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MEMORANDUM

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SUBJECT **Orthosulfamuron (Section 3: Rice Use)**
Tier I Drinking Water Exposure Assessment for Orthosulfamuron on Rice.

TO: James Tompkins
Registration Division (7505P)

Karlyn Bailey
Health Effects Division (7509P)

FROM: Greg Orrick, Environmental Scientist
Environmental Fate and Effects Division (7507P)

THROUGH: Dirk Young, Environmental Engineer
R. David Jones, Senior Agronomist
Betsy Behl, Chief
Environmental Fate and Effects Division (7507P)

EXECUTIVE SUMMARY

This assessment provides estimated drinking water concentrations (EDWC) of orthosulfamuron in surface water and in groundwater in support of human health risk assessment. Tier I EDWCs (Table 1) of orthosulfamuron were generated with the Environmental Fate and Effects Division's (EFED) Interim Rice Model for surface water and SCI-GROW for groundwater. Modeled application rates represent the maximum use pattern of two proposed labels for use on rice (EPA Reg. No. pending). Remaining model input parameters were chosen according to current guidance (EFED, 2002). EDWCs reflect exposure to orthosulfamuron and all potential degradates of concern (USEPA, 2006).

Table 1. Tier I EDWCs in surface water and groundwater from orthosulfamuron use on rice.		
Source	Peak Exposure (ppb)	Annual Mean Exposure (ppb)
Surface water	40.5 ppb	<40.5 ppb
Groundwater	0.611 ppb	<0.611 ppb

Orthosulfamuron is a foliarly-applied, post-emergent sulfamoylurea herbicide. The compound has low volatility and is soluble in water. It readily hydrolyzes at low pH (<7), but is relatively stable to hydrolysis at high pH (>7). Moderate biodegradation also occurs in aquatic environments. The compound is relatively stable to photolysis. Orthosulfamuron is moderately mobile in soil and may be a groundwater concern in alkaline sandy soils with low organic carbon content. While the compound shows affinity to organic carbon, variability in adsorption to soil is not well explained by soil organic carbon content.

Major degradates include hydrolysis products N-(4,6-dimethoxypyrimidin-2-yl)urea (DOP urea; S12), 2-dimethylcarbamoylphenyl sulfamic acid (DBS acid; S1), and 2-amino-N,N-dimethylbenzamide (DB amine; S4) and demethylation product 1-(4-hydroxy-6-methoxypyrimidin-2-yl)-3-[2-(dimethylcarbamoyl)phenylsulfamoyl]urea (o-desmethyl orthosulfamuron; S9). Both orthosulfamuron and its o-desmethyl degradate hydrolyze to DBS acid, which, in turn, hydrolyses to DB amine. The toxicities of orthosulfamuron's major degradates are not determined; the Office of Pesticide Programs (OPP) assumes in the circumstances that these major degradates are of equal or less toxicity than the parent (USEPA, 2006). EDWCs reflect exposure to orthosulfamuron and its organic degradates in order to include any potential residues of concern.

EDWCs for orthosulfamuron use on rice are conservative due to the screening design of the exposure models and assumptions resulting from the limited number of available and acceptable environmental fate studies.

PROBLEM FORMULATION

This drinking water assessment uses environmental modeling to provide estimates of surface water and groundwater concentrations in drinking water source water (pre-treatment) resulting from orthosulfamuron use. Primary routes of transport to source water include runoff and leaching. The Interim Rice Model was used to assess high-end exposure in released flood water (tailwater) due to runoff from the proposed orthosulfamuron use. Degradation processes and potential dilution with uncontaminated water outside the paddy were not modeled, due to the inability of the Interim Rice Model to account for them and lack of dilution data. These estimates assume that the active ingredient concentrations are evenly distributed throughout the water in the rice paddy. Exposure in groundwater due to leaching was assessed with the screening model SCI-GROW. The single exposure estimates produced with both the Interim Rice Model and SCI-GROW were used as both acute and chronic exposure estimates for drinking water exposure assessment.

ANALYSIS

Use Characterization

Orthosulfamuron is a systemic sulfamoylurea herbicide used for selective post-emergent control of annual and perennial broadleaf weeds and sedges and barnyard grass. Orthosulfamuron's mode of action is inhibition of the plant enzyme acetolactate synthase, which is active in the biosynthesis of three branched-chain amino acids, valine, leucine, and isoleucine, involved in plant growth processes.

Orthosulfamuron has not been previously registered for use as a pesticide. Proposed labels for granular (GR) and water dispersible granular (WG) formulations specify foliar, post-emergent application to rice at the 1-2 leaf stage and 2-4 leaf stage, respectively. Orthosulfamuron WG is applied once per year as a liquid spray via ground or aerial application to dry-seeded or wet-seeded rice. Orthosulfamuron GR is applied once per year as a solid via ground or aerial application to permanently flooded rice that is held static for at least 7 days after application.

The maximum annual application rates for orthosulfamuron WG and GR formulations are similar at 0.0689 pounds of active ingredient per acre (lbs a.i./A) and 0.0663 lbs a.i./A, respectively. Therefore, the maximum annual application rate of 0.0689 lbs a.i./A characterizes the maximum use pattern for orthosulfamuron on rice. As no uses of orthosulfamuron have yet been registered, the proposed use on rice is the maximum use pattern from which EDWCs were generated in order to characterize national drinking water exposure from orthosulfamuron use.

Fate and Transport Characterization

Our understanding of the environmental fate and transport properties of orthosulfamuron is limited by the sparse data set available. Many chemical properties and environmental fate parameters of orthosulfamuron listed in **Table 2** are based on single, supplemental studies. Model parameters listed in **Tables 4 and 5** were generated from the fate and transport properties given below.

Chemical/Fate Parameter	Value	Source
Molecular mass	424.44	MRID 46219074
Vapor pressure (20°C)	8.4×10^{-7} torr	MRID 46219010
Water solubility (20°C)	26.2 mg/L (pH 4) 629 mg/L (pH 7) 38,900 mg/L (pH 8.5)	MRID 46219009
Octanol-to-water partition coefficient (K_{ow})	104 (pH 4) 20.5 (pH 7) <2.0 (pH 10)	MRID 46219012
Range of Freundlich soil-to-water partition coefficients (K_f) for adsorption	2.0 (1/n = 1.02) – 20 (1/n = 0.98)	MRID 46219074 MRID 46578971

Chemical/Fate Parameter	Value	Source
Range of organic carbon normalized partition coefficients (K_{OC}) ¹	217 - 2330 L/kg _{OC}	MRID 46219074 MRID 46578971
Hydrolysis half life (25°C)	0.442 d (pH 5) 24.4 d (pH 7) 228 d (pH 9)	MRID 46219015 MRID 46578970
Aqueous photolysis half life	180 d	MRID 46588509
Aerobic aquatic metabolism half-life range	30.3 – 70.5 d	MRID 46219017
Anaerobic aqueous metabolism half-life range	88 – 217 d	MRID 46578967

¹ K_{OC} values were calculated based on K_f values for adsorption. *E.g.*, $K_{OC} = K_f$ (adsorption) ÷ % organic carbon.

The persistence of orthosulfamuron appears to depend on the pH of its environment. The compound readily undergoes acid catalyzed hydrolysis; however, as alkaline conditions increase, the compound is increasingly hydrolytically stable in water. Biodegradation in aerobic and anaerobic aquatic conditions occurs on a scale of months. Due to its low vapor pressure and high water solubility, volatilization from water and soil surfaces is not expected to be an important route of environmental dissipation. Orthosulfamuron is moderately to slightly mobile in soils, presenting a groundwater concern in alkaline sandy soils with low organic carbon content. Some of the transformation products of the compound appear to be more susceptible to leaching than the parent.

Transport and Mobility

Volatilization of orthosulfamuron from water and soil systems is not expected to be an important environmental fate process based on a water solubility range of 26 mg/L to 39,000 mg/L and a vapor pressure of 8.4×10^{-7} torr, both at 20°C (MRID 46219009; 46219010). The compound is expected to be mobile to slightly mobile in soils based upon K_f values from ten soils ranging from 2.0 (1/n=1.02) to 20 (1/n=0.98) (MRID 46219074; 46578971). Mobility is partially but not well explained by affinity to organic matter, as the organic carbon content of soils correlates with K_f values ($p < 0.05$), however the coefficient of variation (CV) across ten soils for K_{OC} (115%) is larger than that for K_f (78%). The mobility of orthosulfamuron may also be partially explained by affinity to clay particles, as K_f values correlate with the clay content of soils ($p < 0.05$) but not soil pH ($p > 0.05$; pH range 4.7 to 8.0). Affinity to clay may be due to an anion congener of orthosulfamuron, the presence of which could explain the ranges found in orthosulfamuron's water solubility (26 mg/L to 39,000 mg/L) and octanol-to-water partition coefficients (<2 to 104) across a range of pH values. The ionic nature of this congener would give orthosulfamuron greater mobility than expected for the neutral compound alone in high-pH, low-clay soils.

Compounds with K_f values less than five present a groundwater concern. Therefore, orthosulfamuron presents a groundwater concern in some soils, especially those that are alkaline, sandy, and of low organic carbon content.

Column leaching studies were inconclusive as to the leaching potential of orthosulfamuron, as they were conducted with only two pore volumes of water (MRID 46219018; 46578968). These

studies indicated that some of the compound's metabolites are capable of greater mobility than the parent and are more likely to leach to groundwater. Unaged orthosulfamuron ranged from not detected to 2.1% of the applied in leachate from five columns (MRID 46578968). Aged orthosulfamuron was not detected in leachate from four soil columns; however, pyrimidinyl-labeled degradates (mostly o-desmethyl orthosulfamuron) averaged 4.6% of the applied in leachate and phenyl-labeled degradates (mainly DBS acid) averaged 62.5% of the applied in leachate (MRID 46219018).

Degradation

Orthosulfamuron hydrolyzed with half-lives of 0.442, 24.4, and 228 days at 25°C in pH 5, pH 7, and pH 9 buffered solutions, respectively (MRID 46219015, 46578970). Major identified hydrolysis degradates were DBS acid and DB amine, as shown in Table 3. Orthosulfamuron degraded with a calculated phototransformation half-life of approximately 180 solar days when irradiated continuously under a filtered xenon lamp for 6.92 days at 25°C in a pH 9 buffered solution (MRID 46588509). The environmental phototransformation half-life of orthosulfamuron could not be determined because the intensity of the artificial light was not compared to natural sunlight. No major photodegradates were detected.

An anaerobic aquatic metabolism study was conducted for orthosulfamuron using a pond water-loam sediment system from Arkansas for 180 days and a pond water-clay sediment system from California for 150 days (MRID 46578967). Half-lives were calculated by counting non-extracted residues as parent compound, since multiple solvent systems were not employed in a reasonable extraction attempt. Half-lives in the microcosm from Arkansas were 217 and 142 days for pyrimidinyl- and phenyl-labeled orthosulfamuron, respectively. Dissipation occurred more rapidly in the microcosm from California; half-lives were 161 and 88 days for pyrimidinyl- and phenyl-labeled orthosulfamuron, respectively. Half-lives calculated with extracted parent compound alone were 89 and 80 days for the pyrimidinyl and phenyl labels in Arkansas, respectively, and 41 days for both labels in California. The major identified degradates were DBS acid, o-desmethyl orthosulfamuron, and DOP urea, as displayed in Table 3.

The aerobic aquatic metabolism of orthosulfamuron was also studied using the pond water-sediment systems from Arkansas and California (MRID 46219017). Once again, half-lives were calculated by counting non-extracted residues as parent compound, since multiple solvent systems were not employed in a reasonable extraction attempt. Degradation appeared to occur more rapidly under aerobic conditions as compared to anoxic conditions. Half-lives in the microcosm from Arkansas were 70.5 and 59.2 days for pyrimidinyl- and phenyl-labeled orthosulfamuron, respectively. Similar to the anaerobic aquatic metabolism study, degradation appeared to occur more rapidly in the California pond-sediment system; half-lives were 65.4 and 30.3 days for pyrimidinyl- and phenyl-labeled orthosulfamuron, respectively. It is possible that degradation occurred more rapidly in the California system since it was more acidic (pH = 6 in sediment) than the Arkansas system (pH = 6.9 in sediment) and hydrolysis of orthosulfamuron appears to be an acid catalyzed reaction. Half-lives calculated with extracted parent compound alone were 40.6 and 36.3 days for the pyrimidinyl and phenyl labels in Arkansas, respectively, and 24.1 and 21.3 days for the pyrimidinyl and phenyl labels in California, respectively. The major identified degradates were DBS acid and DOP urea, as shown in Table 3.

Field Studies

A single aquatic field dissipation study was conducted for orthosulfamuron using a bare ground plot of silt loam soil in Arkansas and a bare ground plot of silty clay loam soil in California (MRID 46578985). The plots were flooded with water 1–2 days following the application of orthosulfamuron to approximate the timing of herbicide application to rice. Soil samples (0-60 cm depth) were collected through 550-553 days post-treatment. Water and the underlying soil (“sediment”; 0-5 cm depth) samples were collected from 3 days to 2 months post-treatment. At the Arkansas test site, orthosulfamuron dissipated in the soil (0-15 cm depth), sediment (0-5 cm depth), and water with half-lives of 4.1 days, 6.2 days, and 6.4 days, respectively. At the California site, orthosulfamuron dissipated in the soil (0-15 cm depth) and sediment (0-5 cm depth) with half-lives of 2.3 days and 4.3 days, respectively. No half-life was calculated for the water phase because the chemical was not detected in the water.

Both test sites were analyzed for DOP urea, DBS acid, and o-desmethyl orthosulfamuron. The latter was below detection limits in all samples from both sites. At the Arkansas site, DOP urea and DBS acid were major degradates in the soil and sediment; DBS acid was the major degradate in water. At the California site, DOP urea was the major degradate in soil and sediment; no major degradates were detected in water.

Degradates

The transformation products resulting from the environmental degradation processes of orthosulfamuron are: 2-dimethylcarbamoylphenyl sulfamic acid (DBS acid; IR7863; S1); 2-sulfamoylamino-N,N-dimethylbenzamide (DBS amide; S2); 2-amino-N,N-dimethylbenzamide (DB amine; S4); 1-(4-hydroxy-6-methoxypyrimidin-2-yl)-3-[2-(dimethylcarbamoyl)phenylsulfamoyl]urea (o-desmethyl orthosulfamuron; IR8181; S9); N-(4,6-dimethoxypyrimidin-2-yl)urea (DOP urea; IR7825; S12); 4,6-dimethoxypyrimidin-2-yl amine (DOP amine; S13); and N-(4-hydroxy-6-methoxypyrimidin-2-yl)urea (o-desmethyl DOP urea; S15). Two other degradation products S5 and S11 are composed of multiple unresolved compounds. DBS acid (S1), DB amide (S4), o-desmethyl orthosulfamuron (S9), and DOP urea (S12) are the major degradates identified. The chemical structures of these degradation products are shown in Appendix II.

The main degradation pathway of orthosulfamuron appears to be the hydrolytic cleavage of the sulfamoylurea linkage to give DOP urea (S12) and DBS acid (S1). Another degradation pathway is the demethylation of orthosulfamuron to its o-desmethyl degradate (S9), which may also degrade to DBS acid. Orthosulfamuron and DBS acid may hydrolyze to form DB amine (S4). The maximum amounts of these degradation products observed in hydrolysis, aerobic aquatic metabolism, anaerobic aquatic metabolism, soil column leaching, and aquatic field dissipation studies are summarized in Table 3. No degradation products were detected in the photolysis study. Degradation products were not analyzed for in the hydrolysis study at pH 5.

Table 3. Maximum Reported Amounts of Orthosulfamuron Degradation Products.			
Degradate	Maximum % of Applied	Study Type	MRID
S1 (DBS acid)	58 (37 d, pH 7) 39 (150 d) 61 (61 d) 25 (55 d); 51 (leachate) 62 (% of initial, 7 d, soil); 23 (% of initial, 7 d, sediment); 34 (% of initial, 15 d, water)	Hydrolysis Anaerobic aquatic metabolism Aerobic aquatic metabolism Soil column leaching Aquatic Field Dissipation	MRID 46239015 MRID 46578967 MRID 46219017 MRID 46219018 MRID 46578985
S2 (DBS amide)	8.7 (113 d, pH 9) 4.2 (180 d) 4.0 (61 d) 7.2 (34 d); 8.0 (leachate)	Hydrolysis Anaerobic aquatic metabolism Aerobic aquatic metabolism Soil column leaching	MRID 46239015 MRID 46578967 MRID 46219017 MRID 46219018
S4 (DB amine)	13 (176 d, pH 9) 6.4 (71 d) <3 (55 d), 3.7 (leachate)	Hydrolysis Anaerobic aquatic metabolism Soil column leaching	MRID 46239015 MRID 46578967 MRID 46219018
S5 (multiple unresolved compounds)	22 (180 d) 2.5 (leachate)	Anaerobic aquatic metabolism Soil column leaching	MRID 46578967 MRID 46219018
S9 (o-desmethyl orthosulfamuron)	19 (100 d) 9.2 (90 d) 5.2 (55 d), 4.6 (leachate)	Anaerobic aquatic metabolism Aerobic aquatic metabolism Soil column leaching	MRID 46578967 MRID 46219017 MRID 46219018
S11 (multiple unresolved compounds)	23 (180 d) 0.26 (leachate)	Anaerobic aquatic metabolism Soil column leaching	MRID 46578967 MRID 46219018
S12 (DOP urea)	19 (59 d) 47 (90 d) 36 (55 d), 52 (soil column) 38 (soil column) 64 (% of initial, 186 d, soil); 111 (% of initial, 7 d, sediment)	Anaerobic aquatic metabolism Aerobic aquatic metabolism Soil column leaching Soil column leaching Soil column leaching Aquatic Field Dissipation	MRID 46578967 MRID 46219017 MRID 46219018 MRID 46578968 MRID 46578985
S13 (DOP amine)	1.1 (71 d) <3 (leachate) 1.4 (soil column)	Anaerobic aquatic metabolism Soil column leaching Soil column leaching	MRID 46578967 MRID 46219018 MRID 46578968
S15 (o-desmethyl DOP urea)	4.9 (150 d) <3 (leachate)	Anaerobic aquatic metabolism Soil column leaching	MRID 46578967 MRID 46219018

The toxicities of orthosulfamuron's major degradates are not determined; however, none are expected to be more toxic than the parent (USEPA, 2006). EDWCs were modeled to reflect exposure to orthosulfamuron and all of its organic degradates so as to include any potential residues of concern.

Drinking Water Exposure Modeling

Models

The Environmental Fate and Effects Division (EFED) Interim Rice Model was used to estimate the concentration of orthosulfamuron expected in surface water potentially used for drinking water (USEPA, 2002). The model calculates exposure as a function of the amount of pesticide applied and the pesticide's soil-to-water partition coefficient. Concentrations are assumed to immediately and evenly distribute throughout the rice paddy water. Dissipation processes, such as degradation, volatilization, and potential dilution with uncontaminated water outside the paddy are not accounted for by the model. The single exposure estimate produced with the Interim Rice Model is used as both an acute and a chronic screening exposure estimate that is expected to represent high-end exposure in released flood water (tailwater). The EDWC produced by the Interim Rice Model was not adjusted by an agricultural Percent Cropped Area (PCA) value because these values may not apply to semi-aquatic agriculture. PCA values were designed for exposure estimates in runoff from terrestrial crops, not for exposure estimates in tailwater from flooded crops.

Screening Concentration in Ground Water (SCI-GROW v2.3, Jul. 29, 2003) is a regression model used as a screening tool to estimate pesticide concentrations found in groundwater used as drinking water. SCI-GROW was developed by fitting a linear model to groundwater concentrations with the Relative Index of Leaching Potential (RILP) as the independent variable. Groundwater concentrations were taken from 90-day average high concentrations from Prospective Ground Water studies. The RILP is a function of aerobic soil metabolism and the soil-water partition coefficient. The output of SCI-GROW represents the concentrations that might be expected in shallow unconfined aquifers under sandy soils, which is representative of the groundwater most vulnerable to pesticide contamination and likely to serve as a drinking water source. The SCI-GROW model and user's manual may also be downloaded from the EPA Water Models web-page (USEPA, 2006a). Both the Interim Rice Model and SCI-GROW were run to estimate screening-level exposure of drinking water sources to orthosulfamuron.

Input Parameters

Input parameters for the Interim Rice Model follow in **Table 4**; data source and justification descriptors accompany values for each parameter.

Input Parameter	Value	Justification	Source
Application Rate (kg a.i./ha)	0.0772	0.0689 lbs a.i./A x 1.121 kg·A/ha·lbs = 0.0772 kg a.i./ha	Proposed label
Soil-to-Water Partition Coefficient (K_d) (L/kg)	6.48	Mean K_f is used when mean K_{oc} use is not appropriate.	MRID 46219074 MRID 46578971

The mean Freundlich soil-to-water partition coefficient (K_f) from ten soils (6.48) was used to model partitioning to rice paddy sediment. The organic carbon partition coefficient (K_{OC}) was not used, as it is a less precise descriptor of orthosulfamuron's mobility.

Input parameters for the SCI-GROW model appear in **Table 5**. Data source and justification descriptors accompany values for each parameter.

Input Parameter	Value	Justification	Source
Application Rate (lbs a.i./A)	0.0689	Label directions	Proposed label
Applications per Year	1	Label directions	Proposed label
Organic Carbon Partition Coefficient (K_{OC}) (L/kg _{OC})	221	Represents the lowest reported K_{OC} , as values show greater than three-fold variation.	MRID 46219074 MRID 46578971
Aerobic Soil Metabolism Half-life (days)	10 000	Approximates the stable kinetics of total toxic residues in aerobic aquatic metabolism systems.	MRID 46219017

The lowest reported Freundlich organic carbon partition coefficient ($K_{OC} = 221 \text{ L/kg}_{OC}$) was selected because the range of values from ten soils showed greater than three-fold variation. The aerobic soil metabolism half-life model input value (10 000 d) was selected to approximate the stable kinetics of total toxic residues in aerobic aquatic metabolism systems used in the absence of soil metabolism data.

Modeling Results

Screening estimates generated for drinking water exposure assessment are listed in **Table 6**. The surface water estimate was not adjusted by a Percent Cropped Area (PCA) factor, as these factors may not apply to semi-aquatic agriculture. The proposed use pattern for the WG formulation was the maximum use pattern modeled for surface water and groundwater exposure estimates, as described above. Modeled estimates are used as both peak and annual mean exposure values. Model input/output data for these estimates are attached in **Appendix I**.

Source	Peak Exposure (µg/L)	Annual Mean Exposure (µg/L)
Surface Water	40.5	<40.5
Groundwater	0.611	<0.611

¹ Surface water concentrations calculated by the Interim Rice Model and groundwater concentrations calculated by SCI-GROW do not distinguish between peak and chronic concentrations.

Peak concentrations found at drinking water facilities impacted by rice culture in California, Texas, and Louisiana may be less than these screening values due to degradation and dilution processes. Annual mean concentrations are expected to be lower than these screening values due to the same processes over a greater period of time. Where orthosulfamuron exposure occurs in acidic water bodies, peak concentrations are expected to be significantly lower and annual mean concentrations are expected to be negligible due to acid catalyzed hydrolysis.

The environmental fate data for orthosulfamuron are sparse. Soil metabolism data would increase confidence in and refine groundwater exposure estimates. Other factors inherent in the modeling that may have affected the accuracy and precision of this analysis include the quality of the input data, the ability of the models to represent the real world, and the use of maximum label practices that can be greater than actual agricultural practices.

SCI-GROW is a regression model that generates EDWCs based on two fate parameters that affect leaching. The Interim Rice Model generates EDWCs based on one fate parameter and the standardized dimensions of a rice paddy, which also results in a simple analysis. Screening models are not designed to simulate real events or typical exposure. These models should simply indicate which chemicals surpass high-end levels of concern and warrant refinement of dietary risk.

Drinking Water Treatment

The Office of Pesticide Programs (OPP) does not have direct data on the effects of drinking water treatment on orthosulfamuron. Flocculation and sedimentation removal may be effective at reducing orthosulfamuron concentrations. Carbon filtering may also reduce orthosulfamuron concentrations due to the compound's moderate affinity to organic carbon. Because of the absence of data on orthosulfamuron, the effects of drinking water treatment were not considered in this assessment.

CONCLUSIONS

Tier I drinking water exposure estimates for orthosulfamuron use on rice are represented by the maximum use pattern on rice (**Tables 1 and 6**). The environmental fate data used to generate these estimates are sparse. Aerobic soil metabolism data would increase confidence in and enable refinement of groundwater exposure estimates. The toxicities of orthosulfamuron's degradates have not been determined. Lastly, use of a refined surface water exposure model would increase confidence in and could potentially reduce surface water exposure estimates.

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Appendix I. Interim Rice Model and SCI-GROW Input/Output Data.

Interim Rice Model (USEPA, 2002).

$$EEC_0 = 10^9 \times M_T / [V_T + (m_{sed} \times K_d)]$$

where,

- M_T = total mass of pesticide applied per area = 0.0772 kg a.i./ha
- V_T = paddy water volume, assuming 4-inch depth = 1.067×10^6 L/ha
- K_d = mean K_f from ten soils = 6.48 L/kg
- m_{sed} = mass of top 1 cm of sediment = 130000 kg/ha
- 10^9 = $\mu\text{g}/\text{kg}$ conversion for estimate of ppb
- EEC = the dissolved concentration in water ($\mu\text{g}/\text{L}$, ppb)

Example calculation:

$$EEC = (10^9 \mu\text{g}/\text{kg}) \times (0.0772 \text{ kg a.i./ha}) / [1.067 \times 10^6 \text{ L/ha} + (130000 \text{ kg/ha} \times 6.48 \text{ L/kg})]$$

$$= \underline{40.5 \mu\text{g}/\text{L (ppb)}}$$

SCI-GROW Input/Output File.

SciGrow version 2.3
 chemical:Orthosulfamuron
 time is 9/21/2006 14:16:23

Application rate (lb/acre)	Number of applications	Total Use (lb/acre/yr)	Koc (ml/g)	Soil Aerobic metabolism (days)
0.069	1.0	0.069	2.21E+02	10000.0

groundwater screening cond (ppb) = 6.11E-01

Appendix II. Chemical Structures of Orthosulfamuron and Its Degradates.

Table II. Chemical Structures of Orthosulfamuron and Degradation Products Detected in Submitted Environmental Fate Studies.

Name	Structure
Orthosulfamuron (S3); IR5878; 1-(4,6-dimethoxypyrimidin-2-yl)-3-[2-(dimethylcarbamoyl)phenylsulfamoyl]urea	
DBS acid (S1); 2-dimethylcarbamoylphenyl sulfamic acid	
DBS amide (S2); 2-sulfmoylamino-N,N-dimethylbenzamide	
DB amine (S4); 2-amino-N,N-dimethylbenzamide	

Table II. Chemical Structures of Orthosulfamuron and Degradation Products Detected in Submitted Environmental Fate Studies.

Name	Structure
O-desmethyl orthosulfamuron (S9) 1-(4-hydroxy-6-methoxypyrimidin-2-yl)-3-[2-(dimethylcarbamoyl)phenylsulfamoyl]urea	
DOP urea (S12); IR7825; N-(4,6-dimethoxypyrimidin-2-yl)urea	
DOP amine (S13); 4,6-dimethoxypyrimidin-2-yl amine	
O-desmethyl DOP urea (S15); N-(4-hydroxy-6-methoxypyrimidin-2-yl)urea	