CENTRAL VALLEY SALINITY ALTERNATIVES FOR LONG-TERM SUSTAINABILITY (CV–SALTS)

Aquatic Life Study Final Report

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Prepared for
SAN JOAQUIN VALLEY DRAINAGE AUTHORITY

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Section 1
Introduction

Central Valley Salinity Alternatives for Long Term Sustainability (CV-SALTS) is developing a comprehensive regulatory and programmatic approach to the management of salt and nitrate in the Central Valley that is consistent with the State Recycled Water Policy (SRWP). This work is being carried out collaboratively with the Central Valley Regional Water Quality Control Board (Central Valley Water Board), the State Water Resources Control Board (State Water Board), the Central Valley Salinity Coalition and Stakeholders. As stated in the CV-SALTS Strategy and Framework document, the strategy to fulfill the requirements of the SRWP is to adopt a Central Valley Salt and Nitrate Management Plan (SNMP) and revise the Basin Plans applicable to the Central Valley to facilitate implementation of the SNMP. This effort includes a technical review of the state of science with regards to the establishment of appropriate water quality objectives (WQO) to protect surface water beneficial uses. This Aquatic Life Study was undertaken to evaluate the technical basis for the establishment of salinity-related WQOs to protect aquatic life in the Central Valley. The findings of this analysis will be considered by CV-SALTS during development and adoption of the SNMP.

1.1 Project Background and Purpose

This report is intended to meet the following objectives of the Aquatic Life Study:

- Identify salinity-related water quality criteria that could be used as the basis for establishing WQOs in Central Valley surface waters for protection of aquatic life beneficial uses;
- Identify salinity-related WQOs, standards, goals, procedures and/or policies that have been established elsewhere (state, federal or international) and can be used to inform efforts to establish WQOs to protect aquatic life beneficial uses, consistent with the Central Valley Water Board biological-related beneficial uses (i.e., Warm Freshwater Habitat [WARM], Cold Freshwater Habitat [COLD], and Preservation of Biological Habitats of Special Significance [BIOL] – collectively referred to as "aquatic life beneficial uses"); and
- Prepare technical recommendations for adoption of salinity-related WQOs to protect aquatic life beneficial uses for consideration by the Central Valley Water Board.

1.2 Scope of Work

To accomplish the goals listed above, five key tasks were identified as follows:

- ***Task 1 – Review Selected Literature Regarding Salinity and Protection of Aquatic Life Beneficial Uses.*** The following documents were identified by CV-SALTS as pertinent for review and summarized in a technical memorandum. Key findings of this review are presented in Section 2 of this report.

- Central Valley Regional Water Quality Control Board. 1999. *Boron: A Literature Summary for Developing Water Quality Objectives (Draft)*. Central Valley Regional Water Quality Control Board, January 1999. http://www.swrcb.ca.gov/centralvalley/water_issues/swamp/historic_reports_and_faq_sheets/info_supt_rec_guidelines/boron_literature_sum_draft.pdf

- Central Valley Regional Water Quality Control Board. 2000. *Salinity: A Literature Summary for Developing Water Quality Objectives (Draft)*. Central Valley Regional Water Quality Control Board, January 2000. http://www.swrcb.ca.gov/rwqcb5//water_issues/swamp/historic_reports_and_faq_sheets/info_supt_rec_guidelines/davis2000_salinity_litsum_ar07019.pdf.

- Department of Interior (DOI). 1998. *Guidelines for the Interpretation of Biological effects of Selected Constituents in Biota, Water and Sediment*. Department of Interior, National Irrigation Water Quality Program, Information Report No. 3. November 1998. In particular, review the sections titled introduction, boron, and salinity. http://www.usbr.gov/niwqp/guidelines/index.html.

- Evans, M. and C. Frick. 2001. *The Effects of Road Salts on Aquatic Ecosystems*. NWRI Contribution Series No. 02-308, National Water Research Institute and University of Saskatchewan, Saskatoon, SK, Canada.

- Iowa Department of Natural Resources (IDNR). 2009. Water Quality Standards Review: Chloride, Sulfate and Total Dissolved Solids. Iowa Department of Natural Resources, February 9, 2009. http://www.dnr.mo.gov/env/wpp/rules/rir/so4-cl-ws_review_idnr_so4-cl.pdf

- Nagpal, N.K., D.A. Levy, and D.D. MacDonald. 2003. *Water Quality: Ambient Water Quality Guidelines for Chloride - Overview Report*. Ministry of Environment, British Columbia, Canada. http://www.env.gov.bc.ca/wat/wq/BCguidelines/chloride/chloride.html.

- Stroud Water Research Center. 2010. *Expert Report on the Proposed Rulemaking by the Pennsylvania Environmental Quality Board for Ambient Water Quality Criterion Chloride (Cl).* Stroud Water Research Center, Avondale, Pennsylvania. Stroud Report #2010004. June 14, 2010. http://www.sierraclub.org/naturalgas/rulemaking/documents/PA.Chapter93/2010.6.14.StroudReport.pdf.

- Weber-Scannell, P.K. and L.K. Duffy. 2007. *Effects of Total Dissolved Solids on Aquatic Organisms: A Review of Literature and Recommendation for Salmonid Species*. American Journal of Environmental Sciences 3: 1-6.

- ***Task 2 – Supplemental Review of Key Technical References.*** Based on the review of Task 1 documents, the following documents were identified for additional review as pertinent to the overall goals of this project. Additional documents reviewed are listed below and findings are included in Section 2 of this report, as appropriate:

  - Birge, W.J. and J.A. Black. 1977. *Sensitivity of vertebrate embryos to boron compounds*, April 1977 Final Report. EPA-560/1-76-008. U.S. Environmental Protection Agency, Office of Toxic Substances. Washington, DC. 66p.

  - Black, J.A., J.B. Barnum, and W.J. Birge. 1993. *An integrated assessment of the biological effects of boron to the rainbow trout*. Chemosphere 26: 1383-1413.
  - Bradley, T.J. and J.E. Phillips. 1977. *Regulation of rectal secretion in saline-water mosquito larvae living in waters of diverse ionic composition*. Journal of Experimental Biology 66:83-96.
  - Canadian Council of Ministers of the Environment (CCME). 2011. *Canadian water quality guidelines for the protection of aquatic life: Chloride*. In: Canadian Environmental Quality Guidelines, 1999.
  - Goetsh, P.A. and C.G. Palmer. 1997. *Salinity tolerance of selected macroinvertebrates of the Sabie River, Kruger national Park, South Africa*. Archive of Environmental Contaminant Toxicology 32: 32-41.
  - Goodfellow, W.L., L.W. Ausley, D.T. Burton, D.L. Denton, P.B. Dorn, D.R. Grothe, M.A. Heber, T.J. Norberg-King, and J.H. Rodgers, Jr. 2000. *Major ion toxicity in effluents: A review with permitting recommendations*. Environmental Toxicology and Chemistry 19: 175-182.
  - Leland, H.V. and S.V. Fend. 1998. *Benthic invertebrate distributions in the San Joaquin River, California, in relation to physical and chemical factors*. Canadian Journal of Fisheries and Aquatic Sciences 55: 1051-1067.
  - Loewengart, G. 2001. *Toxicity of boron to Rainbow Trout: A weight-of-the-evidence assessment*. Environmental Toxicology and Chemistry 20: 796-803.
  - Rowe, R.I., C. Bouzan, S. Nabili, and C.D. Eckert. 1998. *The response of trout and zebrafish embryos to low and high boron concentrations is U-shaped*. Biological Trace Elements Research 66: 237-259.
  - Schoderboeck, L., S. Muhlegger, A. Losert, C. Gausterer, and R. Hornek. 2011. *Effects assessment: Boron compounds in the aquatic environment*. Chemosphere 82: 883-487.
  - Soucek, D. J., A. Dickinson, and B.T. Koch. 2011. *Acute and chronic toxicity of boron to a variety of freshwater organisms*. Environmental Toxicology and Chemistry 30: 1906-1914.

- ***Task 3 – Supplemental Review of Federal Procedures, Policies and Guidelines.*** This task largely focused upon the review of documents related to the development of field based benchmarks for salinity/total dissolved solids (TDS) and review of the EPA Ecotoxicology database (ECOTOX; http://cfpub.epa.gov/ecotox/). Findings are incorporated into Sections 2 and 3 of this report, as applicable.

- ***Task 4 – Supplemental Review of California Plans, Procedures, Policies and Guidelines.*** Efforts under this task included contacting the California Regional Water Quality Control Boards to identify potentially relevant documents, determine the status of biological assessment in California, and evaluate the applicability of California's emerging Biological Objectives for determining salinity related impacts on aquatic biota in the Central Valley. Contact with the California Regional Water Quality Control Boards did not result in the identification of any new information relevant to the purposes of this study. Information regarding Central Valley aquatic biota within the context of the purposes of this study is incorporated into Section 3 of this report.

- ***Task 5 – Supplemental Review of Selected International Procedures, Policies and Guidelines***. Reviews of specific international regulatory approaches are summarized in Section 2 of this report; their applicability to the Central Valley is summarized in subsequent sections, as appropriate. International approaches reviewed included:
  - *Canada* – Summarize any applicable water quality guidelines for protection of aquatic life published by the CCME. http://www.ccme.ca/publications/ceqg_rcqe.html.
  - *Australia/New Zealand* - Summarize any applicable water quality guidelines for protection of aquatic life in Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZECC and AMCANZ 2000). http://www.environment.gov.au/resource/australian-and-new-zealand-guidelines-fresh-and-marine-water-quality-volume-1-guidelines.
  - *South Africa* - Summarize any applicable water quality guidelines for protection of aquatic life included in South African Water Quality Guidelines, Volume 7: Aquatic Ecosystems Department of Water Affairs and Forestry, 1996). http://www.capetown.gov.za/en/CSRM/Documents/Aquatic_Ecosystems_Guidelines.pdf.

1.3 Definitions and Acronyms

The following definitions and acronyms are provided to support understanding of the material presented in subsequent section.

1.3.1 Definitions

The following definitions have been provided consistent with the United States Environmental Protection Agency (EPA) Guidelines (Stephens et al., 1985):

Acute Toxicity: Toxicity elicited immediately following short-term exposure to a chemical. EPA derives acute criteria from 48- to 96-hour tests of lethality or immobilization.

Acute to Chronic Ratio (ACR): ACR = Acute value/chronic value. Three ACR values are required, and typically they are averaged to produce a Final Acute-Chronic Ratio (FACR).

Chronic Toxicity: Toxicity resulting from long-term exposure to a toxicant generally at exposure levels below those that elicit acute toxicity. Exposure durations are considered in relation to the lifespan of the organism tested.

Chronic Value (CV): Can be the geometric mean of the No Observed Effect Concentration (NOEC) and the Lowest Observed Effect Concentration (LOEC), or a statistically defined value (e.g. EC_{50}).

Criterion Continuous Concentration (CCC) (or chronic criterion): This value is usually equivalent to the FCV (see below).

Criterion Maximum Concentration (CMC) (or acute criterion): This value is derived by dividing the FAV by a safety factor (usually 2).

EC_{50}: The effective toxicant concentration that results in a 50% decrease in a test population for a non-lethal response (e.g., growth, reproductive output).

Final Acute Value (FAV): A value used to estimate an acute concentration that would be protective of 95% of species. These values can be based on a species sensitivity distribution (SSD) or extrapolated from the acute toxicity data from the four most sensitive GMAVs.

Final Chronic Value (FCV): A value used to estimate a chronic concentration that would be protective of 95% of species. If sufficient chronic data are not available (from 8 families), chronic values are estimated from acute values via the application of ACRs.

Genus Mean Acute Value (GMAV): When data for more than one species within a given genus is available, the geometric mean of these values is used to calculate a GMAV.

HC_{05}: An estimate of the concentration of a toxicant that will not harm 95% of species in a community.

LC_{50}: Median lethal concentration; the concentration of a toxicant that results in a 50% reduction of survival in the test population.

1.3.2 Acronyms

The following acronyms are used in this document:

- **ACR** – Acute to Chronic Ratio
- **ANZ** – Australia/New Zealand
- **CAM** – Central Appalachian Mountains
- **CCC** – Criterion Continuous Concentration (or chronic criterion)
- **CCME** – Canadian Council of Ministers of the Environment
- **CEDEN** – California Environmental Data Exchange Network
- **Central Valley Water Board** – Central Valley Regional Water Quality Control Board
- **CMC** – Criterion Maximum Concentration (or acute criterion)
- **CV** – Chronic Value
- **CV-SALTS -** Central Valley Salinity Alternatives for Long Term Sustainability
- **CWQG** – Canadian Water Quality Guidelines
- **DOI** – Department of Interior
- **EC –** Electrical Conductivity
- **ECOTOX** – EPA Ecotoxicology Database
- **EPA** – U.S. Environmental Protection Agency
- **FACR** – Final Acute-Chronic Ratio
- **FAV** – Final Acute Value

- **FCV** – Final Chronic Value
- **GLEC** – Great Lakes Environmental Center
- **GMAV** – Genus Mean Acute Value
- **IDNR** – Idaho Department of Natural Resources
- **INHS** – Illinois Natural History Survey
- **LOAEL –** Lowest Observed Adverse Effect Level
- **NOAEL -**No Observed Adverse Effect Level
- **PNEC** – Predicted No Effect Concentration
- **SC** – Specific Conductance
- **SCCWRP** – Southern California Coastal Water Research Project
- **SNMP** – Salt and Nitrate Management Plan
- **SRWP** – State Recycled Water Policy
- **SSD** – Species Sensitivity Distribution
- **State Water Board** – State Water Resources Control Board
- **SWAMP** – Surface Water Ambient Monitoring Program
- **TDS** – Total Dissolved Solids
- **WQO –** Water Quality Objective

Section 2

Analysis of Salinity-Related Constituents

For the purposes of this project, salinity-related water quality constituents refers to the following parameters: TDS, electrical conductivity (EC), sodium (Na^+), chloride (Cl^-), calcium (Ca), magnesium (Mg), potassium (K), sulfate (SO_4^-), carbonate (CO_3), bicarbonate (HCO_3), hardness (as $CaCO_3$), and boron (B). The following sections provide information primarily on four of these constituents: TDS/EC, chloride, sulfate and boron. As will be noted below information on the relationship between other salts and potential impacts to aquatic life are not well studied or documented.

2.1 Total Dissolved Solids

2.1.1 Background

As a general point of reference, Wetzel (1983) reports that the mean salinity concentrations in the world's rivers is 120 milligrams/liter (mg/L), and McKee and Wolf (1963) state that 95% of inland United States waters are < 400 mg/L. This report treats TDS and EC jointly because in practicality, EC is the readily attainable measurement used to estimate TDS. The DOI National Irrigation Program Information Report No 3. (1998) provides a summary of the relationships between these parameters as follows:

- For specific conductance (SC) less than 5,000 microsiemens/centimeter (µS/cm) at 25°C, TDS = 0.584 x SC + 22.1

- For specific conductance between 5,000 and 9,000 µS/cm at 25°C, TDS = 0.682 x SC - 269

  Where: TDS = mg/L total salt

Linsley and Franzini (1979) report that for most waters, TDS concentrations typically range from 0.55 – 0.7 times the specific conductance, whereas Hem (1985) reports regression slopes of 0.54 - 0.96 for dilute waters. The U.S. Bureau of Reclamation (1993) reports the use of 0.64 as a "rule of thumb", though this value can be higher if the primary driver of conductivity is sulfate (> 0.7). The U.S. Department of Agriculture (1954) recommends a ratio of 0.64. For portions of the San Joaquin River, reported conversion values range from 0.59 to 0.69 (State Water Board, 1987). Thus, while there is coarse agreement on the conversion of EC values to TDS concentrations, it is clear that individual ionic constituents, temperature, and ionic strength and can alter the utility of a single conversion value.

2.1.2 State of the Science with Respect to Understanding the Toxicity of Complex TDS Matrices

TDS toxicity is complex and remains poorly understood because (a) individual ions vary in their toxicities (e.g., Saiki et al., 1992; Mount et al., 1997); (b) other ions in solution can play profound modifying roles in the toxicity of individual ions (e.g., Soucek and Kennedy, 2005; Soucek, 2007; Soucek et al., 2011); and (c) the relative toxicities of individual ions are not consistent across species (e.g., Kunz et al., 2013). Further, it remains unclear to what extent the "balance" of various ions determines the toxicity of complex ion mixtures, specifically:

- Individual ions vary in their toxicities. The work of Saiki et al. (1992) is particularly relevant in illustrating this point. When seawater (dominated by Na+ and Cl-) was diluted to match the TDS concentrations in agricultural drain water, both juvenile Chinook salmon and striped bass survived well in 28-day exposures. However, even diluted drain water (50% dilutions) reduced performance. This poor performance was attributed to either the high sulfate concentrations or unusual ratios of major ions. A study by Bradley and Phillips (1977) with saline tolerant mosquitos demonstrated that these insects did not tolerate the substitution of sulfate for chloride in their rearing water. The authors speculated that sulfate was a more difficult ion to deal with (more bulky) than chloride, and potentially more difficult to eliminate. Similar results were observed for a South African Mayfly (Goetsh and Palmer, 1997).

- Other ions in solution can play profound modifying roles in the toxicity of individual ions. For example, Soucek et al. (2011) report that chloride reduces the toxicity of boron to the crustacean *Hyallela azteca*. Similarly, Soucek (2007) report on the ameliorating effects of water hardness and chloride on sulfate toxicity. Several other studies explore such relationships and are discussed in later sections of this report.

The relative toxicities of individual ions are not consistent across species. For example, Mount et al., (1997) described sulfate as being the least toxic of the major ions tested with *Ceriodaphnia dubia, Daphnia magna* and *Pimephales promelas*, while the Saiki et al. (1992), Bradley and Phillips (1977), and Goetsh and Palmer (1997) studies suggest sulfate to be more toxic than chloride in other species.

A very important and under-appreciated issue that needs to be considered in any water quality criteria or WQO development process is the disconnect between the species commonly found in toxicity datasets and the species that occur in freshwater ecosystems. In particular, insects tend to dominate freshwater invertebrate communities and are thus the focal group for ecological assessment and monitoring efforts. However, insects remain tremendously under-represented in toxicity databases. Thus while a toxicity dataset may meet EPA requirements (eight families represented per the methods developed by Stephan et al. (1985)) for criteria development, considerable uncertainty will remain with respect to the protection of aquatic communities if the predominant taxa in those communities are not well represented in the datasets used to generate regulatory guidelines, criteria, or WQOs.

While there are many examples in the literature of freshwater organisms tolerating extremely high acute exposures to high TDS waters, there are also examples of species poorly tolerating environmentally relevant high TDS concentrations. In their brief review of TDS toxicity, Weber-Scannell and Duffy (2007) compiled data from several aquatic species showing sensitivity of aquatic organisms (**Table 2-1**)[1]. They conclude that fertilization and egg development are generally the most sensitive physiological processes disrupted by high TDS concentrations in fish. It is noteworthy that the toxicity values compiled by Weber-Scannell and Duffy (2007) are considerably lower than those compiled in DOI (1998).

Goodfellow et al. (2000) state that, "In general terms, it is more important to match the salinity tolerance for chronic versus acute toxicity testing, given the fact that the growth and reproductive

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[1] Note that EC_{50} values for Khangarot (1991) in Table 2-1 are based on immobility. The original Baudoin (1974) reference in Table 2-1 was not found; therefore, the basis for the EC_{50} endpoints could not be determined.

endpoints are more sensitive to energy-taxing requirements of osmoregulation than is the acute endpoint of survival." Work with aquatic insects in our laboratory also supports the need for chronic exposures to adequately evaluate toxicity due to high TDS exposures (Kunz et al., 2013).

Table 2-1. TDS effect concentrations from Weber-Scannell and Duffy (2007) (References in table are not provided in this document – see original references).

Species			TDS Components	Effects Unit	Effects Concentration mg L^{-1}	Reference
Chironomus tentans	Diptera larvae		$CaSO_4$	Growth reduced by 45%	2,089	Chapman *et al*.[9]
C tentans	Diptera larvae		$CaSO_4$	Reduced survival	1,750 and 2,240	Chapman *et al*.[9]
C tentans	Diptera larvae		$CaSO_4$	10 day, LC50[1]	2,035	USEPA[22]
C tentans	Diptera larvae		$CaSO_4$	IC_{20}	1,598	USEPA[23]
Cricotopus trifascia	Diptera larvae		K^+	LC50	1567	Hamilton 1975, cited in ENSR[24]
C. trifascia	Diptera larvae		CL^-	LC50	1406	Hamilton 1975, cited in ENSR[24]
Hexagenia bilineata	Insect: mayfly		K, Li, Mg, Mo, Na, SO_4, NO_3	15 day test, 80% survival	2,270	Woodward *et al*.[25]
H. bilineata	Insect: mayfly		K, Li, Mg, Mo, Na, SO_4, NO_3	30 day test, 70% survival	1,230	Woodward *et al*.[25]
Hydroptila angusta	Insect: caddisfly		K^+	LC50	2316	Hamilton 1975, cited in ENSR[24]
Hydroptila angusta	Insect: caddisfly		Cl^-	LC50	2077	Hamilton 1975, cited in ENSR[[24]
Dugesia gonocephala	flatworm		Cl^-	Mortality	1230	Palladina 1980, cited in ENSR[24]
Tubifex tubifex	segmented worm		K^+	EC50[1]	2000	Khangarot 1991, cited in ENSR[24]
Tubifex tubifex	segmented worm		Ca^{+2}	EC50	814	Khangarot 1991, cited in ENSR[24]
Cyclops abyssorum prealpinus	cyclopoid copepod		Mg^{+2}	EC50	280	Baudoin 1974, cited in ENSR[24]
C.abyssorum prealpinus	cyclopoid copepod		Ca^{+2}	EC50	7000	Baudoin 1974, cited in ENSR[24]
C. dubia	zooplankton			LC50	1,692	Tietge and Hockett[26]
C. dubia	zooplankton		NaCl	48-hr, LC50	835	Hoke *et al*.[10]
C. dubia	zooplankton		NaCl	48-hr, LC50	735	Hoke *et al*.[10]
Cladoceran	zooplankton		$CaSO_4$	LC50, 48-h	>1,910	Mount *et al.* [7]
D. pulex	zooplankton		Ca, ion	EC50, 48-h	499	Goodfellow *et al*.[27]
D. magna	zooplankton			LC50	1,692	Tietge and Hockett[25]
D. magna	zooplankton	<24 h	NaCl	48-hr, LC50	5015	Hoke *et al*.[10]
D. magna	zooplankton	<24 h	NaCl	48-hr, LC50	5000	Hoke *et al*.[10]
D. magna	zooplankton	4th instar	NaCl	48-hr, LC50	4000	Hoke *et al*.[10]
D. magna	zooplankton	<24 h	$NaHCO_3$	48-hr, LC50	1400	Hoke *et al*.[10]
D. magna	zooplankton	<24 h	$NaHCO_3$	48-hr, LC50	1150	Hoke *et al*.[10]
D. magna	zooplankton	7 day	$NaHCO_3$	48-hr, LC50	1780	Hoke *et al*.[10]
D. magna	zooplankton	7 day	$NaHCO_3$	48-hr, LC50	2200	Hoke *et al*.[10]
D. magna	zooplankton	7 day	$NaHCO_3$	48-hr, LC50	1250	Hoke *et al*.[10]
D. magna	zooplankton	<24 h	$NaHCO_3$	48-hr, LC50	1160	Hoke *et al*.[10]
D. magna	zooplankton	<24 h	$NaHCO_3$	48-hr, LC50	1000	Hoke *et al*.[10]
Mysidopsis bahia	mysid shrimp		Ca, ion	LC50, 96-h	927	Goodfellow *et al*.[27]

LC50 = Lethal Concentration 50, or concentration causing 50% mortality
IC0 = Inhibition Concentration 0, or concentration causing inhibition of 0% of the population.
EC50 = Effects Concentration, or concentration effecting 50% of the population.

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IDNR (2009) Appendix B is a Draft Justification for Changing Water Quality Standards for Sulfate, Total Dissolved Solids and Mixing Zones produced by the Illinois Environmental Protection Agency (2006). The justification includes the claim that, "Unlike many toxicants that exert toxic effects over both short term and long term periods (acute and chronic toxicity), sulfate has been demonstrated to affect only short term survival of test organisms. In other words, organisms that survive the initial osmotic shock of exposure will survive indefinitely at that concentration". However, analysis of sulfate toxicity data suggests much lower effect levels resulting from chronic exposures than from acute exposures (see below). Thus, the Illinois justification does not appear to be supported by the data[2].

The concept of high TDS exposures as energy-taxing is an important one. Pimentel and Bulkley (1983) report that low temperatures, "reduce the ability of fish to regulate internal salt balance". There are anecdotal reports of marked increases in mayfly food consumption rates under high TDS exposures accompanying significant developmental delays (it takes longer for larvae to develop to adulthood if they survive the exposure regime)(David Funk, entomologist, Stroud Water Research Center, Avondale, PA, personal communication; David Buchwalter, direct observation).

The issues described above coupled with the lack of clarity within the scientific community regarding the effects of ion balance or ionic ratios on aquatic life make the establishment of rational regulations based on un-characterized TDS or EC challenging. Weber-Scannell and Duffy (2007) conclude that, "It is recommended that different limits for individual ions, rather than TDS, be used for salmonid species". While this may be more scientifically defensible in theory, in practice it creates the need for well-established criteria for all components of TDS. This requires the development of a more complete understanding of the toxicities of several ions and the modifying effects of other interacting ions in solution. Our scientific understanding is considerably lacking in this regard. The State of Iowa has essentially decided to abandon its 1,000 mg/L TDS standard for the development of individual chloride and sulfate standards (see below). Yet how the State of Iowa will proceed with high TDS discharges when constituents other than chloride or sulfate dominate the matrix remains unclear.

2.1.3 Regulatory Approaches to Total TDS/Salinity/EC for the Protection of Aquatic Life

In the U.S. and Canada, the predominant manner by which criteria are developed for aquatic life protection is through the analysis of toxicity datasets containing data for numerous species (e.g., see Stephan et al., 1985). The complexity of TDS matrices and the variability of toxicity described above have precluded the development of lab generated criteria values for total salinity/TDS at the federal and state levels. Other nations have developed field-based methods based on statistical distributions of salinity in natural systems to form the basis of regulatory approaches. Examples of each of these approaches are described below.

United States Field-based Regulatory Approach[3]

The absence of federal or state regulatory TDS/EC values in Central Appalachia Mountains (CAM) prompted EPA to develop a field derived conductivity benchmark in Ecoregions 68-70, where aquatic life is reported to be highly impaired below surface coal mining operations (EPA, 2011). The benchmark employed a field-derived genus sensitivity distribution based on the probability of

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[2] The agency was not contacted for the purposes of this study; additional follow-up is recommended if these data were to be used for WQO development.

[3] The document titled, *A Field-Based Aquatic Life Benchmark for Conductivity in Central Appalachian Streams* was briefly used in the permitting process for the coal industry. Currently, the benchmark is in legal review.

occurrence of a given genus as a function of conductivity from 2,210 biological samples taken with accompanying conductivity data. The 5th centile of the distribution of occurrence was taken for each genus to reflect its extirpation concentration. The method creates a sensitivity distribution of extirpation concentrations of TDS across genera. The benchmark is based on the 5th centile of the distribution of genus specific extirpation concentrations and calculated to be 300 µS/cm (**Figure 2-1**). Importantly, this benchmark is intended only to apply to Ecoregions 68-70 where the ionic composition is dominated by sulfate/bicarbonate salts.

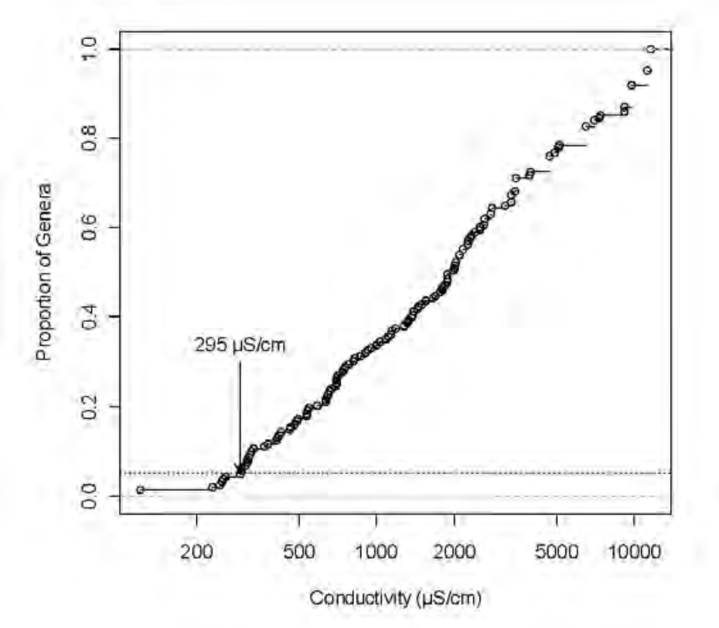


Figure 2-1. The distribution of genus-specific XC_{95} values for 163 CAM genera (from Figure 8, EPA 2011).

Australia/New Zealand (ANZ)

The ANZ approach to regulating salinity is through the use of "trigger values" (ANZECC and AMCANZ, 2000). Trigger values are concentrations that may pose potential environmental risk. As such, they "trigger" the need for further testing and/or analysis. Rather than employing a "one size fits all" approach for all freshwaters, the ANZ approach uses statistical distributions of salinity based on waterbody types (e.g., wetlands, small streams) and ecological settings (equivalent to an Ecoregion approach). For example, the 80th percentile of salinity distributions for a given waterbody type in a given region is used as a "trigger value". As such, they "trigger" the need for further testing and/or analysis. Another component to this approach is the consideration of the conservation value of individual waterbodies or stream segments. Here, the natural status of surface waters are assigned to one of three different classes with respect to conservation. Section 3.1.3.1 of the ANZ Guidelines document titled "Ecosystem Condition and Levels of Protection" recognizes three categories of ecosystem conditions: (1) High conservation/ecological value systems; (2) Slightly to moderately disturbed systems; and (3) Highly disturbed systems. "The third ecosystem condition recognises that degraded aquatic ecosystems still retain, or after rehabilitation may have, ecological or conservation values, but for practical reasons it may not be feasible to return them to a slightly–moderately disturbed condition."

South Africa

The South African approach for regulating TDS shares some similarities with the ANZ approach in that is it based on a statistical approach (Department of Water Affairs and Forestry, 1996). For all inland waters, TDS concentrations should not be changed by >15% from normal cycles of the of the waterbody under un-impacted conditions at any time of the year; and the amplitude and frequency of natural cycles in TDS concentrations should not be changed. The regulatory document recognizes different natural salinities as a function of underlying geologies, and considers that high "evapoconcentration" rates have an elevating influence on TDS concentrations.

2.2 Chloride

Chloride toxicity is perhaps the best studied/understood of all of the major ions. Robust acute and chronic datasets have been used to generate criteria at the national level in Canada and the U.S., the Canadian provincial level (British Columbia), and at the state level (e.g., the state of Iowa). A review of the ECOTOX database did not identify any new data relevant for the purposes of this study. Updated U.S. federal criteria are being considered based on the development of Iowa's criteria, but at the current time, the 1988 values are still the criteria recommended for use by the EPA (see below).

2.2.1 Chloride Water Quality Criteria

In 1988, EPA recommended an Ambient Water Quality Criterion for Chloride (EPA, 1988) based primarily on available data for sodium chloride (it is interesting to note that EPA acknowledged that chloride salts of K, Mg and Ca were generally more toxic than Na salts, but decided not to include these data in their criteria derivation). EPA's CMC (for acute exposure) was set at 860 mg/L, and the CCC (for chronic exposure) was set (based on the mean ACRs of only three taxa) at 230 mg/L (**Table 2-2**). Since that time, there has been a considerable volume of data added to our knowledge base of chloride toxicity, and recently, new criteria have been developed for British Columbia, Canada (Nagpal et al., 2003) and Iowa (2009) (Table 2-2). A nationwide standard for Canada was established in 2011 (Table 2-2). Several other entities are currently considering chloride criteria.

Evans and Frick (2001) provide a thorough and detailed evaluation of chloride in support of Canada's decision to regulate chloride as a priority 2 toxic substance. Though they did not establish regulatory values *per se*, their compilation of toxicity data has been instrumental in subsequent Canadian regulatory approaches to chloride (See **Appendix A, Table A-1**). Though their focus was on road salts (primarily Na salts), the authors provide tabulations of toxicity data for Ca, Mg and K chloride salts as well.

Recent research conducted by the Great Lakes Environmental Center (GLEC) and the Illinois Natural History Survey (INHS) on the interactions of water chemistry (hardness and sulfate) on chloride toxicity have been incorporated into chloride standards developed by the State of Iowa (2009). Briefly, it was found that for some test species, chloride toxicity was decreased with increasing water hardness ($CaCO_3$) and sulfate concentrations. Details of those finding and their utility in developing site specific chloride criteria are described below.

Table 2-2. Chloride criteria for the U.S., British Columbia, Canada and Iowa (see text for rationale)

Entity	Acute (mg/L)	Chronic (mg/L)	Notes
US EPA (1988)	860[a]	230[b]	Based on NaCl only
British Columbia, Canada	600[c]	150[d]	Based on *T. tubifex* data only
Canada	640	120	Based on data for sodium and calcium chloride salts only
Iowa	$= 287 * [\text{hardness}]^{0.205797} * [\text{sulfate}]^{-0.07452}$	$= 177.87 * [\text{hardness}]^{0.205797} * [\text{sulfate}]^{-0.07452}$	See text below

[a] CMC (not to be exceeded for more than 1 hour every three years)

[b] CCC (typically to be implemented as a 4-day average)

[c] Instantaneous maximum

[d] The average of 5 weekly measurements taken over a 30-day period

2.2.2 Various Rationale for Acute and Chronic Chloride Criteria

The following sections provide the rationale or basis for the establishment of chloride criteria summarized in Table 2-2:

U.S. EPA Rationale

- *Acute*: The EPA CMC is driven by the lowest four GMAVs from the water flea *Daphnia* (1,974), snail *Physa* (2,540), isopod *Lirceus* (2,950) and midge *Cricotopus* (3,795) and a FAV of 1,720. A safety factor of two was applied to establish a CMC of 860.

- *Chronic*: The EPA CCC is based on chronic tests with fathead minnows, rainbow trout and *Daphnia pulex*, with ACRs of 15.17, 7.31, and 3.93, respectively. EPA applied the geometric mean of these three values (7.594) to the CMC (860/7.594) and rounded to the nearest 10.

British Columbia Rationale

The following language in quotations is taken directly from Nagpal et al. (2003) describing the rationale for the British Columbia chronic and acute values:

- *Acute*: "The guideline for maximum chloride concentration was derived by applying a safety factor of two to the 96-h EC_{50} of 1,204 mg/L for the tubificid worm, *Tubifex tubifex* – the most sensitive species in their data set (**Appendix A, Table A-2**), and rounding the number to nearest tenth. Safety factor of two is applied to the acute data because of the relative strength of the acute data set."

- *Chronic*: "The recommended water quality guideline was derived by dividing the lowest LOEC (lowest observed effect concentration) from a chronic toxicity test by a safety factor of five. The lowest LOEC for a chronic toxicity test is 735 mg/L for *Ceriodaphnia dubia* (**Appendix A, Table A-3**); this chloride concentration resulted in a 50% reduction in reproduction over the 7 day test duration. Utilizing this value and following application of a safety factor of five, the chronic guideline is 150 mg/L (rounded to nearest tenth place)…The safety factor of five in the derivation of the chronic guideline was justified as follows: (a) Chronic data (Appendix A, Table

A-2) available from the literature were scant; (b) in a recent study, Diamond et al. (1992) found a LOEC/NOEC ratio for reproduction of 3.75 in *C. dubia* exposed to NaCl for 7 days. Also, LC_{50}/LC_0 of 3 and LC_{100}/LC_0 of 4 were obtained by Hughes (1973), whereas the DeGreave et al. (1992) data yielded LC_{50}/NOEC ratios that ranged from about 1.0 to 6.9; (c) additional protection may be required for those species that are more sensitive but have not yet been tested in the literature."

Canadian Rationale

It was determined that toxicity resulting from tests using $MgCl_2$ or KCl resulted from Mg or K toxicity, and thus were not included in the derivation of chloride criteria for Canada. For chloride, the specific rationale is as follows:

- *Acute***:** "Derived with severe-effects data (such as lethality) and are not intended to protect all components of aquatic ecosystem structure and functions but rather to protect most species against lethality during severe but transient events (e.g., inappropriate application or disposal of the substance of concern").

  The Canadian approach was to use a SSD of acute toxicity values (**Figure 2-2**, see **Appendix A, Table A-4** for a table of values). A log-normal distribution fits these data best, with 640 mg chloride/L representing the 5th percentile of the distribution. The lower and upper 90% confidence bands about the 5th percentile were 605 mg/L and 680 mg/L, respectively.

  One noteworthy observation from these data, is that glochidia (mussel larval stage) tests with endangered freshwater mussel species *Epioblasma torulosa rangiana* (244 mg/L) (Gillis, 2011), and a 48-hour EC_{50} for immobilization in *Daphnia magna* (621 mg/L) (Khangarot and Ray, 1989) fall below the 640 mg/L criterion. Data from other freshwater mussel tests (*Lampsilis fasciola* and *Lampsilis siliquoidea*) suggest that freshwater mussels may be the most sensitive group to chloride yet studied.

- *Chronic:* "Derived with mostly no- and some low-effect data and are intended to protect against negative effects to aquatic ecosystem structure and function during indefinite exposures…Long-term exposure guidelines identify benchmarks in the aquatic ecosystem that are intended to protect all forms of aquatic life for indefinite exposure periods. Long-term exposure guidelines are derived using long-term data (≥ 7-day exposures for fish and invertebrates, ≥ 24-hour for aquatic plants and algae)."

  The Canadian approach was to use the SSD of chronic toxicity values (**Figure 2-3**, see **Appendix 2, Table A-4** for a table of values). A log-logistic distribution fit these data best, with 120 mg chloride/L representing the 5th percentile of the distribution. The lower and upper 90% confidence bands about the 5th percentile of the distribution were 90 mg/L and 150 mg/L, respectively.

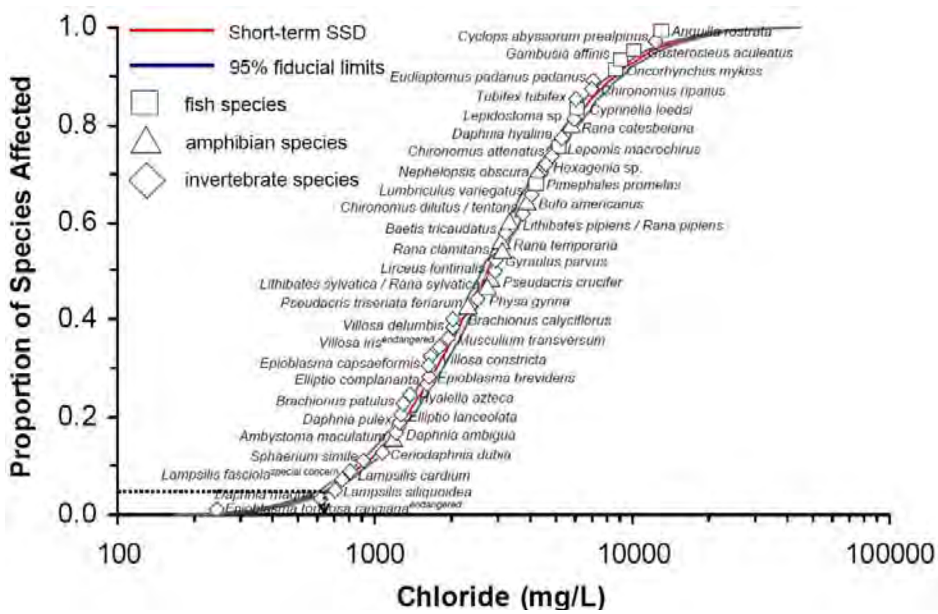


Figure 2-2. SSD of short-term L/EC_{50} toxicity data for the chloride ion in freshwater derived by fitting the Normal model to the logarithm of acceptable toxicity data for 51 aquatic species versus Hazen plotting position (proportion of species affected). The arrow at the bottom of the graph denotes the 5th percentile and the corresponding short-term benchmark concentration value (Source: Figure and caption from CCME, 2011)

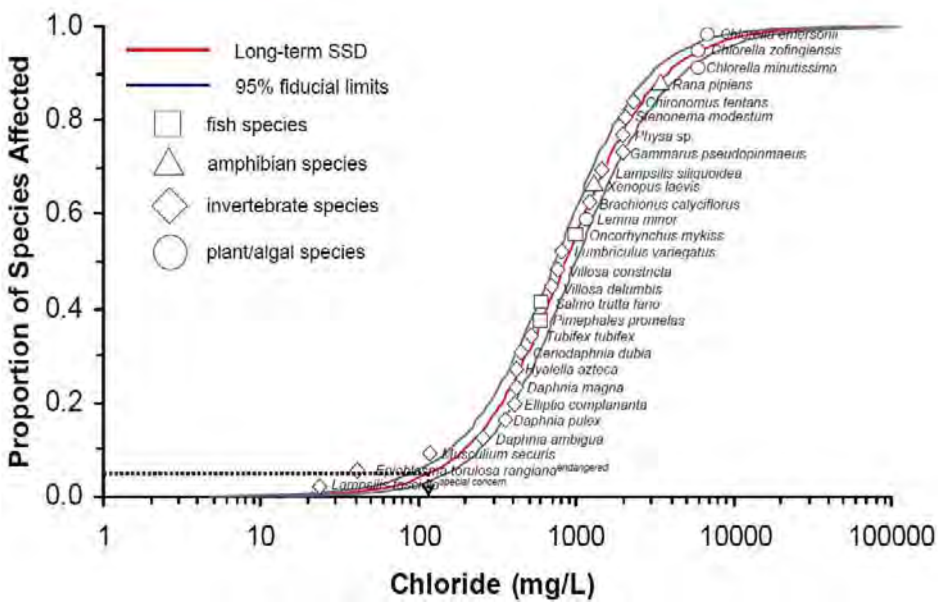


Figure 2-3. SSD of long-term no- and low-effect endpoint toxicity data for the chloride ion in freshwater (where mussels are present) derived by fitting the Logistic model to the logarithm of acceptable data for 28 aquatic species versus Hazen plotting position (proportion of species affected). The arrow at the bottom of the graph denotes the 5th percentile and the corresponding long-term Canadian Water Quality Guideline value (Source: Figure and caption from CCME, 2011).

As was the case with the acute data, tests with the glochidia stages of freshwater mussels (Gillis, 2011; Bringolf, 2010) fell below the 120 mg Cl^-/L criterion. The Protection Clause in the Canadian Guidelines gives entities the option of establishing lower criteria values to protect commercially, recreationally, or ecologically important species. Thus, this value may not apply (it may be lowered) when endangered mussels are present in a given waterbody.

Another study (Karraker and Gibbs, 2011)based on mass changes in spotted salamander (*Ambystoma maculatum*) eggs transferred to clean water following exposure to chloride was not included in the calculation of the Canadian chronic value.

With regards to toxicity modifying factors, the following paragraph was taken verbatim from CCME, 2011 Canadian Water Quality Guidelines (CWQG) for the Protection of Aquatic Life - Chloride document:

> "Some studies have indicated that increased hardness may have an ameliorating effect on the toxicity of chloride. One long-term study by Elphick et al*.* (2011) assessed the effect of hardness (10, 20, 40, 80, 160, 320 mg/L as $CaCO_3$) on sodium chloride toxicity to the water flea *Ceriodaphnia dubia* during a 7-day exposure. An approximate 4-fold difference was observed in the 7-day $IC_{25}/_{50}$ (reproduction) effect concentrations, and a 9-fold difference in 7-day LC_{50} concentrations over the hardness range tested. Gillis (2011) exposed glochidia of the freshwater mussel *Lampsilis siliquoidea* to water of varying hardness (47, 99, 172, 322 mg/L as $CaCO_3$). An approximate 2.5-fold difference in 24-hour EC_{50} (glochidia survival) values was observed over the hardness range tested. GLEC and INHS (2008) also conducted some short-term exposures indicating the existence of a hardness-chloride toxicity relationship for the water flea *Ceriodaphnia dubia*, the fingernail clam *Sphaerium simile*, the oligochaete *Tubifex tubifex* and the aquatic snail *Gyraulus parvus*. Insufficient data were available to develop a hardness relationship for chronic toxicity and thus, a hardness-based CWQG was not developed. CCME will re-visit the chloride guidelines when sufficient studies are available. Jurisdictions may choose to derive site-specific hardness-adjusted water quality criteria (or objectives) where appropriate."

Iowa Rationale

Prior to 2009, Iowa did not have a chloride standard for the protection of aquatic life, but used EPA (1988) chronic (230 mg/L) and acute (860 mg/L) chloride values to trigger whole effluent toxicity testing. The IDNR worked with Dr. Charles Stephan (EPA) to compile data for updated criteria. That literature review not only provided several studies conducted post-1988, but also identified an important early study by Wurtz and Bridges (1961) that had not been used in EPA's initial criteria development. The Wurtz and Bridges study included data for two species – a snail (*Gyraulus circumstriatus*), and the fingernail clam (*Sphaerium tenue*) which appeared to be sensitive to chloride. Additionally, the Khangarot (1991) study with *Tubifex tubifex* (the study that drives the British Columbia criteria) suggested that this species may also be sensitive to chloride. The addition of new data resulted in an FAV of 1,364 mg/L, which would result in a new CMC (acute criterion) of 682 mg/L after a safety factor of two is applied (but Iowa did not choose this option).

EPA contracted with the GLEC and the INHS to explore the influences of hardness and sulfate on chloride toxicity to sensitive aquatic taxa – *Ceriodaphnia dubia, Sphaerium simile, Gyraulus parvus* and *Tubifex tubifex* (**Figures 2-4 and 2-5**).

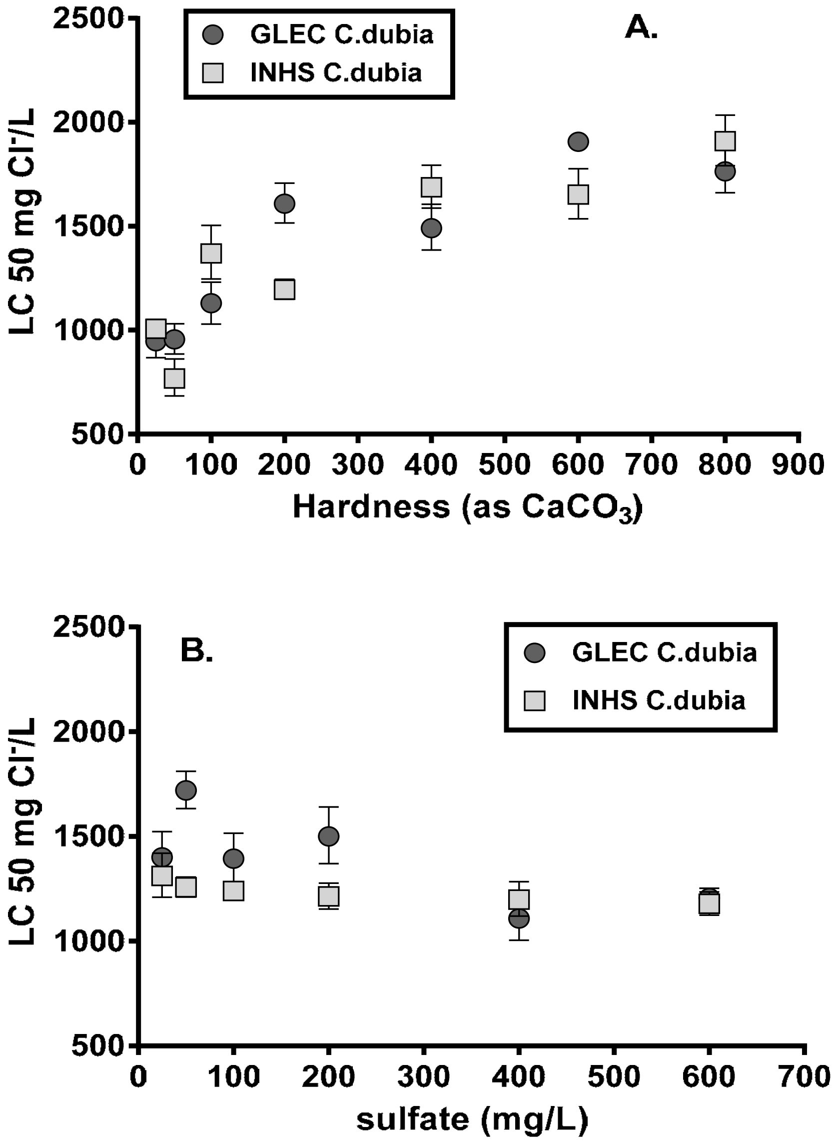


Figure 2-4. Influence of water hardness (as $CaCO_3$) (Panel A) and sulfate (Panel B) on the toxicity of chloride in parallel studies conducted by GLEC and the INHS. The exponents associated with hardness in the Iowa criteria (0.205797) are primarily based on these data (but see Figure 2-5 below). The exponent associated with sulfate in the Iowa criteria (-0.07452) is based primarily on the data shown in Panel B. These data were plotted from the Table 2 data in the IDNR (2009) report (see Appendix A, Table A-5).

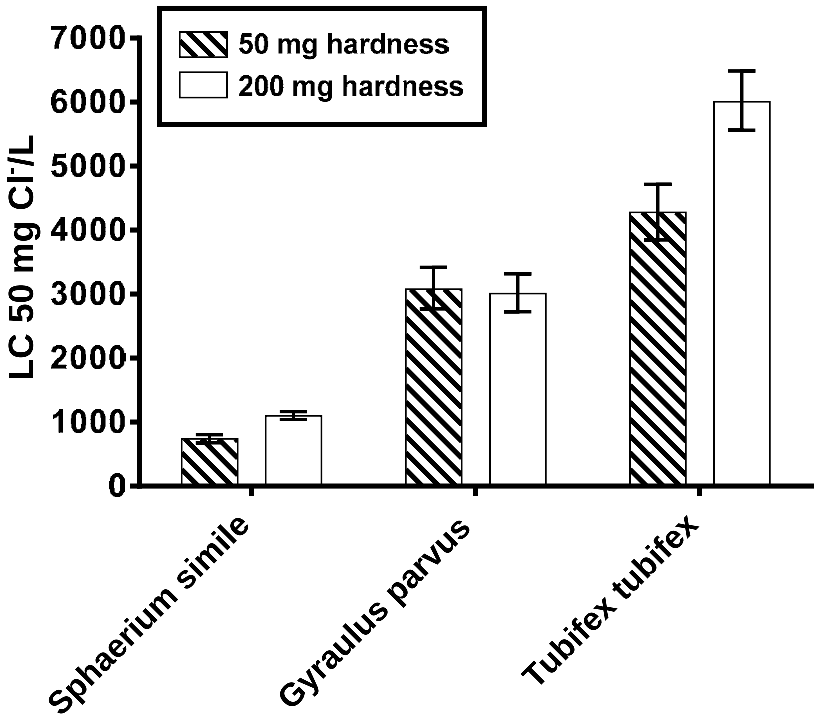


Figure 2-5. Influence of water hardness (as $CaCO_3$) on the acute toxicity of chloride to the fingernail clam (*Sphaerium simile),* planorbid snail *(Gyraulus parvus),* and tubificid worm *(Tubifex tubifex).*

- *Acute*: Based on the relationships between chloride toxicity and hardness (Figures 2-4A and 2-5) and sulfate (Figure 2-4B), Iowa evaluated the following four options below, and chose Option C to establish an acute criterion:
  - Option A**:** Acute values were not normalized for either hardness or sulfate and the criterion is not dependent on either hardness or sulfate;
  - Option B**:** Acute values were not normalized for either hardness or sulfate, but the criterion is dependent on both hardness and sulfate;
  - Option C: Acute values were normalized for both hardness and sulfate and the criterion is dependent on both hardness and sulfate (**Table 2-3**);
  - Option D: Acute values were normalized for hardness (but not sulfate) and the criterion is dependent on hardness (but not sulfate).
- *Chronic*: EPA's 1988 chronic chloride criteria relied upon chronic data for only three species (see above), applying the geometric mean value (7.594) in acute to chronic ratios. For Iowa's chronic value, the fathead minnow ACR (15.17) was discarded, and newer ACR values were considered from studies with *Ceriodaphnia dubia* (1.508, 2.601 and 3.841), *Daphnia ambigua* (4.148), and *Daphnia magna* (1.974). These values generated a CCC (chronic criterion) of 417 mg/L (substantially higher than EPA's original value of 230 mg/L). Thus, Iowa considered different options for applying ACRs as follows:

- CCC1: Derived using ACR = 4.826, which is the geometric mean of the ACRs for Rainbow Trout and *Daphnia*. CCC1 is too high for species at the 5th percentile.
- CCC2: Derived using ACR = 3.187, which is the ACR for *Daphnia*. CCC2 is appropriate for species at the 5th percentile.
- CCC3: Derived from predicted Genus Mean Chronic Values that were calculated using ACR = 7.308 of Rainbow Trout for all vertebrates and ACR = 3.187 of *Daphnia* for all invertebrates. Then the similar procedure for deriving acute criterion was used to derive the chronic criterion (**Table 2-4**).

Iowa opted for CCC3 under Option C.

Table 2-3. Example of relationship between hardness and sulfate and calculated acute criterion[1]

FAV	Hardness	Sulfate	Hardness Exponent	Sulfate Exponent	Adjusted Criteria
287	25	25	0.2057979	-0.07452	**287**
287	50	50	0.2057979	-0.07452	**513**
287	100	100	0.2057979	-0.07452	**553**
287	200	200	0.2057979	-0.07452	**634**
287	400	400	0.2057979	-0.07452	**694**
287	800	800	0.2057979	-0.07452	**777**

[1]The FAV (FAV=574/2 =287) is adjusted based on water hardness and sulfate as follows:
287 * [hardness]$^{0.205797}$ * [sulfate]$^{-0.07452}$

Table 2-4. Example of relationship between hardness and sulfate and calculation of chronic criterion[1]

ACR adjusted FCV	Hardness	Sulfate	Adjusted Toxicity Value
177.87	50	200	**268.1**
177.87	100	200	**309.2**
177.87	200	200	**365.6**
177.87	300	200	**387.6**
177.87	100	300	**300.0**
177.87	100	400	**293.6**
177.87	100	500	**288.8**
177.87	100	600	**284.9**

[1]Acute toxicity values were adjusted via ACRs (adjustment: 177.87*[hardness]$^{0.205797}$ * [sulfate]$^{-0.07452}$

Technically, there are some issues to consider should the Central Valley decide to incorporate water chemistry (e.g., hardness) adjustments to existing datasets as Iowa has opted to. First, the application of the hardness adjustment generated from only one species (*Ceriodaphnia dubia)* to all other taxa is troubling. The justification used by Iowa was that the mean of the slope values taken from *Sphaerium simile, Gyraulus parvus* and *Tubifex tubifex* data (see Figure 2-4) was similar to the *Ceriodaphnia dubia* data. It is important to note that these "slopes" are based on two data points per species, and that one of these species, the planorbid snail *(Gyraulus parvus)* showed no influence of hardness whatsoever. This calls into question whether the hardness adjustment would under-protect sensitive species whose sensitivity to chloride is less affected by water hardness than is *Ceriodaphnia dubia*.

Another issue to consider is the application of ACRs to establish chronic criteria. The EPA and State of Iowa approaches are arbitrary and based on a very small number of species. The Canadian approach to chronic criteria is much more transparent – and thus is easier to communicate to stakeholders.

2.2.3 Chloride Summary

- Different approaches have been established for deriving chloride criteria in the U.S. and Canada at both national and state/provincial levels *(Note: EPA may update its national criteria recommendations in the near future)*.
- Of the approaches outlined, it appears that the Canadian approach is the most transparent and considers adjustments on a site-specific basis based on the presence/absence of endangered mussel populations.
- The Iowa approach makes adjustment for chloride and sulfate, whereas Canada did not consider that sufficient data existed to make such adjustments.

2.3 Boron

2.3.1 Overview

The Central Valley Water Board (1999) report provides a broad overview of boron with respect to different beneficial uses. At a first approximation, it would appear that criteria developed to protect sensitive crops should also be protective of aquatic life uses. The DOI Guidelines document (DOI, 1998) concurs with this notion in their table of predicted boron effects, with levels of concern for: (a) crops and aquatic plants listed at 0.5 - 10 mg/L (No Observed Adverse Effect Level [NOAEL] - Lowest Observed Adverse Effect Level [LOAEL]); (b) aquatic invertebrates (*Daphnia magna*) at 6 - 13 mg/L (NOAEL – LOAEL); and (c) fish at 5 - 25 mg/L (NOAEL – LOAEL). However, data for boron is still fairly limited (**Table 2-5**).

There appears to be considerable consternation regarding data generated by Birge and Black (1977), where a 0.1 mg/L concentration was reported to cause toxicity in rainbow trout embryo-larval tests. Follow up studies (Black et al., 1993) conclude that, "the flat concentration-response curve observed for boron (i.e., small changes in effects relative to large increases in boron concentrations) sometimes affected precision in the determination of no-effect or threshold concentrations." They further state, "…a concentration of between 0.75 and 1.0 mg/L is determined to be a reasonable, environmentally acceptable limit for boron in aquatic systems." In his 1998 review, Howe (1998) lists 1-2 mg/L as a common NOEC for community level studies, and 1 mg/L as a NOEC for studies with fish.

An analysis of ECOTOX for the toxicity of boron is provided below (raw data are provided in **Appendix A, Tables A-6 and A-7)**. Borates are the predominant form of boron in natural waters, and ECOTOX contained substantial data for the toxicity of boron as sodium borate ($NaBO_4$). Species sensitivity distributions were generated by ranking the toxicity values and creating centiles such that the cumulative percentage of species affected could be plotted against concentration. Analyzing the data in this manner allows for the estimate of an HC_{05} – the concentration of contaminant expected to be protective of 95% of the species in a given community. The inherent assumption in this EPA methodology is that the population of species in a toxicity dataset is a reasonable proxy for the population of species in real ecosystems.

The following four analyses were conducted from the existing boron data:

(1) Only short term exposures (acute studies) with lethality endpoints (**Figure 2.6A**).

(2) Only short term exposures (acute studies) with both lethality and non-lethality endpoints (e.g., immobility, growth) combined (**Figure 2.6B**).

(3) Only long term exposures (chronic studies) with lethality endpoints (**Figure 2.7A**).

(4) Only long term exposures (chronic studies) with both lethality and non-lethality endpoints (e.g., growth, reproductive output) (**Figure 2.7B**).

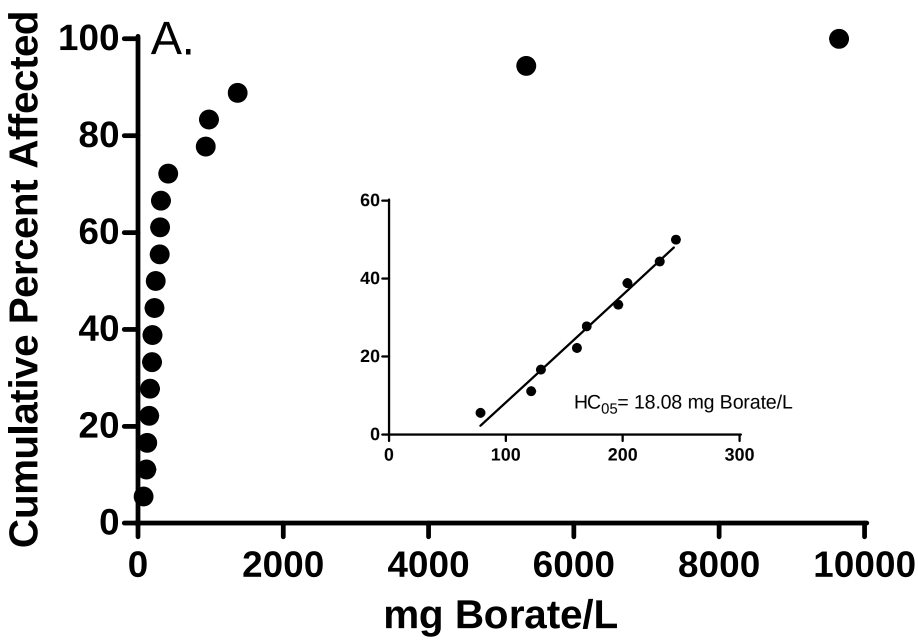
Table 2-5. Predicted boron effect levels (Source: DOI, 1998; see original reference for references in table).

Medium	No Effect (NOAEL)	Level of Concern	Toxicity Threshold (LOAEL)	Explanation
Water (mg/L)	0.5	0.5 - 19	10	For crops and aquatic plants (Perry et al., 1994)
	6	6 – 13	13	For aquatic invertebrates (NOAEL and LOAEL for *Daphnia magna*)
	5	5 – 25	25	For fish *(viz.,* catfish and trout embryos; Birge and Black, 1977; Perry et al., 1994)
	--	--	< 200	For amphibians (LC_{100} for leopard frog embryos)
Bird Eggs (mg/kg fw)	13	13 – 20	20	Smith and Anders (1989), Stanley et al. (1996); 20 = EC_{10} for viability of mallard eggs
Waterfowl Diet (mg/kg)	--	> 30	--	LOAEL for mallards; impaired growth of ducklings
Mammal Diet (mg/kg bw/day)	--	> 80	--	LOAEL for rodents; decreased fetal body weight

For Figures 2.6 and 2.7 the larger plot shows the entire distribution of the toxicity data. The smaller insets show only the sensitive tail of the sensitivity distribution and the linear regressions used to estimate HC_{05} concentrations. No assumptions were made about the overall shapes of the distributions as linear fits to the truncated data fit the data well.

The HC_{05} estimate based on acute lethality only is 18.08 mg/L, whereas the inclusion of all acute endpoints results in an HC_{05} estimate of 3.99 mg/L. Chronically, HC_{05} estimates are similar whether lethality only (1.18 mg/L) or if all chronic endpoints are considered (1.57 mg/L).

A review of available boron literature by Loewengart (2001) makes a compelling argument that the shape of boron dose-response curves in rainbow trout suggests that boron may be essential, and low dose effects could be the result of boron deficiency. To support this argument, U-shaped dose response curves reported by Rowe et al. (1998), and the stimulatory effects of low concentrations of boron on growth (Eckert, 1998) are summarized in addition to field observations of vibrant trout populations occurring in high boron environments.



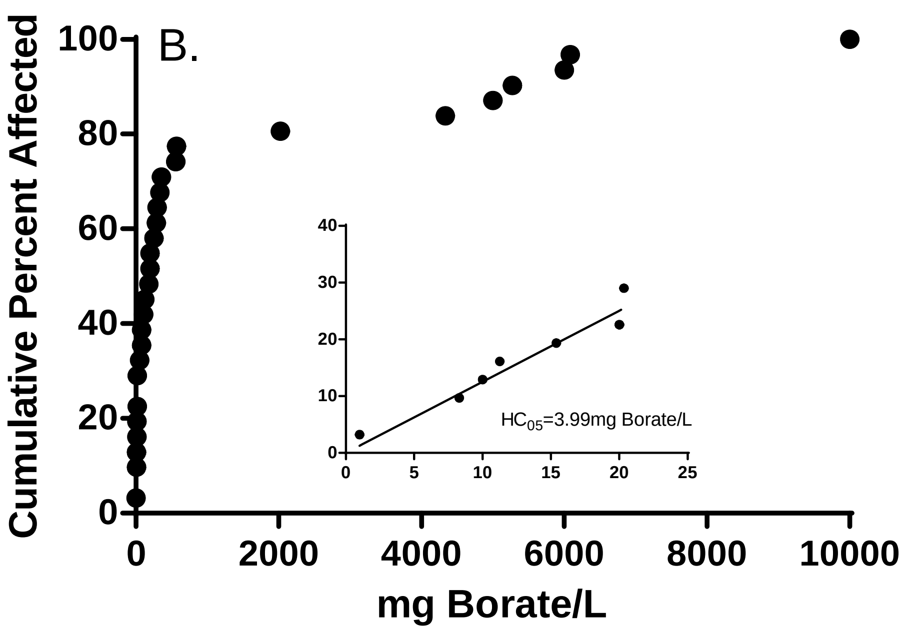


Figure 2.6. Distribution of the acute toxicity data of borate to aquatic organisms (lethal and non-lethal endpoints) based on ECOTOX data: A (top) - Distribution of the data based only on a lethality endpoint; B (bottom) - Distribution of data based on a combination of lethality and other endpoints (data from Figure 2.6A included).

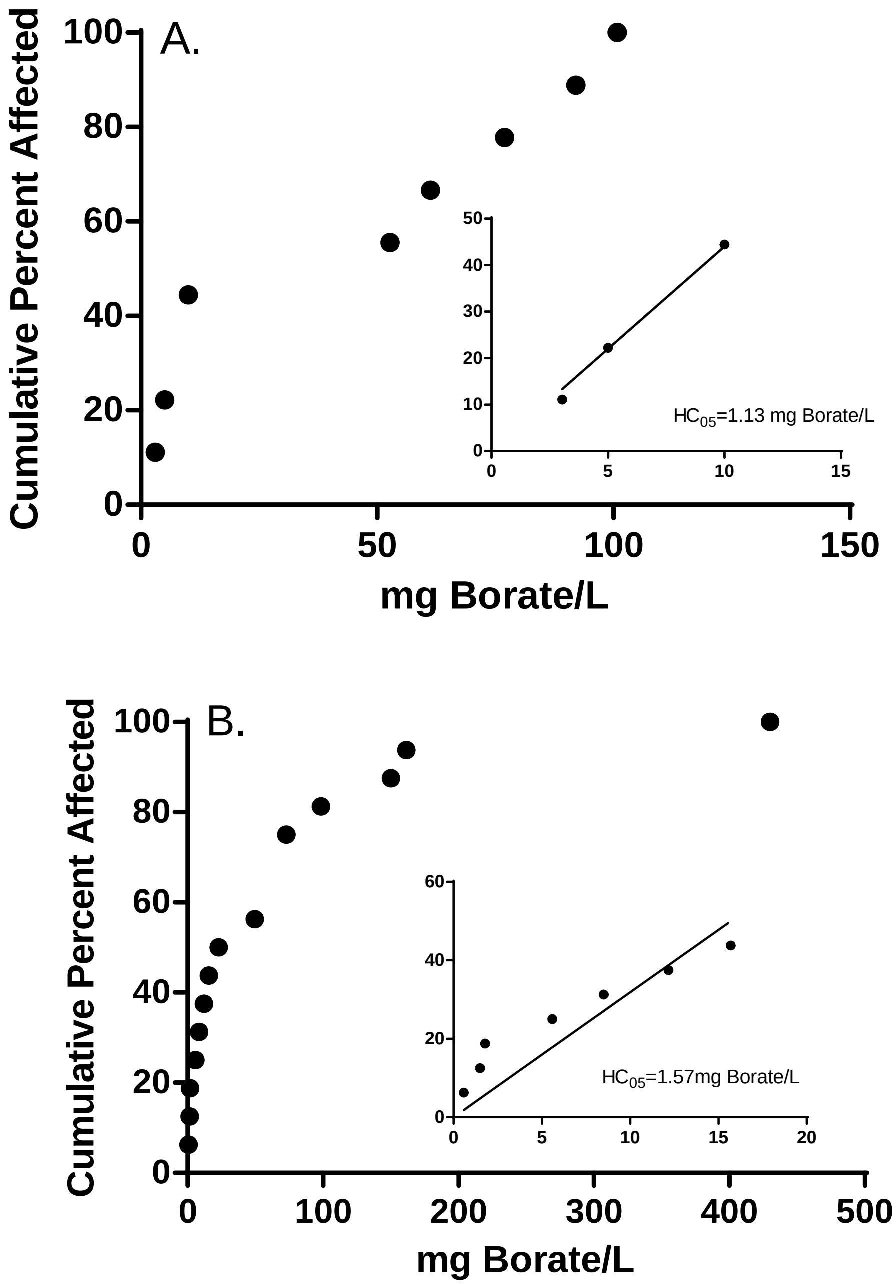


Figure 2.7. Distribution of the chronic toxicity data of borate to aquatic organisms (lethal and non-lethal endpoints based on ECOTOX data: A (top) - Distribution of data based on lethal endpoint; B (bottom) - Distribution of data based on both lethal and other endpoints (data from Figure 2.7A included).

A recent compilation of boron data in support of European Union REACH initiatives (dealing with the **R**egistration, **E**valuation, **A**uthorisation and Restriction of **Ch**emical substances) compared the traditional assessment (safety) factor and SSD approaches to infer a Predicted No Effect Concentration (PNEC) for boron in aquatic environments (Schoderboeck et al., 2011) (**Figures 2-8 and 2-9**). Their standard approach generated a PNEC of 0.18 mg/L and their SSD approach generated a PNEC of 0.34 mg/L.

A relatively new study by Soucek et al. (2011) tested boron with eight species, all of which resulted in typically high LC_{50} values for boron. pH and hardness did not affect boron toxicity, though chloride provided an ameliorative effect in studies with the crustaceans *Ceriodaphnia dubia* and *Hyallela Azteca*. Aspects of the Soucek et al. (2011) data are similar to those in other studies, where there are suggestions of subtle toxic affects at lower boron concentrations, but significant increases in boron concentration do not seem to increase toxicity. These authors conclude that the current standard for boron in Illinois (1 mg/L) is conservative.

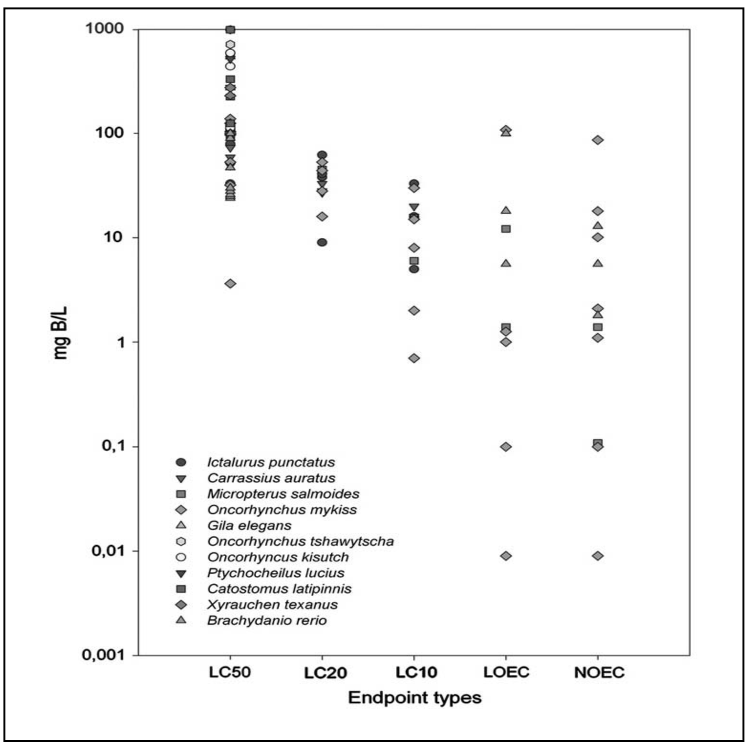


Figure 2-8. Compiled acute and chronic toxicity values for boron in fish (Source: Figure 2 in Schoderboeck et al., 2011).

At the request of CV-SALTS, additional review was conducted to evaluate boron toxicity for fish species that move freely from freshwater to the ocean. Relevant studies of such salmonids not cited in Schoderboeck et al. (2011) include the following:

- Hamilton and Buhl (1990): Acute boron LC_{50} values for chinook salmon and coho salmon fry were > 100 mg/L.

- Thompson et al. (1976): Tests conducted for 283 hours in "underyearling" coho salmon resulted in an LC_{50} value of 113 mg/L in freshwater and 12.2 mg/L in saltwater.

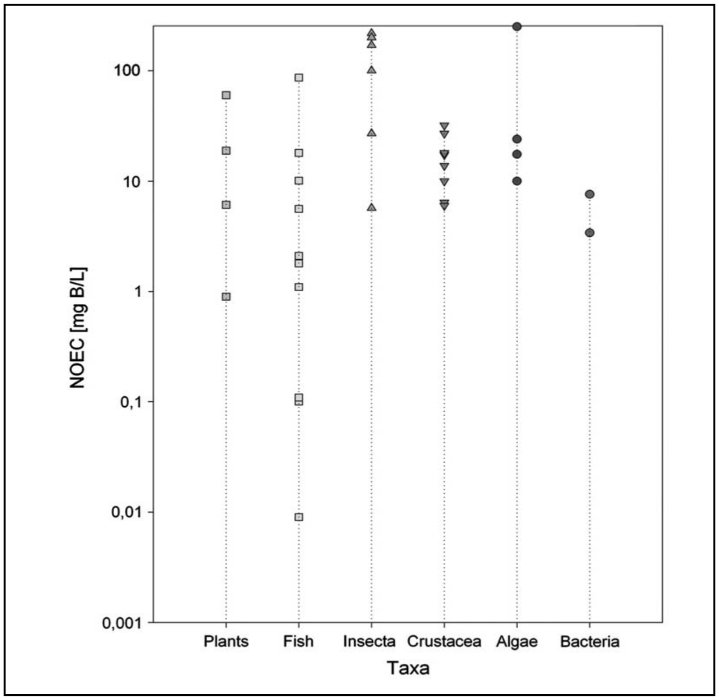


Figure 2-9. Available chronic toxicity values for boron in aquatic species of diverse taxonomic groups (Source: Figure 1 in Schoderboeck et al., 2011).

2.3.2 Boron Summary

- There is enough toxicity data by EPA standards (i.e., Stephan et al., 1985) to establish a boron criterion:
  - Acute toxicity data suggest that a concentration of 4 mg/L could be used to establish an acute criterion (see Figures 2.6 A, B).
  - Chronic toxicity data suggests that a chronic criterion of 1.0-1.3 mg/L could be established (see Figures 2.7 A, B).
- However, there remains considerable uncertainty regarding an appropriate criterion because of the following issues with the available toxicity data:
  - There is some disagreement about the nature of the few low toxicity values for boron (see discussion above), which may require re-evaluation of those data with respect to essentiality (deficiency) arguments.
  - Aquatic insects, an important part of the biological community, remain under-studied with respect to boron toxicity.

2.4 Sulfate

2.4.1 Overview

There is sufficient toxicity data available (by EPA standards) to generate water quality criteria/ objectives for sulfate, but few entities have established them to date. The IDNR (2009) report largely follows the lead of efforts in Illinois to establish hardness and chloride adjusted sulfate criteria, as shown in **Table 2-6**. This approach is based on an extremely limited set of species and as such, much uncertainty remains with respect to its protectiveness of aquatic communities. In particular, available data for sulfate toxicity is extremely limited for aquatic insects, which may be more sensitive to sulfate than other commonly used species. The limited available data suggests that aquatic insects may be more sensitive to sulfate than chloride:

- Bradley and Phillips (1977) report a higher sensitivity to sulfate than chloride in a salt tolerant mosquito. There is growing evidence that the same may be true for some mayflies. Goetsch and Palmer (1997) report that sodium sulfate was, "considerably more toxic to *Tricorythus sp.*, than sodium chloride." They further reiterate that "mortality cannot be linked only to conductivity or total dissolved solid (TDS) concentrations", but that the nature of the salt was important.

- Recent research (Kunz et al., 2013) used reconstituted high TDS waters to mimic water chemistries found in Central Appalachian streams affected by mountaintop coal mining. The data show that the mayfly *Centroptilum triangulifer* and freshwater mussel *Lampsilis siliquoidea* were highly sensitive to elevated TDS dominated by sulfate salts whereas commonly used test species *Ceriodaphnia dubia* and *Hyalella azteca* were relatively unaffected.

- Insects will undoubtedly be the predominant faunal group used in biological assessments of freshwater habitats in the San Joaquin River basin. It therefore makes sense to consider the potential for sulfate to be an issue in light of the apparent sensitivity of insects to sulfate.

- Sulfate and/or bicarbonate are the likely drivers of reduced macroinvertebrate (particularly mayflies) in West Virginia streams receiving high TDS inputs from surface coal mining.

The IDNR (2009) document points towards "the Illinois approach" (Illinois EPA, 2006), which asserts that, "unlike many toxicants that exert toxic effects over both short term and long term periods (acute and chronic toxicity), sulfate has been demonstrated to affect only short term survival of organisms. In other words, organisms that survive the initial osmotic shock of exposure will survive indefinitely at that concentration." Analysis of sulfate data from the ECOTOX database suggests that this line of reasoning is questionable[4].

Table 2-6. Proposed Iowa sulfate criteria based on binned chloride and hardness categories.

Hardness (mg/L as $CaCO_3$)	Chloride Concentration (mg/L)		
	$Cl^- < 5$	$5 \le Cl^- \le 25$	$25 \le Cl^- \le 500$
H < 100	500	500	500
100 ≤ H ≤ 500	500	[-57.478 + 5.79 (hardness) + 54.163 (chloride)] *0.65	[1276.7 + 5.508 (hardness) – 1.457 (chloride)] *0.65
H > 500	500	2,000	2,000

[4] The agency was not contacted for the purposes of this study; additional follow-up is recommended if these data were to be used for WQO development.

An analysis of the ECOTOX database for the toxicity of sulfate is provided below (raw data are provided in **Appendix A, Tables A-8 and A-9**). The ECOTOX database contained substantial data for the toxicity of sulfate as $NaSO_4$. Species sensitivity distributions were generated by ranking the toxicity values and creating centiles such that the cumulative percentage of species affected could be plotted against concentration. Analyzing the data in this manner allows for the estimate of an HC_{05} – the concentration of contaminant expected to be protective of 95% of the species in a given community. The inherent assumption in this EPA methodology is that the population of species in a toxicity dataset is a reasonable proxy for the population of species in real ecosystems.

The following four analyses were conducted from the existing sulfate data:

(1) Only short term exposures (acute studies) with lethality endpoints (**Figure. 2.10A**).

(2) Only short term exposures (acute studies) with both lethality and non-lethality endpoints (e.g., immobility, growth) combined (**Figure 2.10B**).

(3) Only long term exposures (chronic studies) with lethality endpoints (**Figure 2.11A**).

(4) Only long term exposures (chronic studies) with both lethality and non-lethality endpoints (e.g., growth, reproductive output) (**Figure 2.11B**).

For Figures 2.10 and 2.11, the larger plot shows the entire distribution of the toxicity data. The smaller insets show only the sensitive tail of the sensitivity distribution and the linear regressions used to estimate HC_{05} concentrations. No assumptions were made about the overall shapes of the distributions as linear fits of the truncated data forced through the origin fit the data well.

The HC_{05} for acute lethality was calculated as 464 mg SO_4/L. The inclusion of other non-lethal acute data lowered the HC_{05} calculation to 234 mg SO_4/L. The HC_{05} for chronic lethality data was estimated as 154 mg SO_4/L, whereas the inclusion of other chronic non-lethal endpoints lowered the HC_{05} estimate slightly to 124 mg SO_4/L.

2.4.2 Sulfate Summary

- There is enough toxicity data by EPA standards (i.e., Stephan et al., 1985) to establish a sulfate criterion:
    - Existing acute toxicity data for sulfate could be used to support an acute criterion of 234 mg SO4/L (see Figures 2.10 A,B).
    - Existing chronic toxicity data for sulfate could be used to support a chronic criterion of 124 mg SO4/L (see Figures 2.11 A,B).
- However, there remains considerable uncertainty regarding an appropriate criterion because of the following issues with the available toxicity data:
    - The lack of available aquatic insect data for sulfate raises considerable uncertainty about the protectiveness of these potential objectives to resident central valley aquatic communities.
    - The hardness and chloride effects on sulfate toxicity need to be examined in greater detail.

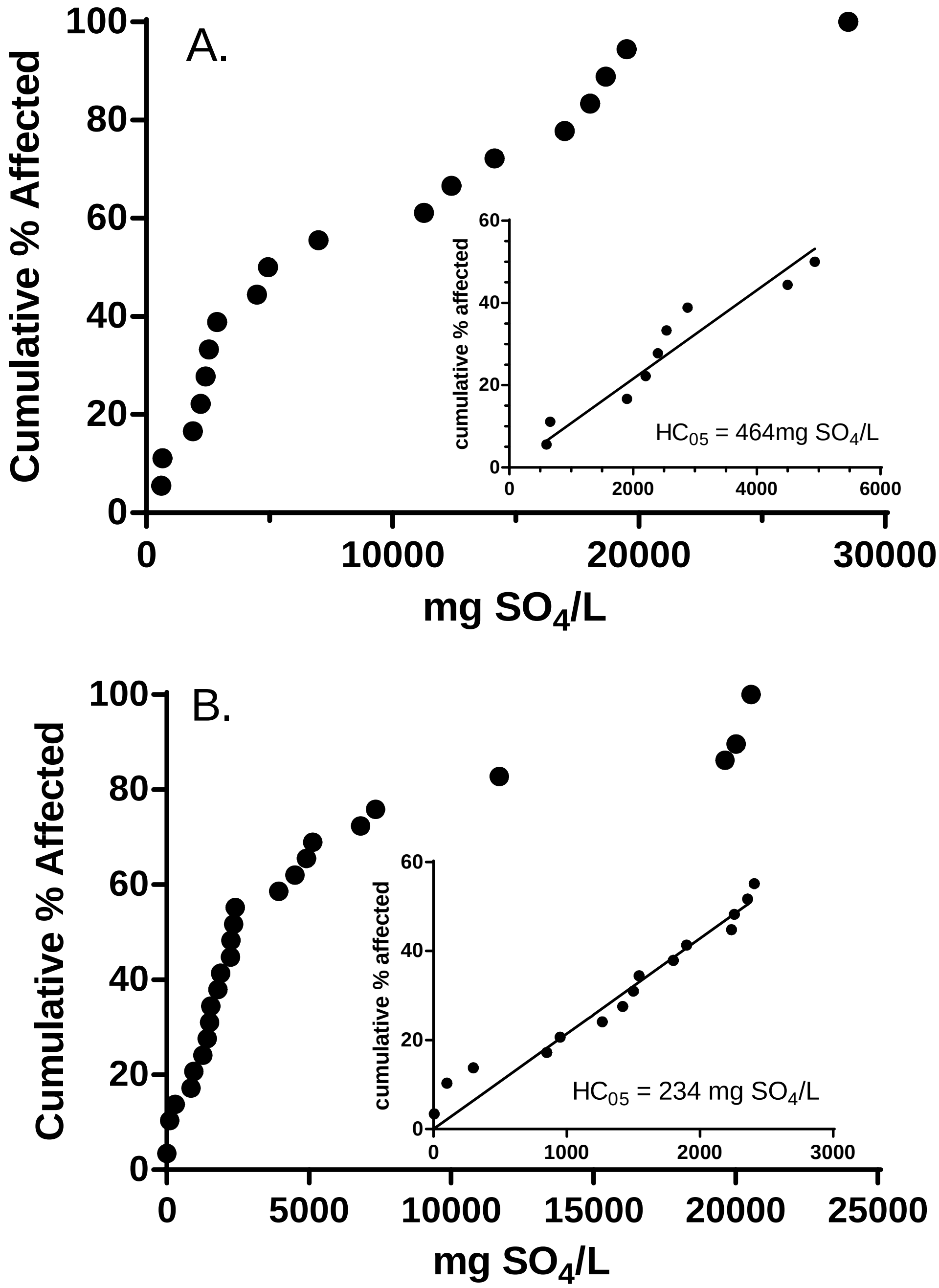


Figure 2.10. Distribution of the acute toxicity data of sulfate to aquatic organisms (lethal and non-lethal endpoints) based on ECOTOX data: A (top) - Distribution of the data based only on a lethality endpoint; B (bottom) - Distribution of data based on a combination of lethality and other endpoints (data from Figure 2.10A included).

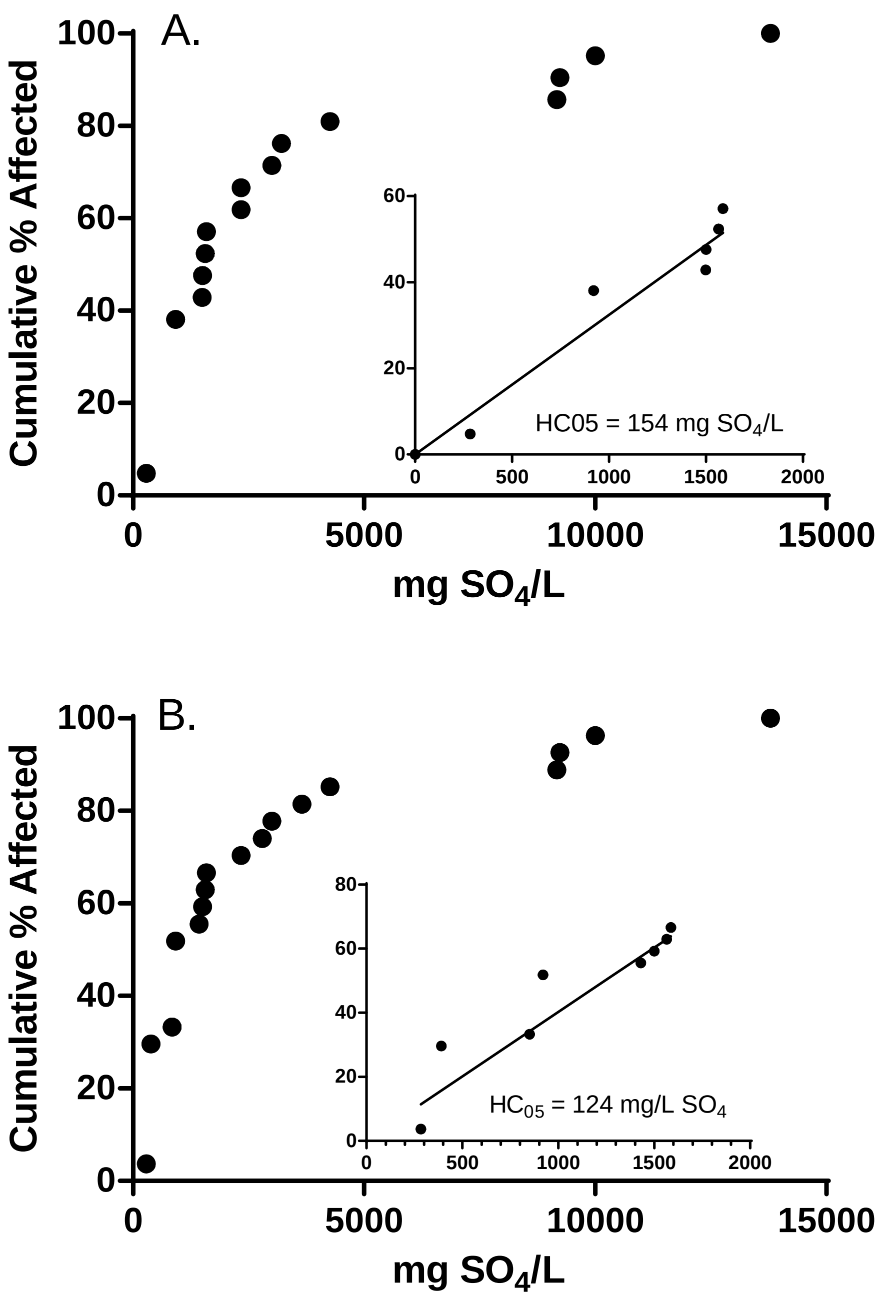


Figure 2.11. Distribution of the chronic toxicity data of sulfate to aquatic organisms (lethal and non-lethal endpoints based on ECOTOX data: A (top) - Distribution of data based on lethal endpoint; B (bottom) - Distribution of data based on both lethal and other endpoints (data from Figure 2.11A included).

2.5 Other Salinity-Related Constituents

Documents provided for review were limited with respect to the toxicity of other major ions (Na, Ca, Mg, K, CO_3, HCO_3 and hardness. In the case of Na, the most commonly used salt for toxicity testing has been NaCl, with toxicity generally being ascribed to the Cl ion rather than the Na ion. Evans and Frick (2001) provide limited data for other chloride salts ($CaCl_2$, $MgCl^2$, KCl), but in general, these salts remain relatively under-studied relative to the salts summarized above. Carbonate, bicarbonate and hardness toxicity remain poorly understood. Considerable uncertainty remains regarding the relative toxicity of the anions vs. cations in salt exposures.

Section 3

Applicability of Findings to the Central Valley

3.1 Applicability of Toxicity Data to Central Valley Fauna

The extrapolation of toxicity values from a handful of laboratory tested species to values intended to protect aquatic communities in nature has a high degree of uncertainty associated with it. Arguments that such approaches are protective cite the use of safety factors and conservative thresholds (extrapolation to protect 95% of species). Arguments that such approaches may not be protective point towards the fact that aquatic communities tend to be dominated by insects, yet toxicity datasets are dominated by crustaceans and fish. Regardless, the application of toxicity values to set water quality criteria is the norm in the U.S. and Canada, and local fauna are rarely given special consideration unless they have special economic, conservation or cultural value.

There may be compelling reasons to question whether the fauna from a given region are differentially adapted to particular salinities. For example, one could argue that taxa evolved in low TDS environments (e.g. Central Appalachian headwater streams, Sierra Nevada streams) may be more sensitive to elevated TDS than species that evolved in higher natural salinities (e.g., West slope of the Central Valley). While direct evidence for this concept is lacking, it is important to consider the status of the fauna intended to be protected by the development of WQOs here.

The wholesale conversion of the Central Valley to agricultural (and urban) land uses and the development of water supply and water conveyance systems has essentially rendered streams in this entire region of the state as significantly altered (degraded) from natural or reference conditions. Ode et al. (personal communication) report that only one Central Valley reference stream currently exists in California's Surface Water Ambient Monitoring Program (SWAMP) database of reference streams, and this stream is likely better classified as a Sierra Nevada foothills stream than a valley floor stream (Dr. Raphael Mazor, Southern California Coastal Water Research Project [SCCWRP] personal communication, 4/29/2013). Thus, the regional species pool in the Central Valley floor exists in highly degraded streams and likely represents a generally more tolerant/facultative fauna. It is likely that TDS sensitive fauna, if any were originally present, have already been excluded from streams with elevated TDS in the Central Valley.

3.2 Toxicity of Chloride, Boron and Sulfate in Relation to Water Chemistry Concentrations in the Central Valley

3.2.1 Interpretation of Resident Biological Communities in the Central Valley

The lack of streams in reference (or near reference) condition provides a major challenge to practitioners of biological assessment for the State of California, because scoring tools for streams are reliant on suitable suites of reference streams to adequately assess ecological conditions at a given site. What this means practically for the Central Valley, is that biological monitoring efforts will (at least in the short term) not be able to incorporate the newly developed scoring tools associated with the emerging biological objectives that may be implemented in the remainder of the state. This does not mean that monitoring of biological communities in the Central Valley is not feasible. The work of Leland and Fend (1998) (summarized below) provides an example of a well-executed monitoring study. What is important to consider however, is that biological communities are responsive to habitat and other water chemistry (e.g., pesticides and nutrients) that may co-occur with elevated natural or anthropogenic TDS. Biomonitoring efforts by themselves cannot adequately ascribe cause(s) of ecological impairment. Thus, any TDS/salinity objectives that are eventually established will initially be extremely difficult to link directly to changes in biological assemblages. However, should activities associated with the implementation of TDS/salinity objectives successfully reduce TDS concentrations in sites with historically elevated TDS concentrations, increases in biodiversity measures may be evident over time.

A study by Leland and Fend (1998) titled “Benthic invertebrate distributions in the San Joaquin River, California, in relation to physical and chemical factors” has several relevant pieces of information that should be considered by CV-SALTS as discussed below.

- Salinity has important geographic signatures:

  Samples were taken along the San Joaquin River and tributaries (**Figure 3-1**). Samples were collected every other week for one year, and monthly for an additional 16 months. Values are depth integrated across the river cross sections, and thus more robust than single grab samples.

  Results of chemical analysis (**Table 3-1**) show that the tributaries feeding the San Joaquin River from the east (Stanislaus River, Tuolumne River, Merced River) have a significant diluting effect on mainstem chemistry. In contrast, Mud Slough and Salt Slough (northeasterly flowing tributaries) contribute highly elevated salt concentrations to the San Joaquin River. Tributaries feeding the San Joaquin River from the west were not measured, but can be assumed to be higher in naturally occurring salinities based on the underlying geologies in that region.

- Biological communities respond to salinity gradients

  - Salinity was identified as a primary determinant of species assemblages, and is dominated by sulfate/bicarbonate. It remains uncertain whether sulfate, bicarbonate, combined ionic interactions or general ionic imbalance is the primary driver of species responses.

  - Substrate (sand grain size) was also an important determinant of invertebrate assemblages. This is important because the lack of reference systems in the Central Valley coupled with heterogeneity of substrate sizes seriously limits the ability to biomonitor effectively. Leland and Fend (1998) used standardized substrates to tease apart water chemistry vs. substrate effects. While an appropriate technical approach, my concern here is that the development of salinity guidelines for aquatic life uses cannot at this time be effectively tied to any meaningful biological monitoring efforts unless the use of standardized substrates are routinely employed.

  - Table 3 of Leland and Fend (1998) lists species-specific TDS optima for 73 species based on the densities of individual taxa collected as a function of salinity. A tolerance score is also provided and defined as one standard deviation (variance) around the optimal value. In many cases, the variances are relatively small, suggesting the species performance is strongly affected by salinity. Species with wider variances would appear to have a broader range of “comfortable” salinities. **Figure 3-2** below shows a cumulative ranking of species based on both the reported optima and two standard deviations from their optima (Leland and Fend, 1998) using the equation: Cumulative Percentile = (100 * Rank)/(n+1).

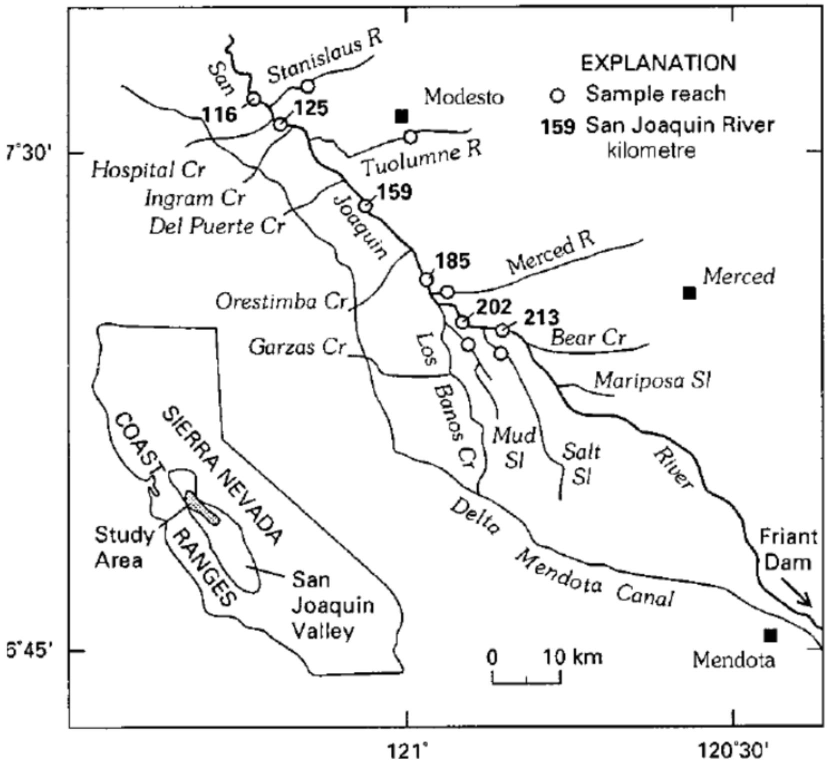


Figure 3-1. Sampling locations for water chemistry measurements reported from the San Joaquin River and tributaries from Leland and Fend (1998) (Data reported in Table 3-1).

Table 3-1. Major ion concentrations in the San Joaquin River and tributaries from Leland and Fend (1998). Total TDS values from Table 1 of that publication; remaining values were converted from mequiv/L units.

Ion	Central Valley Waterbody										
	San Joaquin River (River Km) (see Figure 3-1)						Stanislaus River	Tuolumne River	Merced River	Mud Slough	Salt Slough
	116	125	159	185	202	213					
Ca	22-34	24-46	36-54	32-60	36-86	19-68	8-12	4-15	9-16	60-142	54-112
Mg	10-17	9-22	16-29	16-30	15-49	7-30	3-5	2-7	3-5	42-70	24-60
Na	44-78	41-108	78-163	80-170	76-253	34-170	3-6	3-16	10-22	252-437	124-321
Cl	49-106	53-135	85-191	89-191	103-312	22-191	2-5	8-15	8-20	248-425	152-389
SO_4	53-91	57-110	105-187	101-207	101-331	20-269	3-8	6-9	10-14	394-672	172-442
HCO_3	67-104	73-128	98-152	73-128	104-183	79-159	33-52	28-79	40-73	140-238	128-195
Total TDS	250-350	270-540	420-690	420-740	410-970	200-620	54-78	55-140	74-140	1300-1700	650-1200

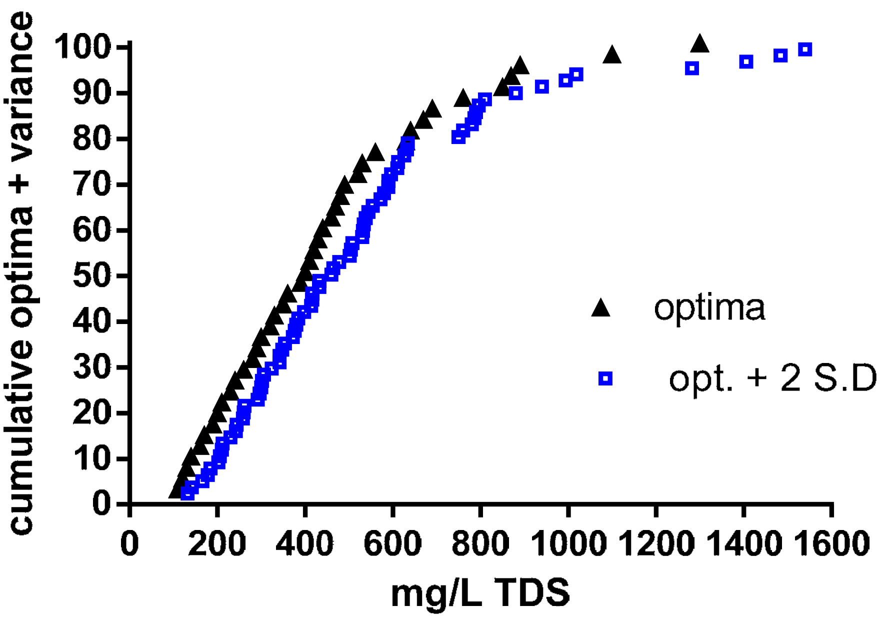


Figure 3-2. Cumulative percentiles for optima and variances (two standard deviations) for San Joaquin Valley invertebrates reported from Leland and Fend (1998).

Key features of the relationship shown in Figure 3-2 include:

- Approximately 50% of the taxa from this dataset have salinity optima of < 400 mg/L.
- The sampling coverage and intensity was likely not powerful enough to use these data on their own to make regulatory decisions because the sampling was not randomly/statistically determined.

At a first approximation, these data would appear similar to the data used by EPA to create field-based SSDs in Appalachian streams affected by surface coal mining (see Section 2). However, certain conditions associated with the CAM example do not hold true for California's Central Valley:

- The CAM situation is unique in that the affected habitats range from headwaters to relatively low order streams with little previous history of impairment and an ample pool of suitable reference streams available for comparative purposes. Central Valley streams have no such appropriate reference conditions from which to develop a regional scale, field-based criterion.
- The CAM example relies upon the idea that there are signature water chemistry changes (elevated sulfate and bicarbonate) exclusively associated with mining land uses, and that there are no natural sources of elevated TDS to confound the analysis. Further, the CAM situation and associated EPA benchmark implies that TDS related toxicity is derived from these relatively predictable water chemistry changes. However, there may be flaws in this assumption. In the Central Valley, the naturally higher conductivities of West slope tributaries can be confounded with the elevated conductivities associated with irrigation practices, resulting in a mixture of natural and anthropogenic sources of elevated TDS.

- The CAM example is set amid a backdrop of relatively high water quality, and the relative lack of other proximal stressors that may affect biological communities. In the Central Valley, both physical habitat alterations (sediments) and other water chemistry issues (e.g., pesticides, nutrients, temperature, selenium) preclude the ability of bioassessment tools to elucidate or assign causes of biological impairment to strictly TDS related issues (though Leland and Fend [1998] devised ways to control for physical habitat differences among sites, and note that, "Ephemeropterans in the San Joaquin River rarely occurred at salinities > 1,000 mg/L TDS"). Thus, the development of biotic survey approaches to infer TDS effects in streams appears possible with some effort, but is unlikely to be successful at this point without additional research effort.
- Another major difference is that here we use two standard deviations from the optimal score, whereas EPA essentially used extirpation salinities (salinities at which taxa failed to be effectively present in samples). It is unclear how the application of variances (e.g., one or two times the standard deviation) to optima scores would relate to an extirpation concentration for each taxon. Moreover, EPA's analysis was far more robust with respect to sampling intensity and included a substantial reference population.

For this study a list of 72 taxa for the single Central Valley reference site (Deer Creek) was obtained from the California reference database, where most taxa are identified to the genus level. This site is likely better thought of as "Sierra foothill site than a valley floor site" (R. Mazor, SCCWRP, personal communication). Of these 72 taxa occurring in the reference site, only 25 appear in the Leland and Fend (1998) San Joaquin study area. While it is not possible to determine why a relatively small number of reference taxa occurred in the San Joaquin study, it is perhaps instructive that the mean salinity optima reported by Leland and Fend for these taxa in the San Joaquin study area was 336 mg/L TDS. This suggests that the taxa present in both reference and San Joaquin sites had a relatively high salinity tolerance.

3.2.2 Water Chemistry in the Central Valley

The analysis of water chemistry in the Central Valley with respect to TDS or other salinity-related constituents was not part of the scope of work for this review. However, to assess the feasibility/attainability of any future WQOs, it would be useful to assess some of the spatial aspects of salinity related constituents in the Central Valley as well as the magnitudes of different constituents. Attempts to summarize existing California Environmental Data Exchange Network (CEDEN) data were hampered by a lack of documentation regarding the nature of the samples reported. Specifically, it could not be adequately determined if many reported values for salinity-related constituents were total (unfiltered) or dissolved (filtered) values. This greatly reduced the available dataset. **Table 3-2** summarizes observed water quality concentrations of dissolved sulfate, dissolved chloride, dissolved boron, and TDS as percentiles of the available data set for the Central Valley. With the exception of TDS, it is notable that the majority of dissolved data for salinity-related constituents is from Sierra Nevada foothill locations rather than from the floor of the Central Valley (**Figures 3-3, 3-4, 3-5 and 3-6** for TDS, chloride, sulfate and boron , respectively). Based on these limited data it difficult to make any useful findings regarding typical water quality in the Central Valley for these constituents. As a result it is not possible to reliably assess the feasibility of attaining potential WQOs for these constituents at this time.

Table 3-2. Water quality characteristics of salinity-related constituents in the Central Valley in terms of percentiles

Statistic	Dissolved Sulfate (mg/L)	Dissolved Chloride (mg/L)	Dissolved Boron (mg/L)	TDS (mg/L)
N	186	222	35	4711
10%	0.585	0.351	0.0266	39
25%	1.285	0.6625	0.041	85
50%	3.04	2.65	0.135	220
75%	9.68	9.2775	0.225	580
90%	19.95	42.58	0.902	1,100
95%	32.325	63.455	1.083	1,700

Note: Percentiles calculated from all available CEDEN database (provided courtesy of Melissa Turner, Michael L. Johnson, LLC)

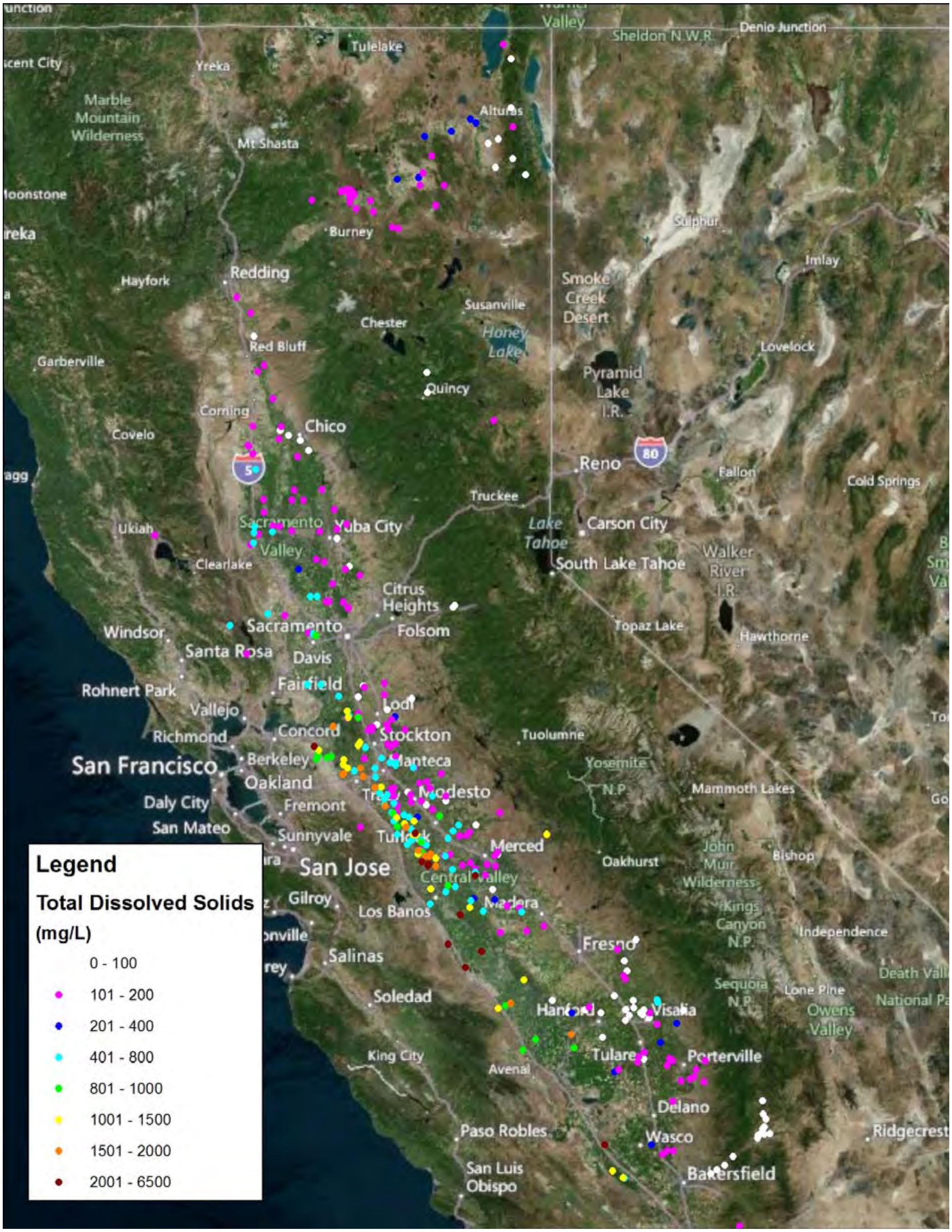


Figure 3-3. Sample locations of Central Valley TDS data summarized in Table 3-2.

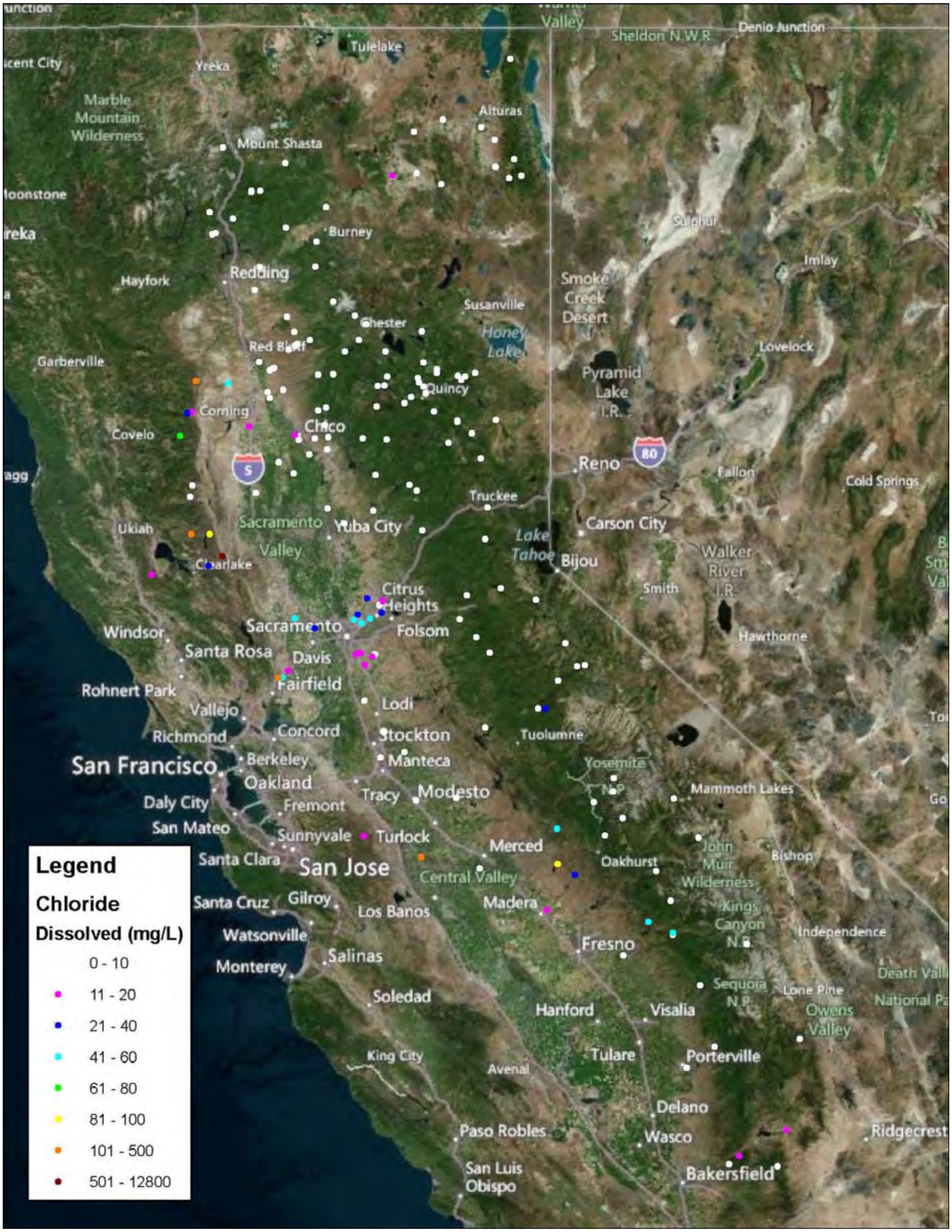


Figure 3-4. Sample locations of Central Valley dissolved chloride data summarized in Table 3-2.

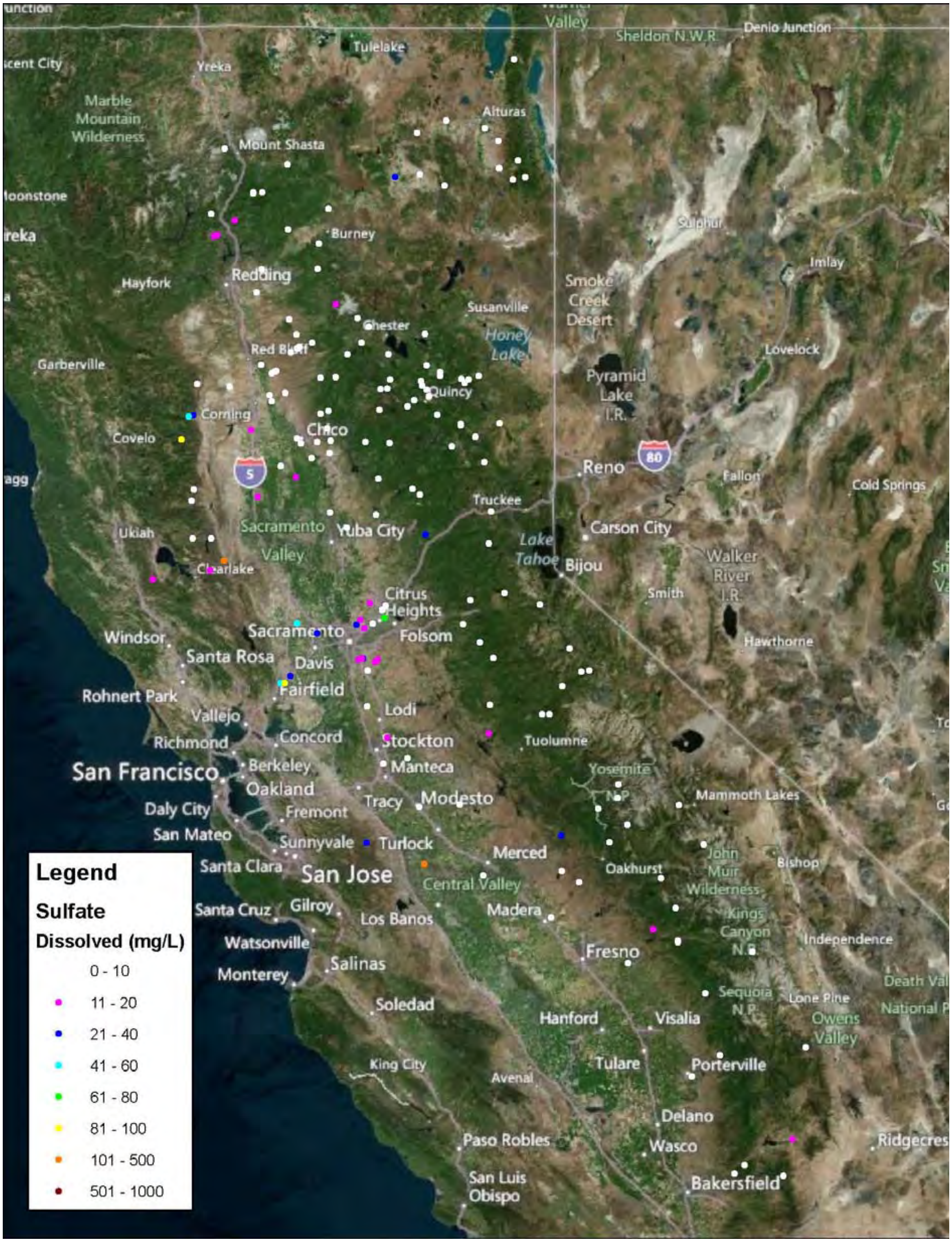


Figure 3-5. Sample locations of Central Valley dissolved sulfate data summarized in Table 3-2.

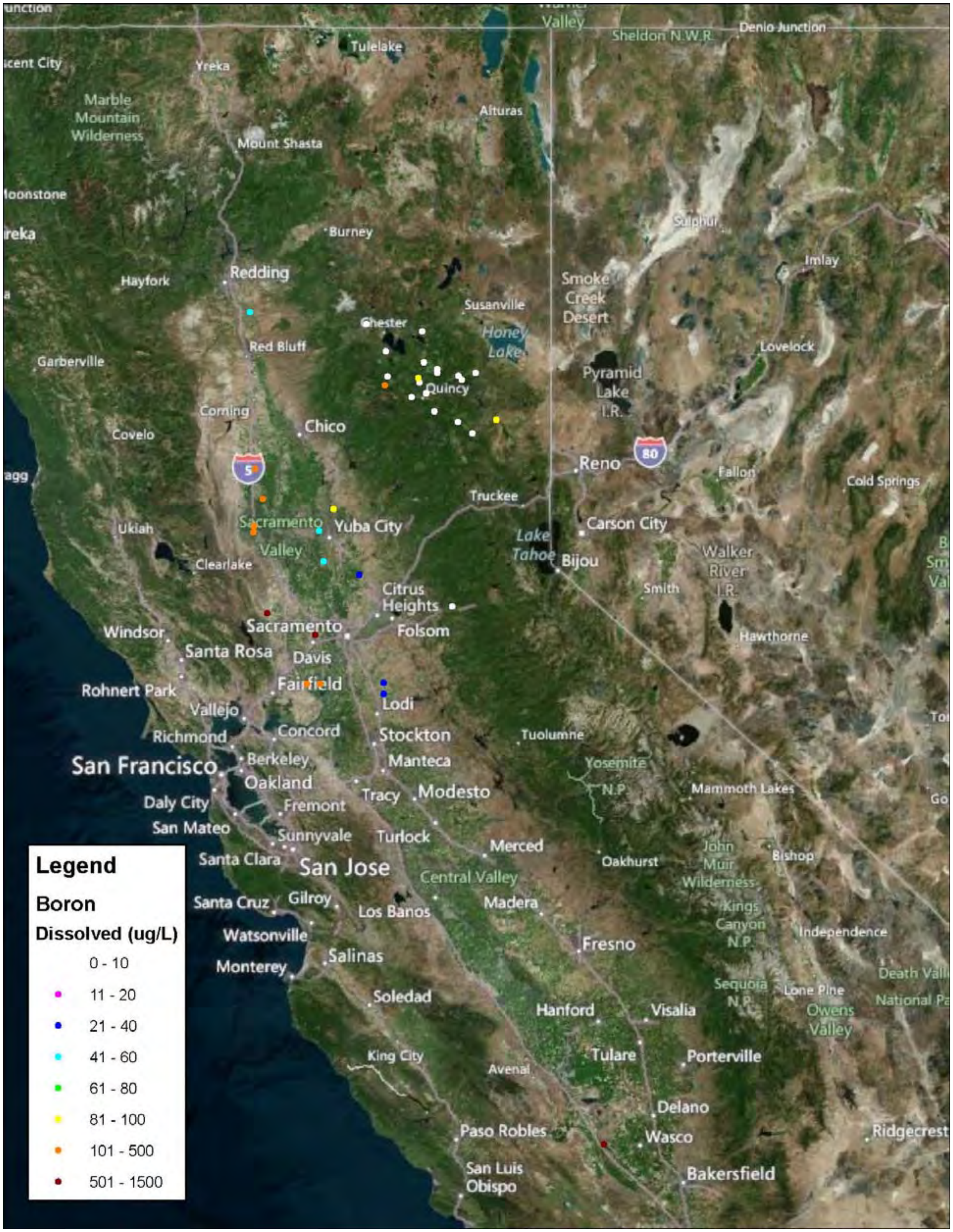


Figure 3-6. Sample locations of Central Valley dissolved boron data summarized in Table 3-2.

Section 4

Conclusions and Recommendations

4.1 Conclusions

4.1.1 Salinity-related Toxicity

- Toxicity of complex TDS matrices is considerably more variable (and less predictable) than the toxicity of simple salts. This variability may stem from ionic interactions and ionic balance issues as well as complex physiological responses of aquatic organisms to these different mixtures. Thus while ample toxicity data exist for simple salts, there will remain considerable uncertainty in the application of these values as the basis for developing WQOs.

- There is sufficient data (by EPA standards) for the toxicity of individual ions (chloride, boron, and sulfate) to produce WQOs based on toxicity. However, any such WQOs will contain a significant amount of uncertainty for the following reasons:

  - Methods for inferring the ions responsible for toxicity in complex mixtures are not well developed at this time;

  - Interactions among ions and their effects on toxicity remain poorly understood;

  - It remains unclear whether the toxicity of TDS mixtures results from individual ions causing toxicity or whether ions in combination elicit toxicity; and

  - The predominant species found in freshwater ecosystems are usually insects, and these species are largely absent from most toxicity databases.

4.1.2 Water Chemistry in the Central Valley

- There appears to be a large difference between natural salinities in streams flowing from the east and west slopes of the Central Valley.

- Other water chemistry issues (i.e., pesticides, nutrients) likely play a large role in species composition but were not considered here; therefore, it is not possible to establish with any certainty the degree to which salinity is a factor in establishment of aquatic communities in the Central Valley.

4.1.3 Central Valley Biota and Biological Monitoring

- The lack of streams in reference (or near reference) conditions suggests that the regional species pool is comprised of largely tolerant/facultative species.

- The lack of streams in reference (or near reference) conditions greatly hinders biomonitoring approaches that could be used to assess the success/failure of any adopted salinity-related WQOs unless trends (changes in species composition/biodiversity over time) at a given site are the basis of assessment.

4.2 Regulatory Options

Given the findings of this study, there are three general approaches/options for generating salinity/TDS related WQOs for the protection of aquatic life in the Central Valley based on (a) toxicity; (b) chemistry; and (c) biology (**Figure 4-1**).

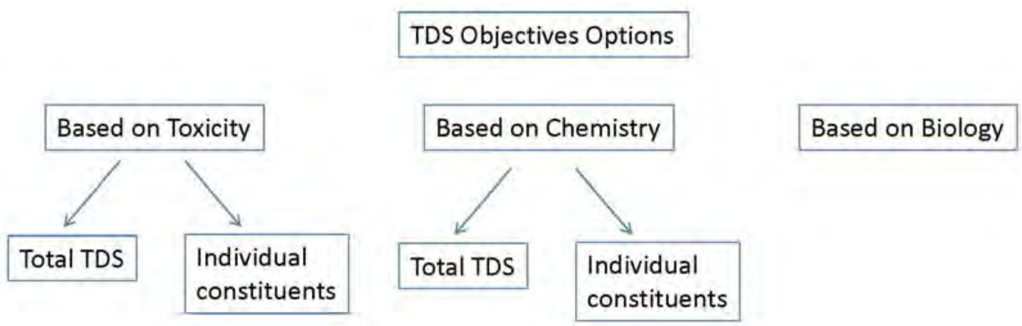


Figure 4-1. General approaches for generating salinity/TDS related WQOs for the protection of aquatic life in the Central Valley.

4.2.1 Water Quality Objectives Based on Toxicity

Toxicity datasets for chloride, boron and sulfate could be used to generate WQOs (**Figure 4-2**); however, it is important to recognize that there are a number of factors that may need to be considered if these datasets were used to develop WQOs. For example the Canadian Guidelines which provide the basis for the Chloride values in Table 4-1 identify how these guidelines should be interpreted[5]. **Table 4-1** lists acute and chronic HC_{05} estimates based on existing toxicity datasets.

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[5] ***From CCME (2011) -*** *Guidance on the Use of Guidelines:* These guidelines for the chloride ion are only intended to protect against direct toxic effects of chloride, based on studies using NaCl and $CaCl_2$ salts. The guideline should be used as a screening and management tool to ensure that chloride does not lead to the degradation of the aquatic environment…The short-term benchmark concentration [acute] and long-term [chronic] CWQG for chloride are set to provide protection for short- and long-term exposure periods, respectively. They are based on generic environmental fate and behaviour and toxicity data. The long-term water quality guideline is a conservative value below which all forms of aquatic life, during all life stages and in all Canadian aquatic systems, should be protected – with one exception. As noted earlier, the CWQG may not be protective of the early (glochidia) life-stage [of] certain species of…endangered and special concern freshwater mussels…Because the guideline is not corrected for any toxicity modifying factors (e.g., hardness), it is a generic value that does not take into account any site-specific factors. Moreover, since the guideline is mostly based on toxicity tests using naïve (i.e., non-tolerant) laboratory organisms, the guideline may be over-protective for areas with a naturally-elevated concentration of chloride and associated adapted ecological community (CCME, 2007). Thus, if an exceedence of the guideline is observed (due to anthropogenically-enriched water or because of elevated natural background concentrations), it does not necessarily suggest that toxic effects will be observed, but rather indicates the need to determine whether or not there is a potential for adverse environmental effects…In general, CWQGs are numerical concentrations or narrative statements that are recommended as levels that should result in negligible risk of adverse effects to aquatic biota. As recommendations, the CWQGs are not legally enforceable limits, though they may form the scientific basis for legislation, regulation and/or management at the provincial, territorial, or municipal level. CWQGs may also be used as benchmarks or targets in the assessment and remediation of contaminated sites, as tools to evaluate the effectiveness of point-source controls, or as “alert levels” to identify potential risks.

Table 4-1. HC_{05} estimates for the acute and chronic toxicity of major ions.

Constituent	Acute (mg/L)	Chronic (mg/L)
Chloride[1]	640	120
Boron[2]	4	1.13
Sulfate[2]	234	124

[1] Based on Canadian Guidelines (CCME, 2011)

[2] Based on use of ECOTOX data and EPA standard methods (Stephan et al., 1985)

Advantages of adopting WQOs based on available toxicity data include:

- Approach is transparent and objective; it uses readily available data facilitating communication with stakeholders
- Approach is based on EPA-established methodologies

Primary disadvantages of adopting WQOs based on available toxicity data include:

- Data are for simple matrices (sodium salts of the given ions) and do not represent the complex mixtures of ions found in natural surface waters
- Mixture toxicity of major ions in Central Valley waters remains poorly understood
- Datasets used to generate these HC_{05} estimates are poor with respect to aquatic insect data. Since insects will predominate the species pool in the ecosystems in question, major uncertainty exists with respect to protectiveness.

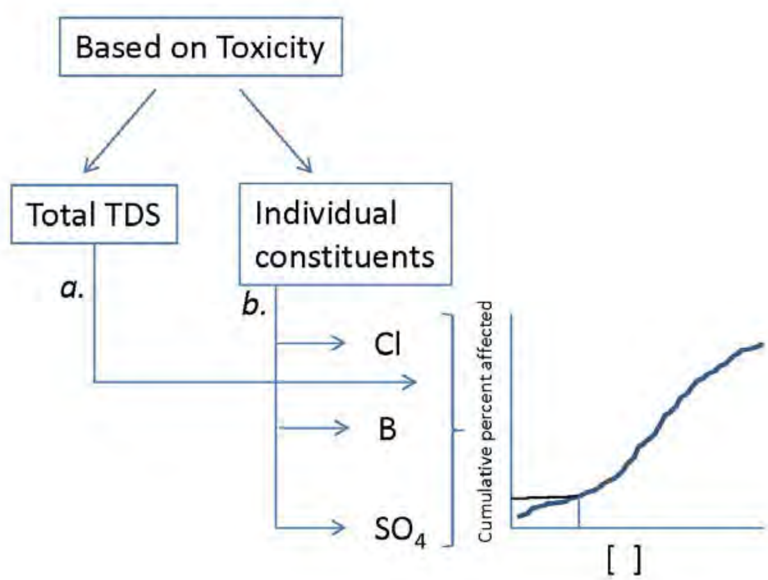

Figure 4-2. Options for developing WQOs based on toxicity.

4.2.2 Water Quality Objectives Based on Chemistry

WQOs based on allowable deviations from natural salinities (the South African approach, see Section 2.1.3) could be developed. Such an approach could take into consideration the broad range of natural salinities present in the Central Valley source waters, and apply an allowable degree of change (salinity increase) on a geographic (east vs. west slope originating waters; valley floor originating waters) or on a case by case basis (a more site-specific approach). It is highly recommended that a statistically based, spatially explicit evaluation of available water chemistry data be performed to better understand both natural distributions of salinities in source waters and anthropogenic changes to those chemistries on a geographic basis, regardless of whether or not WQOs are based on water chemistry.

Should WQOs be developed based on deviations from "natural" conditions, the allowable degree of salinity change would be policy decision. Such an approach could be applied to total TDS/salinity/EC or to individual constituents. The advantages of this approach are that it would remove the situation where a one-size-fits-all WQO could render naturally lower TDS sites as under-protected, whereas naturally high TDS sites could never reach attainable status. The major difficulty associated with this approach is the determination of what should be the natural salinities of larger river systems (segments containing inputs from several sources of salinity). The AZN approach of tailoring local criteria to the perceived conservation value could also be considered here (see Section 2.1.3).

4.2.3 Water Quality Objectives Based on Biology

The lack of suitable reference systems renders this option very difficult to develop, much less defend. Additionally, the soft substrates of many Central Valley streams (and the heterogeneity associated with particle sizes) practically affects the ability to effectively compare samples from site to site. Leland and Fend (1998) were able to get around this issue by using artificial substrates. However, as noted above, this type of data is not normally collected.

4.2.4 Hybrid Approaches for Setting Water Quality Objectives

A hybrid approach could borrow the "trigger value" concept from the AZN Approach (**Figure 4-3**). Exceedance of a total TDS/salinity/EC "trigger value" could prompt the need for further chemical analysis to quantify the concentrations of individual major ions and/or whole effluent type toxicity testing. Only where a concern was identified for a particular ion, e.g., chloride or sulfate, would it be necessary to consider implementation of a control measure.

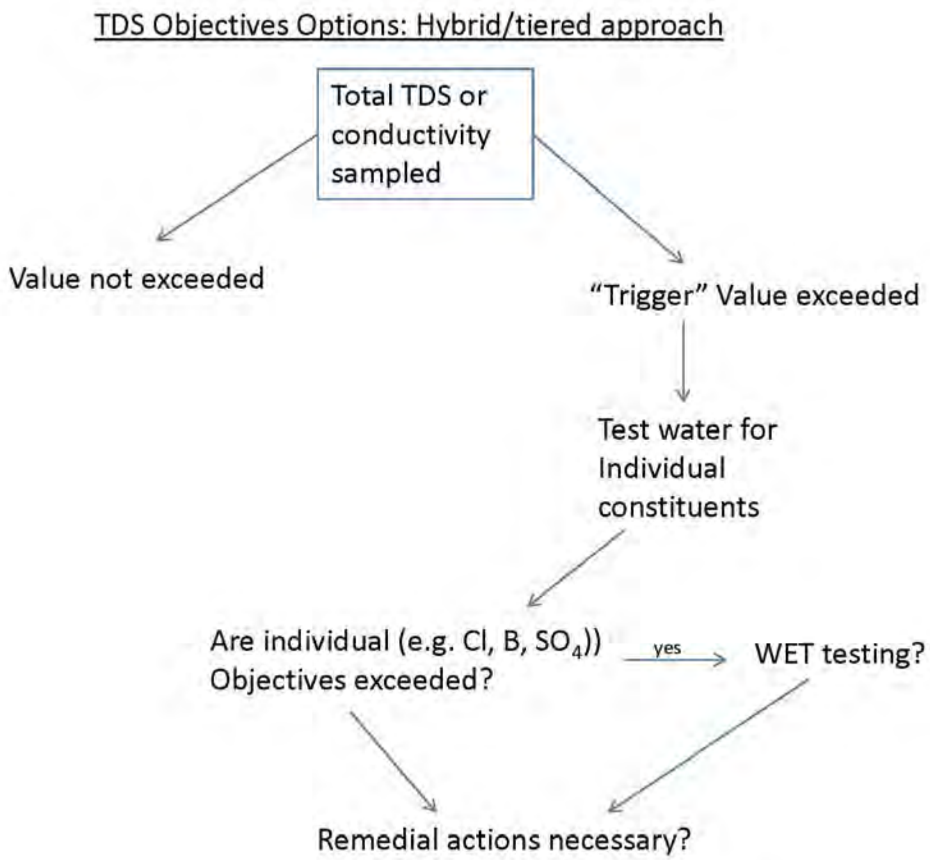

Figure 4-3. A hybrid approach combining a total TDS/salinity/EC trigger value with WQOs for individual ions.

4.3 Final Thoughts

This report provides potential directions that could be taken for the development of WQOs for the protection of aquatic life for TDS/salinity related constituents in the Central Valley. There are several reasons why explicit recommendations for WQOs were not made here. First, (and obviously), there are considerable scientific/technical uncertainties that are highlighted throughout this report. In addition, there is a general lack of dissolved water chemistry data in much of the Central Valley floor. That said, there will always be scientific uncertainty, and this alone should not necessarily be a basis for inaction. Based on both field and toxicity studies outlined here, there is the potential for salinity to be causing ecological impairment in at least some stream segments in the Central Valley. In those situations, setting some regulatory limits on TDS/salinity components for the protection of aquatic life would make sense. However, there remains a surprisingly incomplete picture of TDS related chemistry on the ground in the Central Valley. Before any new regulatory tools are considered (e.g., other than the use of the narrative WQOs), there should be a strong understanding of both spatial and temporal chemistry dynamics in the Central Valley, such that the attainability of any future agreed upon numeric WQOs are well understood, as these could have regulatory and economic consequences Additionally, there should be some articulation (ideally based on stakeholder consensus) regarding the conservation goals for aquatic life in the Central Valley. The degree to which the whole ecosystem has been modified places the ecology well outside of any historic or comparable ecosystem. This renders wadable streams in the Central Valley outside of the current capacity to adequately biomonitor using traditional reference approaches. If there are species/ecosystem processes that are identified as of particular importance/conservation value, this should be articulated.

Given the uncertainties and data gaps described above, it is not recommended that WQOs for aquatic life be established at this time. However, if the Central Valley Water Board were to implement projects to resolve these data gaps and uncertainties, then it is recommended that the following activities be carried out:

- Generate a robust assessment of water chemistry (dissolved) on the valley floor (this includes both spatial and seasonal coverage).
- Determine the extent to which elevated salts arise from natural (seeps, springs and streams draining salt bearing geology) vs. anthropogenic sources (irrigation drain water).
- If salinity is deemed to come from primarily anthropogenic activities, consider implementing toxicity based objectives after performing an analysis of attainability.
- Articulate which biological communities/species are priorities for protection and how salinity based objectives would be applied.

Section 5
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Appendix A

Data Tables

Table A-1 - Predicted cumulative percentage of species affected by chronic exposures to chloride (from Evans and Frick, 2001).

Cumulative % of Species Affected	Mean Chloride Concentration (mg/L)	Lower Confidence Limit (mg/L)	Upper Confidence Limit (mg/L)
5	213	136	290
10	238	162	314
25	329	260	397
50	563	505	622
75	964	882	1,045
90	1,341	1,254	1,428

Table A-2 - Four-day LC_{50} values of various taxa exposed to sodium chloride (adapted from Table 7-5 in Evans and Frick, 2001 and Table B.6 in Bright and Addison, 2002) (References in table not provided in this document – see original references).

Species	Common Name	96 h LC_{50} (mg Cl/L)	References
Tubifex tubifex	Tubificid worm	1,204	Khangarot, 1995
Ceriodaphnia dubia	Cladoceran	1,400	Cowgill and Milazzo, 1990
Daphnia pulex	Cladoceran	1,470	Birge et al., 1985
Ceriodaphnia dubia	Cladoceran	1,596	WI SLOH, 1995
Daphnia magna	Cladoceran	1,853	Anderson, 1948
Daphnia magna	Cladoceran	2,390	Arambasic et al., 1995
Physa gyrina	Snail	2,480	Birge et al., 1985
Lirceus fontanels	Isopod	2,970	Birge et al., 1985
Cirrhinius mrigalo	Indian carp fry	3,021	Gosh and Pal, 1969
Labeo rohoto	Indian carp fry	3,021	Gosh and Pal, 1969
Catla catla	Indian carp fry	3,021	Gosh and Pal, 1969
Daphnia magna	Cladoceran	3,658	Cowgill and Milazzo, 1990
Cricotopus trifascia	Chironomid	3,795	Hamilton et al., 1975
Chironomus attenatus	Chironomid	4,026	Thorton and Sauer, 1972
Hydroptila angusta	Caddisfly	4,039	Hamilton et al., 1975
Daphnia magna	Cladoceran	4,071	WI SLOH, 1995
Limnephilus stigma	Caddisfly	4,255	Sutcliffe, 1961
Anaobolia nervosa	Caddisfly	4,255	Sutcliffe, 1961
Carassius auratus	Goldfish	4,453	Adelman et al., 1976
Pimephales promelas	Fathead minnow	4,600	WI SLOH, 1995
Pimephales promelas	Fathead minnow	4,640	Adelman et al., 1976
Lepomis macrochirus	Bluegill	5,840	Birge et al., 1985
Culex sp.	Mosquito	6,222	Dowden and Bennett, 1965
Pimephales promelas	Fathead minnow	6,570	Birge et al., 1985
Lepomis macrochirus	Bluegill	7,864	Trama, 1954
Gambusia affinis	Mosquito fish	10, 616	Wallen et al., 1957
Anguilla rostrata	American eel	10,900	Hinton and Eversole, 1978
Anguilla rostrata	American eel	13,085	Hinton and Eversole, 1978

Table A-3 - Results of chronic toxicity tests (> 7 day duration) conducted on freshwater organisms exposed to sodium chloride (adapted from Table 7-6 in Evans and Frick, 2001 and Table B.6 in Bright and Addison, 2002) (References in table not provided in this document – see original references).

Species	Common Name	LC_{50}/EC_{50} (mg Cl/L)	Measured Endpoint	References
Ceriodaphnia dubia	Cladoceran	735	brood size	Degreave et al., 1985
Pimephales promelas	Fathead minnow	874	survival	Beak, 1999
Ceriodaphnia dubia	Cladoceran	1,068	brood size	Cowgill and Milazzo, 1990
Oncorhynchus mykiss	Rainbow trout	1,456	survival	Beak, 1999
Nitschia linearis	Diatom	1,475	cell numbers	Gonzales-Moreno et al., 1997
Xenopus leavis	Frog	1,524	survival	Beak, 1999
Oncorhynchus mykiss	Rainbow trout	1,595	survival	Beak, 1999
Daphnia magna	Cladoceran	2,451	brood size	Cowgill and Milazzo, 1990
Pimephales promelas	Larvae	3,029	growth	Beak, 1999
Lemna minor	Duckweed	3,150	population	Buckley et al., 1996
Myriophyllum spicatum	Eurasian Watermilfoil	4,291	population	Stanley, 1974
Myriophyllum spicatum	Eurasian Watermilfoil	4,681	growth	Stanley, 1974

Table A-4 – Chloride toxicity endpoint data from Canadian Water Quality Guidelines for the Protection of Aquatic Life (2011) (References in table not provided in this document – see original reference).

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Table 3. Endpoints used to determine the freshwater short-term CWQG for the chloride ion.

Species	Endpoint	Concentration (mg Cl⁻/L)	Reference
Fish			
Pimephales promelas Fathead minnow	96-hour LC_{50} (geomean)	4,223	Mount *et al* 1997; Birge *et al* 1985
Lepomis macrochirus Bluegill sunfish	96-hour LC_{50} (geomean)	5,272	Birge *et al.* 1985; Trama 1954
Cyprinella leedsi Bannerfin shiner	96-hour LC_{50}	6,070	Environ 2009
Oncorhynchus mykiss Rainbow trout	96-hour LC_{50} (geomean)	8,634	Elphick *et al* 2011; Vosyliene *et al.* 2006
Gambusia affinis Mosquito fish	96-hour LC_{50}	9,099	Al-Daham & Bhatti 1977
Gasterosteus aculeatus Threespine stickleback	96-hour LC_{50}	10,200	Garibay & Hall 2004
Anguilla rostrata American eel	96-hour LC_{50}	13,012	Hinton and Eversol 1979
Amphibians			
Ambystoma maculatum Spotted salamander	96-hour LC_{50}	1,178	Collins & Russell 2009
Pseudacris triseriata feriarum Chorus frog	96-hour LC_{50}	2,320	Garibay & Hall 2004
Lithibates sylvatica (previously *Rana sylvatica*) Wood frog	96-hour LC_{50} (geomean)	2,716	Sanzo & Hecnar, 2006; Collins & Russell 2009; Jackman 2010
Pseudacris crucifer Spring peeper	96-hour LC_{50}	2,830	Collins & Russell 2009
Rana clamitans Green frog	96-hour LC_{50}	3,109	Collins & Russell 2009
Rana temporaria Common frog	96-hour LC_{50}	3,140	Viertel 1999
Lithibates pipiens (previously *Rana pipiens*) Leopard frog	96-hour LC_{50}	3,385	Jackman 2010
Bufo americanus American toad	96-hour LC_{50}	3,926	Collins & Russell 2009
Rana catesbeiana Bullfrog	96-hour LC_{50}	5,846	Environ 2009
Invertebrates			
Epioblasma torulosa rangiana Northern riffleshell mussel (COSEWIC[a] endangered)	24-hour EC_{50} (survival of glochidia)	244	Gillis 2011
Daphnia magna Water flea	48-hour EC_{50} (immobilization)	621	Khangarot and Ray 1989
Lampsilis siliquoidea Freshwater mussel	24-hour EC_{50} (survival of glochidia) (geomean)	709	Bringolf *et al* 2007; Gillis 2011
Lampsilis fasciola Wavy-rayed lampmussel (COSEWIC[a] special concern)	24-hour EC_{50} (survival of glochidia) (geomean)	746	Valenti *et al.* 2007; Bringolf *et al* 2007; Gillis 2011
Lampsilis cardium Plain pocketbook	24-hour EC_{50} (survival of glochidia)	817	Gillis 2011
Sphaerium simile Fingernail clam	96-hour LC_{50} (geomean)	902	GLEC & INHS 2008
Ceriodaphnia dubia	48-hour LC_{50}	1,080	Valenti *et al* 2007;

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Species	Endpoint	Concentration (mg Cl^-/L)	Reference
Water flea	(geomean)		Hoke *et al* 1992; Mount *et al* 1997; GLEC & INHS 2008; Elphick *et al* 2011; Cowgill & Milazzo 1990
Daphnia ambigua Water flea	48-hour EC_{50} (immobilization)	1,213	Harmon *et al* 2003
Daphnia pulex Water flea	48-hour LC_{50} (geomean)	1,248	Palmer *et al* 2004 ; Birge *et al* 1985
Elliptio lanceolata Yellow lance mussel	96-hour LC_{50}	1,274	Wang & Ingersoll 2010
Brachionus patulus Rotifer	24-hour LC_{50}	1,298	Peredo-Alvarez *et al* 2003
Hyalella azteca Amphipod	96-hour LC_{50}	1,382	Elphick *et al* 2011
Elliptio complanata Freshwater mussel	24-hour EC_{50} (survival of glochidia)	1,620	Bringolf *et al* 2007
Epioblasma brevidens Cumberlandian combshell (endangered in USA)	24-hour EC_{50} (survival of glochidia)	1,626	Valenti *et al* 2007
Epioblasma capsaeformis Oyster mussel (endangered in USA)	24-hour EC_{50} (survival of glochidia)	1,644	Valenti *et al* 2007
Villosa constricta Freshwater mussel	24-hour EC_{50} (survival of glochidia)	1,674	Bringolf *et al* 2007
Villosa iris Rainbow mussel (COSEWIC[a] endangered)	96-hour EC_{50} (geomean)	1,815	Wang & Ingersoll 2010
Musculium transversum Fingernail clam	96-hour LC_{50}	1,930	US EPA 2010
Villosa delumbis Freshwater mussel	24-hour EC_{50} (survival of glochidia)	2,008	Bringolf *et al* 2007
Brachionus calyciflorus Rotifer	24-hour LC_{50} (geomean)	2,026	Elphick et al 2011; Peredo-Alvarez *et al* 2003; Calleja *et al* 1994
Physa gyrina Snail	96-hour LC_{50}	2,540	Birge *et al*. 1985
Lirceus fontinalis Isopod	96-hour LC_{50}	2,950	Birge *et al*. 1985
Gyraulus parvus Snail	96-hour LC_{50} (geomean)	3,043	GLEC & INHS 2008
Baetis tricaudatus Mayfly	48-hour EC_{50} (immobilization) (geomean)	3,266	Lowell *et al* 1995
Chironomus dilutus / tentans Midge	96-hour LC_{50}	3,761	Wang & Ingersoll 2010
Lumbriculus variegates Oligochaete	96-hour LC_{50} (geomean)	4,094	Elphick *et al* 2011; Environ 2009
Nephelopsis obscura Leech	96-hour LC_{50}	4,310	Environ 2009
Hexagenia sp. Mayfly	48-hour LC_{50}	4,671	Wang & Ingersoll 2010
Chironomus attenatus Midge	48-hour LC_{50}	4,850	Thornton & Sauer 1972
Daphnia hyalina Water flea[b]	48-hour LC_{50}	5,308	Baudouin & Scoppa 1974
Lepidostoma sp.	96-hour LC_{50}	6,000	Williams *et al* 1999

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Species	Endpoint	Concentration (mg Cl⁻/L)	Reference
Caddisfly			
Tubifex tubifex Oligochaete	96-hour LC_{50} (geomean)	6,119	Elphick *et al* 2011; Wang & Ingersoll 2010; GLEC & INHS 2008
Chironomus riparius Midge	48-hour LC_{50}	6,912	Wang & Ingersoll 2010
Eudiaptomus padanus padanus Copepod[b]	48-hour LC_{50}	7,077	Baudouin & Scoppa 1974
Cyclops abyssorum prealpinus Copepod[b]	48-hour LC_{50}	12,385	Baudouin & Scoppa 1974

[a]Committee on the Status of Endangered Wildlife in Canada; Canadian occurrence in Ontario.
[b]Based on testing with $CaCl_2$ salt (all others based on testing with NaCl salt).

Table A-5 – Chloride toxicity values from Tables 2 and 3, IDNR (2009).

Table 2. Chloride acute toxicity to C. dubia at different water harnesses and single sulfate concentration

Chloride Toxicity Test	*C. dubia* 48 h LC50 (95%CI) GLEC (mg Cl/L)	*C. dubia* 48 h LC50 (95%CI) INHS (mg Cl/L)	Mean LC50 value (mg Cl/L)
Acclimated to and Tested at Various Total Hardness Levels (and 65 mg/L Sulfate)			
25 mg/L Hardness	947 (868-1034)	1007 (964-1052)	977
50 mg/L Hardness	955 (885-1031)	767 (684-861)	861
100 mg/L Hardness	1130 (1029-1231)	1369 (1246-1505)	1250
200 mg/L Hardness	1609 (1516-1707)	1195 (1148-1245)	1402
400 mg/L Hardness	1491 (1385-1606)	1687 (1587-1794)	1589
600 mg/L Hardness	1907 (Estimates not Reliable)	1652 (1536-1776)	1779
800 mg/L Hardness	1764 (1661-1874)	1909 (1791-2034)	1836
Acclimated to and Tested at Various Sulfate Levels (and 300 mg/L Hardness)			
25 mg/L Sulfate	1400 (1287-1523)	1311 (1210-1421)	1356
50 mg/L Sulfate	1720 (1634-1811)	1258 (1211-1306)	1489
100 mg/L Sulfate	1394 (1281-1516)	1240 (1203-1278)	1317
200 mg/L Sulfate	1500 (1370-1641)	1214 (1153-1278)	1357
400 mg/L Sulfate	1109 (1004-1225)	1199 (1120-1284)	1154
600 mg/L Sulfate	1206 (1161-1253)	1179 (1125-1235)	1192

Table 3. Chloride acute toxicity for fingernail clam, snail and tubificid worm

Test species	96 h LC50 (95%CI) at 50 mg/L total hardness (mg Cl/L)	96 h LC50 (95%CI) at 200 mg/L total hardness (mg Cl/L)
Fingernail clam (juveniles), *Sphaerium simile*	740 (678-807)	1100[a] (1040-1164)
Planorbid snail (mixed ages), *Gyraulus parvus*	3,078 (2,771-3,418)	3,009 (2,728-3,318)
Tubificid worm (mixed ages), *Tubifex tubifex*	4,278 (3,848-4,717)	6,008 (5,563-6,489)

[a] Result is from a repeat test because control mortality in the first test slightly exceeded maximum acceptable mortality of 10% (15% mortality recorded). LC50 was similar to the LC50 of the failed test (1098 mg Cl/L) which was based on nominal concentrations.

Table A-6. Acute toxicity data for borate (ECOTOX).

Table A. Acute borate LC_{50} toxicity data

Genus	Common Name	Mean Concentration (mg/L)
Limanda	Flounder	78.43
Carassius	Ray-finned Fish	121.75
Americamysis	Opossum Shrimp	130.01
Menidia	Silverside Fish	160.92
Ictalurus	Channel Catfish	169.25
Daphnia	Water Flea	196.20
Xyrauchen	Sucker Fish	204.00
Rasbora	Minnow	231.67
Cyprinodon	Sheepshead Minnow	245.56
Ptychocheilus	Pike Minnow	302.00
Gila	Chub Fish	310.67
Lepomis	Bluegill Fish	321.75
Catostomus	Sucker Fish	421.75
Oncorhynchus	Silver Salmon	934.96
Xenopus	African Clawed Frogs	978.11
Chironomus	Midge	1,376.00
Spirostomum	Ciliate Protist	5,348.00
Gambusia	Western Mosquitofish	9,650.00

Table B. Acute borate data – Non-LC_{50} data

Genus	Common Name	Mean Concentration (mg/L)
Entosiphon	Protist Flagellates	1.00
Tetrahymena	Ciliated Protozoan	1.00
Chilomonas	Algae	8.30
Dugesia	Flatworm	10.00
Anacystis	Algae	11.25
Pseudokirchneriella	Green Algae	15.40
Chironomus	Midge	20.00
Kuhlia	Flagtail Fish	20.00
Lemna	Duckweed	20.34
Dreissena	Zebra Mussel	50.00
Ceriodaphnia	Water Flea	79.16
Simocephalus	Water Flea	80.75
Uronema	Algae	109.00
Anthocidaris	Sea Urchin	127.00
Colpidium	Ciliate	182.97
Rasbora	Minnow	196.67
Americamysis	Opossum Shrimp	199.72
Menidia	Silverside Fish	251.43
Xyrauchen	Sucker Fish	288.00
Xenopus	African Clawed Frogs	296.88
Daphnia	Water Flea	336.99
Ptychocheilus	Pike Minnow	360.00
Spirostomum	Ciliate Protist	558.00
Danio	Zebra Fish	568.84
Cypris	Ostracod	2,022.50
Poecilia	Molly Fish	4,333.33
Oncorhynchus	Silver Salmon	5,000.00
Pimephales	Fathead Minnow	5,275.00
Tubifex	Sludge Worm	6,000.00
Anguilla	Freshwater Eel	6,083.33
Gammarus	Scud	10,000.00

Table A-7. Chronic toxicity data for borate (ECOTOX).

Table A. Chronic borate LC_{50} toxicity data

Genus	Common Name	Mean Concentration (mg/L)
Hyalella	Scud	3.04
Elodea	Waterweed	5.00
Myriophyllum	Eurasian Watermilfoil	5.00
Ranunculus	Buttercup Plant	10.00
Daphnia	Water Flea	52.70
Carassius	Ray-finned Fish	61.25
Oncorhynchus	Silver Salmon	76.94
Micropterus	Largemouth Bass	92.00
Ictalurus	Channel Catfish	100.75

Table B. Chronic borate data – Non-LC_{50} data

Genus	Common Name	Mean Concentration (mg/L)
Scenedesmus	Green Algae	0.58
Ranunculus	Buttercup Plant	1.50
Elodea	Waterweed	1.80
Chlorella	Green Algae	5.60
Lemna	Duckweed	8.50
Micropterus	Largemouth Bass	12.17
Ceriodaphnia	Water Flea	15.70
Daphnia	Water Flea	22.99
Ambystoma	Salamander	49.50
Bufo	Toad	49.50
Lithobates	Bullfrog	49.50
Anacystis	Algae	73.00
Myriophyllum	Eurasian Watermilfoil	98.43
Xenopus	African Clawed Frogs	150.00
Oncorhynchus	Silver Salmon	161.60
Phragmites	Common Reed Grass	430.00
Typha	Common Cattail	430.00

Table A-8. Acute toxicity data for sulfate (ECOTOX).

Table A. Acute sulfate LC_{50} toxicity data

Genus	Common Name	Mean Concentration (mg/L)
Morone	Striped Bass	600.44
Tricorythus	Mayfly	660.00
Nitzschia	Diatom	1,900.00
Hyalella	Scud	2,204.03
Sphaerium	Grooved Fingernail Clam	2,402.86
Ceriodaphnia	Water Flea	2,541.56
Lampsilis	Lamp-Mussel	2,882.30
Daphnia	Water Flea	4,498.54
Lymnaea	Pond Snail	4,938.50
Pimephales	Fathead Minnow	6,985.16
Lepomis	Bluegill Fish	11,271.25
Culex	Mosquito	12,390.00
Chironomus	Midge	14,134.00
Gambusia	Western Mosquitofish	17,000.00
Poecilia	Sailfin Molly Fish	18,018.00
Americamysis	Opossum Shrimp	18,659.94
Cyprinodon	Sheepshead Minnow	19,502.80
Menidia	Inland Silverside Fish	28,507.43

Table B. Acute sulfate data – Non-LC_{50} data

Genus	Common Name	Mean Concentration (mg/L)
Artemia	Brine Shrimp	6.60
Ophryotrocha	Polychaete	6.60
Notropis	Shiner Fish	100.00
Gammarus	Scud	299.75
Bulinus	Snail	850.00
Lymnaea	Pond Snail	950.00
Brachionus	Rotifer	1,267.75
Dreissena	Zebra Mussel	1,420.40
Corbicula	Asiatic Clam	1,500.00
Hyalella	Scud	1,544.89
Americamysis	Opossum Shrimp	1,800.00
Navicula	Diatom	1,900.00
Morone	Striped Bass	2,237.50
Pseudokirchneriella	Green Algae	2,258.41
Pimephales	Fathead Minnow	2,359.95
Ceriodaphnia	Water Flea	2,408.35
Oncorhynchus	Silver Salmon	3,942.72
Cyprinidae	Minnow	4,500.00
Daphnia	Water Flea	4,915.50
Biomphalaria	Freshwater Snail	5,133.33
Polycelis	Planarian	6,817.92
Chimarra	Caddisfly	7,340.00
Tricorythus	Mayfly	7,340.00
Chironomus	Midge	11,682.00
Cyprinodon	Sheepshead Minnow	19,613.46
Ictalurus	Channel Catfish	20,000.00
Lepomis	Bluegill Fish	20,000.00
Micropterus	Largemouth Bass	20,000.00
Menidia	Inland Silverside Fish	20,538.98

Table A-9. Chronic toxicity data for sulfate (ECOTOX).

Table A. Chronic sulfate LC_{50} toxicity data

Genus	Common Name	Mean Concentration (mg/L)
Calla	Water Arum	284.08
Equisetum	Water Horsetail	284.08
Glyceria	Reed Mannagrass	284.08
Juncus	Rush	284.08
Menyanthes	Buck-bean	284.08
Potamogeton	Pondweed	284.08
Thelypteris	Eastern Marsh Fern	284.08
Fontinalis	Common Water Moss	920.25
Pseudacris	Pacific Chorus Frog	1,498.19
Corbicula	Asiatic Clam	1,500.00
Hyalella	Scud	1,565.00
Ceriodaphnia	Water Flea	1,586.83
Myriophyllum	Eurasian Watermilfoil	2,341.00
Caridina	Common Water Shrimp	2,341.43
Pseudokirchneriella	Green Algae	3,000.00
Oncorhynchus	Silver Salmon	3,208.81
Rana	Bullfrog	4,261.20
Ictalurus	Channel Catfish	9,166.67
Americamysis	Opossum Shrimp	9,236.15
Lepomis	Bluegill Fish	10,000.00
Micropterus	Largemouth Bass	13,800.00

Table B. Chronic sulfate data – all lethal and non-lethal data

Genus	Common Name	Mean Concentration (mg/L)
Calla	Water Arum	284.08
Equisetum	Water Horsetail	284.08
Glyceria	Reed Mannagrass	284.08
Juncus	Rush	284.08
Menyanthes	Buck-bean	284.08
Potamogeton	Pondweed	284.08
Thelypteris	Eastern Marsh Fern	284.08
Anabaena	Blue-Green Algae	390.00
Chlorella	Green Algae	850.00
Dunaliella	Green Algae	850.00
Platymonas	Green Flagellate	850.00
Porphyridium	Red Algae	850.00
Tetraselmis	Prasinophyte	850.00
Fontinalis	Common Water Moss	920.25
Pseudacris	Pacific Chorus Frog	1,430.24
Corbicula	Asiatic Clam	1,500.00
Hyalella	Scud	1,565.00
Ceriodaphnia	Water Flea	1,586.83
Myriophyllum	Eurasian Watermilfoil	2,341.00
Caridina	Common Water Shrimp	2,798.75
Pseudokirchneriella	Green Algae	3,000.00
Oncorhynchus	Silver Salmon	3,657.83
Rana	Bullfrog	4,261.20
Ictalurus	Channel Catfish	9,166.67
Americamysis	Opossum Shrimp	9,236.15
Lepomis	Bluegill Fish	10,000.00
Micropterus	Largemouth Bass	13,800.00