Water Quality Criteria Report for Diuron

Phase III: Application of the pesticide water quality criteria methodology

Prepared for the Central Valley Regional Water Quality Control Board

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List of acronyms and abbreviations

AF	Assessment factor
APHA	American Public Health Association
ASTM	American Society for Testing and Materials
BAF	Bioaccumulation Factor
BC	Black carbon
BCF	Bioconcentration Factor
BMF	Biomagnification Factor
CAS	Chemical Abstract Service
CDFG	California Department of Fish and Game
CSIRO	Commonwealth Scientific and Industrial Research Organization, Australia
CVRWQCB	Central Valley Regional Water Quality Control Board
DPR	Department of Pesticide Regulation
EC_x	Concentration that affects x% of exposed organisms
FDA	Food and Drug Administration
FT	Flow-through test
GMAV	Genus Mean Acute Value
IC_x	Inhibition concentration; concentration causing *x*% inhibition
ICE	Interspecies Correlation Estimation
IUPAC	International Union of Pure and Applied Chemistry
K	Interaction Coefficient
K_H	Henry's law constant
K_{ow}	Octanol-Water partition coefficient
K_p or K_d	Solid-Water partition coefficient
LC_x	Concentration lethal to *x*% of exposed organisms
LD_x	Dose lethal to *x*% of exposed organisms
LL	Less relevant, Less reliable study
LOEC	Lowest-Observed Effect Concentration
LOEL	Lowest-Observed Effect Level
LR	Less relevant, Reliable study
MATC	Maximum Acceptable Toxicant Concentration
N	Not relevant or Not reliable study
n/a	Not applicable
NOAEL	No-Observed Adverse Effect Level
NOEC	No-Observed Effect Concentration
NR	Not reported
OECD	Organization for Economic Co-operation and Development
QSAR	Quantitative Structure Activity Relationship
pK_a	Acid dissociation constant
RL	Relevant, Less reliable study
RR	Relevant and Reliable study
S	Static test
SMAV	Species Mean Acute Value
SR	Static renewal test
SSD	Species Sensitivity Distribution

TES	Threatened and Endangered Species
US	United States
USEPA	United States Environmental Protection Agency

1. Introduction

A new methodology for deriving freshwater water quality criteria for the protection of aquatic life was developed by the University of California - Davis (TenBrook *et al.* 2009a). The need for a new methodology was identified by the California Central Valley Regional Water Quality Control Board (CVRWQCB 2006) and findings from a review of existing methodologies (TenBrook & Tjeerdema 2006, TenBrook *et al.* 2009b). The UC-Davis methodology is currently being used to derive aquatic life criteria for several pesticides of particular concern in the Sacramento River and San Joaquin River watersheds. The methodology report (TenBrook *et al.* 2009a) contains an introduction (Chapter 1); the rationale of the selection of specific methods (Chapter 2); detailed procedure for criteria derivation (Chapter 3); and a chlorpyrifos criteria report (Chapter 4). This criteria report for diuron describes, section by section, the procedures used to derive criteria according to the UC-Davis methodology. Also included are references to specific sections of the methodology procedure detailed in Chapter 3 of the report so that the reader can refer to the report for further details (TenBrook *et al.* 2009a).

2. Basic information

Chemical: Diuron (Fig. 1)
CAS: N'-(3,4-dichlorophenyl)-N,N-dimethylurea
CAS Number: 330-54-1
USEPA PC Code: 035505
CA DPR Chem Code: 231
IUPAC: 3-(3,4-dichlorophenyl)-1,1-dimethylurea
Chemical Formula: $C_9H_{10}Cl_2N_2O$

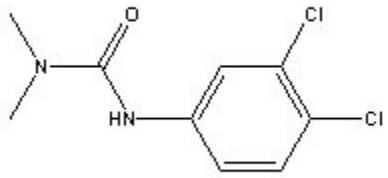

Figure 1. Structure of diuron (source: http://sitem.herts.ac.uk/aeru/footprint/structure/260.jpg)

Trade names: AF 101, Cekiuron, Crisuron, Dailon, DCMU, Di-on, Diater, Dichlorofonidim, Direx, Diurex, Diurol, Diuron, Drexel, Dynex, Herbatox, Karmex, Krovar, Marmer, NA 2767, Telvar, Unidron, Urox D, Vonduron (Mackay *et al.* 2006).

3. Physical-chemical data

Molecular Weight
233.10 (ExToxNet 1996)

Density
1.5 g/mL (IUPAC 2008)

Water Solubility
42 mg/L at 25°C (Tomlin 1994)
35.6 mg/L at 20°C (IUPAC 2008)
Geometric mean: 38.7 mg/L

Melting Point
158°C (Lide 2003)

Vapor Pressure
1.15 E -3 mPa at 25°C (IUPAC 2008)

Henry's constant (K_H)
1.5 x 10^{-4} Pa m^3 mol^{-1} (20-25°C, calculated-P/C) (Mackay *et al.* 2006; Montgomery 1993)
2.00 x 10^{-6} Pa m^3 mol^{-1} at 25°C (IUPAC 2008)
2.06 x 10^{-8} dimensionless at 20°C (IUPAC 2008)
Geometric mean: 173,205 Pa m^3 mol^{-1}

Organic Carbon Sorption Partition Coefficients (log K_{oc})
All values and references from Mackay *et al.* (2006).

2.60 soil (Farmer 1976; Hamaker & Thompson 1972; Hance 1976)
2.59 average of 3 soils, HPLC-RT correlation (McCall *et al.* 1980)
2.21 soil, converted from reported K_{om} multiplied 1.724 (Briggs 1981)
2.58 average of 84 soils (Rao & Davidson 1982)
2.18 soil (Thomas 1982)
2.83 Webster soil (Nkedikizza *et al.* 1983)
2.49 soil slurry method (Swann *et al.* 1983)
2.48 RP-HPLC-RT correlation (Swann *et al.* 1983)
2.94 25°C, Semiahmoo soil, batch equilibrium method-LSS (Madhun *et al.* 1986)
2.68 25°C, Adkins soil, batch equilibrium method-LSS (Madhun *et al.* 1986)
2.58 soil, screening model calculations (Jury *et al.* 1987a; Jury *et al.* 1987b; Jury & Ghodrati 1989)
2.35 subsurface soil from Oklahoma (Bouchard & Wood 1988)
2.57 subsurface soil from Oklahoma (Bouchard & Wood 1988)
2.94 mucky peat soil, quoted (Howard 1991)
2.68 loam sand soil, quoted (Howard 1991)
2.68 soil, 20-25°C, selected (Hornsby *et al.* 1996; Wauchope *et al.* 1992)
2.40 soil (Sabljic *et al.* 1995)
2.44 soil, organic carbon OC ≥ 0.1%, average (Delle Site 2001)
2.43 soil, OC ≥ 0.5%, average (Delle Site 2001)
2.57 soil, 0.1 ≤ OC ≤ 0.5%, average (Delle Site 2001)
2.78 sediment, OC ≥ 0.5%, average (Delle Site 2001)

GeoMean of log K_{oc} values: 2.61

Log K_{ow}

2.68 recommended by Hansch (Hansch *et al.* 1995; Mackay *et al.* 2006)
2.78 recommended by Sangster Research Laboratories (2008)
2.87 at pH 7, 20°C (IUPAC 2008)
Geometric mean: 2.78

Bioconcentration Factor

Table 1. Bioconcentration factors (BCF) for diuron; FT: flow-through, SR: static renewal, S: static; values are on a wet weight basis and are not lipid-normalized.

Species	BCF	Exposure	Reference
Gambusia affinis	290	S	Isensee 1976
Physa sp.	40	S	Isensee 1976
Daphnia magna	260	S	Isensee 1976
Oedogonium cardiacum	90	S	Isensee 1976
Pimephales promelas	2.00	FT	Call *et al.* 1983, 1987

Environmental Fate

Table 2. Diuron hydrolysis and photolysis and other degradation. (NR: not reported).

	Half- life (d)	Water	Temp (°C)	pH	Reference
Hydrolysis	> 4 months	Phosphate buffer	20	5-9	Mackay *et al.* 2006
	Stable	Sterile buffer	25	5, 7, 9	USEPA 2003
Aqueous Photolysis	2.25 h	Distilled	NR	NR	Mackay *et al.* 2006
	43 d	NR	NR	NR	USEPA 2003
Biodegradation (aerobic)	~20 d	Filtered sewage water	20	NR	Mackay *et al.* 2006

4. Human and wildlife dietary values

There are no FDA action levels for diuron in food (USFDA 2000), but there is an EPA pesticide tolerance for farm-raised freshwater finfish tissue of 2.0 mg/kg (USEPA 2007).

Wildlife LC_{50} values (dietary) for animals with significant food sources in water

Toxicity tests on mallards are available in a report from the U.S. Fish and Wildlife Service that summarizes similar avian tests of 131 compounds (Hill *et al.* 1982). The dietary intake LC_{50} (lethal concentration for 50% of organisms tested) was reported to be 5000 ppm for mallards (Hill *et al.* 1982). The US EPA Environmental Risk Assessment for the Reregistration of Diuron (USEPA 2003) states that diuron is practically nontoxic to mallard duck on an acute oral basis, and slightly toxic to mallard duck on a subacute dietary basis. The subacute LC_{50} for mallard duck is 1730 mg/kg feed (USEPA 2003). The acute oral toxicity of diuron to mallard duck was reported as $LD_{50} > 2000$ mg/kg (USEPA 2003).

Wildlife dietary NOEC values for animals with significant food sources in water

No NOEC (no observed effect concentration) data was available for wildlife species with significant food sources in water.

5. Ecotoxicity data

Approximately 86 original studies on the effects of diuron on aquatic life were identified and reviewed. In the review process, many parameters are rated for documentation and acceptability for each study, including, but not limited to: organism source and care, control description and response, chemical purity, concentrations tested, water quality conditions, and statistical methods (see Tables 3.6, 3.7, 3.8 in TenBrook *et al.* 2009a). Single-species effects studies that were rated as relevant (R) or less relevant (L) according to the method (Table 3.6) were summarized in data summary sheets. Information in these summaries was used to evaluate each study for reliability, using the rating systems described in the methodology (Tables 3.7 and 3.8, section 3-2.2, TenBrook *et al.* 2009a), to give a reliability rating of reliable (R), less reliable (L), or not reliable (N). Copies of completed summaries for all studies are included in the Appendix of this report. All data rated as acceptable (RR) or supplemental (RL, LR, LL) for criteria derivation are summarized in Tables 3 - 8, found at the end of this report. Acceptable studies rated as RR are used for numeric criteria derivation, while supplemental studies rated as RL, LR or LL are used for evaluation of the criteria to check that they are protective of particularly sensitive species and threatened and endangered species. These considerations are reviewed in section 12 and 14 of this report, respectively. Studies that were rated not relevant (N) or not reliable (RN or LN) were not used for criteria derivation.

Eleven mesocosm, microcosm and ecosystem (field and laboratory) studies were identified and reviewed. Ten of these studies were rated reliable (R) or less reliable (L), and are listed in Table 9. These studies were used as supporting data in section 13 to evaluate the derived criteria to ensure that they are protective of ecosystems. Two relevant toxicity values for terrestrial wildlife were identified in section 4, and are further reviewed for consideration of bioaccumulation in section 15.

Evaluation of aquatic animal data

Using the data evaluation criteria (section 3-2.2, TenBrook *et al.* 2009a), two acute studies yielding four toxicity values from two taxa were judged reliable and relevant for acute criterion derivation (Tables 3 and 4). Twenty-four acute toxicity animal values from nine studies were rated RL, LL, or LR and were used as supplemental information for evaluation of the derived acute criteria in the Sensitive Species section 12 (Table 5). Three studies yielding ten chronic animal toxicity values were rated RR (Table 6b). Eight chronic toxicity animal values from five studies were rated RL, LL, or LR (Table 8b).

Evaluation of aquatic plant data

Plant data were used to derive the chronic criterion instead of chronic animal data because diuron is an herbicide and plants are the most sensitive taxa (section 3-4.3, TenBrook *et al.* 2009a). All plant studies were considered chronic because the typical endpoints of growth or reproduction are inherently chronic. Three studies yielding seven plant toxicity values were rated RR for the chronic criterion derivation (Tables 6a and 7). Supplemental information for the derived chronic criteria includes 70 plant toxicity values from 21 studies (Table 8a).

Plant studies are much more difficult to interpret than animal data because a variety of endpoints may be used, but the significance of each one is not clear. In this methodology, only endpoints of growth or reproduction (measured by biomass) and tests lasting at least 24-h had the potential to be rated highly and used for criteria calculation, which is in accordance with standard methods (ASTM 2007a, 2007b; USEPA 1996). The plant studies were rated for quality using the data evaluation criteria described in the methodology (section 3-2.2, TenBrook *et al.* 2009a).

There are several endpoints listed in the tables for plant data. The endpoints are explained here for clarity and the description includes if the endpoint is clearly linked to survival, growth, or reproduction.

Growth inhibition: All of these endpoints are relative to a control growth measurement. Depending on the plant it may have been measured by direct cell counts with a hemacytometer, cell counts with a spectrophotometer, cell counts with an electronic particle counter, chlorophyll concentration measured by absorbance, turbidity measured by absorbance, or number of fronds (*Lemna spp.*). In all cases, growth of exposed samples was compared statistically to controls.

Relative Growth Rate: Biomass of macrophytes was measured before and after exposure to calculate a growth rate as (final mass-initial mass)/initial mass x 100. This endpoint is very similar to growth inhibition, except it is expressed as a positive effect, while growth inhibition is expressed a negative effect. In all cases, growth rate of exposed samples was compared statistically to controls.

Change in chlorophyll fluorescence ratio: Chlorophyll fluorescence was measured at a maximal fluorescence and either a variable or steady-state fluorescence and a ratio was computed. An increase in the ratio indicates a disruption of photosystem II, which may lead to a decrease in carbohydrate production and thus decreased growth. This endpoint measures physiological stress in plants (Lambert *et al.* 2006). This ratio is a valid

measurement that is related to algal growth according to ASTM Standard Method E1218-04 (ASTM 2004), but is described as less definitive than measuring chlorophyll *a* content, and is therefore not a preferred endpoint if an endpoint more directly related to growth is available.

Reduced oxygen evolution: Plants evolve oxygen during photosynthesis, and reduced photosynthesis has been shown to correlate well with the concentrations that inhibit growth by Walsh (1972), but it is not clear that this endpoint is a good predictor of growth inhibition across all plant species. This endpoint is always calculated as relative to controls.

6. Data reduction

Multiple toxicity values for diuron for the same species were reduced down to one species mean acute value (SMAV) or one species mean chronic value (SMCV) according to procedures described in the methodology (section 3-2.4, TenBrook *et al.* 2009a). Acceptable acute and chronic data that were reduced, and the reasons for their exclusion, are shown in Tables 4 and 7, respectively. Reasons for reduction of data included: a test with a more sensitive exposure duration for the same species was available, flow-through tests are preferred over static tests, a test with a more sensitive life-stage of the same species was available, and tests with more sensitive endpoints were available. The final acute animal, chronic plant, and chronic animal data sets are shown in Tables 3, 6a, and 6b, respectively. The final acute data set contains three SMAVs, and the final chronic plant data set contains three SMCVs.

7. Acute criterion calculation

An acute criterion was calculated with acute animal toxicity data only, because plant toxicity tests are always considered chronic (section 3-2.1.1.1, TenBrook *et al.* 2009a). Since acceptable acute toxicity values were not available from the five required taxa, the acute criterion was calculated using the Assessment Factor (AF) procedure (section 3-3.3, TenBrook *et al.* 2009a). Diuron is an organic pesticide, and the AFs given in the methodology (Table 3.13, TenBrook *et al.* 2009a) are the most specific AFs available for organic pesticides. The methodology points out that the AFs are limited in that they are based on organochlorine and one organophosphate pesticides, which are neurotoxic insecticides, while diuron is an herbicide that inhibits photosynthesis. However, diuron is a chlorinated compound that does exhibit toxicity to animals with an unclear mechanism, and diuron is an organic pesticide, thus, it is reasonable to use the AF procedure for diuron.

The AFs given in the methodology will be used for diuron with the understanding that AFs based on measured pesticide toxicity data are likely more accurate than choosing an arbitrary AF. The methodology points out that AFs are recognized as a conservative approach for dealing with uncertainty in assessing risks posed by chemicals (section 2-3.2, TenBrook *et al.* 2009a). Using an AF to calculate a criterion always involves a high degree of uncertainty and there is potential or under- or over-protection, which is strongly dependent on the representation of sensitive species in the available

data set. The methodology instructs that the derived criterion should be compared to all available ecotoxicity data to ensure that it will be protective of all species (section 3-6.0, TenBrook *et al.* 2009a).

There are two available taxa in the acceptable (RR) data set shown in the in Table 3: planktonic crustaceans (*Daphnia magna* and *D. pulex*) and a benthic invertebrate (*Hyalella azteca*). Missing from the taxa requirements for use of a species sensitivity distribution (SSD) are a fish from the family Salmonidae, a warm water fish, and an insect. The AF method calculates the criterion by dividing the lowest SMAV from the acceptable (RR) data set by an AF, which is determined by the number of taxa available in the data set (section 3-3.3, TenBrook *et al.* 2009a). The lowest SMAV was the 48-h *Daphnia magna* LC_{50} value of 12 mg/L. This value was divided by an AF of 36 because there are acceptable data from two taxa (Table 3.13, TenBrook *et al.* 2009a). The acute value calculated using the AF represents an estimate of the median 5^{th} percentile value of the SSD, which is the recommended acute value. The recommended acute value is divided by a factor of 2 to calculate the acute criterion (section 3-3.3, TenBrook *et al.* 2009a). Because the toxicity data used to calculate the criterion only reported two significant figures, the criterion is rounded to two significant figures (section 3-3.2.6, TenBrook *et al.* 2009a).

Acute value = lowest SMAV ÷ assessment factor
= 12 mg/L ÷ 36
= 0.333 mg/L

Acute criterion = acute value ÷ 2
= 0.333 mg/L ÷ 2
= 0.167 mg/L

Acute criterion = 0.17 mg/L
= 170 µg/L

8. Chronic criterion calculation

Diuron is an herbicide and the chronic data in Table 6a demonstrate that plants are the most sensitive taxa; therefore, the procedure for derivation of the chronic criterion of an herbicide was followed (section 3-4.3, TenBrook *et al.* 2009a). The chronic criterion is derived to be protective of plants, but will also likely be protective of animals, which are less sensitive to diuron. Acceptable chronic toxicity values were not available for five different species of vascular plants or alga, so a distribution could not be fit to the available toxicity data (part 1, section 3-4.3, TenBrook *et al.* 2009a). The methodology instructs that in the absence of acceptable data to fit a distribution, the chronic criterion is equal to the lowest NOEC from an important alga or vascular aquatic plant species that has measured concentrations and a biologically relevant endpoint (part 2, section 3-4.3, TenBrook *et al.* 2009a). Acceptable toxicity data for the green algae *Pseudokirchneriella subcapitata* (formerly *Selenastrum capricornutum)* is shown in Table 6a, and the NOEC value reported for this species serves as the chronic criterion.

Chronic criterion = 1.3 µg/L

9. Bioavailability

Few studies were found concerning the bioavailability of diuron, and only one study was found pertaining to bioavailability to organisms in the water column. Knauer *et al.* (2007) found that the presence of black carbon (BC) in the water column can reduce the toxicity of diuron to the freshwater green algae *Pseudokirchneriella subcapitata* (formerly *Selenastrum capricornutum)* due to sorption of diuron to BC. BC is ubiquitous in the environment because it is a product of incomplete combustion and can act as a supersorbent for some organic contaminants, but it is only a small fraction of total organic carbon, which is usually responsible for the majority of sorption to solids. BC in its native form, compared to isolated and re-combusted BC, was much less effective at sorbing diuron, and subsequently reducing toxicity. This study indicates that sorption of diuron to BC reduces bioavailability, but it does not provide enough information about sorption to recommend basing compliance on less than the whole water concentrations. Studies investigating the sorption of diuron to dissolved organic carbon and clays are not currently available in the literature, but sorption to these materials is also likely to inhibit bioavailability in a similar manner as sorption to BC. No other information about bioavailability of diuron in the water column that differentiates when diuron is sorbed to solids, sorbed to dissolved solids, or freely dissolved was found. Until there is more information that discusses the bioavailability of these three phases, compliance must be based on the total concentration of diuron in water (section 3-5.1, TenBrook *et al.* 2009a).

10. Mixtures

Diuron can occur in the environment with other herbicides of similar or different modes of action. Diuron is a photosystem II (PSII) inhibitor, as are all phenylurea herbicides. Other widely used herbicides, such as the triazines, are also PSII inhibitors, but have different binding sites than the phenylurea herbicides. The concentration addition model and the non-additive interaction model are the only predictive mixture models recommended by the methodology (section 3-5.2, TenBrook *et al.* 2009a), so other models found in the literature will not be considered for compliance.

Several studies have confirmed that toxicity of a mixture of herbicides that are PSII-inhibitors can be predicted by the concentration addition method (Arrhenius *et al.* 2004; Backhaus *et al.* 2004; Knauert *et al.* 2008). Knauert *et al.* (2008) studied the effects of a mixture of the herbicides diuron, atrazine and isoproturon, as well as the single herbicides, in a mesocosm environment using the toxicity unit (TU) approach. In these tests, single herbicides exhibited the same inhibition of phytoplankton photosynthesis as a mixture containing 1/3 the toxicity unit concentration of each herbicide, showing that the TU approach is an accurate for calculating toxicity of a mixture of PSII-inhibitor herbicides. Backhaus *et al.* (2004) tested a mixture of 12 phenylurea herbicides with a unicellular green alga *Scenedesmus vacuolatus* and found that the combined toxicity could be predicted by concentration addition, but also equally well by independent action.

Arrhenius *et al.* (2004) also concluded that the concentration addition method is accurate for predicting mixture toxicity for PSII-inhibitor herbicide mixtures in algal communities. Based on this evidence, the concentration addition method should be used to determine compliance in cases where PSII-inhibitor mixtures occur if the other pesticides considered in the model have numeric water quality criteria. If numeric water quality criteria are not available for the other pesticides the model cannot be used and diuron should be considered alone.

Lydy and Austin (2005) studied the toxicity mixtures of diuron with organophosphate insecticides to *Chironomus tentans* and found some acted as synergists with diuron. The synergistic ratios (K) for diuron in a binary mixture with 50 µg/L chlorpyrifos or 100 µg/L methidathion are 1.5 and 4.8, respectively. Diuron mixed with azinphos methyl or diazinon produced no effect on toxicity. However, because the K value is only for a single species at a single concentration it cannot be used to assess compliance with water quality criteria; it can be used to assess the potential harm for *Chironomus tentans* itself if there are numeric water quality criterion for chlorpyrifos and methidathion.

Teisseire *et al.* (1999) examined the phytotoxicity of diuron combined with two fungicides (copper and folpet) on duckweed (*Lemna minor*) because these pesticides are often used in combination in vineyards. They found that growth inhibition due to the combination of diuron and copper depended on the concentrations of both chemicals, while it only depended on the diuron concentration when combined with folpet. The combination of copper and diuron was found to be additive for most concentrations, but slight antagonism was observed for several concentrations. This data cannot be used to determine compliance because neither the concentration addition nor the non-additivity model can be used. The concentration addition model cannot be used because diuron and copper do not have the same modes of action and a multi-species K value is not available for this mixture so the non-additivity model cannot be used.

Diuron is widely used as an anti-fouling biocide in paint for ship hulls and is often used in combination with other anti-fouling agents. Several articles were found that studied the toxicity of mixtures of diuron and other anti-fouling agents, including: Irgarol (cybutryne) and Sea nine 211 (4, 5-dichloro-2-n-octyl-3(2H)-isothiazolone) (Fernandez-Alba *et al.* 2002); diuron metabolites and copper (Gatidou and Thomaidis 2007); chlorothalonil, copper pyrithione, and zinc pyrithione (Koutsaftis and Aoyama 2007); copper and Irgarol (Manzo *et al.* 2008); Irgarol (Chesworth *et al.* 2004); and tri-*n*-butyltin (Molander *et al.* 1992b). Resulting toxicities were synergistic, additive, and antagonistic for different mixtures, sometimes depending on concentration ratios and how many compounds were in the mixture. None of these studies reported a coefficient of interaction and most were tested with saltwater species, so the data cannot be used to assess mixture toxicity, but they do provide evidence that synergistic, additive and antagonistic effects are all possible with other chemicals commonly used with diuron.

Other studies have focused on mixtures with contaminants or other types of pesticides. Walker (1965) found that diuron combined with trichloroacetate (TCA), used

for aquatic plant control in aquaculture, reduced the EC_{50} of bluegills 4-5 fold compared to diuron alone. The author also states that the carrier in the emulsifiable mixture of diuron and TCA contributed to the increased toxicity. Hernando *et al.* (2003) found that methyl-*tert*-butyl ether (MTBE), a common ground- and surface-water contaminant, had a synergistic effect when used in combination with diuron. The addition of MTBE increased diuron toxicity to the bacterium *Vibrio fischeri* by 50% in a shorter exposure duration than diuron alone. *Daphnia magna* was also tested with the combination, but no change in toxicity was observed compared to diuron alone. A coefficient of interaction was not calculated so this data cannot be used to assess criteria compliance.

In summary, when diuron is detected with other PSII-inhibitor herbicides the toxicity should be predicted by the additive concentration addition model. There are no multi-species coefficients of interaction reported in the literature, so the non-additive interaction model cannot be used to assess water quality criteria compliance when other types of contaminants are present.

11. Temperature, pH, and other water quality effects

Temperature, pH, and other water quality effects on the toxicity of diuron were examined to determine if any effects are described well enough in the literature to incorporate into criteria compliance (section 3-5.3, TenBrook *et al.* 2009a). There were no studies available that examined the effects of temperature or pH on toxicity in the aqueous environment. As diuron is only a very weak base, pH is not expected to have a significant affect on the chemical structure.

12. Sensitive species

The derived criteria are compared to toxicity values for the most sensitive species in both the acceptable (RR) and supplemental (RL, LR, LL) data sets to ensure that these species will be adequately protected (section 3-6.1, TenBrook *et al.* 2009a).

The lowest acute value in the data sets rated RR, RL, LR, or LL (Tables 3, 4, and 5) is 160 µg/L for the amphipod *Gammarus lacustris* (Sanders 1969). This study rated LL because the control response was not reported and many other study details were also not documented. The lack of documentation makes this study less reliable, but it is still a relevant toxicity study. This study tested a freshwater species that resides in North America, the endpoint and exposure duration fit into the acute test definition in the methodology (section 3-2.1.1.1), they used technical grade diuron, and reported toxicity values with 95% confidence intervals. Additionally, data for another amphipod, *Gammarus fasciatus*, is the next lowest acute value in the data set (700 µg/L), indicating that *Gammarus* species are particularly sensitive to diuron (Table 5). The methodology states that criteria should only be adjusted based on data for sensitive species when toxicity values based on measured concentrations are available (section 3-6.1, TenBrook *et al.* 2009a). The Sanders (1969) study used nominal concentrations to calculate toxicity values; therefore, the acute criterion should not be adjusted based on this study. If highly

rated measured data for *Gammarus* species is available in the future, it should be examined to determine if the acute criterion is protective of this sensitive genus.

The derived chronic criterion (1.3 µg/L) is below all chronic data that was highly rated (Table 6a), while there are some values that are lower in the supplemental data set rated RL, LR, or LL (Table 8a). The chronic criterion was not adjusted because the studies reporting lower toxicity values were lacking at least one of the following critical parameters: 1) the use of an endpoint that directly relates to survival, growth, or reproduction (sections 2-2.1.3 and 3-2.1.1.3); 2) the use of an exposure duration of ≥24-h (ASTM 2007a, 2007b; USEPA 1996); 3) proper design of hypothesis tests and reporting of parameters used to evaluate the reasonableness of the resulting toxicity values (section 2-2.1.2); 4) the use of diuron ≥80% purity, and 5) the use of freshwater species (TenBrook *et al.* 2009a). These studies are discussed in detail below.

The lowest measured chronic value in the data sets is an EC_{50} of 0.00026 µg/L for the rooted macrophyte *Apium nodiflorum* for a non-standard endpoint of root growth (Lambert *et al.* 2006). This value was calculated by extrapolation, not interpolation, and is lower than the reported NOEC and below the lowest concentration tested, and therefore is not a toxicity value that should be used for criteria calculation. There are several other NOEC values reported in this study for an appropriate endpoint (relative growth rate) that are below the proposed chronic criterion (0.0005-0.05 µg/L). It is not possible to evaluate the reasonableness of these NOEC values as outlined in the methodology (sections 2-2.1.2, TenBrook *et al.* 2009a) because the control responses are not reported, the p-value selected is not reported, and a minimum significant difference was not calculated. LOEC values were not reported, and cannot be calculated because the control responses are not reported. Preferably, a NOEC, LOEC and MATC would be reported so that the NOEC and LOEC values could be evaluated for reasonableness and the MATC could be compared to the proposed chronic criterion. The dilution factors used in this study were also too large (10 or 100); the diuron test concentrations were 0.5, 50, 500, and 5000 ng/L. The dilution factor should be between 1.5 and 3.2 to ensure that hypothesis test results are reasonable and to decrease uncertainty in the toxicity values (section 2-2.1.2, TenBrook *et al.* 2009). Because of the lack of information available to assess whether the NOEC values reported in this study are reasonable approximations of no-effect levels, and the poor hypothesis test study design, we do not recommend downward adjustment of the chronic criterion based on these data.

Podola and Melkonian (2005) report NOEC and LOEC values of 0.1 and 0.5 µg/L, respectively, for nine different algae. These values are below the proposed criteria, but this study used a less preferred endpoint, change in chlorophyll fluorescence, and a very short, non-standard, exposure duration of 20 min. The authors propose the use of a biosensor to detect and identify herbicides in the environment, and do not discuss the link between the effects they quantify and survival, growth, or reproduction of the algal strains. The endpoint and duration used in this study generate toxicity values that may demonstrate exposure to diuron, but do not directly demonstrate that the exposures adversely effected survival, growth, or reproduction. Therefore, we do not recommend downward adjustment of the chronic criterion based on these data.

Eullaffroy and Vernet (2003) report a toxicity threshold of 1 µg/L for green algae, which is slightly below the chronic criterion. This test used an exposure duration of only 1 min, and its purpose was to rapidly detect herbicides in the environment. This study did not follow a standard method, used an extremely short exposure duration, and does not report an acceptable toxicity value (NOEC, LOEC, MATC, EC_x). The toxicity value from this study cannot be directly related to survival, growth or reproduction, and likely only demonstrates exposure to diuron, not adverse effects. We do not recommend downward adjustment of the chronic criterion based on this toxicity value.

Two studies (Ma *et al.* 2001, Ma 2002a) containing the same data for the alga *Chlorella pyrenoidosa* reported EC_{50} values equal to the derived criterion. These studies used diuron with purity of 50%, did not report a control response and were rated L for reliability because many other standard study details were not reported. Another study by Ma *et al.* (2006) reported an EC_{50} below the derived criterion (0.7 µg/L), but also used diuron of 50% purity and lacked other study details. It is very important to use chemicals of high purity in toxicity testing because impurities or other chemicals present in formulations may cause toxicity effects unrelated to the chemical of interest. Because these tests used wettable powder formulations, containing only 50% diuron, we do not have confidence that the resulting toxicity effects were directly caused by diuron, as the other chemicals in the formulations could have also contributed to toxicity. We do not recommend downward adjustment of the chronic criterion based on these toxicity values.

One study that used saltwater organisms (Ukeles 1962) reported toxicity values below the derived chronic criterion (0.02 and 0.4 µg/L), but saltwater organisms are suspected to have different sensitivities than freshwater organisms; therefore, they are not used to derive freshwater criteria. The values in Table 8a indicate that saltwater organisms may be generally more sensitive to diuron than freshwater organisms.

Overall, it is recommended that the chronic plant toxicity values in the supplemental data that are below the derived chronic criterion are not used to adjust the criterion, because these studies were not found to be relevant and reliable for criteria generation for the various reasons described in this section.

13. Ecosystem and other studies

The derived criteria are compared to acceptable laboratory, field, or semi-field multispecies studies (rated R or L) to determine if the criteria will be protective of ecosystems (section 3-6.2, TenBrook *et al.* 2009a). Eleven mesocosm, microcosm or ecosystem (field and laboratory) studies were identified. Two studies tested saltwater species and can only be used as supplemental information (Molander & Blanck 1992a; Devilla *et al.* 2005). Eight freshwater studies rated as acceptable (R or L; Table 9). Three of the studies were rated R (Hartgers *et al.* 1998; Sumpono *et al.* 2003; Tlili *et al.* 2008), and seven were rated L (Devilla *et al.* 2005; Dorigo *et al.* 2007; Flum & Shannon 1987; Molander & Blanck 1992a; Perschbacher & Ludwig 2004; Pesce *et al.* 2006; Zimba *et al.* 2002) and are used as supporting data. These studies were almost all indoor or laboratory

studies mimicking small river or pond natural environments and examining microbial, phytoplanktonic, or bacterial communities. Most of these studies noted an initial drop in phytoplankton biomass, which led to a decrease in dissolved oxygen due to the decay of the phytoplankton. Two studies report a community EC_{50} (Dorigo *et al.* 2007; Flum & Shannon 1987), and one study reported a NOEC (Hartgers *et al.* 1998) to which the calculated criteria may be compared.

Plankton communities have displayed varying degrees of response to diuron, depending on, among other things, the concentrations applied. Hartgers *et al.* (1998) set up microcosms containing phyto-, peri-, bacterio- and zooplankton and monitored them for a 28-d chronic exposure to a mixture of diuron, atrazine, and metolachlor, and a 28-d recovery period. A NOEC for the mixture based on phytoplankton was determined to be 1.5 µg/L diuron, 5.4 µg/L atrazine, and 5.6 µg/L metolachlor. The derived chronic criterion is slightly lower than the diuron NOEC, and thus the criterion would likely be protective of phytoplankton based solely on diuron. Flum and Shannon (1987) reported an 96-hr EC_{50} of 2205 µg/L (1630-3075 µg/L 95% CI) for an artificial microecosystem containing zooplankton, amphipods, ostracods, unicellular and filamentous algae, protozoans, and microbes, which is much higher than the derived chronic criterion. The EC_{50} value was based on monitoring the redox potential, pH, and dissolved oxygen as measure of toxicity.

Plankton and algae communities exposed to diuron have been studied in regard to the aquaculture industry because some algae give fish an "off" flavor, yet plankton is necessary for healthy ponds. Zimba *et al.* (2002) assessed the effect of 9 weeks of diuron application (10 µg/L) on catfish pond ecology. Algae, phyto-, zoo-, and ultra-plankton composition and biomass were examined as well as water quality. The only significant effect of the diuron exposure was a change in the phytoplankton composition; the phytoplankton biomass was not altered. Perschbacher and Ludwig (2004) also studied plankton communities in outdoor pool mesocosms simulating aquaculture ponds. Three diuron concentrations were tested and monitored for 4 weeks post-application. Diuron depressed primary production and biomass of phytoplankton for at least 4 weeks post-application, which in turn caused a decrease in dissolved oxygen to levels that are potentially lethal to fish. The concentrations were reported as field rate (1.4 kg a.i./ha), 1/10 field rate, and 1/100 field rate of Direx without adjuvants, but were not measured. Low dissolved oxygen (< 4 ppm) occurred only for the two highest diuron applications at 10 d post-application, and it took until 3 weeks post-application for dissolved oxygen levels to return to close to that of the control ponds. Fish were not used in this study, but it is known that low dissolved oxygen can be potentially lethal to fish.

Tlili *et al.* (2008) studied biofilm communities in a small river with chronic exposure to 1 µg/L diuron, as well as 3-hour pulses of 7 µg/L or 14 µg/L diuron with and without prior exposure. The results indicate that photosynthesis was never significantly inhibited by any of the treatments, but the pulses did alter the community structure of the microalgae. The pulses affected the eukaryotic community structure in microcosms that did not have prior chronic diuron exposure, but had no significant impact on those that did have prior exposure. Dorigo *et al.* (2007) assessed prokaryotic and eukaryotic

communities and microalgae exposed to vineyard runoff water in a small stream containing diuron concentrations of 0.09 and 0.43 µg/L. The diuron tolerance in these communities increased in the downstream direction and the pristine control site had the lowest tolerance, following the concept that contaminant exposure increases the tolerance of biofilms either by adaptation or species changes. The endpoints in these studies are not clearly linked to survival, growth and reproduction and do not exhibit a clear dose-response relationship, so it is not clear if diuron exposure at these levels impacted the diversity of species of biofilm communities. Biofilm community restructuring may have long-term effects on an ecosystem, however, the studies available only provide preliminary data on this subject. If more in-depth data becomes available on this topic, it should be incorporated into criteria derivation.

Several other studies also look at the impact of diuron on microbes. Pesce *et al.* (2006) reported that a 21-d exposure of 10 µg/L prevented the implementation and development of a productive microbial community in a riverine microcosm, but the derived chronic criterion is well below this concentration. Sumpono *et al.* (2003) studied the effects of diuron on aquatic bacteria in a wastewater treatment pond model ecosystem. The single concentration exposure was 12.5 mg/L, which is well above the acute and chronic criteria. Photosynthetic microorganisms decreased, but bacteria proliferated with diuron exposure, likely due to the bacteria using the detritus as a new carbon source.

The literature shows that herbicides in aquatic ecosystems may have detrimental effects on the bottom of the food chain, which may indirectly impact species up the food chain via changes in water quality or decreased food supply. However, many of these studies only tested a single concentration, and no dose-response relationship can be inferred and no-effect concentrations are not available. Considering the available studies, it appears that the derived acute and chronic criteria could be protective of these types of negative effects because most studies used much higher exposure concentrations. The only studies that reported effects at concentrations lower than the derived chronic criterion examined biofilm community restructuring, and provide preliminary data that cannot be incorporated into criteria derivation until more in-depth studies are available (Dorigo *et al.* 2007; Tlili *et al.* 2008).

14. Threatened and endangered species

The derived criteria are compared to measured toxicity values for threatened and endangered species (TES), as well as to predicted toxicity values for TES, to ensure that they will be protective of these species (section 3-6.3, TenBrook *et al.* 2009a). Current lists of state and federally listed threatened and endangered plant and animal species in California were obtained from the California Department of Fish and Game website (CDFG 2008). Several listed animal species are represented in the dataset. The California red-legged frog (*Rana aurora draytonii*) is represented in the data set by *Rana aurora* from a study rated RR with an LC_{50} of 22.2 mg/L for a 14-d test, well above the derived criteria. Five Evolutionarily Significant Units of *Oncorhynchus mykiss* are listed as federally threatened or endangered throughout California. The acute data set include two

96-hr LC_{50} values for *O. mykiss* of 4.9 (4.1-5.9) mg/L and 16 (11.3-22.7) mg/L (Johnson and Finley 1980). Data is also available for Cutthroat trout (*Oncorhynchus clarki*), of which the subspecies Lahontan cutthroat trout (*O. c. henshawi)* is listed as federally threatened. The *O. clarki* LC_{50} is reported by Johnson and Finley (1980) is 1.4 (1.1-1.9) mg/L. These data indicate that the acute criterion of 170 µg/L would be protective of these two species.

The USEPA interspecies correlation estimation (ICE v. 3.1; USEPA 2010) software was used to estimate toxicity values for the listed animals or plants represented in the acute data set by members of the same family or genus. Table 10 summarizes the results of the ICE analyses. The estimated toxicity values in Table 10 range from 1.673 – 8.086 mg/L for Coho salmon, 5.983 mg/L for Chinook salmon, and 4.758 mg/L for Lahontan cutthroat trout.

No plant studies used in the criteria derivation were of state or federal endangered, threatened or rare species. Plants are particularly sensitive to diuron because it is an herbicide, but there are no aquatic plants listed as state or federal endangered, threatened or rare species so they could not be considered in this section.

Based on the available data and estimated values for animals, there is no evidence that the calculated acute and chronic criteria will be underprotective of threatened and endangered species.

15. Bioaccumulation

Bioaccumulation was assessed to ensure that the derived criteria will not lead to unacceptable levels of diuron in food items (section 3-7.1, TenBrook *et al.* 2009a). Diuron has a log K_{ow} of 2.78 (Sangster Research Laboratories 2008), and a molecular weight of 233.1, which indicates a low bioaccumulative potential. There is a USEPA pesticide tolerance for farm-raised freshwater finfish tissue of 2.0 mg/kg (USEPA 2007), but there are no FDA food tolerances for diuron (USFDA 2000). Bioconcentration of diuron has been measured in fathead minnow, mosquito fish, snails, daphnids, and algae (Table 1).

Isensee (1976) measured bioconcentration in model ecosystems of mosquito fish (*Gambusia affinis)*, snails *(Physa spp*.), daphnids (*Daphnia magna),* and algae (*Oedogonium cardiacum).* The model ecosystem was designed to simulate contamination due to erosion in a static system. Soil was spiked with ^{14}C-labeled diuron and clean water was added and allowed to equilibrate 1-d before all organisms were added, except fish, which were added after 30 d, when daphnids were removed and analyzed. All other animals were harvested and analyzed at 33 d. Bioconcentration factors (BCFs) for the four organisms range from 40-290 and are listed in Table 1.

Bioconcentration of diuron was measured in fathead minnow by Call *et al.* (1983, 1987) in a flow-through system. Test aquaria water was spiked with ^{14}C-labeled diuron containing 30-d old fathead minnows at two aqueous concentrations (3.15 and 30.4

µg/L). Fish were removed and analyzed at nine time points up to 24 d. A mean BCF of 2.0 was determined for diuron from the two test concentrations. Call *et al.* (1983, 1987) also documented rapid metabolism and elimination of diuron in fathead minnows and rainbow trout. The available studies show that diuron has a low potential for bioaccumulation in the environment.

To check that these criteria are protective of terrestrial wildlife that may consume aquatic organisms, a bioaccumulation factor (BAF) was used to estimate the water concentration that would roughly equate to a reported toxicity value for such terrestrial wildlife ($LC_{50,\ oral\ predator}$). These calculations are further described in section 3-7.1 of the methodology (TenBrook *et al.* 2009a). The BAF of a given chemical is the product of the BCF and a biomagnification factor (BMF), such that BAF=BCF*BMF. No BAF or BMF values were found for diuron. Chronic dietary toxicity values are preferred for this calculation, but none were identified in the literature. A subacute dietary LC_{50} of 1730 mg/kg feed for mallard (USEPA 2003) was the lowest dietary toxicity value available. While several BCF values are available, the value given by Call *et al.* (1983, 1987) is the most reliable because they used a flow-through test and the study rated RR for their chronic test. The subacute dietary LC_{50} of 1730 mg/kg feed for mallard (USEPA 2003) and the BCF of 2.0 L/kg for *Pimephales promelas* (Call *et al.* 1983, 1987) were used as an example estimation of bioaccumulation in the environment. A default BMF of 1 was chosen based on the log K_{ow} of diuron (Table 3.15, TenBrook *et al.* 2009a) because no biomagnification data was found in the literature.

$$NOEC_{water} = \frac{LC_{50,oral\_predator}}{BCF_{food\_item} * BMF_{food\_item}}$$

Mallard:
$$NOEC_{water} = \frac{1730\,^{mg}/_{kg}}{2.0\,^{L}/_{kg} * 1} = 865\,^{mg}/_{L} = 865{,}000\,^{\mu g}/_{L}$$

The EPA pesticide tolerance for farm-raised freshwater finfish tissue of 2.0 mg/kg (USEPA 2007) was used to make a similar estimation for human health. This is an attempt to anticipate if concentrations allowed by the derived chronic criterion could bioaccumulate in fish to a level that could be toxic to humans that consume fish.

Human:
$$NOEC_{water} = \frac{2.0\,^{mg}/_{kg}}{2.0\,^{L}/_{kg} * 1} = 1.0\,^{mg}/_{L} = 1000\,^{\mu g}/_{L}$$

In this example, the calculated chronic criterion is more than five orders of magnitude below the estimated $NOEC_{water}$ value for wildlife and is not expected to cause adverse effects due to bioaccumulation. The chronic criterion is a factor of 770 below the estimated $NOEC_{water}$ value for human health, and is not expected to cause adverse effects to humans due to bioaccumulation in food sources.

16. Harmonization with air and sediment criteria

This section addresses how the maximum allowable concentration of diuron might impact life in other environmental compartments through partitioning (section 3-7.2, TenBrook *et al.* 2009a). The only available sediment criterion for diuron is estimated based on partitioning from water using empirical K_{oc} values. There are no other federal or state sediment or air quality standards for diuron (CARB 2008; CDWR 1995), nor is diuron mentioned in the NOAA sediment quality guidelines (NOAA 1999). For biota, the limited data on bioconcentration or biomagnification of diuron is addressed in section 15.

17. Limitations, assumptions, and uncertainties

The assumptions, limitations and uncertainties involved in criteria generation are available to inform environmental managers of the accuracy and confidence in criteria (section 3-8.0, TenBrook *et al.* 2009a). Chapter 2 of the methodology (TenBrook *et al.* 2009a) discusses these points for each section as different procedures were chosen, such as the list of assumptions associated with using an SSD (section 2-3.1.5.1), and reviews them in section 2-7.0. This section summarizes any data limitations that affected the procedure used to determine the final diuron criteria.

One major limitation was the lack of highly rated acute toxicity data for diuron, which prevented the use of a SSD for criterion derivation. Only two of the five taxa required for use of a SSD were available; the three missing taxa were a warm water fish, a fish from the family Salmonidae, and an insect. Due to this lack of data, an AF was used to calculate the acute criterion. Uncertainty cannot be quantified using the AF procedure, as it is based on only one toxicity value. There were no highly rated amphipod data available, which is an important data gap, as this taxon appears to be the most sensitive animal taxa to diuron. There were no acceptable measured data for amphipods, so the acute criterion could not be adjusted. If highly rated measured data for amphipods becomes available in the future, the diuron acute criterion should be re-evaluated to ensure protection of this sensitive taxon.

The most important limitation is the lack of acceptable plant data because diuron is an herbicide. Plant and algal data is difficult to interpret and do not use consistent endpoints. The assumptions that went into evaluation of plant studies are described in section 5. The chronic data set only contained three plant values, precluding the use of a SSD, and only two of the studies reported a NOEC, LOEC, and MATC, which are the appropriate toxicity values for chronic tests. The methodology requires that MATC values are used to derive chronic criterion by the SSD procedure, unless studies are available with EC_x values that show what level of x is appropriate to represent a no-effect level (section 3-2.1.1.2, TenBrook *et al.* 2009a). The chronic criterion was derived with the absolute minimum amount of data according to the methodology (section 3-4.3, TenBrook *et al.* 2009a), and uncertainty in the chronic criterion cannot be quantified because it is based on only one toxicity value.

Chronic animal taxa requirements were almost met, only data on a cold water fish was missing, but chronic animal data is not used for chronic criterion derivation of an herbicide, or when plants are the most sensitive taxa to a particular pesticide (3-4.3, TenBrook *et al.* 2009a). Although diuron is an herbicide, some animals do show sensitivity to it.

Other limitations include the lack of information about diuron and mixture toxicity and ecosystem-level effects. There is evidence that diuron exhibits synergism with some other chemicals, including organophosphate pesticides, but there is a lack of multispecies interaction coefficients available to incorporate the presence of chemical mixtures into criteria compliance (section 10). Biofilms displayed sublethal effects to low-level diuron exposures, but these effects need to be further investigated to determine if the exposures are linked to survival, growth or reproduction of organisms in biofilms (section 13). Another issue to consider is the averaging periods of the acute and chronic criteria. The chronic 4-d averaging period should be protective based on available data. However, the acute criterion is very high when compared to plant data, and it may allow for a pulse that could kill off a large amount of algae, resulting in increased biological demand and potential fish kills due to low dissolved oxygen, as discussed in section 13. Necessary information on the timing and concentrations that could cause this effect is not obvious from the data found.

Confidence intervals or other measures of uncertainty could not be calculated for either criterion because they are each based on only one value.

18. Comparison to national standard methods

This section is provided as a comparison between the UC-Davis methodology for criteria calculation (TenBrook *et al.* 2009a) and the current USEPA (1985) national standard. The following example diuron criteria were generated using the USEPA (1985) methodology with the data set generated in this diuron criteria report.

The USEPA acute methods have three additional taxa requirements beyond the five required by the SSD procedure of the UC-Davis methodology (section 3-3.1, TenBrook *et al.* 2009a). They are:

1. A third family in the phylum Chordata (e.g., fish, amphibian);
2. A family in a phylum other than Arthropoda or Chordata (e.g., Rotifera, Annelida, Mollusca);
3. A family in any order of insect or any phylum not already represented.

None of the three additional requirements could be met; only two of the eight total taxa requirements are available in the data set. A planktonic crustacean (*Daphnia magna* or *Daphnia pulex*) and a benthic invertebrate (*Hyalella azteca*) are available, while a fish from the family Salmonidae, a warm water fish, an insect and the three additional taxa requirements of the USEPA (1985) methodology are all missing. Because of this lack of data, no acute criterion could be calculated according to the USEPA (1985) methodology.

According to the USEPA (1985) methodology, the chronic criterion is equal to the lowest of the Final Chronic Value, the Final Plant Value, and the Final Residue Value. To calculate the Final Chronic Value, animal data is used and the same taxa requirements must be met as in the calculation of the acute criterion. Seven of the eight taxa requirements are available in the RR chronic animal data set (Table 6b). The missing taxon is a fish from the family Salmonidae; the seven available taxa are as follows:

1. A planktonic crustacean (*Daphnia pulex*)
2. A benthic invertebrate (*Hyalella azteca*)
3. An insect (*Chironomus tentans*)
4. A warm water fish (*Pimephales promelas*)
5. A third family in the phylum Chordata (either *Pseudacris regilla, Rana aurora or catesbeiana, or Xenopus laevis*)
6. A family in a phylum other than Arthropoda or Chordata (*Physa* sp.)
7. A family in any order of insect or any phylum not already represented (*Lumbriculus variegatus*).

The California Department of Fish and Game has derived criteria using the USEPA (1985) SSD method with fewer than the eight required families, using professional judgment to determine that species in the missing categories were relatively insensitive and their addition would not lower the criteria (Menconi & Beckman 1996; Siepmann & Jones 1998). It is not clear that a fish from the family Salmonidae would be relatively insensitive to diuron, because the lowest animal chronic toxicity value is for a fish (*Pimephales promelas*). As an example, the data in Table 6b were used to calculate genus mean chronic values from the given SMCVs, and the log-triangular distribution was employed to yield a 5^{th} percentile estimate.

Final Chronic Value = 5^{th} percentile estimate
= 23 μg/L

The Final Plant Value is calculated as the lowest result from a 96-hr test conducted with an important plant species in which the concentrations of test material were measured and the endpoint was biologically important. None of the plant toxicity values in the RR data set (Table 6a) are for a 96-hr test, and two use measured concentrations. The closest test that fits this description is the 120-hr NOEC of 1.3 μg/L reported for *Pseudokirchneriella subcapitata* (Blasburg *et al.* 1991). This test has an exposure duration that is only 4-hr longer than the specified duration.

Final Plant Value = lowest result from a plant test
= 1.3 μg/L

The Final Residue Value is calculated by dividing the maximum permissible tissue concentration by an appropriate bioconcentration or bioaccumulation factor. A maximum allowable tissue concentration is either (a) a FDA action level for fish oil or for the edible portion of fish or shellfish, or (b) a maximum acceptable dietary intake based on observations on survival, growth, or reproduction in a chronic wildlife feeding

study or long-term wildlife field study. While no FDA action level exists for fish tissue, there is an EPA pesticide tolerance for farm-raised freshwater finfish tissue of 2.0 mg/kg (USEPA 2007). There is no relevant study that meets the requirement of part (b) above. A BCF of 2.0 for *Pimephales promelas* (Table 1) is used to calculate the Final Residue Value.

Final Residue Value = maximum permissible tissue concentration ÷ BCF
= 2.0 mg/kg ÷ 2.0 L/kg
= 1 mg/L
= 1000 µg/L

The Final Plant Value is lower than both the Final Chronic Value and the Final Residue Value, therefore the chronic criterion by the USEPA (1985) methodology would be 1.3 µg/L. The example chronic criterion is equivalent to the one recommended by the UC-Davis methodology.

19. Final criteria statement

The final criteria statement is:

Aquatic life in the Sacramento River and San Joaquin River basins should not be affected unacceptably if the four-day average concentration of diuron does not exceed 1.3 µg/L (1300 ng/L) more than once every three years on the average and if the one-hour average concentration does not exceed 170 µg/L more than once every three years on the average.

Although the criteria were derived to be protective of aquatic life in the Sacramento and San Joaquin Rivers, these criteria would be appropriate for any freshwater ecosystem in North America, unless species more sensitive than are represented by the species examined in the development of these criteria are likely to occur in those ecosystems.

The acute criterion is based only on acute animal data and was derived to protect animals from acute pulses of diuron. Details of the acute criterion calculation are described in section 7 and the acute data are shown in Tables 3 - 5. An assessment factor was used instead of a distribution to calculate the acute criterion because there were not sufficient data from the five required taxa for use of a SSD.

Details of the chronic criterion calculation are described in section 8 and chronic plant data are shown in Tables 6a, 7 and 8a. The chronic criterion was derived to only be protective of plants, but will also likely be protective of animals, which are less sensitive to diuron. The lowest NOEC of a highly rated plant study was used as the criterion because there were insufficient data for use of a SSD for criterion calculation. The chronic criterion was calculated with the absolute minimum amount of data, and uncertainty cannot be quantified. Some plant toxicity values in the supplemental data set are lower than the derived chronic criterion, but the studies were not appropriate for

criteria derivation or adjustment; these studies are discussed in detail in section 12. Thus, it is not currently recommended that the criteria be adjusted downward based on these data. Plant toxicity data is essential when considering diuron usage and regulations because plants and algae are the most sensitive taxa, however, plant data are difficult to interpret. The chronic criterion was derived using the best data available, and firm evidence that could support lowering criteria was not found. The criteria should be updated whenever new relevant and reliable data is available.

There are no established water quality criteria for diuron with which to compare the criteria derived in this report. The US EPA has several aquatic life benchmarks established for diuron, shown in Table 11, to which the derived criteria in this report can be compared with caution (USEPA 2003). According to the USEPA (2003), aquatic life benchmarks are not calculated following the same methodology used to calculate water quality criteria. Water quality criteria can be used to set water quality standards under the Clean Water Act, but aquatic life benchmarks may not be used for this purpose (USEPA 2003).

Table 11. US EPA Aquatic Life Benchmarks (USEPA 2003). All units are µg/L.

Acute Fish	Chronic Fish	Acute Invertebrates	Chronic Invertebrates	Acute nonvascular plants
355	26	80	160	2.4

The derived acute criterion of this report is below the acute fish benchmark, and about a factor of 2 above the acute invertebrate benchmark. The derived chronic criterion of this report is below the chronic benchmarks for fish and invertebrates, as well as the acute nonvascular plant benchmark. Because the chronic criterion was derived using only plant data, it is most comparable to the acute nonvascular plant benchmark. The Environmental Risk Assessment for the Reregistration of Diuron (USEPA 2003) cites the same green algae study used in this report as the only acceptable plant data for diuron, but the authors use the EC_{50} value of 2.4 µg/L (reported as 2.9 µg/L in the study, see Table 6a) as a benchmark, instead of the NOEC value of 1.3 µg/L. The use of the EC_{50} value is required according to the EPA methodology for calculation of an acute benchmark (USEPA 2003), but it is not clear if the discrepancy in EC_{50} values in the original study and the EPA benchmark was a mistake or the product of a calculation. The USEPA (1985) criteria derivation methodology requires the use of the lowest result from a test conducted with an important plant species, which would be the NOEC, not the EC_{50}. The use of the NOEC value as the chronic criterion is recommended by the UC-Davis method, and the USEPA (1985) method, in order to be protective of nonvascular plants.

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Data Tables

Table 3. Final acute toxicity data set for diuron. All studies were rated RR and were conducted at standard temperature. S: static; SR: static renewal; FT: flow-through.

Species	Common Identifier	Family	Test type	Meas/ Nom	Chemical grade	Duration	Temp (°C)	Endpoint	Age/ size	LC/EC_{50} (µg/L) (95% CI)	Reference
Daphnia magna	Daphnid	Daphniidae	S	Nom	80.0%	48-h	19.9	Mortality/ Immobility	< 24-h	12000 (10000-13000)*	Baer 1991a
Daphnia pulex	Daphnid	Daphniidae	SR	Meas	99.8%	96-h	22	Mortality	5-d	17900 (14200-22600)	Nebeker & Schuytema 1998
Hyalella azteca	Amphipod	Hyalellidae	SR	Meas	99.8%	96-h	22	Mortality	<11 d	19400 (17700-21300)	Nebeker & Schuytema 1998

* Lowest value used for criteria calculation because not enough data available for a distribution

Table 4. Acceptable reduced acute data rated RR with given reason for exclusion. S: static; SR: static renewal; FT: flow-through.

Species	Common Identifier	Family	Test type	Meas/ Nom	Chemical grade	Duration	Temp (°C)	Endpoint	Age/size	LC/EC_{50} (µg/L) (95% CI)	Reference	Reason
Daphnia magna	Daphnid	Daphniidae	S	Nom	80.0%	24-h	19.9	Mortality/ Immobility	< 24-h	68000 (55000-86000)	Baer 1991a	A

Reduction Reasons

A. Not the most sensitive or appropriate duration

Table 5. Supplemental acute data rated RL, LR, LL with given reason for rating and exclusion. S: static; SR: static renewal; FT: flow-through. NR: not reported. 95% CI: 95% confidence interval. Exclusion reasons are listed at the end of the table.

Species	Common Identifier	Family	Test type	Meas/ Nom	Chemical grade	Duration	Temp (°C)	Endpoint	Age/ size	LC/EC_{50} (µg/L) (95% CI)	Reference	Rating/ Reason
Artemia salina	Brine Shrimp	Artemiidae	S	NR	NR	24-h	25	Mortality	Instar II-III larvae	12010 (11420-12100)	Koutsaftis & Aoyama 2007	LL 2, 5
Asellus brevicaudus	Aquatic sow bug	Asellidae	S	Nom	95.0%	96-h	15	Mortality	Mature	15500 (7200-33400)	Johnson & Finley 1980	LL 5, 6
Ctenopharyngodon idella	Grass carp	Cyprinidae	FT	NR	100.0%	96-h	13	Mortality	1+ year, 15.8 g, 9.5 cm	31000 (28000-34000)	Tooby et al. 1980	LL 1, 5, 6
Daphnia magna	Daphnid	Daphniidae	S	Nom	Technical grade	26-h	21.1	Mortality/ Immobility	1st instar	47000 (41600-53100)	Crosby & Tucker 1966	LL 1, 5, 6
Daphnia pulex	Daphnid	Daphniidae	S	Nom	95.0%	48-h	15	Mortality/ Immobility	1st instar	1400 (1000-1900)	Johnson & Finley 1980	LL 5, 6
Gammarus fasciatus	Scud (amphipod crustacean)	Gammaridae	S	Nom	Technical grade	24-h	15.5	Mortality	early instar	2500 (1000-5500)	Sanders 1970	LL 1, 5, 6
Gammarus fasciatus	Scud (amphipod crustacean)	Gammaridae	S	Nom	Technical grade	48-h	15.5	Mortality	early instar	1800 (800-5200)	Sanders 1970	LL 1, 5, 6
Gammarus fasciatus	Scud (amphipod crustacean)	Gammaridae	S	Nom	Technical grade	96-h	15.5	Mortality	early instar	700 (190-8200)	Sanders 1970	LL 1, 5, 6
Gammarus lacustris	Scud (amphipod crustacean)	Gammaridae	S	Nom	Technical grade	24-h	21.1	Mortality	2 months old	700 (590-8300)	Sanders 1969	LL 1, 5, 6
Gammarus lacustris	Scud (amphipod crustacean)	Gammaridae	S	Nom	Technical grade	48-h	21.1	Mortality	2 months old	380 (290-500)	Sanders 1969	LL 1, 5, 6

Species	Common Identifier	Family	Test type	Meas/ Nom	Chemical grade	Duration	Temp (°C)	Endpoint	Age/ size	LC/EC_{50} (µg/L) (95% CI)	Reference	Rating/ Reason
Gammarus lacustris	Scud (amphipod crustacean)	Gammaridae	S	Nom	Technical grade	96-h	21.1	Mortality	2 months old	160 (130-190)	Sanders 1969	LL 1, 5, 6
Lepomis macrochirus	Bluegill Sunfish	Centrarchidae	S	Nom	Technical grade	96-h	12.7	Mortality	0.6-1.5 g	8900 (8200-9600)	Macek et al. 1969	LL 1, 5, 6
Lepomis macrochirus	Bluegill Sunfish	Centrarchidae	S	Nom	Technical grade	96-h	18.3	Mortality	0.6-1.5 g	7600 (7000-8200)	Macek et al. 1969	LL 1, 5, 6
Lepomis macrochirus	Bluegill Sunfish	Centrarchidae	S	Nom	Technical grade	96-h	23.8	Mortality	0.6-1.5 g	5900 (5300-6500)	Macek et al. 1969	LL 1, 5, 6
Lymnaea spp.	Snail	Lymnaeidae	S	Nom	NR	96-h	NR	Mortality	Adult	15300	Christian & Tate 1983	LL 1, 3, 6
Oncorhynchus clarki (Salmo clarki)	Cutthroat Trout	Salmonidae	S	Nom	95.0%	96-h	10.0	Mortality	3.00 g	1400 (1100 - 1900)	Johnson & Finley 1980	LL 5, 6
Oncorhynchus mykiss (Salmo gairdneri)	Rainbow Trout	Salmonidae	S	Nom	95.0%	96-h	13	Mortality	0.8 g	4900 (4100-5900)	Johnson & Finley 1980	LL 5, 6
Oncorhynchus mykiss (Salmo gairdneri)	Rainbow Trout	Salmonidae	S	Nom	80.0%	96-h	13	Mortality	1.2 g	16000 (11300-22700)	Johnson & Finley 1980	LL 5, 6
Pimephales promelas	Fathead minnow	Cyprinidae	FT	Meas	98.6%	96-h	24.3	Mortality	30-d	14200 (13400-15000)	Call et al. 1983, 1987	RL 1, 5
Pteronarcys californica	Stonefly Naiad	Pteronarcidae	S	Nom	95.0%	96-h	15	Mortality	2nd year class	1200 (900-1700)	Johnson & Finley 1980	LL 5, 6
Pteronarcys californica	Stonefly Naiad	Pteronarcidae	S	Nom	Technical grade	24-h	15.5	Mortality	30-35 mm	3600 (2800-4700)	Sanders & Cope 1968	LL 1, 5, 6

Species	Common Identifier	Family	Test type	Meas/ Nom	Chemical grade	Duration	Temp (°C)	Endpoint	Age/ size	LC/EC$_{50}$ (µg/L) (95% CI)	Reference	Rating/ Reason
Pteronarcys californica	Stonefly Naiad	Pteronarcidae	S	Nom	Technical grade	48-h	15.5	Mortality	30-35 mm	2800 (2100-3800)	Sanders & Cope 1968	LL 1, 5, 6
Pteronarcys californica	Stonefly Naiad	Pteronarcidae	S	Nom	Technical grade	96-h	15.5	Mortality	30-35 mm	1200 (870-1700)	Sanders & Cope 1968	LL 1, 5, 6
Salvelinus namaycush	Lake Trout	Salmonidae	S	Nom	95.0%	96-h	10	Mortality	1.5 g	2700 (2400-3000)	Johnson & Finley 1980	LL 5, 6
Simocephalus serrulatus	Water fleas, daphnid	Daphniidae	S	Nom	95.0%	48-h	15	Mortality	1st instar	2000 (1400-2800)	Johnson & Finley 1980	LL 5, 6

Exclusion Reasons

1. Not a standard method
2. Saltwater
3. Low chemical purity or purity not reported
4. Toxicity value not calculable
5. Control response not reported
6. Low reliability score

Table 6a. Final chronic plant toxicity data set for diuron. All studies were rated RR. S: static; SR: static renewal; FT: flow-through. NR: not reported, n/a: not applicable. SMCV is in bold.

Species	Common identifier, Family	Test type	Meas/ Nom	Chemical grade	Duration	Temp (°C)	Endpoint	Age/size	NOEC (µg/L)	LOEC (µg/L)	MATC (µg/L)	EC_{50} (µg/L)	Reference
Lemna gibba G3	Duckweed, Araceae	S	Meas	99.1%	7-d	24.7	Growth inhibition (Biomass yield), Relative growth rate (Biomass)	Plant with 4 fronds	2.47	8.11	**4.48**	14.4 (9.26-19.6)*	Ferrell 2006
Pseudokirchneriella subcapitata (Selenastrum capricornutum Printz)	Green algae	S	Meas	96.8%	120 h	24	Growth inhibition	2-d old algal cells	1.3	2.5	**1.8**	2.9 (2.5-3.5; 95% CI)	Blasburg et al. 1991
Scenedesmus obliquus	Microalgae, Scenedesmaceae	S	Nom	Technical	24 h	21	Growth inhibition	Algal cells	NR	NR	NR	**10**	Geoffroy et al. 2002

*EC_{50} based on biomass yield endpoint, not the growth rate endpoint.

Table 6b. Final chronic animal toxicity data set for diuron. All studies were rated RR. S: static; SR: static renewal; FT: flow-through. NR: not reported

Species	Common identifier	Test type	Meas/ Nom	Chemical grade	Duration	Temp (°C)	Endpoint	Age/size	NOEC (µg/L)	LOEC (µg/L)	MATC (µg/L)	Reference
Chironomus tentans	Midge	SR	Meas	99.8%	10-d	24	Mortality	2-d, 1st instar larvae	1900	3400	2540	Nebeker & Schuytema 1998
Daphnia pulex	Daphnid	S	Meas	99.8%	7-d	NR	Reduced # of young/ mortality	5-d old	4000.0	7700	5550	Nebeker & Schuytema 1998
Hyalella azteca	Amphipod	SR	Meas	99.8%	10-d	22	Mortality/ Reduced weight	< 11-d	7900	15700	11140	Nebeker & Schuytema 1998
Lumbriculus variegatus	Annelid worm	SR	Meas	99.8%	10-d	23	Reduced weight	small, short adults	1800	3500	2510	Nebeker & Schuytema 1998
Physa gyrina	Snail	SR	Meas	99.8%	10-d	24	Reduced weight	2-d 1st instar larvae	13400	22800	17480	Nebeker & Schuytema 1998
Pimephales promelas	Fathead minnow	FT	Meas	98.6%	64-d	25	Deformity, Mortality	Eggs < 24-h, hatched fry	33.4	78	51	Call et al. 1983, 1987
Pseudacris regilla	Pacific treefrog	SR	Meas	99.8%	14-d	20	Growth inhibition (Length)	Tadpole	14500	21100	17490	Schuytema & Nebeker 1998
Rana aurora	Red-legged frog	SR	Meas	99.8%	7-d	20	Growth inhibition (Wet weight)	Tadpole	7600	14500	10500	Schuytema & Nebeker 1998
Rana catesbeiana	Bullfrog	SR	Meas	99.8%	21-d	24	Growth inhibition (Dry weight)	Tadpole	11690*	16430*	12450*	Schuytema & Nebeker 1998
Xenopus laevis	African clawed frog	SR	Meas	99.8%	4-d	24	Growth inhibition (Length)	Embryo	10490**	20540**	14680**	Schuytema & Nebeker 1998

*SMCV calculated from 3 values

** SMCV calculated from 2 values

Table 7. Acceptable reduced chronic data rated RR with reason for exclusion given below. S: static; SR: static renewal; FT: flow-through. NR: not reported

Species	Common identifier	Test type	Meas/ Nom	Chemical grade	Duration	Temp (°C)	Endpoint	Age/size	NOEC (µg/L)	LOEC (µg/L)	MATC (µg/L)	Reference	Reason for exclusion
Chironomus tentans	Midge	SR	Meas	99.8%	10-d	24	Reduced weight	2-d, 1st instar larvae	3400	7100	4910	Nebeker & Schuytema 1998	A
Lemna gibba G3	Duckweed, Araceae	S	Meas	99.1%	7-d	24.7	Growth inhibition (Biomass)	Plant with 4 fronds	2.47	8.11	EC_{50}=15.7 (10.06-20.8)	Ferrell 2006	A
Lemna gibba G3	Duckweed, Araceae	S	Meas	99.1%	7-d	24.7	Growth inhibition (Frond count)	Plant with 4 fronds	8.11	25.8	EC_{50}=19.1 (13.4-24.8)	Ferrell 2006	A
Lemna gibba G3	Duckweed, Araceae	S	Meas	99.1%	7-d	24.7	Growth inhibition (Frond count yield)	Plant with 4 fronds	8.11	25.8	EC_{50}=17.5 (11.8-23.2)	Ferrell 2006	A
Lemna gibba G3	Duckweed, Araceae	S	Meas	99.1%	7-d	24.7	Relative growth rate (frond count)	Plant with 4 fronds	8.11	25.8	14.5	Ferrell 2006	A
Pimephales promelas	Fathead minnow	SR	Meas	99.8%	7-d	25	Reduced weight	2.5 d embryo	4200	8300	5900	Nebeker & Schuytema 1998	C
Pimephales promelas	Fathead minnow	SR	Meas	99.8%	10-d	24	Mortality	1.5 month old juvenile	20000	27100	23280	Nebeker & Schuytema 1998	B
Pseudacris regilla	Pacific treefrog	SR	Meas	99.8%	10-d	20	Increased Deformity	Embryo	14500	29100	20540	Schuytema & Nebeker 1998	A
Pseudacris regilla	Pacific treefrog	SR	Meas	99.8%	14-d	20	Growth inhibition (Wet weight)	Tadpole	21000	29100	24720	Schuytema & Nebeker 1998	A
Pseudacris regilla	Pacific treefrog	SR	Meas	99.8%	14-d	20	Growth inhibition (Dry weight)	Tadpole	21100**	29100**	24750**	Schuytema & Nebeker 1998	A

Rana catesbeiana	Bullfrog	SR	Meas	99.8%	21-d	24	Growth inhibition (length)	Tadpole	14500**	24780**	18950**	Schuytema & Nebeker 1998	A
Rana catesbeiana	Bullfrog	SR	Meas	99.8%	21-d	24	Growth inhibition (Wet weight)	Tadpole	17490**	29100**	22560**	Schuytema & Nebeker 1998	A
Xenopus laevis	African clawed frog	SR	Meas	99.8%	4-d	24	Deformity	Embryo	17490	29100	22560	Schuytema & Nebeker 1998	A

Reasons for Exclusion

A. Less sensitive endpoint

B. Less sensitive life-stage

C. Test type not preferred (static vs. flow-through)

* SMCV calculated from 3 values

** SMCV calculated from 2 values

Table 8a. Supplemental chronic plant toxicity data set for diuron of studies rated RL, LR, or LL. S: static; SR: static renewal; FT: flow-through. NR: not reported, n/a: not applicable; 95% CI: 95% confidence interval; SE: standard error.

Species	Common identifier	Test type	Meas/ Nom	Chemical grade	Duration	Temp (°C)	Endpoint	Age/size	NOEC (µg/L)	LOEC (µg/L)	EC_{50} (µg/L) (95% CI)	Reference	Rating/ Reason for exclusion
Achnanthes brevipes	Bacillariophyceae family	S	Nom	Tech.	3-d	20	Reduced Oxygen Evolution	Algal cells	NR	NR	24 (SE=1.0)	Hollister & Walsh 1973	LL 1, 2, 6
Amphora exigua	Bacillariophyceae family	S	Nom	Tech.	3-d	20	Reduced Oxygen Evolution	Algal cells	NR	NR	31 (SE=4)	Hollister & Walsh 1973	LL 1, 2, 6
Apium nodiflorum	Rooted macrophyte	S	Nom	> 99%	14-d	NR	Relative growth rate	Single stem node w/ leaf	0.05	NR	2.808	Lambert et al. 2006	LL 1, 5, 6
Apium nodiflorum	Rooted macrophyte	S	Nom	> 99%	14-d	NR	Growth inhibition (roots)	Single stem node w/ leaf	<0.0005	NR	0.00026	Lambert et al. 2006	LL 1, 5, 6, 7
Apium nodiflorum	Rooted macrophyte	S	Nom	> 99%	14-d	NR	Change in chlorophyll fluorescence ratio	Single stem node w/ leaf	5	NR	> 5.0	Lambert et al. 2006	LL 1, 5, 6
Chara vulgaris	Macrophytic alga	S	Nom	> 99%	14-d	NR	Relative growth rate	Terminal lengths of shoots w/ 3 nodes	0.0005	NR	0.35	Lambert et al. 2006	LL 1, 5, 6
Chara vulgaris	Macrophytic alga	S	Nom	> 99%	14-d	NR	Change in chlorophyll fluorescence ratio	Terminal lengths of shoots w/ 3 nodes	0.5	NR	4.033	Lambert et al. 2006	LL 1, 5, 6
Chlamydomonas moewusii Gerloff	Algae, Chlamydomonadaceae	S	Nom	80.0%	7-d	21	Growth inhibition	7-d old algal cell stock	NR	NR	559.44	Cain & Cain 1983	RL 1, 6
Chlamydomonas sp.	Chlorophyceae family	S	Nom	Tech.	3-d	20	Reduced Oxygen Evolution	Algal cells	NR	NR	37 (SE=3)	Hollister & Walsh 1973	LL 1, 2, 6

Species	Common identifier	Test type	Meas/ Nom	Chemical grade	Duration	Temp (°C)	Endpoint	Age/size	NOEC (µg/L)	LOEC (µg/L)	EC_{50} (µg/L) (95% CI)	Reference	Rating/ Reason for exclusion
Chlamydomonas sp.	Chlorophyceae family	S	Nom	99.8%	20 min	21.5	Change in chlorophyll fluorescence ratio	2-4 week old algal cells	0.1	0.5	10.8 (8.5-13.6)	Podola & Melkonian 2005	RL 1, 5, 8
Chlorella pyrenoidosa	Green algae	S	Nom	95.0%	4-d	25	Growth inhibition	Algal cells	NR	NR	25	Maule & Wright 1984	LR 1, 6
Chlorella pyrenoidosa	Green algae	S	Nom	50.0%	96-h	25	Growth inhibition	Algal cells	NR	NR	1.3	Ma et al. 2001, Ma 2002a	LL 1, 3, 6
Chlorella sp.	Nonmotile unicell phytoplankton	S	Nom	Tech.	10-d	20.5	Growth inhibition	Algal cells	NR	NR	EC_{66}= 4	Ukeles 1962	LL 1, 2, 6
Chlorella sp.	Nonmotile unicell phytoplankton	S	Nom	Tech.	3-d	20	Reduced Oxygen Evolution	Algal cells	NR	NR	19 (SE=2)	Hollister & Walsh 1973	LL 1, 2, 6
Chlorella vulgaris	Green algae	S	Nom	50.0%	96-h	25	Growth inhibition	Algal cells	NR	NR	4.3	Ma et al. 2002b	LL 1, 3, 6
Chlorella vulgaris SAG211-11b	Green algae	S	Nom	99.8%	20 min	21.5	Change in chlorophyll fluorescence ratio	2-4 week old algal cells	0.1	0.5	27.4 (21.1-35.5)	Podola & Melkonian 2005	RL 1, 8
Chlorococcum sp.	Chlorophyte algae	S	Nom	Tech.	7-d	20	Growth inhibition	Algal cells	< 1.0	NR	EC_{62} = 10	Walsh & Grow 1971	RL 1, 2
Chlorococcum sp.	Chlorophyte algae	S	Nom	Tech.	10-d	20	Growth inhibition	Algal cells	NR	NR	10	Walsh 1972	RL 1, 2
Chlorococcum sp.	Chlorophyte algae	S	Nom	Tech.	90 min	20	Reduced Oxygen Evolution	Algal cells	NR	NR	20	Walsh 1972	RL 1, 2
Chlorococcum sp.	Chlorophyte algae	S	Nom	Tech.	3-d	20	Reduced Oxygen Evolution	Algal cells	NR	NR	20 (SE=4)	Hollister & Walsh 1973	LL 1, 2, 6

Species	Common identifier	Test type	Meas/ Nom	Chemical grade	Duration	Temp (°C)	Endpoint	Age/size	NOEC (µg/L)	LOEC (µg/L)	EC_{50} (µg/L) (95% CI)	Reference	Rating/ Reason for exclusion
Cyclotella nana	Bacillariophyceae family	S	Nom	Tech.	3-d	20	Reduced Oxygen Evolution	Algal cells	NR	NR	39 (SE=7)	Hollister & Walsh 1973	LL 1, 2, 6
Cryptomonas sp.	Algae	S	Nom	99.8%	20 min	21.5	Change in chlorophyll fluorescence ratio	2-4 week old algal cells	0.1	0.5	6.4 (5.3-7.8)	Podola & Melkonian 2005	RL 1, 5, 8
Dunaliella euchlora Lerche	Motile flagellate phytoplankton	S	Nom	Tech.	10-d	20.5	Growth inhibition	Algal cells	NR	NR	EC_{56}=0.4	Ukeles 1962	LL 1, 2, 6
Dunaliella tertiolecta	Green algae	S	Nom	99.0%	96-h	20	Growth inhibition	Algal cells	NR	NR	5.9	Gatidou & Thomaidis 2007	LL 2, 5
Dunaliella tertiolecta	Chlorophyceae family	S	Nom	Tech.	3-d	20	Reduced Oxygen Evolution	Algal cells	NR	NR	10 (SE=3)	Hollister & Walsh 1973	LL 1, 2, 6
Dunaliella tertiolecta Butcher	Green algae	S	Nom	Tech.	10-d	20	Growth inhibition	Algal cells	NR	NR	20	Walsh 1972	RL 1, 2
Dunaliella tertiolecta Butcher	Green algae	S	Nom	Tech.	90 min	20	Reduced Oxygen Evolution	Algal cells	NR	NR	10	Walsh 1972	RL 2, 6, 8
Eudorina elegans	Algae	S	Nom	99.8%	20 min	21.5	Change in chlorophyll fluorescence ratio	2-4 week old algal cells	0.1	0.5	13.2 (10.4-16.9)	Podola & Melkonian 2005	RL 1, 5, 8
Isochrysis galbana	Chrysophyte	S	Nom	Tech.	3-d	20	Reduced Oxygen Evolution	Algal cells	NR	NR	10 (SE=3)	Hollister & Walsh 1973	LL 1, 2, 6
Isochrysis galbana Parke	Chrysophyte	S	Nom	Tech.	90 min	20	Reduced Oxygen Evolution	Algal cells	NR	NR	10	Walsh 1972	RL 1, 2, 8

Species	Common identifier	Test type	Meas/ Nom	Chemical grade	Duration	Temp (°C)	Endpoint	Age/size	NOEC (µg/L)	LOEC (µg/L)	EC_{50} (µg/L) (95% CI)	Reference	Rating/ Reason for exclusion
Isochrysis galbana Parke	Chrysophyte	S	Nom	Tech.	10-d	20	Growth inhibition	Algal cells	NR	NR	10	Walsh 1972	RL 1, 2
Lemna gibba G3	Duckweed	S	Nom	98.0%	7-d	25	Growth inhibition	NR	NR	NR	29 (27-31)	Okamura et al. 2003	LR 6
Lemna minor	Duckweed	S	Nom	98.0%	48 h	21	Reduced oxygen evolution	Plant fronds	NR	5	NR	Eullaffroy et al. 2007	LL 1, 6, 7
Lemna minor 1769	Duckweed	S	Nom	98.0%	7-d	25	Growth inhibition	NR	NR	NR	30 (28-31)	Okamura et al. 2003	LR 6
Lemna minor	Duckweed	S	Nom	98.0%	7-d	25	Growth inhibition	Plant fronds	NR	5	25	Teisseire et al. 1999	RL 1, 6
Monochrysis lutheri	Motile flagellate phytoplankton	S	Nom	Tech.	3-d	20	Reduced Oxygen Evolution	Algal cells	NR	NR	18 (SE=3)	Hollister & Walsh 1973	LL 1, 2, 6
Monochrysis lutheri Droop	Motile flagellate phytoplankton	S	Nom	Tech.	10-d	20.5	Growth inhibition	Algal cells	NR	NR	EC_{100}= 0.02	Ukeles 1962	LL 1, 2, 6
Monochrysis lutheri Droop	Motile flagellate phytoplankton	S	Nom	Technical grade	10-d	20.5	Mortality	early instar	NR	NR	2500 (1000-5500)	Sanders 1970	LL 1, 5, 6
Myriophyllum spicatum	Rooted macrophyte	S	Nom	> 99%	14-d	NR (green house)	Relative growth rate	Terminal lengths of shoots w/ 3 nodes	0.0005	NR	5	Lambert et al. 2006	LL 1, 5, 6
Myriophyllum spicatum	Rooted macrophyte	S	Nom	> 99%	14-d	NR (green house)	Change in chlorophyll fluorescence ratio	Terminal lengths of shoots w/ 3 nodes	5	NR	> 5	Lambert et al. 2006	LL 1, 5, 6
Navicula forcipata	Diatom	S	Nom	99.0%	96-h	20	Growth inhibition	Algal cells	NR	NR	27	Gatidou and Thomaidis 2007	LL 2, 5

Species	Common identifier	Test type	Meas/ Nom	Chemical grade	Duration	Temp (°C)	Endpoint	Age/size	NOEC (µg/L)	LOEC (µg/L)	EC_{50} (µg/L) (95% CI)	Reference	Rating/ Reason for exclusion
Navicula inserta	Bacillariophyceae family	S	Nom	Tech.	3-d	20	Reduced Oxygen Evolution	Algal cells	NR	NR	93 (SE=12)	Hollister & Walsh 1973	LL 1, 2, 6
Neochloris sp.	Chlorophyceae family	S	Nom	Tech.	3-d	20	Reduced Oxygen Evolution	Algal cells	NR	NR	28 (SE=5)	Hollister & Walsh 1973	LL 1, 2, 6
Nitzschia (Ind. 684)	Bacillariophyceae family	S	Nom	Tech.	3-d	20	Reduced Oxygen Evolution	Algal cells	NR	NR	169 (SE=17)	Hollister & Walsh 1973	LL 1, 2, 6
Nitzschia closterium	Bacillariophyceae family	S	Nom	Tech.	3-d	20	Reduced Oxygen Evolution	Algal cells	NR	NR	50 (SE=6)	Hollister & Walsh 1973	LL 1, 2, 6
Oscillatoria cf. chalybea	Cyanobacterium	S	Nom	80.0%	96-h	25	Growth inhibition	Algal cells	NR	280	28	Schrader et al. 1998	LR 1, 6
Phaeodactylum tricornutum	Chrysophyte	S	Nom	Tech.	3-d	20	Reduced Oxygen Evolution	Algal cells	NR	NR	10 (SE=3)	Hollister & Walsh 1973	LL 1, 2, 6
Phaeodactylum tricornutum Bohlin	Chrysophyte	S	Nom	Tech.	90 min	20	Reduced Oxygen Evolution	Algal cells	NR	NR	10	Walsh 1972	RL 1, 2, 8
Phaeodactylum tricornutum Bohlin	Chrysophyte	S	Nom	Tech.	10-d	20	Growth inhibition	Algal cells	NR	NR	10	Walsh 1972	RL 1, 2
Phaeodactylum tricornutum Bohlin	Chrysophyte	S	Nom	Tech.	10-d	20.5	Growth inhibition	Algal cells	NR	NR	EC_{21}=0.4	Ukeles 1962	LL 1, 2, 6
Platymonas sp.	Chlorophyceae family	S	Nom	Tech.	3-d	20	Reduced Oxygen Evolution	Algal cells	NR	NR	7 (SE=3)	Hollister & Walsh 1973	LL 1, 2, 6
Porphyridium cruentum	Rhodophyceae family	S	Nom	Tech.	3-d	20	Reduced Oxygen Evolution	Algal cells	NR	NR	24 (SE=3)	Hollister & Walsh 1973	LL 1, 2, 6

Species	Common identifier	Test type	Meas/ Nom	Chemical grade	Duration	Temp (°C)	Endpoint	Age/size	NOEC (µg/L)	LOEC (µg/L)	EC_{50} (µg/L) (95% CI)	Reference	Rating/ Reason for exclusion
Protcoccus sp.	Nonmotile unicell phytoplankton	S	Nom	Tech.	10-d	20.5	Growth inhibition	Algal cells	NR	NR	EC_{48}=0.02	Ukeles 1962	LL 1, 2, 6
Pseudokirchneriella subcapitata (Selenastrum capricornutum)	Green algae	S	Nom	80.0%	96-h	25	Growth inhibition	Algal cells	NR	280	36.4	Schrader et al. 1998	LR 1, 6
Pseudokirchneriella subcapitata (Selenastrum capricornutum)	Green algae	S	Nom	98.0%	3-d	25	Growth inhibition	Algal cells	NR	NR	6.6 (5.9-7.2)	Okamura et al. 2003	LL 5, 6
Pseudokirchneriella subcapitata (Selenastrum capricornutum)	Green algae	S	Nom	99.8%	20 min	21.5	Change in chlorophyll fluorescence ratio	2-4 week old algal cells	0.1	0.5	13.8 (9.3-20.4)	Podola & Melkonian 2005	RL 1, 8
Raphidocelis subcapitata	Green algae	S	Nom	50.0%	96-h	25	Growth inhibition	Algal cells	NR	NR	0.7	Ma et al. 2006	LL 3, 5, 6
Scenedesmus obliquus	Green algae	S	Nom	50.0%	96-h	25	Growth inhibition	Algal cells	NR	NR	4.09	Ma et al. 2002a	LL 1, 3, 6
Scenedesmus obliquus	Green algae	S	Nom	98.0%	1 min	22	Change in chlorophyll fluorescence ratio	Algal cells	NR	NR	1†	Eullaffroy & Vernet 2003	LL 1, 4, 6, 8
Scenedesmus quadricauda	Green algae	S	Nom	50.0%	96-h	25	Growth inhibition	Algal cells	NR	NR	2.7	Ma et al. 2003	LL 1, 3, 6
Scenedesmus subspicatus	Green algae	S	Nom	Tech.	24-h	20	Growth inhibition	Algal cells, 3-d old	4	NR	NR	Schafer et al. 1994	LR 5, 6
Scenedesmus subspicatus	Green algae	S	Nom	Tech.	72-h	20	Growth inhibition	Algal cells, 3-d old	10	NR	36	Schafer et al. 1994	LR 5, 6
Scherffelia dubia	Algae	S	Nom	99.8%	20 min	21.5	Change in chlorophyll fluorescence ratio	2-4 week old algal cells	0.1	0.5	3.9 (2.5-6.2)	Podola & Melkonian 2005	RL 1, 8

Species	Common identifier	Test type	Meas/ Nom	Chemical grade	Duration	Temp (°C)	Endpoint	Age/size	NOEC (µg/L)	LOEC (µg/L)	EC_{50} (µg/L) (95% CI)	Reference	Rating/ Reason for exclusion
Staurodesmus convergens	Algae	S	Nom	99.8%	20 min	21.5	Change in chlorophyll fluorescence ratio	2-4 week old algal cells	0.1	0.5	4.1 (2.5-6.9)	Podola & Melkonian 2005	RL 1, 5, 8
Stauroneis amphoroides	Bacillariophyceae family	S	Nom	Tech.	3-d	20	Reduced Oxygen Evolution	Algal cells	NR	NR	31 (SE=2)	Hollister & Walsh 1973	LL 1, 2, 6
Synechocystis sp.	Cyanobacterium	S	Nom	99.8%	20 min	21.5	Change in chlorophyll fluorescence ratio	2-4 week old algal cells	0.1	0.5	7.6 (5.5-10.5)	Podola & Melkonian 2005	RL 1, 5, 8
Tetraselmis elegans	Phytoplankton	S	Nom	99.8%	20 min	21.5	Change in chlorophyll fluorescence ratio	2-4 week old algal cells	0.1	0.5	3.0 (2.3-3.8)	Podola & Melkonian 2005	RL 1, 8
Thalassiosira fluviatilis	Bacillariophyceae family	S	Nom	Tech.	3-d	20	Reduced Oxygen Evolution	Algal cells	NR	NR	95 (SE=10)	Hollister & Walsh 1973	LL 1, 2, 6
Ulothrix fimbriata	Green algae	S	Nom	95.0%	7-d	25	Growth inhibition	Algal cells	NR	NR	540	Maule & Wright 1984	LR 1, 6

Exclusion Reasons
1. Not a standard method
2. Saltwater
3. Low chemical purity or purity not reported
4. Toxicity value not calculable
5. Control not described and/or response not reported
6. Low reliability score
7. Endpoint not linked to growth, reproduction or survival (Ch. 3, Section 3-2.1.3)
8. Inappropriate test duration (Ch. 3, Section 3-2.1.1)

† Value reported as toxicity threshold, which is conceptually very similar to a MATC, but calculated differently than a MATC or an ECx value.
‡ Growth inhibition of roots is not a standard endpoint.

Table 8b. Supplemental chronic animal toxicity data set for diuron of studies rated RL, LR, or LL. S: static; SR: static renewal; FT: flow-through. NR: not reported; 95% CI: 95% confidence interval.

Species	Common identifier	Test type	Meas /Nom	Chemical grade	Duration	Temp (°C)	Endpoint	Age/size	NOEC (µg/L)	LOEC (µg/L)	MATC (µg/L) (95% CI)	Reference	Rating/ Reason for exclusion
Crassostrea virginica	Eastern oyster	FT	Meas	96.8%	96-h	23	Shell deposition	Neonates, <24-h	2400	NR	EC_{50}=4800 (4400-5200)	Ward & Boeri 1991	RL 2
Cyprinodon variegates	Sheepshead minnow	FT	Meas	96.8%	32-d	30	Mortality	< 24-h	1700	3600	2500	Ward & Boeri 1992a	RL 2
Mysidopsis bahia	Mysid	FT	Meas	96.8%	28-d	25.3	# of young surviving	< 24-h, juvenile	960	1900	1400	Ward & Boeri 1992b	RL 2
											LC_{50} (µg/L)		
Oncorhynchus mykiss	Rainbow Trout	S	Nom	95%	7-d	10	Mortality	juveniles, hatched <24h ago	NR	NR	74000 (29000-3681000)	Okamura et al. 2002	LR 1, 6
Oncorhynchus mykiss	Rainbow Trout	S	Nom	95%	14-d	10	Mortality	juveniles, hatched <24h ago	NR	NR	15000 (11000-29000)	Okamura et al. 2002	LR 1, 6
Oncorhynchus mykiss	Rainbow Trout	S	Nom	95%	21-d	10	Mortality	juveniles, hatched <24h ago	NR	NR	5900 (4700-7700)	Okamura et al. 2002	LR 1, 6
Oncorhynchus mykiss	Rainbow Trout	S	Nom	95%	28-d	10	Mortality	juveniles, hatched <24h ago	NR	NR	230 (8.9-590)	Okamura et al. 2002	LR 1, 6

Exclusion Reasons
1. Not a standard method
2. Saltwater
3. Low chemical purity or purity not reported
4. Toxicity value not calculable
5. Control response not reported
6. Low reliability score
7. Endpoint not linked to growth, reproduction or survival (Ch. 3, Section 3-2.1.3)
8. Inappropriate test duration (Ch. 3, Section 3-2.1.1)

Table 9. Acceptable multispecies field, semi-field, laboratory, microcosm, mesocosm studies; R= reliable; L= less reliable.

Reference	Habitat	Rating
Devilla *et al.* (2005)	Laboratory model ecosystem	L
Dorigo *et al.* (2007)	Lotic outdoor stream	L
Flum & Shannon (1987)	Laboratory microcosm	L
Hartgers *et al*. (1998)	Laboratory microcosm	R
Molander & Blanck (1992a)	Laboratory microcosm	L
Perschbacher & Ludwig (2004)	Outdoor pond	L
Pesce *et al.* (2006)	Laboratory microcosm	L
Sumpono *et al.* (2003)	Indoor pond	R
Tlili *et al*. (2008)	Laboratory microcosm	R
Zimba *et al.* (2002)	Outdoor pond	L

Table 10. Threatened, Endangered, or Rare Species Predicted values by ICE.

Surrogate		Predicted	
Species	**LC_{50} (mg/L)**	**Species**	**LC_{50} (95% confidence interval) (mg/L)**
Rainbow trout (*Oncorhynchus mykiss)*	4.9	Chinook salmon *(O. tshawytscha)*	5.983 (3.225-11.097)
		Coho salmon (*O. kisutch)*	8.086 (6.104-4.016)
		Lahontan cutthroat trout (*O. clarki henshawi)*	4.758 (3.545-6.387)
Cutthroat trout (*O. clarki)*	1.4	Coho salmon (*O. kisutch)*	1.673 (1.156-2.421)