops affinis*) and jacksmelt (*A. californiensis*). The next most abundant prey species was the northern anchovy (*Engraulis mordax*). These three species were the most abundant in both studies, comprising around 75-80% of dropped prey items. Other relatively abundant species included Pacific herring (*Clupea pallasii*) in the San Francisco Bay region, and California killifish (*Fundulus parvipinnis*) at the Southern California sites (Atwood and Kelly 1984; Elliott 2008).

Based on this information in U.S. FWS (2017) report, the dietary composition of the least tern was assumed to be 100% fish. The least tern diet was modeled as 30% topsmelt, 30% jacksmelt, 20% northern anchovy, 10% Pacific herring, and 10% California killifish. Although California least-tern have been shown to consume up to 50 fish species, these five species represent the most abundant species in the California least tern diet, and are all small, relatively shallow bodied species that are regularly consumed.

An empirical *TTF* was not available for the least tern, so the surrogate species *TTF* value was the pied-billed grebe *TTF* of 0.78 (Table B-6), a piscivorous bird with a similar diet. Composite *TTF*s for the fish species modeled to comprise the least tern diet (topsmelt, jacksmelt, Pacific herring, northern anchovy, California killifish) are not available in Table B-14, so $TTF^{composite}$ values were calculated below.

Dietary information for topsmelt was obtained from FishBase (https://www.fishbase.de/summary/Atherinops-affinis.html), which notes adults tend to feed on zooplankton while juveniles will feed on algae and kelp fly larvae. Based on this information, the topsmelt diet was modeled as 50% zooplankton and 50% insects. No *TTF* for topsmelt was available, or for a closely related surrogate species, so a class-level (Actinopterygii) *TTF* of 1.21 was used to represent this species (Table 5-2). The *TTF* for zooplankton is 1.89 and the median *TTF* for all insect orders is 2.14 (Table 5-1). The $TTF^{composite}$ for topsmelt is calculated as follows.

$$TTF^{composite} = [1.21 \times (1.89 \times 0.50)] + [1.21 \times (2.14 \times 0.50)] = \mathbf{2.44}$$

Dietary information for jacksmelt (https://www.fishbase.se/summary/Atherinopsis-californiensis.html) and Pacific herring (https://www.fishbase.se/summary/1520) were obtained from FishBase, which note these species feed predominantly on zooplankton. Based on this information, the jacksmelt and Pacific herring diets were calculated as 100% zooplankton, using a *TTF* of 1.89 (Table 5-1). No *TTF* for jacksmelt or Pacific herring were available, or for a closely related surrogate species, so a class-level (Actinopterygii) *TTF* of 1.21 was used to represent this species (Table 5-2). The $TTF^{composite}$ for jacksmelt and Pacific herring is calculated as follows.

$$TTF^{composite} = [1.21 \times 1.89 \times 1.0] = \mathbf{2.29}$$

Dietary information for northern anchovy is described above in the description of the brown pelican diet. The $TTF^{composite}$ for northern anchovy is 2.29. Finally, dietary information for California killifish is described above in the description of the light-footed Ridgway's rail diet. The $TTF^{composite}$ for California killifish is 2.40. The $TTF^{composite}$ for the least tern is 1.84 and is calculated as follows.

$$TTF^{composite} = \big[0.78 \times \big((2.44 \times 0.30) + (2.29 \times 0.30) + (2.29 \times 0.20) + (2.29 \times 0.10) + (2.40 \times 0.10)\big)\big] = \mathbf{1.84}$$

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. Table B-14 (below) provides the fish $TTF^{composite}$ values for the 32 fish species with an available *TTF* (Table 5-2). These *TTF*s encompass a diverse range of species, representing eight fish orders. For bird species that consume a wide range of fish species, the median $TTF^{composite}$ of the 32 fish species (2.35) was used as a representative fish $TTF^{composite}$. For bird species that consume a relatively small number of fish species, representative $TTF^{composite}$ values were used if available, or $TTF^{composite}$ values were calculated for fish species not listed in the table below. Fish $TTF^{composite}$ were obtained from Appendix B of U.S. EPA (2016a).

Table B-14. Fish $TTF^{composite}$ for the 32 species with empirically derived *TTF*.

Common name	Scientific name	*TTF*	$TTF^{composite}$
Cypriniformes			
blacknose dace	*Rhinichthys atratulus*	0.71	1.29
bluehead sucker	*Catostomus discobolus*	1.04	1.24
longnose sucker	*Catostomus catostomus*	0.90	1.34
white sucker	*Catostomus commersonii*	1.11	1.58
flannelmouth sucker	*Catostomus latipinnis*	0.98	1.52
common carp	*Cyprinus carpio*	1.20	1.58
creek chub	*Semotilus atromaculatus*	1.06	1.55
fathead minnow	*Pimephales promelas*	1.57	2.78
red shiner	*Cyprinella lutrensis*	1.31	2.27
redside shiner	*Richardsonius balteatus*	1.08	2.48
sand shiner	*Notropis stramineus*	1.56	2.43
Cyprinodontiformes			
western mosquitofish	*Gambusia affinis*	1.21	2.37
northern plains killifish	*Fundulus kansae*	1.27	2.44
Esociformes			
northern pike	*Esox Lucius*	1.78	4.02
Gasterosteiformes			
brook stickleback	*Culaea inconstans*	1.79	2.47
Perciformes			
black crappie	*Pomoxis nigromaculatus*	2.67	5.66
bluegill	*Lepomis macrochirus*	1.03	2
green sunfish	*Lepomis cyanellus*	1.12	2.29
largemouth bass	*Micropterus salmoides*	1.39	3.04
smallmouth bass	*Micropterus dolomieu*	0.86	1.93
striped bass	*Morone saxatilis*	1.48	3.23
walleye	*Sander vitreus*	1.60	3.21
yellow perch	*Perca flavescens*	1.42	2.47
Salmoniformes			

Common name	Scientific name	*TTF*	$TTF^{composite}$
brook trout	*Salvelinus fontinalis*	0.88	1.96
brown trout	*Salmo trutta*	1.38	2.78
mountain whitefish	*Prosopium williamsoni*	1.38	2.97
cutthroat trout	*Oncorhynchus clarkii*	1.12	2.29
rainbow trout	*Oncorhynchus mykiss*	1.07	2.33
Scorpaeniforme			
mottled sculpin	*Cottus bairdi*	1.38	2.72
sculpin	*Cottus sp.*	1.29	2.66
Siluriformes			
black bullhead	*Ameiurus melas*	0.85	1.72
channel catfish	*Ictalurus punctatus*	0.68	1.35

Appendix C TOTAL SELENIUM AND DISSOLVED SELENIUM CONCENTRATIONS IN CALIFORNIA WATER BODIES

Information Summary

Data Source: California Environmental Data Exchange Network (CEDEN).

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.

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).

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.

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).

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 final default 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. The 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 Table C-1 and Table C-2 above.

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). Figure C-2 maps the distributions and abundances of the reported dissolved selenium concentrations (µg/L) in surface water samples.

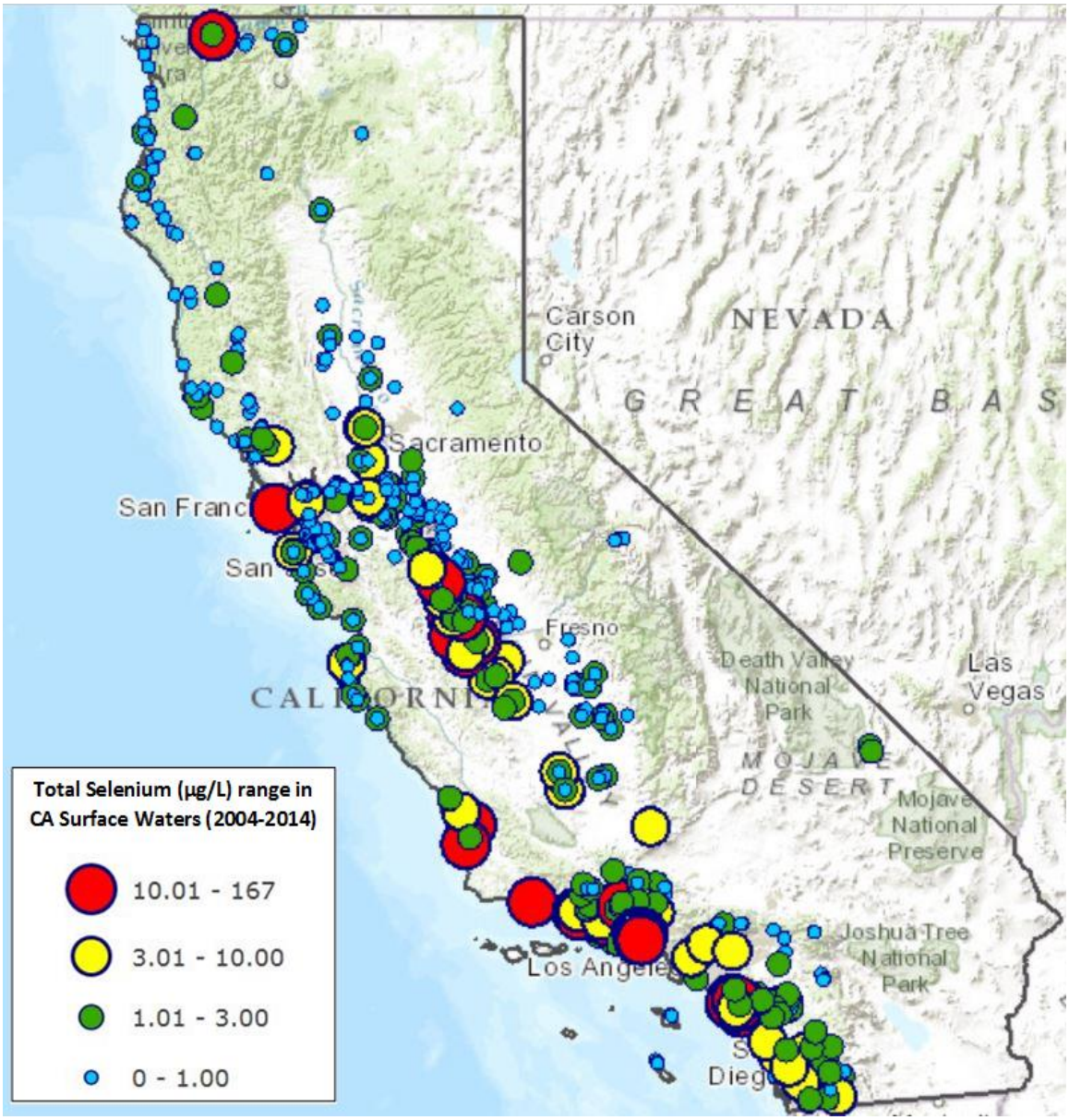


Figure C-1. Distributions and abundances of total selenium concentrations (µg/L) in surface water samples collected from October 5, 2004 through June 3, 2014.

The environmental data was last accessed through the California Environmental Data Exchange Network website (**CEDEN: http://www.ceden.org/**) on February 4, 2015.

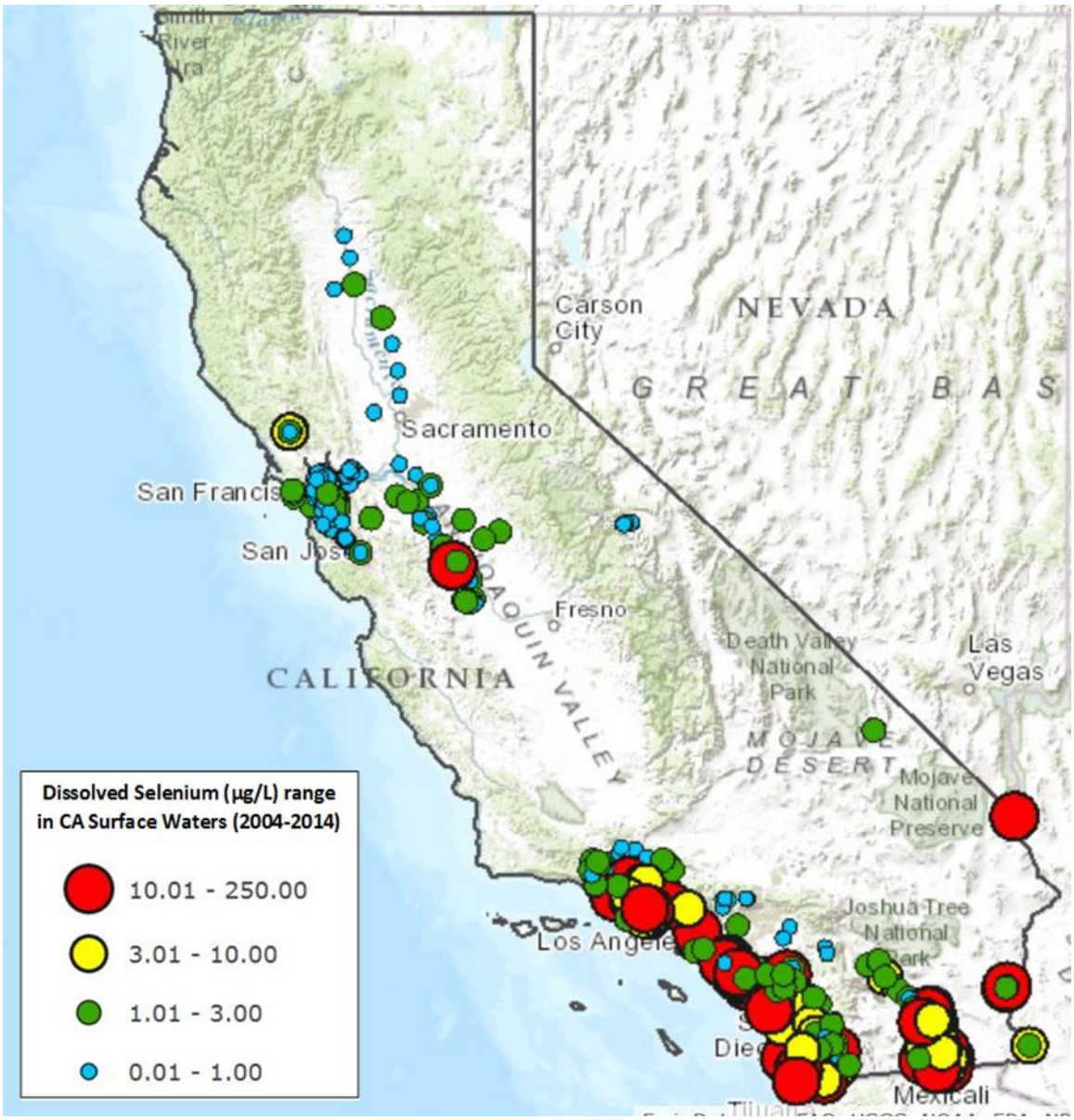


Figure C-2. Distributions and abundances of dissolved selenium concentrations (µg/L) in surface water samples collected from October 5, 2004 through June 3, 2014.

The environmental data was last accessed through the California Environmental Data Exchange 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 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 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.

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).
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.

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).
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.

Appendix D COMPARISON OF MEASURED AND PAIRED BIRD AND FISH TISSUE SELENIUM CONCENTRATIONS RELATIVE TO THEIR RESPECTIVE TISSUE CRITERIA

In 2016, the U.S. EPA published a national freshwater selenium aquatic life criterion (ALC) including elements for fish tissue of 15.1 mg Se/kg dw for eggs and ovaries, 8.5 mg Se/kg dw for whole body, and 11.3 mg Se/kg dw for muscle (U.S. EPA 2016a). The current document presents a final aquatic-dependent wildlife selenium criterion of 11.2 mg Se/kg dw for bird eggs for the State of California. Of interest is information regarding whether the fish tissue criterion elements would be protective of birds, and/or whether the bird criterion element would be protective of fish.

In this analysis measured bird and fish tissue selenium concentrations were used to indicate if one criterion element would be protective of the other. Paired fish and bird tissue data were obtained from ten USGS reconnaissance studies conducted throughout the Western United States, and four reports of monitoring results in the Newport Bay, CA watershed. Collectively, these studies encompass a range of regions across the Western United States, many of which were associated with irrigation projects. An initial literature search was conducted to find supplemental data from eastern states. While these supplemental data from eastern states appear to be much more limited, additional bird and fish data from West Virginia were obtained from West Virginia Department of Environmental Protection (WVDEP 2010, 2019). However, these data were not included in the current analysis because the bird eggs were collected just over one year before the fish tissue samples were collected, and therefore did not meet the data inclusion criteria detailed below.

For purposes of this comparison, data were considered paired if they were collected at the same site (as identified by the respective study authors) within one year of each other. Additionally, only bird species considered to be aquatic-dependent, defined here as species that depend on aquatic prey (e.g., fish and emergent aquatic insects) as a major food source, were included in this analysis (e.g., members of the order Galliformes such as chickens, ring-necked pheasant or Northern bobwhite were excluded from this analysis). Sites were classified as lentic or lotic based on site descriptions provided by the study authors. Fish and bird tissue selenium concentrations were recorded for every unique bird-fish species pair at each site. When multiple selenium measurements for a particular bird or fish species were available for a given pairing,

the final species level concentration was calculated as the geometric mean for all available measurements.

Selenium in birds was always expressed as concentrations in bird eggs. Selenium in fish was expressed as both whole body and egg-ovary concentrations. The majority (> 95%) of fish samples were reported for whole body and converted to egg-ovary concentrations using appropriate egg-ovary to whole body conversion factors for that species following the hierarchical approach based on taxonomic relatedness described in the EPA's 2016 selenium ALC (U.S. EPA 2016a).

When muscle and/or egg-ovary measurements were available at a particular site, appropriate conversion factors, following procedures described above, were applied to convert all measured selenium concentrations to whole body (Figure D-1). This process was then repeated to convert all selenium concentrations to egg-ovary (Figure D-2). For example, if a sample was expressed as a muscle concentration, it was converted to whole body using a muscle to whole body conversion factor, and then separately converted to egg-ovary using a muscle to egg-ovary conversion factor. The objective of this procedure was to minimize uncertainty associated with the application of two conversion factors to the same sample. As noted above, once all fish measurements at a given site were converted to a single tissue type, the final concentration was calculated as the geometric mean for all available measurements.

Paired selenium concentrations in bird eggs and fish tissue for all waterbodies are reported in Figure D-1 (fish whole body - bird egg) and Figure D-2 (fish egg-ovary - bird egg). In each figure, selenium concentrations in fish are plotted along the x-axis and selenium concentrations in birds are plotted along the y-axis. Dashed lines represent the bird (horizontal) and fish (vertical) tissue criterion element concentrations. Bird egg concentrations below the horizontal line meet the bird criterion and those above the line exceed the criterion. Fish tissue concentrations to the left of the line meet the criterion and to the right of the line exceed the criterion. The four quadrants created by these dashed lines show the comparative protectiveness of the bird and fish tissue criteria, based on the numbers of points within each quadrant. For example, points in the lower left represent data pairs that meet both the bird and fish criteria. Points in the upper right exceed both the bird and fish criteria. Points in the upper left meet the fish criterion but exceed the bird criterion, and finally, points in the lower right meet the bird criterion but exceed the fish criterion.

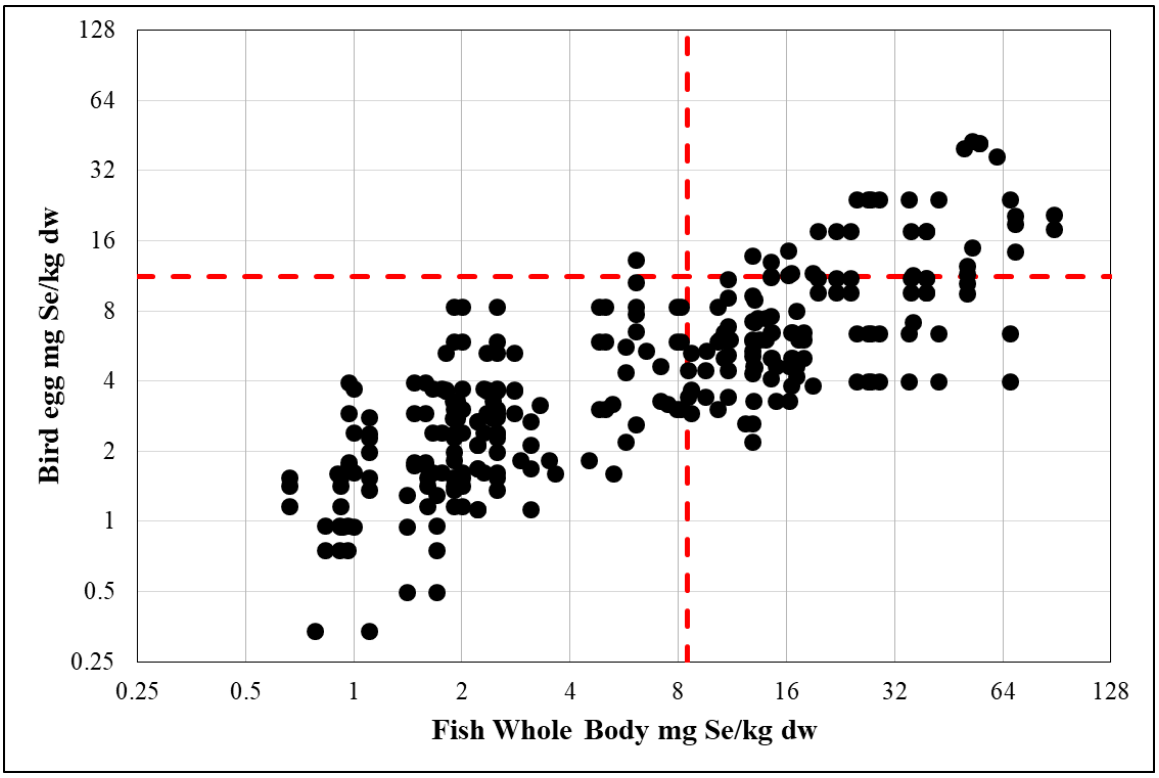


Figure D-1. Selenium concentrations in paired bird egg and fish whole body tissue samples throughout the Western U.S.

The horizontal line is the final bird egg criterion of 11.2 mg Se/kg dw. The vertical line is the whole body fish tissue criterion element of 8.5 mg Se/kg dw (U.S. EPA 2016a). Points in the points in the lower left represent data pairs that meet both the bird and fish criteria. Points in the upper right exceed both the bird and fish criteria. Points in the upper left meet the fish criteria but exceed the bird criterion. Points in the lower right meet the bird criterion but exceed the fish criterion. Each point represents the geometric mean selenium concentrations measured in a unique bird – fish species combination at the same site within one year. Fish tissues not measured as whole body converted using conversion factors based on taxonomic relatedness following the 2016 freshwater selenium criterion document (U.S. EPA 2016a).

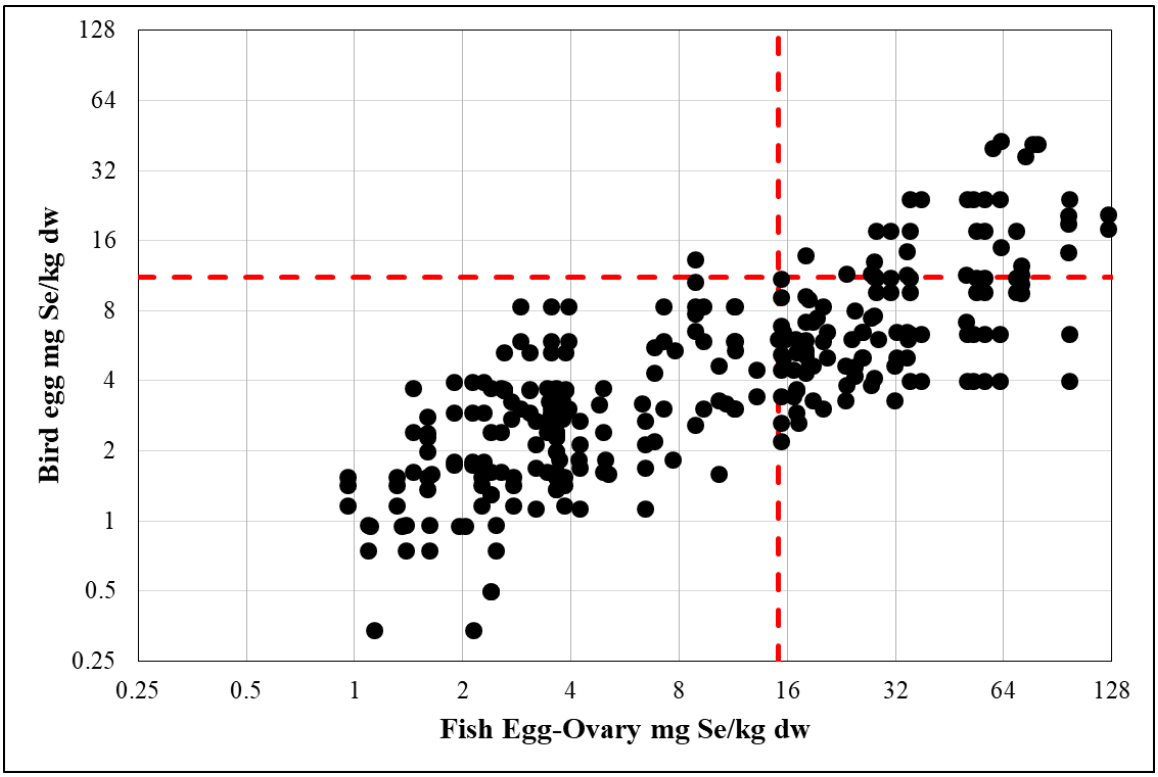


Figure D-2. Selenium concentrations in paired bird egg and fish egg-ovary tissue samples throughout the Western U.S.

The horizontal line is the final bird egg criterion of 11.2 mg Se/kg dw. The vertical line is the egg-ovary fish tissue criterion element of 15.1 mg Se/kg dw (U.S. EPA 2016a). Points in the points in the lower left represent data pairs that meet both the bird and fish criteria. Points in the upper right exceed both the bird and fish criteria. Points in the upper left meet the fish criterion but exceed the bird criterion. Points in the lower right meet the bird criterion but exceed the fish criterion. Each point represents the geometric mean selenium concentrations measured in a unique bird – fish species combination at the same site within one year. Fish tissues not measured as egg-ovary converted using conversion factors based on taxonomic relatedness following the 2016 freshwater selenium criterion document (U.S. EPA 2016a).

The number of values in the lower right quadrant of both figures is much larger than the number of values in the upper left quadrant, meaning that for these data, the fish criterion should be protective of birds (in fact, there was only one site-species pair in the upper left quadrant), but the bird criterion will not necessarily be protective of fish. There were virtually no differences in the relative proportions of points between the two fish tissue types (Table D-1). This is not surprising, as nearly all fish samples were based on whole body measurements, and the fish egg-ovary concentrations primarily reflect whole body concentrations converted to egg-ovary concentrations. Unreported data from West Virginia described above were consistent with

reported data, in that for all available WV species pairs, either birds and fish both attained, or bird eggs attained but fish tissue did not attain (WVDEP 2010, 2019). Paired site and species level selenium concentrations in bird and fish tissue are reported in Table D-2.

Table D-1. Counts and Relative Proportions of Whether Paired Selenium Measurements in Bird Eggs and Fish Whole-Body or Egg-Ovary Tissues Met or Exceeded Their Respective Criterion.

Each pair is a unique bird-fish species combination sampled at the same location within one year.

Comparison	Condition	Counts			Proportions		
		All Sites	Lentic	Lotic	All Sites	Lentic	Lotic
Fish Whole Body - Bird Egg	Total Pairs	287	198	89	1.000	1.000	1.000
	Bird and fish attain	159	91	68	0.554	0.460	0.764
	Bird attains, fish does not attain	93	78	15	0.324	0.394	0.169
	Fish attains, bird does not attain	1	1	0	0.003	0.005	0.000
	Neither bird nor fish attain	34	28	6	0.118	0.141	0.067
Fish Egg-Ovary - Bird Egg	Total Pairs	287	198	89	1.000	1.000	1.000
	Bird and fish attain	161	93	68	0.561	0.470	0.764
	Bird attains, fish does not attain	91	76	15	0.317	0.384	0.169
	Fish attains, bird does not attain	1	1	0	0.003	0.005	0.000
	Neither bird nor fish attain	34	28	6	0.118	0.141	0.067

Paired selenium concentrations in bird eggs and fish tissue for lentic waterbodies are reported in Figure D-3 (fish whole body - bird egg) and Figure D-4 (fish egg-ovary - bird egg). The relative distributions of points in these figures are qualitatively similar to the distributions in the figures representing all waterbodies, as lentic waterbodies comprise nearly 70% of the sites with paired data. Overall, the proportion of lentic sites where both birds and fish meet their respective criteria is slightly lower than for all sites, and the proportion of lentic sites where birds meet the bird criterion but fish do not meet the fish criterion is slightly higher than for all sites (Table D-1).

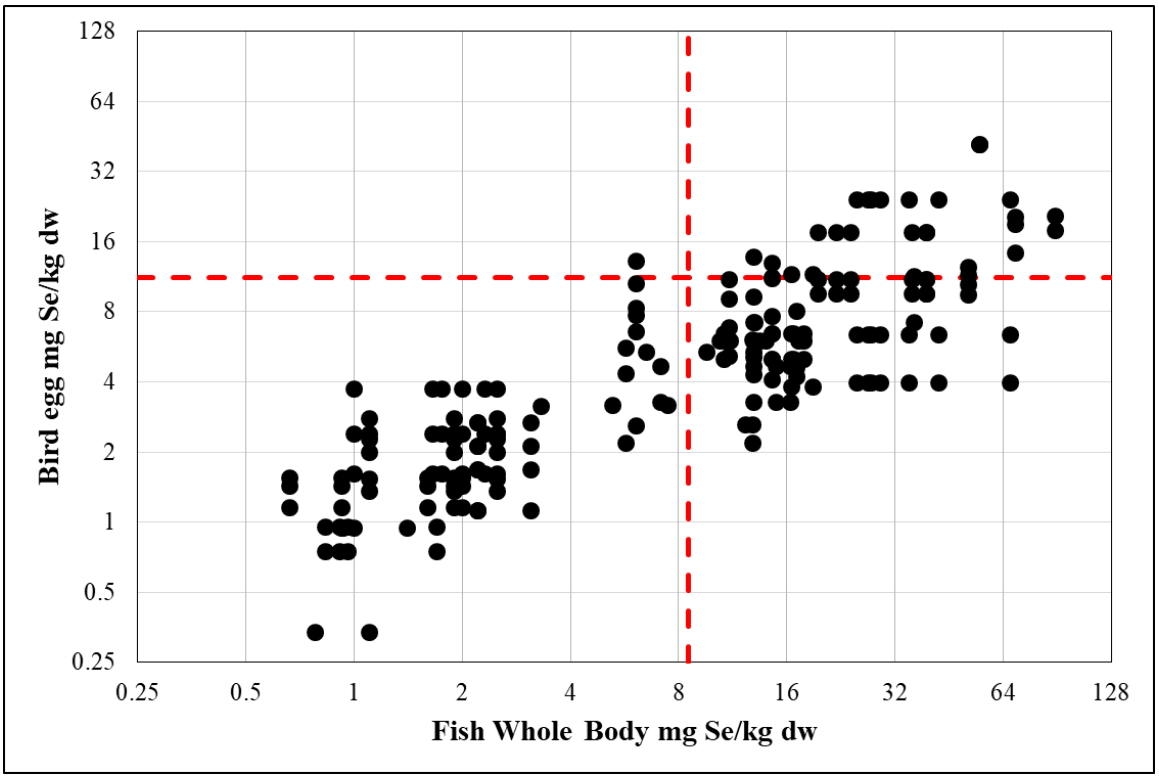


Figure D-3. Selenium concentrations in paired bird egg and fish whole body tissue samples from lentic sites throughout the Western U.S.

The horizontal line is the final bird egg criterion of 11.2 mg Se/kg dw. The vertical line is the whole body fish tissue criterion element of 8.5 mg Se/kg dw (U.S. EPA 2016a). Points in the points in the lower left represent data pairs that meet both the bird and fish criteria. Points in the upper right exceed both the bird and fish criteria. Points in the upper left meet the fish criteria but exceed the bird criterion. Points in the lower right meet the bird criterion but exceed the fish criterion. Each point represents the geometric mean selenium concentrations measured in a unique bird – fish species combination at the same site within one year. Fish tissues not measured as whole body converted using conversion factors based on taxonomic relatedness following the 2016 freshwater selenium criterion document (U.S. EPA 2016a).

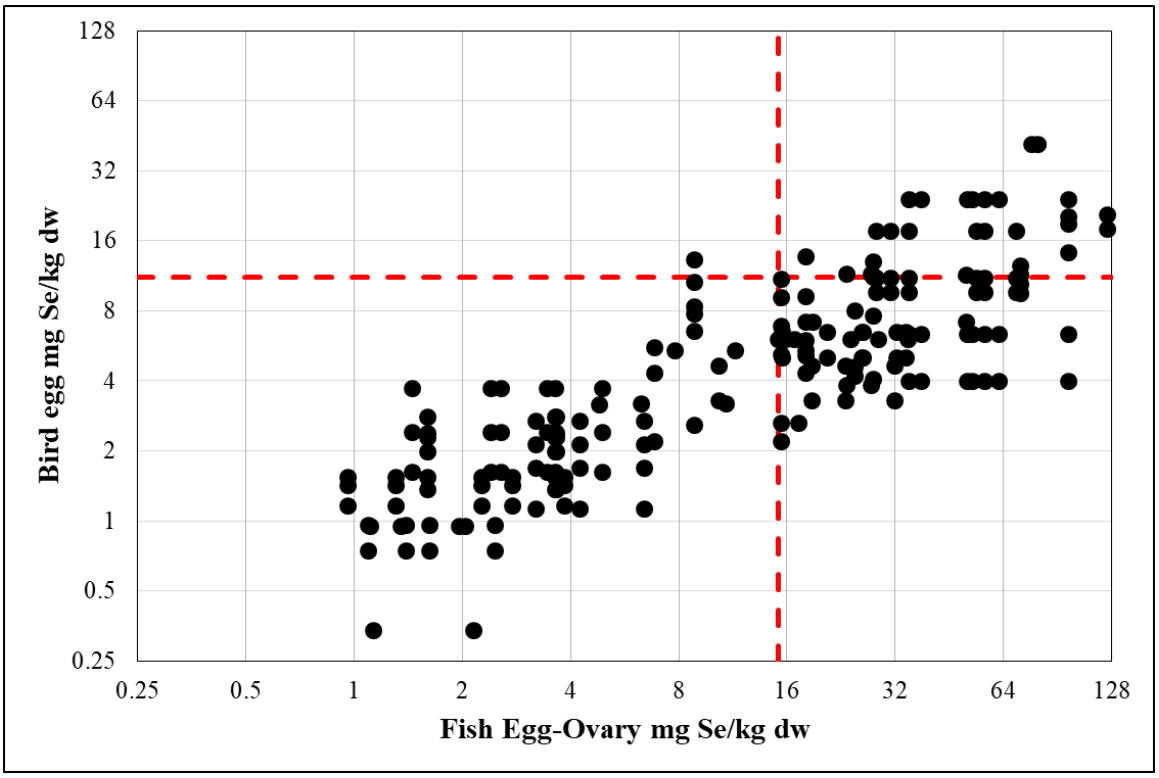


Figure D-4. Selenium concentrations in paired bird egg and fish egg-ovary tissue samples from lentic sites throughout the Western U.S.

The horizontal line is the final bird egg criterion of 11.2 mg Se/kg dw. The vertical line is the egg-ovary fish tissue criterion element of 15.1 mg Se/kg dw (U.S. EPA 2016a). Points in the points in the lower left represent data pairs that meet both the bird and fish criteria. Points in the upper right exceed both the bird and fish criteria. Points in the upper left meet the fish criterion but exceed the bird criterion. Points in the lower right meet the bird criterion but exceed the fish criterion. Each point represents the geometric mean selenium concentrations measured in a unique bird – fish species combination at the same site within one year. Fish tissues not measured as egg-ovary converted using conversion factors based on taxonomic relatedness following the 2016 freshwater selenium criterion document (U.S. EPA 2016a).

Finally, paired selenium concentrations in bird eggs and fish tissue for lentic waterbodies are reported in Figure D-5 (fish whole body - bird egg) and Figure D-6 (fish egg-ovary - bird egg). Overall, the proportion of lotic sites where both birds and fish meet their respective criteria is higher than for all sites. It is unclear whether these lentic-lotic differences represent a general result or are unique to these studies. Despite these differences, however, the general result that the bird egg criterion will most likely be met so long as the fish tissue criterion is being met, but the fish tissue criterion will not necessarily be met if the bird criterion is met, applies to both lentic and lotic waterbodies.

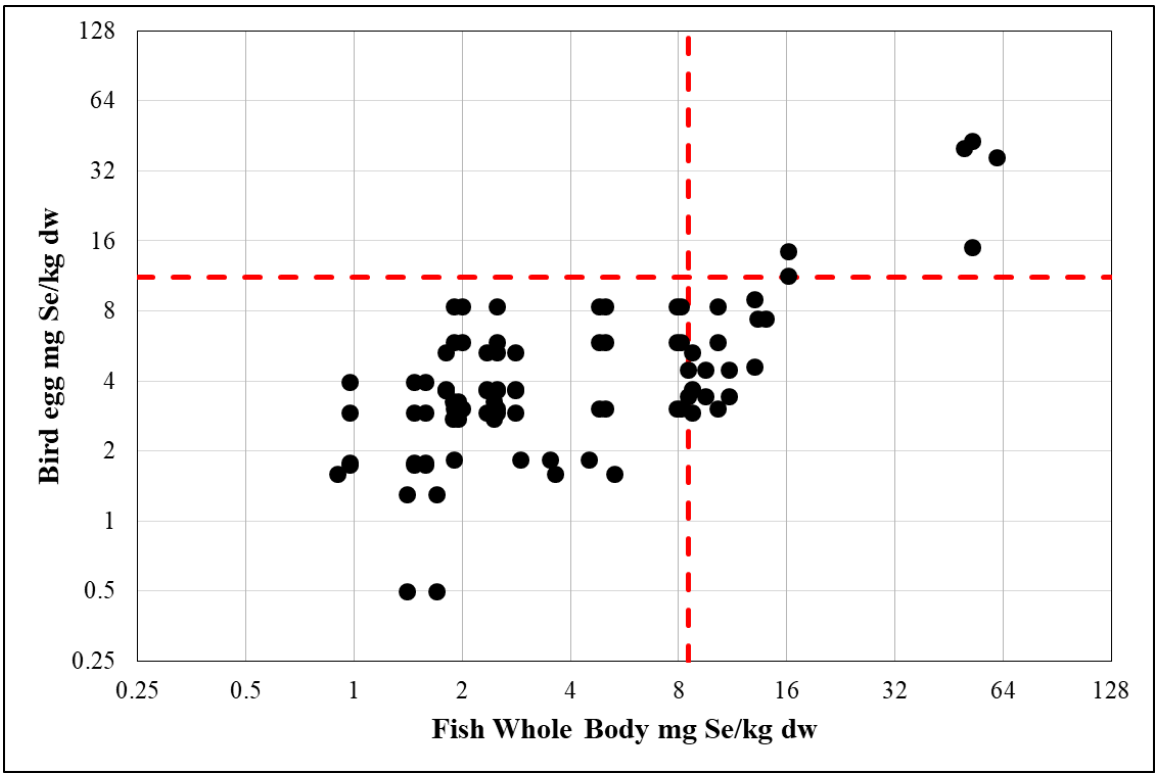


Figure D-5. Selenium concentrations in paired bird egg and fish whole body tissue samples from lotic sites throughout the Western U.S.

The horizontal line is the final bird egg criterion of 11.2 mg Se/kg dw. The vertical line is the whole body fish tissue criterion element of 8.5 mg Se/kg dw (U.S. EPA 2016a). Points in the points in the lower left represent data pairs that meet both the bird and fish criteria. Points in the upper right exceed both the bird and fish criteria. Points in the upper left meet the fish criteria but exceed the bird criterion. Points in the lower right meet the bird criterion but exceed the fish criterion. Each point represents the geometric mean selenium concentrations measured in a unique bird – fish species combination at the same site within one year. Fish tissues not measured as whole body converted using conversion factors based on taxonomic relatedness following the 2016 freshwater selenium criterion document (U.S. EPA 2016a).

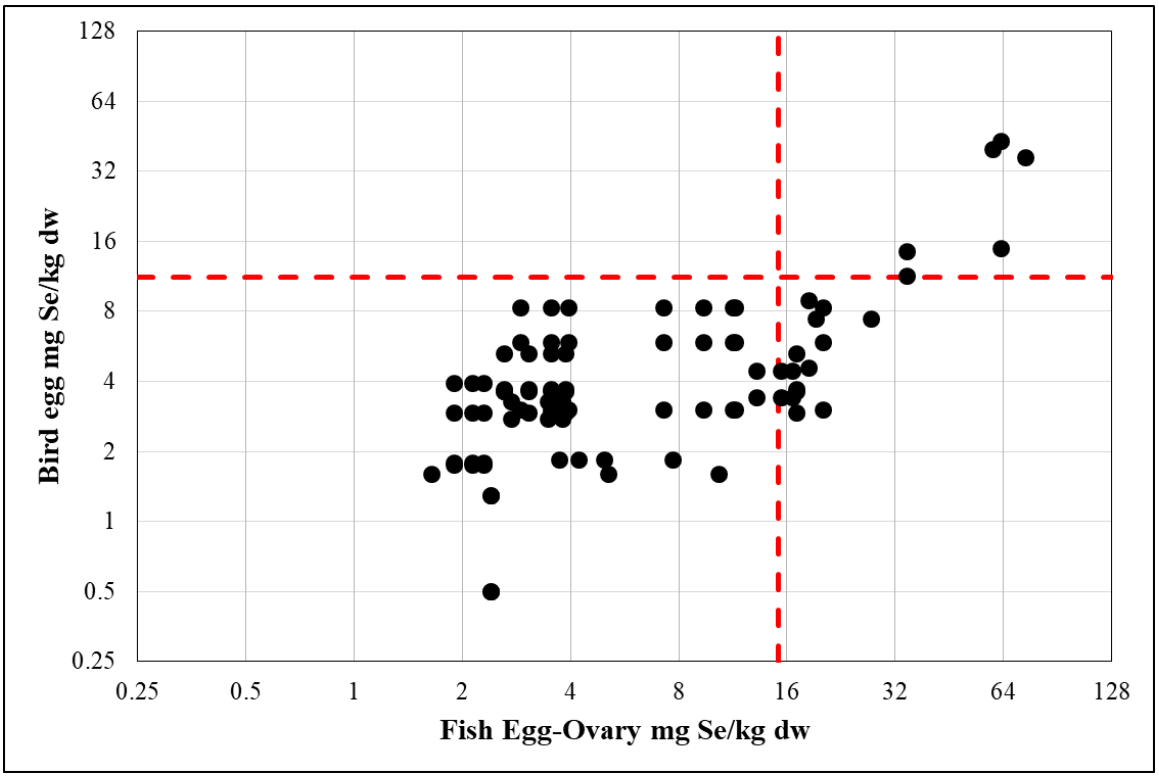


Figure D-6. Selenium concentrations in paired bird egg and fish egg-ovary tissue samples from lotic sites throughout the Western U.S.

The horizontal line is the final bird egg criterion of 11.2 mg Se/kg dw. The vertical line is the egg-ovary fish tissue criterion element of 15.1 mg Se/kg dw (U.S. EPA 2016a). Points in the points in the lower left represent data pairs that meet both the bird and fish criteria. Points in the upper right exceed both the bird and fish criteria. Points in the upper left meet the fish criterion but exceed the bird criterion. Points in the lower right meet the bird criterion but exceed the fish criterion. Each point represents the geometric mean selenium concentrations measured in a unique bird – fish species combination at the same site within one year. Fish tissues not measured as egg-ovary converted using conversion factors based on taxonomic relatedness following the 2016 freshwater selenium criterion document (U.S. EPA 2016a).

Table D-2. Bird and Fish Tissue Species Level Average Selenium Concentrations (mg Se/kg dw).

				Selenium (mg Se/kg dw)		
Reference	**Site**	**Site Type**	**Species**	**Bird Egg**	**Fish WB**	**Fish E-O**
Butler et al. 1991	Sweitzer Lake	lentic	American coot	11.10		
Butler et al. 1991	Sweitzer Lake	lentic	red-winged blackbird	17.60		
Butler et al. 1991	Sweitzer Lake	lentic	yellow-headed blackbird	9.59		
Butler et al. 1991	Sweitzer Lake	lentic	black bullhead		39.00	56.55
Butler et al. 1991	Sweitzer Lake	lentic	carp		35.56	69.35
Butler et al. 1991	Sweitzer Lake	lentic	channel catfish		24.10	34.95
Butler et al. 1991	Sweitzer Lake	lentic	flannelmouth sucker		22.00	31.02
Butler et al. 1991	Sweitzer Lake	lentic	green sunfish		19.53	28.32
Butler et al. 1991	Sweitzer Lake	lentic	white sucker		39.00	53.82
Butler et al. 1991	Gunnison River at Escalante State Wildlife Area	lotic	mallard	5.90		
Butler et al. 1991	Gunnison River at Escalante State Wildlife Area	lotic	Canada goose	3.04		
Butler et al. 1991	Gunnison River at Escalante State Wildlife Area	lotic	red-winged blackbird	8.35		
Butler et al. 1991	Gunnison River at Escalante State Wildlife Area	lotic	brown trout		2.00	2.90
Butler et al. 1991	Gunnison River at Escalante State Wildlife Area	lotic	green sunfish		7.90	11.46
Butler et al. 1991	Gunnison River at Escalante State Wildlife Area	lotic	roundtail chub		1.90	3.93
Butler et al. 1991	Gunnison River at Escalante State Wildlife Area	lotic	carp		10.30	20.09
Butler et al. 1991	Gunnison River at Escalante State Wildlife Area	lotic	fathead minnow		8.10	11.34
Butler et al. 1991	Gunnison River at Escalante State Wildlife Area	lotic	speckled dace		4.80	9.36
Butler et al. 1991	Gunnison River at Escalante State Wildlife Area	lotic	flannelmouth sucker		2.50	3.53

				Selenium (mg Se/kg dw)		
Reference	**Site**	**Site Type**	**Species**	**Bird Egg**	**Fish WB**	**Fish E-O**
Butler et al. 1991	Gunnison River at Escalante State Wildlife Area	lotic	mottled sculpin		5.00	7.25
Butler et al. 1993	Los Pinos River at La Boca	lotic	American bittern	5.30		
Butler et al. 1993	Los Pinos River at La Boca	lotic	mallard	3.65		
Butler et al. 1993	Los Pinos River at La Boca	lotic	red-winged blackbird	2.93		
Butler et al. 1993	Los Pinos River at La Boca	lotic	yellow-headed blackbird	3.69		
Butler et al. 1993	Los Pinos River at La Boca	lotic	brown trout		1.80	2.61
Butler et al. 1993	Los Pinos River at La Boca	lotic	channel catfish		2.34	3.06
Butler et al. 1993	Los Pinos River at La Boca	lotic	flannelmouth sucker		2.50	3.52
Butler et al. 1993	Los Pinos River at La Boca	lotic	speckled dace		8.70	16.97
Butler et al. 1993	Los Pinos River at La Boca	lotic	white sucker		2.80	3.86
Butler et al. 1993	Los Pinos River at La Boca	lotic	mallard	3.43		
Butler et al. 1993	Los Pinos River at La Boca	lotic	yellow-headed blackbird	4.47		
Butler et al. 1993	Los Pinos River at La Boca	lotic	fathead minnow		11.00	15.40
Butler et al. 1993	Los Pinos River at La Boca	lotic	speckled dace		8.50	16.58
Butler et al. 1993	Los Pinos River at La Boca	lotic	white sucker		9.50	13.11
Butler et al. 1993	Rock Creek on the Oxford Tract, near Oxford	lotic	mallard	3.43		
Butler et al. 1993	Rock Creek on the Oxford Tract, near Oxford	lotic	yellow-headed blackbird	4.47		
Butler et al. 1993	Rock Creek on the Oxford Tract, near Oxford	lotic	fathead minnow		11.00	15.40
Butler et al. 1993	Rock Creek on the Oxford Tract, near Oxford	lotic	speckled dace		8.50	16.58
Butler et al.	Rock Creek on the Oxford	lotic	white sucker		9.50	13.11

				Selenium (mg Se/kg dw)		
Reference	**Site**	**Site Type**	**Species**	**Bird Egg**	**Fish WB**	**Fish E-O**
1993	Tract, near Oxford					
Butler et al. 1994	Brozina Pond near F and 2100 Roads, east of Delta	lentic	American avocet	41.84		
Butler et al. 1994	Brozina Pond near F and 2100 Roads, east of Delta	lentic	fathead minnow		54.85	76.80
Butler et al. 1994	Brozina Pond near F and 2100 Roads, east of Delta	lentic	green sunfish		55.00	79.75
Butler et al. 1994	Frontgrowers pond	lentic	American coot	11.20		
Butler et al. 1994	Frontgrowers pond	lentic	pied-billed grebe	13.00		
Butler et al. 1994	Frontgrowers pond	lentic	red-winged blackbird	4.10		
Butler et al. 1994	Frontgrowers pond	lentic	Western grebe	11.32		
Butler et al. 1994	Frontgrowers pond	lentic	yellow-headed blackbird	7.65		
Butler et al. 1994	Frontgrowers pond	lentic	common carp		14.49	27.82
Butler et al. 1994	Ferriers Pond near Austin	lentic	Canada goose	19.00		
Butler et al. 1994	Ferriers Pond near Austin	lentic	American coot	20.44		
Butler et al. 1994	Ferriers Pond near Austin	lentic	yellow-headed blackbird	14.33		
Butler et al. 1994	Ferriers Pond near Austin	lentic	fathead minnow		69.28	96.99
Butler et al. 1994	Gretts Pond near Olathe	lentic	American coot	7.20		
Butler et al. 1994	Gretts Pond near Olathe	lentic	yellow-headed blackbird	11.40		
Butler et al. 1994	Gretts Pond near Olathe	lentic	fathead minnow		36.00	50.40
Butler et al. 1994	Markley Pond on East Mesa, southeast of Olathe	lentic	mallard	9.50		
Butler et al. 1994	Markley Pond on East Mesa, southeast of Olathe	lentic	red-winged blackbird	10.49		
Butler et al. 1994	Markley Pond on East Mesa, southeast of Olathe	lentic	yellow-headed	12.48		

				Selenium (mg Se/kg dw)		
Reference	**Site**	**Site Type**	**Species**	**Bird Egg**	**Fish WB**	**Fish E-O**
			blackbird			
Butler et al. 1994	Markley Pond on East Mesa, southeast of Olathe	lentic	fathead minnow		51.00	71.40
Butler et al. 1994	East Branch of Reed Wash at M Road	lotic	mallard	8.95		
Butler et al. 1994	East Branch of Reed Wash at M Road	lotic	red-winged blackbird	4.60		
Butler et al. 1994	East Branch of Reed Wash at M Road	lotic	flannelmouth sucker		13.00	18.33
Butler et al. 1994	St. George Pond near Austin	lentic	red-winged blackbird	18.00		
Butler et al. 1994	St. George Pond near Austin	lentic	fathead minnow		88.99	124.59
Butler et al. 1994	Thompsons Pond near 12 and O Roads, near Mack	lentic	American avocet	7.20		
Butler et al. 1994	Thompsons Pond near 12 and O Roads, near Mack	lentic	green sunfish		13.00	18.85
Butler et al. 1995	Dawson Draw near Lewis	lotic	red-winged blackbird	1.60		
Butler et al. 1995	Dawson Draw near Lewis	lotic	bluehead sucker		0.90	1.64
Butler et al. 1995	Dawson Draw near Lewis	lotic	fathead minnow		3.63	5.08
Butler et al. 1995	Dawson Draw near Lewis	lotic	speckled dace		5.29	10.31
Butler et al. 1995	Totten Reservoir	lentic	American coot	1.62		
Butler et al. 1995	Totten Reservoir	lentic	mallard	2.41		
Butler et al. 1995	Totten Reservoir	lentic	yellow-headed blackbird	3.73		
Butler et al. 1995	Totten Reservoir	lentic	black crappie		2.50	3.63
Butler et al. 1995	Totten Reservoir	lentic	bluegill		2.30	4.90
Butler et al. 1995	Totten Reservoir	lentic	channel catfish		1.00	1.45
Butler et al. 1995	Totten Reservoir	lentic	northern pike		2.00	3.43
Butler et al. 1995	Totten Reservoir	lentic	walleye		1.75	2.56

				Selenium (mg Se/kg dw)		
Reference	**Site**	**Site Type**	**Species**	**Bird Egg**	**Fish WB**	**Fish E-O**
Butler et al. 1995	Totten Reservoir	lentic	yellow perch		1.65	2.39
Butler et al. 1997	Large pond south of G Road, southern Mancos Valley	lentic	yellow-headed blackbird	5.20		
Butler et al. 1997	Large pond south of G Road, southern Mancos Valley	lentic	American coot	6.88		
Butler et al. 1997	Large pond south of G Road, southern Mancos Valley	lentic	cinnamon teal	11.00		
Butler et al. 1997	Large pond south of G Road, southern Mancos Valley	lentic	ruddy duck	9.13		
Butler et al. 1997	Large pond south of G Road, southern Mancos Valley	lentic	yellow-headed blackbird	5.20		
Butler et al. 1997	Large pond south of G Road, southern Mancos Valley	lentic	fathead minnow		11.00	15.40
Butler et al. 1997	Pond on Woods Canyon at 15 Road	lentic	red-winged blackbird	2.63		
Butler et al. 1997	Pond on Woods Canyon at 15 Road	lentic	fathead minnow		17.15	1.40
Byron and Santolo 2014	Big canyon wash	lotic	American coot	43.00		
Byron and Santolo 2014	Big canyon wash	lotic	pied-billed grebe	15.00		
Byron and Santolo 2014	Big canyon wash	lotic	mosquitofish		52.49	62.99
Byron and Santolo 2014	Irvine Ranch Water District Marsh	lentic	American avocet	6.03		
Byron and Santolo 2014	Irvine Ranch Water District Marsh	lentic	bluegill		13.42	28.59
Byron and Santolo 2014	Irvine Ranch Water District Marsh	lentic	carp		17.80	34.71
Byron and Santolo 2014	Irvine Ranch Water District Marsh	lentic	fathead minnow		17.20	24.08
Byron and Santolo 2014	Irvine Ranch Water District Marsh	lentic	largemouth bass		11.12	15.80
Byron and Santolo 2014	Irvine Ranch Water District Marsh	lentic	mosquitofish		14.00	16.80
Byron and Santolo 2014	Irvine Ranch Water District Marsh	lentic	inland silverside		10.75	15.59
Byron and Santolo 2014	Irvine Ranch Water District Marsh	lentic	threadfin shad		10.40	15.08

				Selenium (mg Se/kg dw)		
Reference	**Site**	**Site Type**	**Species**	**Bird Egg**	**Fish WB**	**Fish E-O**
Byron and Santolo 2014	UC Irvine Marsh	lentic	black-necked stilt	5.60		
Byron and Santolo 2014	UC Irvine Marsh	lentic	pied-billed grebe	4.35		
Byron and Santolo 2014	UC Irvine Marsh	lentic	American coot	2.19		
Byron and Santolo 2014	UC Irvine Marsh	lentic	mosquitofish		5.70	6.84
Byron et al. 2010	Big canyon wash	lotic	pied-billed grebe	36.78		
Byron et al. 2010	Big canyon wash	lotic	mosquitofish		61.30	73.56
Byron et al. 2010	Irvine Ranch Water District Marsh	lentic	American avocet	11.59		
Byron et al. 2010	Irvine Ranch Water District Marsh	lentic	black skimmer	3.82		
Byron et al. 2010	Irvine Ranch Water District Marsh	lentic	largemouth bass		16.50	23.43
Byron et al. 2010	Irvine Ranch Water District Marsh	lentic	green sunfish		18.90	27.41
Byron et al. 2010	San Diego Cr.	lotic	American avocet	11.40		
Byron et al. 2010	San Diego Cr.	lotic	black-necked stilt	14.49		
Byron et al. 2010	San Diego Cr.	lotic	Bluegill		16.16	34.42
Byron et al. 2010	UC Irvine Marsh	lentic	American coot	6.48		
Byron et al. 2010	UC Irvine Marsh	lentic	pied-billed grebe	9.55		
Byron et al. 2010	UC Irvine Marsh	lentic	mosquitofish		5.40	6.48
Byron et al. 2012	Big canyon wash	lotic	pied-billed grebe	40.00		
Byron et al. 2012	Big canyon wash	lotic	mosquitofish		49.74	59.69
Byron et al. 2012	Irvine Ranch Water District Marsh	lentic	American avocet	4.66		
Byron et al. 2012	Irvine Ranch Water District Marsh	lentic	black-necked stilt	3.30		
Byron et al. 2012	Irvine Ranch Water District Marsh	lentic	green sunfish		12.95	18.78

				Selenium (mg Se/kg dw)		
Reference	**Site**	**Site Type**	**Species**	**Bird Egg**	**Fish WB**	**Fish E-O**
Byron et al. 2012	Irvine Ranch Water District Marsh	lentic	bluegill		14.96	31.87
Byron et al. 2012	Irvine Ranch Water District Marsh	lentic	largemouth bass		16.32	23.17
Byron et al. 2012	Irvine Ranch Water District Marsh	lentic	threadfin shad		7.11	10.31
Byron et al. 2012	Peters canyon wash	lotic	black-necked stilt	7.44		
Byron et al. 2012	Peters canyon wash	lotic	green sunfish		13.30	19.28
Byron et al. 2012	Peters canyon wash	lotic	red shiner		14.00	27.30
Byron et al. 2012	UC Irvine Marsh	lentic	American coot	2.20		
Byron et al. 2012	UC Irvine Marsh	lentic	American avocet	6.06		
Byron et al. 2012	UC Irvine Marsh	lentic	black-necked stilt	2.63		
Byron et al. 2012	UC Irvine Marsh	lentic	mosquitofish		12.82	15.38
Byron et al. 2013	Irvine Ranch Water District Marsh	lentic	American avocet	5.04		
Byron et al. 2013	Irvine Ranch Water District Marsh	lentic	black-necked stilt	6.50		
Byron et al. 2013	Irvine Ranch Water District Marsh	lentic	largemouth bass		14.53	20.63
Byron et al. 2013	Irvine Ranch Water District Marsh	lentic	bluegill		16.62	34.31
Byron et al. 2013	Irvine Ranch Water District Marsh	lentic	threadfin shad		10.70	15.52
Byron et al. 2013	Irvine Ranch Water District Marsh	lentic	red shiner		16.50	32.18
Byron et al. 2013	Irvine Ranch Water District Marsh	lentic	redear sunfish		14.50	25.96
Byron et al. 2013	Irvine Ranch Water District Marsh	lentic	green sunfish		17.80	25.81
Byron et al. 2013	UC Irvine Marsh	lentic	American coot	3.20		
Byron et al. 2013	UC Irvine Marsh	lentic	catfish		7.45	10.80
Byron et al. 2013	UC Irvine Marsh	lentic	mosquitofish		5.24	6.29

				Selenium (mg Se/kg dw)		
Reference	**Site**	**Site Type**	**Species**	**Bird Egg**	**Fish WB**	**Fish E-O**
Lambing et al. 1994	Priest Butte Lakes	lentic	American avocet	24.19		
Lambing et al. 1994	Priest Butte Lakes	lentic	gadwall	4.00		
Lambing et al. 1994	Priest Butte Lakes	lentic	northern shoveler	6.39		
Lambing et al. 1994	Priest Butte Lakes	lentic	black crappie		47.01	68.17
Lambing et al. 1994	Priest Butte Lakes	lentic	brassy minnow		29.00	56.55
Lambing et al. 1994	Priest Butte Lakes	lentic	brook stickleback		35.00	50.75
Lambing et al. 1994	Priest Butte Lakes	lentic	carp		26.90	52.45
Lambing et al. 1994	Priest Butte Lakes	lentic	fathead minnow		25.00	35.00
Lambing et al. 1994	Priest Butte Lakes	lentic	white sucker		27.39	37.80
Lambing et al. 1994	Priest Butte Lakes	lentic	yellow perch		67.00	97.15
Lambing et al. 1994	Pond 3	lentic	American Avocet	3.17		
Lambing et al. 1994	Pond 3	lentic	brook stickleback		3.30	4.79
Lambing et al. 1994	Pond 5	lentic	American coot	6.57		
Lambing et al. 1994	Pond 5	lentic	eared grebe	13.27		
Lambing et al. 1994	Pond 5	lentic	lesser scaup	7.78		
Lambing et al. 1994	Pond 5	lentic	northern shoveler	2.60		
Lambing et al. 1994	Pond 5	lentic	redhead	10.62		
Lambing et al. 1994	Pond 5	lentic	ruddy duck	8.35		
Lambing et al. 1994	Pond 5	lentic	brook stickleback		6.10	8.85
Lambing et al. 1994	Freezout Lake - north	lentic	American avocet	4.62		
Lambing et al. 1994	Freezout Lake - north	lentic	lesser scaup	8.05		

				Selenium (mg Se/kg dw)		
Reference	**Site**	**Site Type**	**Species**	**Bird Egg**	**Fish WB**	**Fish E-O**
Lambing et al. 1994	Freezout Lake - north	lentic	mallard	4.21		
Lambing et al. 1994	Freezout Lake - north	lentic	brook stickleback		17.00	24.65
Lambing et al. 1994	Freezout Lake - south	lentic	American avocet	5.40		
Lambing et al. 1994	Freezout Lake - south	lentic	American coot	7.21		
Lambing et al. 1994	Freezout Lake - south	lentic	eared grebe	13.79		
Lambing et al. 1994	Freezout Lake - south	lentic	lesser scaup	9.28		
Lambing et al. 1994	Freezout Lake - south	lentic	mallard	5.14		
Lambing et al. 1994	Freezout Lake - south	lentic	northern shoveler	5.99		
Lambing et al. 1994	Freezout Lake - south	lentic	ruddy duck	4.33		
Lambing et al. 1994	Freezout Lake - south	lentic	fathead minnow		12.88	18.03
Low and Mullins 1990	Spring Creek	lotic	American coot	0.5		
Low and Mullins 1990	Spring Creek	lotic	mallard	1.3		
Low and Mullins 1990	Spring Creek	lotic	Utah sucker		1.4	2.39
Low and Mullins 1990	Spring Creek	lotic	mountain whitefish		1.7	2.40
Ong et al. 1991	18D - Marsh in SE BDNWR	lentic	American coot	0.34		
Ong et al. 1991	18D - Marsh in SE BDNWR	lentic	carp		1.10	2.15
Ong et al. 1991	18D - Marsh in SE BDNWR	lentic	threadfin shad		0.78	1.13
Ong et al. 1991	24B - Marsh in SE BDNWR	lentic	mallard	0.95		
Ong et al. 1991	24B - Marsh in SE BDNWR	lentic	carp		1.00	1.95
Ong et al. 1991	24B - Marsh in SE BDNWR	lentic	centrarchidae		0.93	1.35
Ong et al. 1991	24B - Marsh in SE BDNWR	lentic	mosquitofish		0.92	1.10
Ong et al. 1991	24B - Marsh in SE BDNWR	lentic	brown bullhead		1.40	2.03
Ong et al. 1991	24C - Marsh in SE BDNWR	lentic	American coot	0.75		

				Selenium (mg Se/kg dw)		
Reference	**Site**	**Site Type**	**Species**	**Bird Egg**	**Fish WB**	**Fish E-O**
Ong et al. 1991	24C - Marsh in SE BDNWR	lentic	mallard	0.96		
Ong et al. 1991	24C - Marsh in SE BDNWR	lentic	carp		0.83	1.62
Ong et al. 1991	24C - Marsh in SE BDNWR	lentic	centrarchidae		0.96	1.39
Ong et al. 1991	24C - Marsh in SE BDNWR	lentic	mosquitofish		0.91	1.09
Ong et al. 1991	24C - Marsh in SE BDNWR	lentic	brown bullhead		1.70	2.47
Rinella et al 1994	Ft. Boise WMA	lotic	American avocet	2.93		
Rinella et al 1994	Ft. Boise WMA	lotic	American coot	1.79		
Rinella et al 1994	Ft. Boise WMA	lotic	black-necked stilt	3.95		
Rinella et al 1994	Ft. Boise WMA	lotic	cinnamon teal	1.75		
Rinella et al 1994	Ft. Boise WMA	lotic	bullhead		1.47	2.13
Rinella et al 1994	Ft. Boise WMA	lotic	carp		0.97	1.89
Rinella et al 1994	Ft. Boise WMA	lotic	sunfish		1.58	2.29
Rinella et al 1994	Snake River at Weiser	lotic	black-crowned night heron	3.28		
Rinella et al 1994	Snake River at Weiser	lotic	California gull	2.76		
Rinella et al 1994	Snake River at Weiser	lotic	carp		1.94	3.78
Rinella et al 1994	Snake River at Weiser	lotic	channel catfish		1.88	2.73
Rinella et al 1994	Snake River at Weiser	lotic	smallmouth bass		2.44	3.46
Rinella et al 1994	Snake River mouth of Jump Creek	lotic	mallard	1.84		
Rinella et al 1994	Snake River mouth of Jump Creek	lotic	carp		1.90	3.71
Rinella et al 1994	Snake River mouth of Jump Creek	lotic	channel catfish		2.90	4.21
Rinella et al 1994	Snake River mouth of Jump Creek	lotic	mountain whitefish		4.50	7.70
Rinella et al 1994	Snake River mouth of Jump Creek	lotic	smallmouth bass		3.50	4.97
Rinella and	Harney Lake	lentic	American	1.69		

				Selenium (mg Se/kg dw)		
Reference	**Site**	**Site Type**	**Species**	**Bird Egg**	**Fish WB**	**Fish E-O**
Schuler 1992			avocet			
Rinella and Schuler 1992	Harney Lake	lentic	double crested cormorant	2.13		
Rinella and Schuler 1992	Harney Lake	lentic	gadwall	1.13		
Rinella and Schuler 1992	Harney Lake	lentic	great blue heron	2.69		
Rinella and Schuler 1992	Harney Lake	lentic	common carp		2.20	4.22
Rinella and Schuler 1992	Harney Lake	lentic	tui chub		3.10	6.42
Rinella and Schuler 1992	Harney Lake	lentic	white crappie		2.20	3.19
Rinella and Schuler 1992	North Malheur Lake	lentic	American avocet	2.00		
Rinella and Schuler 1992	North Malheur Lake	lentic	American coot	1.54		
Rinella and Schuler 1992	North Malheur Lake	lentic	double crested cormorant	2.39		
Rinella and Schuler 1992	North Malheur Lake	lentic	gadwall	1.37		
Rinella and Schuler 1992	North Malheur Lake	lentic	great blue heron	2.29		
Rinella and Schuler 1992	North Malheur Lake	lentic	white pelican	2.80		
Rinella and Schuler 1992	North Malheur Lake	lentic	brown bullhead		2.50	3.63
Rinella and Schuler 1992	North Malheur Lake	lentic	common carp		1.90	3.65
Rinella and Schuler 1992	North Malheur Lake	lentic	white crappie		1.10	1.60
Rinella and Schuler 1992	South Malheur Lake	lentic	American avocet	1.55		
Rinella and Schuler 1992	South Malheur Lake	lentic	American coot	1.43		
Rinella and Schuler 1992	South Malheur Lake	lentic	gadwall	1.16		
Rinella and Schuler 1992	South Malheur Lake	lentic	brown bullhead		1.90	2.76
Rinella and	South Malheur Lake	lentic	common		2.00	3.84

				Selenium (mg Se/kg dw)		
Reference	**Site**	**Site Type**	**Species**	**Bird Egg**	**Fish WB**	**Fish E-O**
Schuler 1992			carp			
Rinella and Schuler 1992	South Malheur Lake	lentic	largemouth bass		0.92	1.31
Rinella and Schuler 1992	South Malheur Lake	lentic	sucker		1.60	2.26
Rinella and Schuler 1992	South Malheur Lake	lentic	white crappie		0.66	0.96

Appendix D References

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