Potential Risks of Alachlor Use to Federally Threatened California Red-legged Frog (*Rana aurora draytonii*) and Delta Smelt (*Hypomesus transpacificus*)

Pesticide Effects Determinations

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June 15, 2009

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1.0 Executive Summary

1.1. Purpose of Assessment

The purpose of this assessment is to evaluate potential direct and indirect effects on the California red-legged frog *(Rana aurora draytonii*) (CRLF) and the Delta smelt (*Hypomesus transpacificus*) (DS) arising from FIFRA regulatory actions regarding use of alachlor (PC Code 090501) on agricultural and non-agricultural sites. In addition, this assessment evaluates whether these actions can be expected to result in effects to designated critical habitat for the CRLF and the DS.

The CRLF was listed as a threatened species by USFWS in 1996. The species is endemic to California and Baja California (Mexico) and inhabits both coastal and interior mountain ranges. The DS was listed as threatened on March 5, 1993 (58 FR 12854) by the U.S. Fish and Wildlife Service (USFWS) (USFWS, 2007a). It is only found in Suisun Bay and the Sacramento-San Joaquin estuary near San Francisco Bay.

1.2. Assessed Chemicals

Alachlor, an acetanilide herbicide, is a seedling cell growth inhibitor (Ross and Medlin, 2001) that disrupts the growth of new plant seedlings in areas where it is applied. The physiological details of the mode of action of acetanilide herbicides are not known.

Potential risks to alachlor degradates were quantified for aquatic organisms using the total toxic residues approach for aquatic exposure modeling, as there were no available data on the fate and transport of the three potential equally toxic degradates considered in this assessment (2-chloro-2',6'-diethylacetanilide, 2',6'-diethyl-N-methoxymethylacetanilide, and 2'6'-diethyl-2-hydroxy-N-methoxymethylacetanilide, see section 2.2.1). Potential risks to alachlor degradates were not quantified for terrestrial organisms since only one application was modeled and peak estimated environmental concentrations were used to calculate risk quotients.

1.3. Assessment Procedures

This assessment was completed in accordance with the U.S. Fish and Wildlife Service (USFWS) and National Marine Fisheries Service (NMFS) *Endangered Species Consultation Handbook* (USFWS/NMFS, 1998) and is consistent with procedures and methodology outlined in the Agency's Overview Document (USEPA, 2004).

1.3.1. Toxicity Assessment

The assessment endpoints include direct toxic effects on survival, reproduction, and growth of individuals, as well as indirect effects, such as reduction of the food source and/or modification of habitat. Federally-designated critical habitat has been established for the CRLF and the DS. Primary constituent elements (PCEs) were used to evaluate whether alachlor has the potential to affect designated critical habitat. The Agency evaluated registrant-submitted studies and data from the open literature to characterize alachlor toxicity. The most sensitive toxicity value available from acceptable or supplemental studies for each taxon relevant for estimating potential risks to the CRLF and DS and/or their designated critical habitat was used.

1.3.2. Exposure Assessment

1.3.2.1. Aquatic Exposures

Tier-II aquatic exposure models were used to estimate high-end exposures of alachlor in aquatic habitats resulting from runoff and spray drift from different uses. Peak model-estimated environmental concentrations resulting from different alachlor uses range from 3.2 µg a.i./L (sweet corn, incorporated use) to 56.0 µg a.i./L (woody ornamentals, nursey use). The maximum reported monitoring value from surface water data evaluated in this assessment was 91.5 µg/L. Frequency of detections ranged from 0% to 4.5%. Information from both modeled estimates and monitoring results are considered in this assessment. The study with the 91.5 µg/L value was conducted before the maximum application rates for alachlor were reduced from 6 lb a.i./acre to 4 lb a.i./acre in the 1990's. An application rate of 6 lbs a.i./acre was modeled for comparison purposes using the CAnursery scenario. Modeling output showed peak concentrations that are within a reasonable margin of error to the peak monitoring data (84 µg/L compared to 91.5 µg/L). Therefore, due to the reduced application rate the concentration cannot be used to reflect potential concentrations from current use practices and is not quantitatively used in this risk assessment.

1.3.2.2. Terrestrial Exposures

The T-REX model was used to estimate potential alachlor exposures to terrestrial species including birds (surrogate species for terrestrial phase CRLFs), mammals (CRLF prey), and invertebrates (CRLF prey). The AgDRIFT model was used to estimate deposition of alachlor on terrestrial and aquatic habitats from spray drift and to determine the distance from alachlor use sites the CRLF and the DS may be at risk of direct or indirect effects. The TerrPlant model was used to estimate alachlor exposures to terrestrial-phase CRLF habitat, including plants inhabiting semi-aquatic and dry areas, resulting from uses involving flowable and impregnated bulk fertilizer alachlor applications. The T-HERPS model was used to allow for further characterization of the dietary exposures of terrestrial-phase CRLFs relative to birds, which were used as a surrogate species for the CRLF.

1.3.3. Measures of Risk

Acute and chronic risk quotients (RQs) are compared to the Agency's Levels of Concern (LOCs) to identify instances where alachlor use has the potential to adversely affect the CRLF or DS or adversely modify their designated critical habitat. When RQs for a particular type of effect are below LOCs, the pesticide is considered to have "no effect" on the species and its designated critical habitat. Where RQs exceed LOCs, a potential to cause adverse effects or habitat modification is identified, leading to a conclusion of "may affect". If alachlor use "may affect" the assessed species, and/or may cause effects to designated critical habitat, the best available additional information is considered to refine the potential for exposure and effects, and distinguish actions that are NLAA (not likely to adversely affect) from those that are LAA (likely to adversely affect).

1.4. Alachlor Uses Assessed

All potential uses of alachlor were evaluated as part of this assessment. In the U.S., alachlor is currently registered for use on succulent and dry beans, field and sweet corn, cotton, woody ornamentals, peanuts, sorghum (milo), soybeans, and sunflowers. For the woody ornamentals, there is nothing on the alachlor labels that restricts the use to commercial uses, therefore, both commercial and residential uses will be considered here. Only the end-use products approved for use in California [*i.e.*, Lasso® Herbicide, INTRRO (EPA Reg. No.: 524-314) and Micro-Tech® Herbicide (EPA Reg. No.: 524-344)] are assessed here. These products only contain one active ingredient – alachlor. All of the alachlor end-use products are labeled as Restricted Use Pesticides (RUPs).

Based on California Department of Pesticide Regulation Pesticide Use Reporting (CDPR PUR) data, almost all of the use of alachlor between 1999 and 2006 in CA was on corn (55%) and beans (45%). The remaining alachlor uses listed in the CDPR PUR data made up <1% of alachlor use (*i.e.*, 'preplant', 'landscape', and 'research', as listed in the CDPR PUR data). In California, applications are limited to flowable applications via ground equipment (broadcast boom or banded) or via center pivot irrigation systems. Additionally, applications via impregnated bulk fertilizer are allowed for some uses (corn, sorghum, and soybeans). Application timing includes burndown prior to crop, preplant incorporated, pre-emergence surface, post-emergence surface (corn only), ground-crack surface (peanuts only), and post-transplant (woody ornamentals only).

1.5. Summary of Conclusions

Based on the best available information, the Agency makes a May Affect, and Likely to Adversely Affect (LAA) determination for the CRLF and the DS from the labeled uses of alachlor as described in **Table 1.1**. The effects determination is based on potential direct and indirect effects to terrestrial-phase CRLF and potential indirect effects to aquatic-phase CRLF and the DS. The LAA determination applies to all currently registered alachlor uses in California.

Additionally, the Agency has determined that there is the potential for effects to designated critical habitat of the CRLF and the DS from the use of the alachlor. A summary of the risk conclusions and effects determinations for each listed species assessed and their designated critical habitat is presented in **Tables 1.1** and **1.2**. Further information on the results of the effects determination is included as part of the Risk Description in Section 5.2. Given the LAA determination for the CRLF and the DS and potential effects to designated critical habitat for both species, a description of the baseline status and cumulative effects for the CRLF is provided in **Attachment 2** and the baseline status and cumulative effects for the DS is provided in **Attachment 4**.

Table 1.1. Effects Determination Summary for Effects of Alachlor on the CRLF and the DS.

Species	Effects Determination [1]	Basis for Determination
California red-legged frog (*Rana aurora draytonii*)	**LAA**[1]	**Potential for Direct Effects**
		Aquatic-phase (Eggs, Larvae, and Adults): None of the RQs for freshwater fish (used as a surrogate for aquatic-phase amphibians) exceed the Agency's LOCs for any registered alachlor use.
		Terrestrial-phase (Juveniles and Adults): The risk of direct adverse effects to terrestrial-phase CRLF from acute or sub-acute dietary exposure is low. However, the risk (or potential risk) to terrestrial-phase CRLF from chronic dietary exposure cannot be precluded and exists for all dietary classes relevant to the CRLF (for all of the registered alachlor uses).
		Potential for Indirect Effects
		Aquatic prey items, aquatic habitat, cover and/or primary productivity Alachlor could potentially impact terrestrial and aquatic plants to an extent that could result in indirect effects to the CRLF.
		Terrestrial prey items, riparian habitat CRLFs could be affected as a result of potential impacts to grassy/herbaceous vegetation. Potential effects to amphibian food item abundance that may indirectly affect terrestrial phase CRLFs could not be precluded.
Delta Smelt (*Hypomesus transpacificus*)	**LAA**[1]	**Potential for Direct Effects** None of the RQs for freshwater fish exceed the Agency's LOCs for any registered alachlor use.
		Potential for Indirect Effects Labeled uses of alachlor have the potential to adversely affect the DS by reducing available food (aquatic plants), by impacting the riparian habitat of grassy and herbaceous riparian areas, and/or by impacting water quality via effects to aquatic vegetation.

[1] May affect, likely to adversely affect (LAA)

Table 1.2. Effects Determination Summary for Alachlor Use and CRLF and DS Critical Habitat Impact Analysis.

Assessment Endpoint	Effects Determination	Basis for Determination
Modification of aquatic-phase PCEs (DS and CRLF)	**Habitat Effects**	As described in Table 1.1., the effects determination for the potential for alachlor to affect aquatic-phase CRLFs and the DS is LAA. These determinations are based on the potential for alachlor to indirectly affect the DS and aquatic-phase CRLF. Additionally, the potential areas of effect overlap with critical habitat designated for the CRLF and DS. Therefore, potential effects to aquatic plants and terrestrial (riparian) plants identified in this assessment could result in aquatic habitat modification.
Modification of terrestrial-phase PCE (CRLF)		As described in Table 1.1., the effects determination for the potential for alachlor to affect terrestrial-phase CRLFs is LAA. This determination is based on the potential for alachlor to directly affect terrestrial-phase CRLFs and their food supply and habitat. Additionally, the potential areas of effect overlap with critical habitat designated for the CRLF. Therefore, these potential effects could result in modification of critical habitat.

Based on the conclusions of this assessment, a formal consultation with the U. S. Fish and Wildlife Service under Section 7 of the Endangered Species Act should be initiated.

When evaluating the significance of this risk assessment's direct/indirect and adverse habitat modification effects determinations, it is important to note that pesticide exposures and predicted risks to the listed species and its resources (*i.e*., food and habitat) are not expected to be uniform across the action area. In fact, given the assumptions of drift and downstream transport (*i.e*., attenuation with distance), pesticide exposure and associated risks to the species and its resources are expected to decrease with increasing distance away from the treated field or site of application. Evaluation of the implication of this non-uniform distribution of risk to the species would require information and assessment techniques that are not currently available.

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- Enhanced information on the density and distribution of CRLF and the DS life stages within the action area and/or applicable designated critical habitat. This information would allow for quantitative extrapolation of the present risk assessment's predictions of individual effects to the proportion of the population extant within geographical areas where those effects are predicted. Furthermore, such population information would allow for a more comprehensive evaluation of the significance of potential resource impairment to individuals of the assessed species.

- Quantitative information on prey base requirements for the assessed species. While existing information provides a preliminary picture of the types of food sources utilized by the assessed species, it does not establish minimal requirements to sustain healthy individuals at varying life stages. Such information could be used to establish biologically relevant thresholds of effects on the prey base, and ultimately establish geographical limits to those effects. This information could be used together with the density data discussed above to characterize the likelihood of adverse effects to individuals.

- Information on population responses of prey base organisms to the pesticide. Currently, methodologies are limited to predicting exposures and likely levels of direct mortality, growth or reproductive impairment immediately following exposure to the pesticide. The degree to which repeated exposure events and the inherent demographic characteristics of the prey population play into the extent to which prey resources may recover is not predictable. An enhanced understanding of long-term prey responses to pesticide exposure would allow for a more refined determination of the magnitude and duration of resource impairment, and together with the information described above, a more complete prediction of effects to individual species and potential modification to critical habitat.

2.0 Problem Formulation

Problem formulation provides a strategic framework for the risk assessment. By identifying the important components of the problem, it focuses the assessment on the most relevant life history stages, habitat components, chemical properties, exposure routes, and endpoints. The structure of this risk assessment is based on guidance contained in USEPA's *Guidance for Ecological Risk Assessment* (USEPA 1998), the Services' *Endangered Species Consultation Handbook* (USFWS/NMFS 1998) and is consistent with procedures and methodology outlined in the Overview Document (USEPA 2004) and reviewed by the U.S. Fish and Wildlife Service and National Marine Fisheries Service (USFWS/NMFS 2004).

2.1. Purpose

The purpose of this endangered species assessment is to evaluate potential direct and indirect effects on individuals of the federally threatened California red-legged frog *(Rana aurora draytonii*) (CRLF) and/or the Delta smelt (*Hypomesus transpacificus*) (DS) arising from FIFRA regulatory actions regarding labeled uses of alachlor. In addition, this assessment evaluates whether labeled alachlor use is expected to result in effects to designated critical habitat for the CRLF and/or the DS. This ecological risk assessment has been prepared consistent with the settlement agreement in *Center for Biological Diversity (CBD) vs. EPA et al*. (Case No. 02-1580-JSW(JL)) which addresses the CRLF and was entered in Federal District Court for the Northern District of California on October 20, 2006. This assessment also addresses the DS for which alachlor was alleged to be of concern in a separate suit (*Center for Biological Diversity (CBD) vs. EPA et al.* (Case No. 07-2794-JCS)).

In this assessment, direct and indirect effects to the CRLF and DS and potential modification to their designated critical habitat are evaluated in accordance with the methods described in the Agency's Overview Document (USEPA 2004). The effects determinations for each listed species assessed is based on a weight-of-evidence method that relies heavily on an evaluation of risks to each taxonomic group relevant to assess both direct and indirect effects to the listed species and the potential for modification of their designated critical habitat (*i.e*., a taxon-level approach). Screening level methods include use of standard models such as PRZM, EXAMS, T-REX, TerrPlant, and AgDRIFT, all of which are mentioned in the Overview Document. In addition, T-HERPS has been used to refine estimates of exposure and risk to amphibians. Use of such information is consistent with the methodology described in the Overview Document (USEPA 2004), which specifies that "the assessment process may, on a case-by-case basis, incorporate additional methods, models, and lines of evidence that EPA finds technically appropriate for risk management objectives" (Section V, page 31 of USEPA 2004).

In accordance with the Overview Document, provisions of the ESA, and the Services' *Endangered Species Consultation Handbook*, the assessment of effects associated with registrations of alachlor is based on an action area. The action area is the area directly or indirectly affected by the federal action. It is acknowledged that the action area for a national-level FIFRA regulatory decision associated with a use of alachlor may potentially involve numerous areas throughout the United States and its Territories. However, for the purposes of this assessment, attention will be focused on relevant sections of the action area including those geographic areas associated with locations of the CRLF and DS and their designated critical habitat within the state of California. As part of the "effects determination," one of the following three conclusions will be reached separately for each of the assessed species regarding the potential use of alachlor in accordance with current labels:

- "No effect";
- "May affect, but not likely to adversely affect"; or
- "May affect and likely to adversely affect".

The CRLF and the DS have designated critical habitats associated with them. Designated critical habitat identifies specific areas that have the physical and biological features, (known as primary constituent elements or PCEs) essential to the conservation of the listed species. The PCEs for the CRLF are aquatic and upland areas where suitable breeding and non-breeding aquatic habitat is located, interspersed with upland foraging and dispersal habitat. PCEs for the DS include characteristics required to maintain habitat for spawning, larval and juvenile transport, rearing, and adult migration.

If the results of initial screening-level assessment methods show no direct or indirect effects (no LOC exceedances) upon individuals or upon the PCEs of the species' designated critical habitat, a "no effect" determination is made for use of alachlor as it relates to each species and its designated critical habitat. If, however, potential direct or indirect effects to individuals of a species are anticipated or effects may impact the PCEs of the designated critical habitat, a preliminary "may affect" determination is made for the FIFRA regulatory action regarding alachlor.

If a determination is made that use of alachlor "may affect" a listed species or its designated critical habitat, additional information is considered to refine the potential for exposure and for effects to each species and other taxonomic groups upon which these species depend (*e.g*, prey items). Additional information, including spatial analysis (to determine the geographic proximity of the assessed species' habitat and alachlor use sites) and further evaluation of the potential impact of alachlor on the PCEs is also used to determine whether effects to designated critical habitat may occur. Based on the refined information, the Agency uses the best available information to distinguish those actions that "may affect, but are not likely to adversely affect" from those actions that "may affect and are likely to adversely affect" the assessed listed species and/or result in "no effect" or potential effects to the PCEs of its designated critical habitat. This information is presented as part of the Risk Characterization in Section 5 of this document.

The Agency believes that the analysis of direct and indirect effects to listed species provides the basis for an analysis of potential effects on the designated critical habitat. Because alachlor is expected to directly impact living organisms within the action area (defined in Section 2.7), critical habitat analysis for alachlor is limited in a practical sense to those PCEs of critical habitat that are biological or that can be reasonably linked to biologically mediated processes (*i.e.*, the biological resource requirements for the listed species associated with the critical habitat or important physical aspects of the habitat that may be reasonably influenced through biological processes). Activities that may affect critical habitat are those that alter the PCEs and appreciably diminish the value of the habitat. Evaluation of actions related to use of alachlor that may alter the PCEs of the assessed species' critical habitat form the basis of the critical habitat impact analysis. Actions that may affect the assessed species' designated critical habitat have been identified by the Services and are discussed further in Section 2.6.

2.2. Scope

Alachlor is a chloroacetanilide herbicide currently registered in the U.S. to control broadleaf weeds and grasses in succulent and dry beans, field and sweet corn, cotton, woody ornamentals, peanuts, sorghum (milo), soybeans, and sunflowers. The end result of the EPA pesticide registration process is an approved product label. The label is a legal document that stipulates how and where a given pesticide may be used. Product labels (also known as end-use labels) describe the formulation type, acceptable methods of application, approved use sites, and any restrictions on how applications may be conducted. Thus, the use or potential use of alachlor in accordance with the approved product labels is "the action" being assessed.

This ecological risk assessment is for currently registered uses of alachlor in portions of the action area reasonably assumed to be biologically relevant to the CRLF or the DS habitat and their designated critical habitat. Further discussion of the action area and designated critical habitat is provided in Section 2.4 and 2.5.

2.2.1. Evaluation of Degradates

This ecological risk assessment includes all potential ecological stressors resulting from the use of alachlor, including alachlor and its potential degradates of concern. Degradates of concern may include those that are found at significant concentrations (>10% by weight relative to parent) in available degradation studies or those that are of toxicological concern. Major degradates of alachlor (>10% formation by weight, or are of toxicological concern) are presented below in **Table 2.1**, and are discussed below. A summary of formation pathways of the three degradates of toxicological concern is presented in **Table 2.2**.

Regarding all of the known degradates, the Health and Effects Division (HED) chapter for the RED concluded that alachlor ethane sulfonic acid (alachlor-ESA) is much less toxic than the parent (USEPA, 1998a). Available toxicity data for alachlor oxanilic acid indicate that its toxicity is also much less than the parent (USEPA, 2006). The remaining major water soluble degradates of alachlor, DM-oxanilic acid, 2',6'-diethyl-2-methylsulfinylacetanilide, 2',6'-diethyl-N-methoxymethyloxoanilic acid, and alachlor sulfinylacetic acid, are also polar oxanilic or sulfonic acids and are expected to share the poor *in vivo* absorption and metabolism characteristics of compounds alachlor oxanilic acid and alachlor-ESA (USEPA, 2006). The remaining less polar major degradates, 2-chloro-2',6'-diethylacetanilide, 2',6'-diethyl-N-methoxymethylacetanilide, and 2',6'-diethyl-2hydroxy-N-methoxymethylacetanilide, are more structurally similar to the parent than the water soluble degradates. Because toxicity data for these three compounds are unavailable, HED assumed that they have the same toxicity as alachlor parent and are residues of risk concern (USEPA, 2006). Therefore, a total toxic residues approach was used for this assessment for aquatic exposure to evaluate the potential exposure to the residues of risk concern, *i.e.*, alachlor and the three degradates 2-chloro-2’,6’-diethylacetanilide, 2’,6’-diethyl-N-methoxymethylacetanilide, and 2’,6’-diethyl-2hydroxy-N-methoxymethylacetanilide (identified as compounds II, IX, and XIII in **Table 2.2**). For terrestrial exposures, since only one application is modeled for each use and peak estimated concentrations are used (and, thus, concentrations would not increase even if degradates were considered), only the parent is modeled.

Table 2.1. Major and Minor Degradates of Alachlor*

Compound	Chemical Name	Synonym
	Major Degradates	
II	**2-chloro-2’,6’-diethylacetanilide**	None
III	2',6'-diethyloxanilic acid	DM-oxanilic acid
IV	2’,6’-diethyl-2-sulfoacetanilide	None
VII	2’,6’-diethyl-N-methoxymethyloxoanilic acid	None
VIII	[N-methoxymethyl-N-(2,6-diethylphenyl)-2-amino-2-oxoethyl]sulfinylacetic acid	Alachlor sulfinylacetic acid MON 5768
IX	**2’,6’-diethyl-N-methoxymethylacetanilide**	None
X	2',6'-diethyl-N-methoxymethyloxanilic acid	Alachlor oxanilic acid MON 5760
XI	2',6'-diethyl-N-methoxymethyl-2-sulfoacetanilide	Alachlor sulfonic acid, Alachlor-ESA MON 5775
XIII	**2’,6’-diethyl-2-hydroxy-N-methoxymethylacetanilide**	None
	Minor Degradates	
I	2’,6’-diethylacetanilide	None
V	2’,6’-diethyl-2-methoxyacetanilide	None
VI	2’,6’-diethyl-2-methylsulfinylacetanilide	None
XII	2’,6’-diethyl-N-methoxymethyl-2-oxoacetanilide	None
XV	2’,6’-diethyl-N-methoxymethyl-2-methylthioacetanilide	None
XVI	2’,6’-diethyl-N-methoxymethyl-2-methylsulfinylacetanilide	None
XVII	2’,6’-diethyl-N-methoxymethyl-2-methylsulfonylacetanilide	None
XX	2’,6’-diethylbenzyl alcohol	None
XXII	2’,6’-diethyl-N-hydroxymethyl-2-methoxyacetanilide	None
XXIII	8-ethyl-N-methoxymethyl-4-methyl-2-oxotetrahydroquinoline	None
XXIV	2’-acetyl-2-chloro-6’-ethyl-N-methoxymethylacetanilide	None

*Bold text indicates degradates of toxicological concern previously identified in USEPA, 2006.

Table 2.2. Summary of Formation Pathway of Alachlor Degradates of Concern (assumed equal toxicity to parent)[a]

Degradate	Formation Pathway				
	Photolysis in Water	**Photolysis in Soil**	**Aerobic Metabolism in Soil**	**Anaerobic Metabolism in Soil**	**Anaerobic Metabolism in Water**
2-chloro-2',6'-diethylacetanilide (Compound II)	--	--	X (20%, 18 d t ½)	--	--
2',6'-diethyl-N-methoxymethylacetanilide (Compound IX)	X (1.1%, 48 h t ½)	--	X (2.5%, 14 d t ½)	--	X (35.3%, 21 d t ½)
2',6'-diethyl-2-hydroxy-N-methoxymethylacetanilide (Compound XIII)	--	X (6.5%, 14 d t ½)	X (10.2%, 7 d t ½)	--	--

[a] Values in parentheses are percentage of parent formed; half-lives for these compounds for each formation pathway are presented following the percent formation. See USEPA, 2006 for additional discussion on these degradates.
X = data available
-- = no data available

Some toxicity data are available for aquatic plants and animals on the alachlor degradates alachlor sulfonic acid, alachlor oxanilic acid, and the minor degradate 2,6-diethyaniline. Additionally, rat toxicity data are available for the major alachlor degradates alachlor oxanilic acid, alachlor sulfinylacetic acid, and the minor degradate t-hydroxyalachlor (MON 52707). For the aquatic organisms, in all of the taxa-degradate combinations for which data are available, parent alachlor is more toxic (in many cases, orders of magnitude more toxic) than the degradates (**Table 2.3**). For mammals, the parent compound appears either more toxic or equatoxic with the degradates tested (**Table 2.4**).

Table 2.3. Comparison of Aquatic Organism Toxicity Data for Alachlor and its Degradates.

TAXA/SPECIES	ENDPOINT	COMPOUND			
		Alachlor (parent)	Alachlor Sulfinic Acid (MON 5775)	Oxanilic Acid (MON 5760)	2,6-Diethylaniline
		Endpoint (mg a.i./L) (MRID/Reference)	Endpoint (mg a.i./L) (MRID)	Endpoint (mg a.i./L) (MRID)	Endpoint (mg a.i./L) (Reference)
Aquatic invertebrate Daphnid (*Daphnia magna*)	48-hr EC_{50}	7.7 (40098001)	>104 (43774703)	>95 (43774705)	--
Freshwater fish Rainbow trout (*Oncorhynchus mykiss*)	96-hr LC_{50}	1.8 (C.I.* = 1.5 – 2.1) (00023616)	>104 (43774704)	>100 (43774706)	--
Amphibian African clawed frog (*Xenopus laevis*)	96-hr LC_{50}	6.1 (E66376) (Osano *et al.*, 2002)	--	--	19.4 (E66376) (Osano *et al.*, 2002)
Aquatic plant (nonvascular) Green algae (*Selenastrum capricornutum*)	5-day EC_{50}	0.00164 (C.I. = 0.0015 – 0.0024) (427638-01)	>120 (450460-01)	--	--
	5-day NOAEC	0.00035 (427638-01)	120 (highest conc. tested) (450460-01)		
Aquatic plant (nonvascular) Cyanobacteria (*Anabaena flos-aquae*)	5-day EC_{50}	EC_{50} = >19 (446497-01)	>120 (450460-02)	--	--
	5-day NOAEC	19 (highest conc. tested) (446497-01)	120 (highest conc. tested) (450460-02)		
Aquatic plant (nonvascular) Freshwater diatom (*Navicula pelliculosa*)	5-day EC_{50}	2.63 (C.I. = 2.4 – 3.0) (446497-04)	3.6 (C.I. = 2.9 – 4.1) (450460-03)	--	--
	5-day NOAEC	NOAEC = 1.0 (446497-04)	2.5 (450460-03)		
Aquatic plant (nonvascular) Marine diatom (*Skeletonema costatum*)	5-day EC_{50}	0.21 (C.I. = 0.15 – 0.26) (446497-03)	5.0 (C.I. = 4.6 – 5.6) (450460-04)	--	--
	5-day NOAEC	0.098 (446497-03)	2.0 (450460-04)		
Aquatic plant (vascular) Duckweed (*Lemna gibba*)	14-day EC_{50}	0.0023 (C.I. = 0.0021 – 0.0033) (446497-02)	>120 (450460-05)	--	--
	14-day NOAEC	0.000339 (446497-02)	120 (highest conc. tested) (450460-05)		

-- = No data available
* C.I. = 95% Confidence Interval

Table 2.4. Comparison of Mammalian Acute and Chronic Toxicity Data for Alachlor and its Degradates.

TAXA/ SPECIES	ENDPOINT	COMPOUND				
		Alachlor (parent)	Alachlor Sulfinic Acid (MON 5775)	Oxanilic Acid (MON 5760)	Sulfinylacetic Acid (MON 5768)	t-hydroxyalachlor (MON 52707)
		Endpoint (mg/kg)	Endpoint (mg/kg)	Endpoint (mg/kg)	Endpoint (mg/kg)	Endpoint (mg/kg)
Rat (acute oral)	LD_{50}	930	>6,000	>5,000	>5,000	>500 (males) >500, <2,000 (females)
Rat (90-day dietary)	NOAEL	15*	157 (males) 207 (females)	13,000	4,000	--

* As reported in HEDs assessment (USEPA, 2006), although the study was not acceptable.

2.2.2. Evaluation of Mixtures

The Agency does not routinely include an evaluation of mixtures of active ingredients (either those mixtures of multiple active ingredients in product formulations, or those in the applicator's tank, in its risk assessments. In the case of product formulations of active ingredients (registered products containing more than one active ingredient) each active ingredient is subject to an individual risk assessment for regulatory decision regarding the active ingredient on a particular use site. If effects data are available for a formulated product containing more than one active ingredient, they may be used qualitatively or quantitatively in accordance with the Agency's Overview Document and the Services' Evaluation Memorandum (USEPA, 2004; USFWS/NMFS, 2004). Alachlor does have two end-use products that are co-formulated with atrazine (and atrazine-related compounds), however, neither of these products is registered for use in California. Therefore, none of the alachlor products assessed here contains more than one active ingredient.

Based on the results of the available data, alachlor mixtures have shown additive effects (*e.g.*, when alachlor is mixed with atrazine alone or glyphosate alone), synergistic effects (*e.g.*, when mixed with fluridone alone or with multiple herbicides), and antagonistic effects (*e.g.*, when mixed with imazapyr alone) (see **Appendix A** for details). If chemicals that show more than additive effects with alachlor are present in the environment in combination with alachlor, the toxicity of alachlor could be increased. Conversely, when alachlor is found in combination with chemicals that show antagonistic effects, the toxicity of alachlor could be decreased. The potential increase or decrease in toxicity could be offset by other factors including but not necessarily limited to: (1) the exposed species, (2) the chemicals in the mixture, (3) the ratio(s) of the chemical concentrations, (4) differences in the pattern and duration of exposure to the chemicals, and (5) the differential effects of other physical/chemical characteristics of the receiving waters (*e.g.*, organic matter present in sediment and suspended water). Quantitatively predicting the combined effects of all these variables on mixture toxicity to any given taxa with confidence is beyond current capabilities. However, a qualitative discussion of implications of the available pesticide mixture effects data involving alachlor on the confidence of risk assessment conclusions is addressed as part of the uncertainty analysis for this effects determination.

2.3 Previous Assessments

Alachlor was first registered in the U.S. in 1969. A Reregistration Eligibility Decision (RED) for alachlor was signed in 1998 (USEPA 1998b). In the RED, the following mitigation measures were required: a reduction in application rates and the classification of alachlor as a Restricted Use Pesticide. These mitigation measures have been implemented on current labels. The RED identified potential risk to terrestrial birds from chronic exposure and risk to non-target terrestrial and aquatic plants from exposure to alachlor. Aquatic animals were identified as being at potential risk from chronic exposure to alachlor, but were identified as having low risk from acute exposures.

The following data gaps were identified in the EFED science chapter for the RED (USEPA, 1998b): terrestrial field dissipation studies conducted outside of California, additional aquatic plant toxicity studies, an avian reproduction study, and aquatic plant studies for the alachlor degradate alachlor ethane sulfonic acid (alachlor-ESA). Additionally, the science chapter highlighted concerns about the impact that alachlor and its degradates may have on ground water quality and surface water sources for drinking water.

Subsequent to the RED, EFED conducted an ecological risk assessment for the new use of alachlor on sunflowers and cotton (USEPA, 2006a). The screening-level assessment concluded that the use of alachlor on sunflowers and cotton could result in risk to Federally-listed threatened and endangered (listed) and non-listed terrestrial and aquatic plant species. Additionally, there was potential risk to small, non-listed birds that forage on short grass, broadleaf plants and small insects, and potential acute risk to listed avian species for several size class/forage item combinations. While a no-observed adverse effect concentration (NOAEC) was not determined for birds, using the lowest observed adverse effect concentration (LOAEC) tested indicated potential chronic risk to birds in all forage items. Potential acute risk to listed mammals and potential chronic risk to listed and non-listed mammals was also indicated for most size class/forage item combinations.

A human health risk assessment on potential cumulative effects of alachlor with other chloroacetanilide herbicides was also conducted subsequent to the RED. The cumulative assessment (including the cumulative effects of alachlor and acetochlor) was completed in 2006 (USEPA, 2006b). Similar cumulative assessments have not been conducted for ecological effects.

EPA conducted an assessment of potential effects of alachlor to 26 listed Pacific salmon and steelhead and on May 30, 2002, determined the uses of alachlor would have no effect on those species.

2.4 Stressor Source and Distribution

2.4.1 Environmental Fate Properties

As characterized in the RED (USEPA, 1998b), alachlor is stable to hydrolysis at pH 3, 6, and 9 and stable to photolysis. Alachlor is metabolized at moderate rates ($t_{½}$ = 26 - 34 d) in aerobic soils, with several degradates observed, including DM-oxanilic acid, alachlor-ESA, alachlor oxanilic acid, and alachlor sulfinylacetic acid. Data submitted to the Agency on 4/4/2008 to fulfill the identified data gap for alachlor aquatic metabolism are currently under review. Supplemental batch equilibrium and acceptable column leaching data for alachlor indicate that it is mobile and is not appreciably adsorbed to soils with low organic matter. A batch equilibrium study of alachlor-ESA shows that the degradate does not sorb appreciably and is highly mobile.

Terrestrial field dissipation studies are consistent with laboratory studies and demonstrate that alachlor was mobile and dissipated at moderate rates; dissipation half-lives of 6 and 11 days are within an order of magnitude of aerobic soil metabolism study half-lives ranging from 26 to 34 days. It appears that the persistence and mobility of the chemical may increase as it reaches deeper soil horizons that have lower organic matter content and decreased biological activity, thus, increasing its potential to leach into groundwater.

Degradation and Metabolism

Alachlor is a soluble chemical (240 ppm in water at 24°C), with a moderate vapor pressure of 2.2×10^{-5} torr (24°C; MRID 152209) suggesting the compound could volatilize. Octanol/water partition coefficient (K_{ow}) values were difficult to produce with accuracy and precision; early values included 33.0 and 37.1 (MRID 152209). More recent studies indicate that alachlor's K_{ow} value lies in the range of 1100 to 2800, which is higher than the value of 434 reported in the 1998 RED (MRID 257282, 40396301).

Alachlor is stable to hydrolysis in buffered solutions at pH's 3, 6, and 9, and appears to be relatively stable in natural lake water (MRID 134327). Alachlor does not show any absorption bands above 240 nm; therefore, it is not expected to undergo photolysis in water or on soil (MRID 23012).

In soils, under aerobic microbe-rich conditions, alachlor appears to degrade at a moderate rate. Results of three studies (one acceptable and two supplemental) show that alachlor degrades with first-order half-lives calculated using linear regression on log-transformed data of 26-34 days; the aerobic metabolism rates listed in the RED were faster but were 50% dissipation times (DT_{50}), not half-lives. The terrestrial field dissipation studies include use of different sites, different formulations, and different soil types, and indicate that under aerobic soil metabolism conditions, alachlor degrades to several major metabolites. Major degradates in the aerobic soil metabolism studies were DM-oxanilic acid (with a maximum of 17.0% of the applied), alachlor sulfonic acid (24.9% of the applied), alachlor oxanilic acid (22.4% of the applied), and alachlor sulfinylacetic acid (16.2% of the applied). Alachlor sulfinylacetic acid was not observed in the aerobic soil metabolism study classified as acceptable; however, it was observed in a supplemental study. All four degradates appear to be more persistent than alachlor, since significant concentrations remained in the soils at the end of the studies. Carbon dioxide (complete mineralization) is the ultimate degradate, comprising 16-30% of the mass applied after 175 days. Unextracted residues comprised ≤21% of the mass applied at the same test interval, despite multiple extractions with acetonitrile, ammonium hydroxide, and water (MRIDs 23014, 101531, 134327).

Microbial degradation in aqueous environments is poorly understood; submitted data regarding this degradation pathway are under review.

Degradates

Major degradates of alachlor include compounds 2-chloro-2’,6’-diethylacetanilide, DM-oxanilic acid, 2’,6’-diethyl-2-sulfoacetanilide, 2’,6’-diethyl-N-methoxymethyloxoanilic acid, 2’,6’-diethyl-2-hydroxy-N-methoxymethylacetanilide, 2’,6’-diethyl-N-methoxymethylacetanilide, alachlor oxanilic acid, alachlor-ESA, and 2’,6’-diethyl-2-hydroxy-N-methoxymethylacetanilide [**Table 2.1** (section 2.2.1)]. Minor degradates include compounds 2’,6’-diethylacetanilide , 2’,6’-diethyl-2-methylsulfinylacetanilide, 2’,6’-diethyl-N-methoxymethyl-2-oxoacetanilide, 2’,6’-diethyl-N-methoxymethyl-2-methylthioacetanilide, 2’,6’-diethyl-N-methoxymethyl-2-methylsulfinylacetanilide, 2’,6’-diethyl-N-methoxymethyl-2-methylsulfonylacetanilide, and 2’-acetyl-2-chloro-6’-ethyl-N-methoxymethylacetanilide.

The major water soluble degradates of alachlor, compounds DM-oxanilic acid, 2’,6’-diethyl-2-sulfoacetanilide, alachlor sulfinylacetic acid, alachlor oxanilic acid, and alachlor-ESA have carboxylic or sulfonic acid functional groups that render a negative (anionic) character to the molecule under normal environmental conditions, and, therefore, are expected to be very mobile in soils. This is based on mobility data for the degradates of propachlor (propachlor sulfonic acid and propachlor oxanilic acid), which are structurally similar to the degradates of alachlor (MRID 42485703, 42485704). In addition, a batch equilibrium study on alachlor-ESA shows that this degradate is very weakly adsorbed, although quantitative results could be obtained in only one of the soils (Sable silty clay loam; MRID 44405301). The Freundlich K_f value was 0.45 (1/n=0.95), yielding an organic carbon partition coefficient (K_{OC}) value of 15 L/kg$_{OC}$. Total toxic residues approach was used for this assessment to evaluate the potential exposure to the parent and residues of risk concern (*i.e*., alachlor and 2-chloro-2’,6’-diethylacetanilide, 2’,6’-diethyl-N-methoxymethylacetanilide, and 2’6’-diethyl-2-hydroxy-N-methoxymethylacetanilide). Refer to Section 2.2.1 for a more detailed discussion regarding the toxicity of major degradates of the parent alachlor.

The total residues of concern (TROC) for alachlor were defined as parent plus 3 degradates of concern as listed in **Table 2.2**. Structural analysis suggested that an assumption of additivity is reasonable (USEPA, 2006). This modeling strategy requires an assumption that all residues of concern have similar physical, chemical, and partitioning characteristics. The formation of persistent, toxic degradation products is expected to extend the residual effects of the parent. Therefore, methology, outlined in the “White Paper on Methods for Assessing Ecological Risks of Pesticides with Persistent, Bioaccumulative and Toxic Characteristics” presented to the FIFRA Scientific Advisor Panel, in October, 2008, was followed to calculate total toxic residues of concern (parent, plus 2-chloro-2’,6’-diethylacetanilide, 2’,6’-diethyl-N-methoxymethylacetanilide, and 2'6'-diethyl-2-hydroxy-N-methoxymethylacetanilide). Application rates for alachlor were used to represent the total mass loading of pesticide and its degradation product. This modeling approach does not consider temporal occurrence of degradation products.

Mobility

Based upon supplemental studies, alachlor appears to be mobile in soils (MRID 27139, 27140, 78301, 134327). Freundlich soil partitioning coefficients (K_{ads}) for alachlor ranged from 0.35 to 3.7 L/kg (MRID 152209). Corresponding organic carbon distribution coefficient from the Alachlor Reregistration Eligibility Decision Document was 190 L /kg (USEPA, 1998b). These data range in classification from mobile to moderately mobile according to current guidance (Environmental Fate and Effects Division, 2006). It is expected that the degradates as well as the parent will be very mobile in soils due to the water soluble properties and the carboxylic or sulfonic acid functional groups that render an anionic character to the molecule. Similarly, alalchlor-ESA was found to be very weakly adsorbed to soils (MRID 44405301), yielding a Freundlich K_{ads} value of 0.45 (K_{oc} value of 15).

Volatility

Volatilization is not expected to be an important route of dissipation for alachlor; however it does present a moderate vapor pressure (2.2×10^{-5} torr at 24°C; Beestman and Deming, 1974), and there is some potential for it to volatilize (Section 3.1.3.4).

Bioconcentration

No acceptable bioconcentration data were available for alachlor. The bioconcentration potential of alachlor is unclear, based on conflicting, unacceptable data on the octanol-water partition coefficient. In the 1998 RED, it was indicated that alachlor was not expected to bioconcentrate significantly in fish, based on a relatively low octanol/water partition coefficient (K_{ow} = 434) and the low K_{ow} values of similar chloroacetanilides (USEPA, 1998b). However, a more recently submitted octanol/water partition coefficient value (K_{ow} = 1223) offers less support for the expectation of low bioconcentration (MRID 257282, 40396301).

The screening-level predictive model BCFWIN v2.15 uses alachlor's structural chemistry, an experimental K_{ow} value of 3.3×10^3 (Hansch *et al.*, 1995), and the submitted K_{ow} value of 1223 to estimate a range of alachlor's bioconcentration factor (BCF = 48 - 102). This range of BCF estimates supports the expectation of low alachlor bioconcentration in fish but is not confirmatory.

Furthermore, based on data and a discussion from the most recent Health Effects Division's (HED) risk assessment for alachlor (USEPA, 2006), alachlor is excreted relatively rapidly in mammals. Excretion is via both urine and feces, with about 89% of the administered dose eliminated by 10 days after exposure. Elimination was shown to be biphasic, with an initial rapid phase (half-life = 0.2 - 10.6 hr), followed by a second slower phase (half-life = 5 – 16 days) (MRID 00132045). In rats, alachlor is extensively metabolized (14 metabolites identified in urine and 13 in feces). In urine, the *sec*-amide hydroxymethyl sulfone metabolite was the predominant metabolite (2.1 – 7.4% of the dose). In feces, the tert-amide mercapturic acid and the disulfide appeared to be the major metabolites (<5% of dose).

Field Dissipation

In a terrestrial field dissipation study conducted in Chico, California, alachlor, at 4 lbs a.i./acre, dissipated with a half-life of 11 days from loam/sandy clay loam soil planted to corn. Most of the alachlor was found in the 0- to 18-inch soil layers, with occasional detections in the 18- to 24-, 24- to 36-, and 36- to 48-inch layers (the deepest layer sampled), indicating a large extent of leaching. The four major water-soluble metabolites of alachlor were also monitored in this study. The soil composition data in this study show increasing percent of clay with soil depth (to a maximum of 65% clay in the 24- to 36-inch soil depth). This "clay pan" reduces the flow of water into deeper soils layers, thus, leaching of both parent alachlor and degradates was likely reduced from what may have occurred if a clay pan was not present (MRID 42528001, 42528002).

The oxanilic acid, sulfinylacetic acid, and ethanesulfonic acid degradates were detected in the 0- to 6- and 6- to 12-inch soil depths at average concentrations of 0.010-0.045 ppm. Detections were observed through 36- to 48-inch soil depth for the oxanilic acid, 18- to 24-inch soil depth for the sulfinylacetic and sulfonic acids, and 6- to 12-inch soil depth for the DM-oxanilic acid. Generally, detections occurred through 44-90 days post-treatment in the subsoils. Once in the subsoils, these degradates appeared to persist. Groundwater was not monitored in the study, but detections of alachlor ethanesulfonic acid in groundwater have been confirmed (USEPA, 1998b).

Alachlor, applied once at 4 lbs a.i./acre, dissipated with a half-life of 6 days from the 0- to 6-inch soil depth of a bare-ground plot of sandy loam soil in Hickman, California. Bare ground was used to simulate preemergent application. Alachlor remained mostly in the 0- to 6-inch soil depth. Detections averaging 0.018-0.046 ppm were reported in the 6- to 12-inch soil depth on the day of application and one day afterward (MRID 42528001).

In the Hickman study, the alachlor degradate DM-oxanilic acid was detected in the 0- to 6-inch soil depth from 1 through day 366 post-treatment. The degradate was detected in the 6- to 12-inch soil depth only on day 182 post-application (with an average value of 0.004 ppm). Alachlor oxanilic acid was detected in the 0 to 6-inch soil depth from day 0 through day 366 post-application; in addition, at three test intervals, detections were reported in the 6- to 12-inch soil layer. The degradate was also detected in the 12- to 18- and 18- to 24-inch soil layers on day 182 after application. Alachlor sulfinylacetic acid was observed at low levels in the 0- to 6-inch soil layer from day 1 to 182 after application. In addition, the degradate was detected in the 6- to 12- and 18- to 24-inch soil layers on day 182 after application. Alachlor-ESA was observed at low levels from day 0 through day 366 after application at average levels ranging from 0.003-0.010 ppm. Detections were also reported in the 6- to 12-inch soil depth on two test intervals. Furthermore, alachlor-ESA was detected in the 12- to 18-inch soil depth on day 182 after application, with an average value of 0.003 ppm (MRID 42528001).

General physical/chemical properties of alachlor are summarized in **Table 2.5**.

Table 2.5. General Chemical/Physical Properties of Alachlor.

Parameter	Value	Source
Chemical name	2-chloro-2',6'-diethyl-N-methoxymethylacetanilide	MRID 134327
Molecular Weight	269.77 g/mol	MRID 146114
Solubility	240 mg/L (24°C)	Beestman and Deming, 1974
Vapor Pressure	2.2×10^{-5} torr (24°C)	Beestman and Deming, 1974
Hydrolysis half life (pH 3)	stable (25°C)	MRID 134327
Hydrolysis half life (pH 6)	stable (25°C)	MRID 134327
Hydrolysis half life (pH 9)	stable (25°C)	MRID 134327
Aqueous photolysis half life	Assumed stable	MRID 23012
Soil photolysis half life	Assumed stable	MRID 23012
Aerobic soil metabolism half life	29.7 d (silt loam) 34.0 d (loamy sand) 25.8 d (silt)	MRID 134327
Soil-water distribution coefficient (K_{ads})	0.33 - 3.7 L/kg[1]	MRID 152209
Organic carbon partitioning coefficient (K_{oc})	190 L/kg	USEPA, 1998b
Octanol-water partition coefficient (K_{ow})	1223	MRID 40396301
Bioconcentration factor (BCF)	48 – 102 (estimated)	BCFWIN v2.15

[1] Range of K_{ads} values for four soils, which may be high due to over-sieving of soils.

2.4.2 Mechanism of Action

An acetanilide, alachlor is a seedling cell growth inhibitor (Ross and Medlin, 2001), primarily disrupting the growth of new plant seedlings in areas where it is applied. The physiological details of the mode of action of acetanilide herbicides are not known.

2.4.3 Use Characterization

In the U.S. alachlor is currently registered for use on succulent and dry beans, field and sweet corn, cotton, woody ornamentals, peanuts, sorghum (milo), soybeans, and sunflowers. For the woody ornamentals, there is nothing on the alachlor labels that restricts the use to commercial uses, therefore, both commercial and residential uses will be considered here. There are currently five alachlor products registered in the U.S. (one product is a technical grade for use in the manufacture of end-use products and four are end-use products) (see **Table 2.6**). Of the four end-use products, two [*i.e.*, Lariat® Herbicide (EPA Reg. No.: 524-329 and Bullet® Herbicide (EPA Reg. No.: 524-418)] also contain the herbicide atrazine (and atrazine-related compounds). However, both of these co-formulated products contain label statements that the products are not approved for use in California. Only the end-use products approved for use in California [*i.e.*, Lasso® Herbicide, INTRRO (EPA Reg. No.: 524-314) and Micro-Tech® Herbicide (EPA Reg. No.: 524-344)], which contain only the one active ingredient, *i.e*., alachlor, are assessed here. All of the alachlor end-use products are labeled as Restricted Use Pesticides (RUPs).

Table 2.6. Summary of Alachlor Products Registered in the U.S.

Product Name (EPA Reg. No.)	Registrant	Percent Active Ingredient	Form	Use(s)	Assessed in this Assessment?
Lasso® 94% Stabilized Technical (524-316)	Monsanto Corporation	94 (alachlor)	Technical	Used to make end-use products	**No** - the technical grade chemical is only labeled for use in producing end use products.
Lariat® Herbicide (524-329)	Monsanto Corporation	- 27.2 (alachlor) - 16 (atrazine) - 0.3 (atrazine-related compounds)	Flowable concentrate	- Corn (all types) - Sorghum (milo)	**No** – the label contains a statement that the product is not approved for use in California
Bullet® Herbicide (524-418)	Monsanto Corporation	- 25.4 (alachlor) - 14.5 (atrazine) - 0.8 (atrazine-related compounds)	Emulsifiable concentrate	- Corn (all types) - Sorghum (milo)	**No** – the label contains a statement that the product is not approved for use in California
Lasso® Herbicide, INTRRO (524-314)	Monsanto Corporation	45.1 (alachlor)	Emulsifiable concentrate	- Corn - Corn (sweet) - Sorghum (milo) - Soybeans - Dry beans - Lima beans - Woody ornamentals - Peanuts	**Yes**
Micro-Tech® Herbicide (524-344)	Monsanto Corporation	41.5 (alachlor)	Micro-encapsulated	- Corn - Corn (sweet) - Sorghum (milo) - Soybeans - Dry beans - Lima beans - Woody ornamentals - Peanuts - Cotton - Sunflowers	**Yes**

Alachlor can be used nationally in areas where corn, sorghum, soybeans, dry beans, lima beans, woody ornamentals (junipers and yews), peanuts, cotton, and sunflowers are grown. The two labels assessed here (Lasso® and Micro-Tech®) prohibit the use of alachlor on beans in Kern County, California. The labels also prohibit aerial applications of alachlor in California, therefore, only ground applications will be assessed here. The application rates and application methods are the same for the two products assessed with the exception that cotton and sunflower use are allowed on the Micro-Tech® but not the Lasso® label (see **Table 2.7**).

Both the maximum yearly and maximum single application rates for alachlor are 4 lb a.i./acre (corn, peanuts, sorghum, woody ornamentals, and sunflowers). For peanuts, the Lasso® label allows a maximum application rate of 4 lb a.i./acre/year, whereas the Micro-Tech® label allows a maximum rate of 3 lb a.i./acre/year (therefore, for peanuts, the 4 lb a.i./acre rate will be modeled to represent the maximum application rate). The remaining uses have a maximum yearly and single application rate of 3 lb a.i./acre (soybeans and beans) or 2 lb a.i./acre (cotton). Only a single application per year is allowed on sweet corn, soybeans, beans, cotton and sunflowers, while two applications, not to exceed the yearly maximum application rate, are allowed on corn, peanuts, sorghum, and woody ornamentals (a minimum reapplication interval is not specified on the current labels). In California, applications are limited to flowable applications via ground equipment (broadcast boom or banded) or via center pivot irrigation systems. Additionally, applications via impregnated bulk fertilizer are allowed for some uses (corn, sorghum, and soybeans). Application timing includes burndown prior to crop, preplant incorporated, pre-emergence surface, post-emergence surface (corn only), ground-crack surface (peanuts only), and post-transplant (woody ornamentals only).

The Micro-Tech® label contains restrictions for applications via ground and irrigation equipment. For ground applications, applications are restricted to a maximum 4-ft boom height, a minimum ASAE droplet size distribution of medium – coarse, and a maximum wind speed of 10 mph during application. Application via chemigation is limited to center pivots with a maximum wind speed of 10 mph during application. For dry bulk fertilizer applications, 200 - 450 lb of dry bulk fertilizer per acre is stipulated (with a maximum application rate of 3 to 4 lb a.i./acre, depending on the use, see Table 2.7). Wind speeds and droplet size distributions are not stipulated on the Lasso® label. Both labels also contain the following restriction:

"Do not apply to highly permeable soils (as classified by the USDA Natural Resources Conservation Service) where the depth to ground water is 30 feet or less"

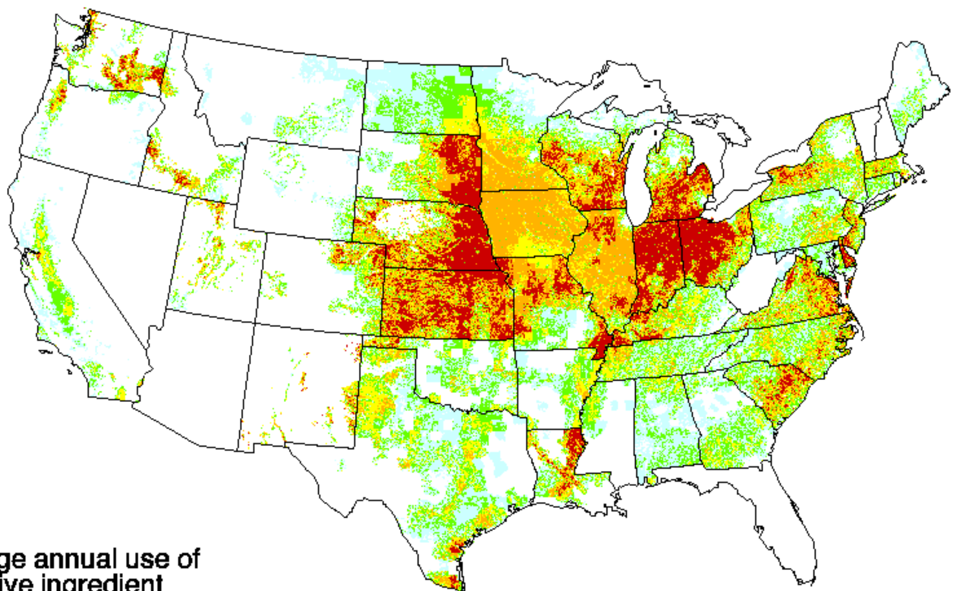
Although this restriction applies to the entire United States including California, it is not expected to impact this assessment because ground water in California tends to be deeper than 30 feet. Alachlor application methods and rates used in this assessment are summarized in **Table 2.7**.

Table 2.7. Summary of Alachlor Application Methods and Rates for California.

Uses	Max Appl. Rate per Year (lbs a.i./acre)	Max Single Appl. Rate (lbs a.i./acre)	Max No. Appl. Per Year	Appl. Methods**	Appl. Timing
Corn	4	4 (no more than 2 can be applied post-mergence)	2	- Ground - Center pivot irrigation - Bulk fertilizer impregnation	- Burndown prior to crop - Preplant incorporated (no deeper than 4 inches into the soil) - Preemergence surface - Postemergence surface
Sweet corn	4	4	1	- Ground - Center pivot irrigation - Bulk fertilizer impregnation	- Preplant incorporated (no deeper than 4 inches into the soil) - Preemergence surface
Grain sorghum (milo)	4	4	2	- Ground - Center pivot irrigation - Bulk fertilizer impregnation	- Burndown prior to crop - Preplant incorporated (no deeper than 4 inches into the soil) - Preemergence surface
Peanuts	4	4	2	- Ground - Center pivot irrigation	- Preplant incorporated (no deeper than 4 inches into the soil) - Preemergence surface - Ground-crack surface
Soybeans	3	3	1	- Ground - Center pivot irrigation - Bulk fertilizer impregnation	- Burndown prior to crop - Preplant incorporated (no deeper than 4 inches into the soil) - Preemergence surface
Dry beans**	3	3	1	- Ground	- Preplant incorporated (no deeper than 4 inches into the soil)
Lima beans (green)**	3	3	1	- Ground	- Preplant incorporated (no deeper than 4 inches into the soil)
Woody ornamentals (Junipers and yews)***	4	4	2 (within 21 days)	- Ground	-Post-transplant
Cotton	2	2	1	- Ground	- Preplant incorporated (into upper 1 – 2 inches of soil) - Preemergence surface
Sunflowers	4	4	1	- Ground	- Preplant incorporated (into upper 1 – 2 inches of soil) - Preemergence surface

* Aerial applications are not allowed in CA.
**Not allowed for use on dry beans in Kern County, CA.
*** Nothing on the label restricts the use of alachlor to commercial uses; therefore, it is assumed that alachlor could be used on woody ornamentals in both commercial and residential settings.
The bolded uses are registered on the Micro-Tech Herbicide label (524-344), but not the Lasso® Herbicide label (524-314); the remaining uses appear on both labels.

According to the United States Geological Survey's (USGS) national pesticide usage data (based on information from 1999 to 2004), an average of 6,221,431 lbs of alachlor per year are applied nationally in the U.S. (see **Fig. 2.1**). Most of the usage (~60%) is for corn, followed by soybeans (~20%), sorghum (~16%), sweet corn (~3%), dry beans (~2%), and peanuts (<1%). The highest usage, geographically, is in the corn-growing regions of the Midwestern U.S.



Crops	Total pounds applied	Percent national use
corn	3687504	59.27
soybeans	1241413	19.95
sorghum	1023185	16.45
sweet corn	157974	2.54
dry beans	107155	1.72
peanuts	4200	0.07

Figure 2.1. Estimated Annual Alachlor Usage in the U.S.
(from http://water.usgs.gov/nawqa/pnsp/usage/maps/show_map.php?year=02&map=m8009) [The pesticide use maps available from this site show the average annual pesticide use intensity expressed as average weight (in pounds) of a pesticide applied to each square mile of agricultural land in a county. The area of each map is based on state-level estimates of pesticide use rates for individual crops that were compiled by the CropLife Foundation, Crop Protection Research Institute based on information collected during 1999 through 2004 and on 2002 Census of Agriculture county crop acreage. The maps do not represent a specific year, but rather show typical use patterns over the five year period 1999 through 2004.]

The Agency's Biological and Economic Analysis Division (BEAD) provides an analysis of both national- and county-level usage information (USEPA, 2009) using state-level usage data obtained from U.S. Department of Agriculture's National Agricultrual Statistics Service (USDA-

NASS[1]), Doane (www.doane.com; the full dataset is not provided due to its proprietary nature) and the CDPR PUR database[2]. CDPR PUR is considered a more comprehensive source of usage data than USDA-NASS or Doane's proprietary database, and, thus, the usage data reported for alachlor by county in this California-specific assessment were generated using CDPR PUR data. Eight years (1999-2006) of usage data were included in this analysis. Data from CDPR PUR were obtained for pesticide applications made on use sites at the section level (approximately one square mile) of the public land survey system. BEAD summarized these data to the county level by site, pesticide, and unit treated. Calculating county-level usage involved summarizing across all applications made within a section and then across all sections within a county for each use site and for each pesticide. The county-level usage data that were calculated include: average annual pounds applied, average annual area treated, and average and maximum application rate across all five years. The units of area treated are also provided where available.

The CDPR PUR data indicate that from 1999 to 2006, an average of 26,060 lbs of alachlor were applied to an average of 10,315 acres per year in CA. This results in an average application rate of 2.5 lb a.i./acre/year (26,060 lbs/10,315 acre). Almost all of the use of alachlor between 1999 and 2006 in CA was on corn (55%) and beans (45%). The remaining alachlor uses listed in the CDPR PUR data made up <1% of alachlor use (*i.e*., 'preplant', 'landscape', and 'research', as listed in the CDPR PUR data).

From 1999 to 2006, alachlor was reportedly used in 24 CA counties (listed in alphabetical order): Alameda, Butte, Fresno, Glenn, Kern, Los Angeles, Madera, Merced, Monterey, Orange, Riverside, Sacramento, San Benito, San Diego, San Joaquin, San Luis Obispo, Santa Barbara, Santa Clara, Solano, Stanislaus, Sutter, Tulare, Ventura, and Yolo (see **Fig. 2.3**). Based on the CA usage data, alachlor use has declined over the past several years. For example, in 1999 a total of 29,327 lbs of alachlor was applied in CA, whereas a total of 13,734 lbs was applied in 2006 (see **Appendix B**).

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[1] United States Depart of Agriculture (USDA), National Agricultural Statistics Service (NASS) Chemical Use Reports provide summary pesticide usage statistics for select agricultural use sites by chemical, crop and state. See http://www.usda.gov/nass/pubs/estindx1.htm#agchem.

[2] The California Department of Pesticide Regulation's Pesticide Use Reporting database provides a census of pesticide applications in the state. See http://www.cdpr.ca.gov/docs/pur/purmain.htm.

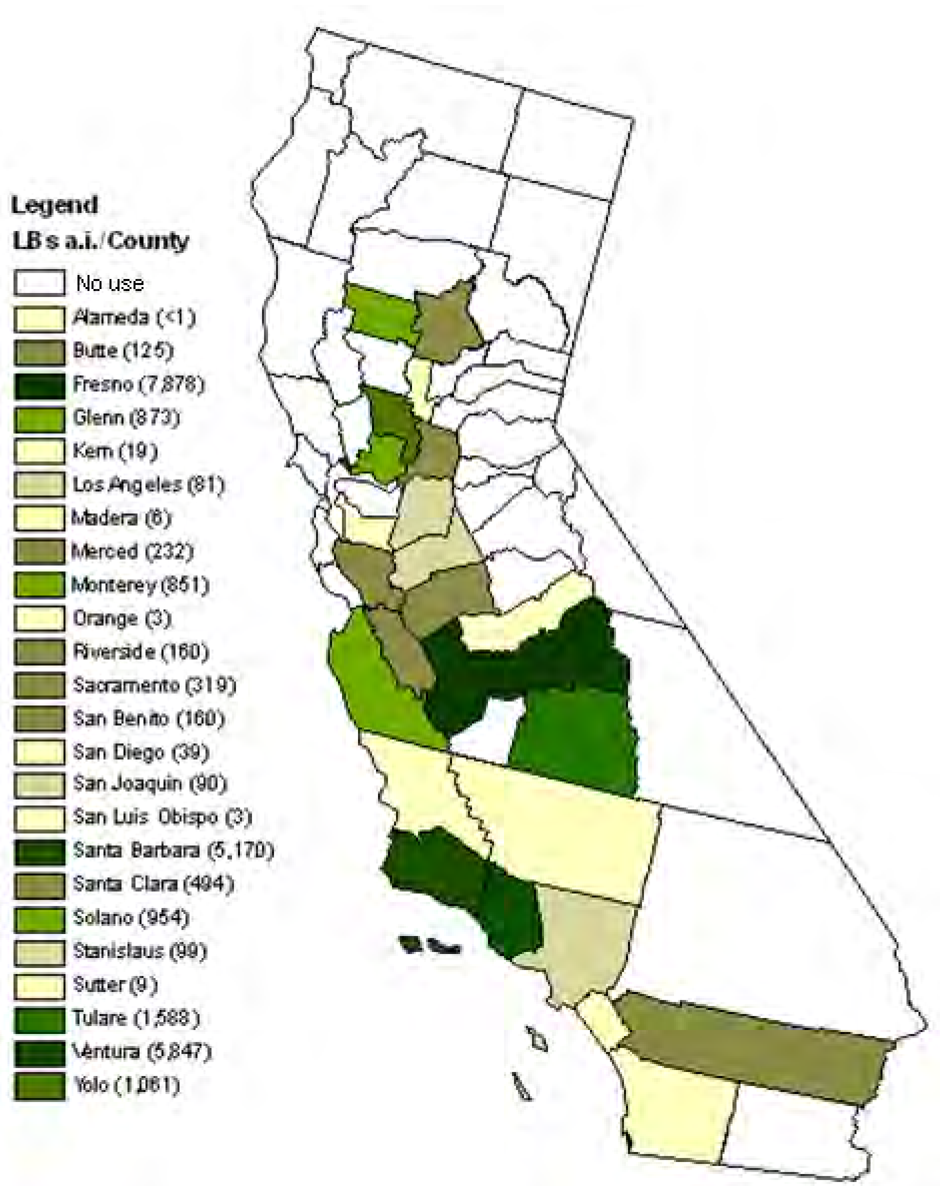

Figure 2.2. Average Pounds of Alachlor Applied/Year/CA County from 1999-2006.

Considering each CA county where alachlor was used, the average application rate per county/year from 1999 to 2006 ranged from <1 to 3.83 lb a.i./acre. The average 95^{th}% and 99^{th}% application rate and the maximum reported application rate per county/year ranged from <1 to 3.99 lb a.i./acre [some counties reported 99^{th}% and maximum application rates higher than the registered use rates (*i.e.*, >4 lb a.i./acre); however, these values are considered misreports or misuses and were not considered in summary calculations] (see **Table 2.8**). These data indicate that, in several CA counties, at least some alachlor users are using the chemical at or near maximum registered application rates.

TABLE 2.8 Summary of California Department of Pesticide Registration (CDPR) Pesticide Use Reporting (PUR) Data from 1999 to 2006 for the Currently Registered Alachlor Uses.

Average Pounds Applied/Year (for All Counties)	County	Avg Annual Area Treated (Acres)	Avg App Rate (lb a.i./Acre per appl.)	Avg 95th% App Rate (lb a.i./Acre per appl.)	Avg 99th% App Rate (lb a.i./Acre per appl.)	Avg Max App Rate (lb a.i./Acre per appl.)
26,060	Alameda	1	0	--	--	--
	Butte	44	2.54	2.99	2.99	2.99
	Fresno	3,283	2.5	3.0	3.99	7.48*
	Glenn	267	3.78	3.99	29.6*	29.6*
	Kern	8	2.85	2.99	2.99	2.99
	Los Angeles	41	1.99	1.99	1.99	1.99
	Madera	3	2.49	2.49	2.49	2.49
	Merced	91	2.51	2.99	2.99	2.99
	Monterey	465	1.87	2.24	2.99	2.99
	Orange	1	2.39	3.99	3.99	3.99
	Riverside	139	1.94	2.99	4.49*	4.49*
	Sacramento	104	3.1	3.49	3.49	3.49
	San Benito	192	0.83	1.0	1.5	3.74
	San Diego	24	0.96	1.99	1.99	1.99
	San Joaquin	24	3.83	3.99	3.99	3.99
	San Luis Obispo	1	2.99	2.99	2.99	2.99
	Santa Barbara	1,892	2.77	3.49	4.79*	29.91*
	Santa Clara	338	1.67	2.49	4.11*	74.97*
	Solano	302	3.44	3.99	4.10*	4.10*
	Stanislaus	35	2.63	2.99	2.99	2.99
	Sutter	4	1.99	1.99	1.99	1.99
	Tulare	656	2.52	2.99	3.0	3.0
	Ventura	2,116	2.98	3.03	3.96	36.29*
	Yolo	378	2.80	3.99	4.70*	4.70*

* These rates are higher than 4 lb a.i./acre (the max registered application rate); therefore, they are considered misreports or misuses, and not included in summary calculations.

2.5 Assessed Species

Table 2.9 provides a summary of the current distribution, habitat requirements, and life history parameters for the listed species being assessed. More detailed life-history and distribution information can be found in Attachments 1 and 3. See **Figures 2.3** and **2.4** for a map of the current range and designated critical habitat of the assessed listed species. Occurrence data at the section level for the delta smelt is based on information provided in the case, Center for Biological Diversity (CBD) vs. EPA *et al*. (Case No. 07-2794-JCS).

Table 2.9. Summary of Current Distribution, Habitat Requirements, and Life History Information for the Assessed Listed Species[1].

Assessed Species	Size	Current Range	Habitat Type	Designated Critical Habitat?	Reproductive Cycle	Diet
California red-legged frog (*Rana aurora draytonii*)	Adult (85-138 cm in length), Females – 9-238 g, Males – 13-163 g; Juveniles (40-84 cm in length)	Northern CA coast, northern Transverse Ranges, foothills of Sierra Nevada, and in southern CA south of Santa Barbara	Freshwater perennial or near-perennial aquatic habitat with dense vegetation; artificial impoundments; riparian and upland areas	Yes	Breeding: Nov. to Apr. Tadpoles: Dec. to Mar. Young juveniles: Mar. to Sept.	Aquatic-phase[2]: algae (tadpoles only), freshwater aquatic invertebrates and fish Terrestrial-phase: terrestrial invertebrates, small mammals, and frogs
Delta smelt (*Hypomesus transpacificus*)	Up to 120 mm in length	Suisun Bay and the Sacramento-San Joaquin estuary (known as the Delta) near San Francisco Bay, CA	The species is adapted to living in fresh and brackish water. They typically occupy estuarine areas with salinities below 2 parts per thousand (although they have been found in areas up to 18parts per thousand). They live along the freshwater edge of the mixing zone (saltwater-freshwater interface).	Yes	Spawns in fresh or slightly brackish water upstream of the mixing zone. Spawning season usually takes place from late March through mid-May, although it may occur from late winter (Dec.) to mid-summer (July-August). Eggs hatch in 9 – 14 days.	Adults forage primarily on planktonic copepods, cladocerans, amphipods, and insect larvae. Larvae feed on phytoplankton; juveniles feed on zooplankton.

[1] For more detailed information on the distribution, habitat requirements, and life history information of the assessed listed species, see Attachment 3

[2] For the purposes of this assessment, tadpoles and submerged adult frogs are considered "aquatic" because exposure pathways in the water are considerably different than those that occur on land.

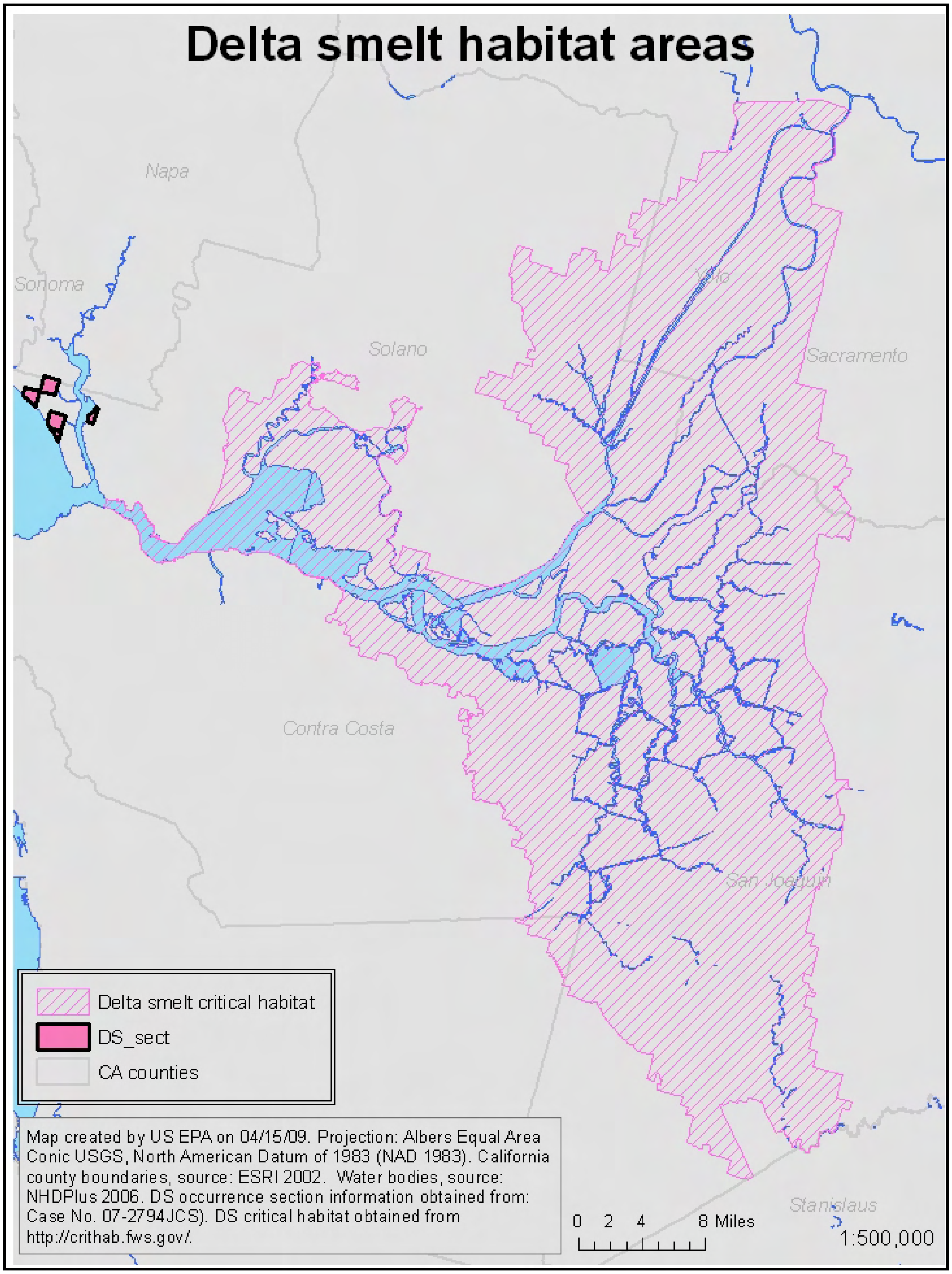

Figure 2.3. Delta Smelt Habitat Areas.

Recovery Units

1. Sierra Nevada Foothills and Central Valley
2. North Coast Range Foothills nd Western Sacramento River Valley
3. North Coast and North San Francisco Bay
4. South and East San Francisc Bay
5. Central Coast
6. Diablo Range and Salinas Valley
7. Northern Transverse Rang nd Tehachapi Mountains
8. Southern Transverse and Peninsular Ranges

Core Areas

1. Feather River
2. Yuba River- S. Fork Feather River
3. Traverse Creek/ Middle Fork/ American R. Rubicon
4. Cosumnes River
5. South Fork Calaveras River*
6. Tuolumne River*
7. Piney Creek*
8. Cottonwood Creek
9. Putah Creek – Cache Creek*
10. Lake Berryessa Tributaries
11. Upper Sonoma Creek
12. Petaluma Creek – Sonoma Creek
13. Pt. Reyes Peninsula
14. Belvedere Lagoon
15. Jameson Canyon – Lower Napa River
16. East San Francisco Bay
17. Santa Clara Valley
18. South San Francisco Bay
19. Watsonville Slough-Elkhorn Slough
20. Carmel River – Santa Lucia
21. Gablan Range
22. Estero Bay
23. Arroyo Grange River
24. Santa Maria River – Santa Ynez River
25. Sisquoc River
26. Ventura River – Santa Clara River
27. Santa Monica Bay – Venura Coastal Streams
28. Estrella River
29. San Gabriel Mountain*
30. Forks of the Mojave*
31. Santa Ana Mountain*
32. Santa Rosa Plateau
33. San Luis Ray*
34. Sweetwater*
35. Laguna Mountain*

* Core areas that were historically occupied by the California red-legged frog are not included in the map

Figure 2.4. Recovery Unit, Core Area, Critical Habitat, and Occurrence Designations for CRLF.

2.6. Designated Critical Habitat

Critical habitat has been designated for the CRLF and the DS. ‘Critical habitat’ is defined in the ESA as the geographic area occupied by the species at the time of the listing where the physical and biological features necessary for the conservation of the species exist, and there is a need for special management to protect the listed species. It may also include areas outside the occupied area at the time of listing if such areas are ‘essential to the conservation of the species.’ Critical habitat receives protection under Section 7 of the ESA through prohibition against destruction or modification with regard to actions carried out, funded, or authorized by a federal Agency. Section 7 requires consultation on federal actions that are likely to result in the destruction or modification of critical habitat.

To be included in a critical habitat designation, the habitat must be ‘essential to the conservation of the species.’ Critical habitat designations identify, to the extent known using the best scientific and commercial data available, habitat areas that provide essential life cycle needs of the species or areas that contain certain primary constituent elements (PCEs) (as defined in 50 CFR 414.12(b)). PCEs include, but are not limited to, space for individual and population growth and for normal behavior; food, water, air, light, minerals, or other nutritional or physiological requirements; cover or shelter; sites for breeding, reproduction, rearing (or development) of offspring; and habitats that are protected from disturbance or are representative of the historic geographical and ecological distributions of a species. **Table 2.10** describes the PCEs for the critical habitats designated for the CRLF and the DS.

Table 2.10. Designated Critical Habitat PCEs for the CRLF and DS.

Species	PCEs	Reference
CRLF	Alteration of channel/pond morphology or geometry and/or increase in sediment deposition within the stream channel or pond.	50 CFR 414.12(b), 2006
	Alteration in water chemistry/quality including temperature, turbidity, and oxygen content necessary for normal growth and viability of juvenile and adult CRLFs and their food source.	
	Alteration of other chemical characteristics necessary for normal growth and viability of CRLFs and their food source.	
	Reduction and/or modification of aquatic-based food sources for pre-metamorphs (*e.g.*, algae)	
	Elimination and/or disturbance of upland habitat; ability of habitat to support food source of CRLFs: Upland areas within 200 ft of the edge of the riparian vegetation or dripline surrounding aquatic and riparian habitat that are comprised of grasslands, woodlands, and/or wetland/riparian plant species that provides the CRLF shelter, forage, and predator avoidance	
	Elimination and/or disturbance of dispersal habitat: Upland or riparian dispersal habitat within designated units and between occupied locations within 0.7 mi of each other that allow for movement between sites including both natural and altered sites which do not contain barriers to dispersal	
	Reduction and/or modification of food sources for terrestrial phase juveniles and adults	
	Alteration of chemical characteristics necessary for normal growth and viability of juvenile and adult CRLFs and their food source.	
DS	Spawning Habitat—shallow, fresh or slightly brackish backwater sloughs and edgewaters to ensure egg hatching and larval viability. Spawning areas also must provide suitable water quality (*i.e*., low "concentrations of pollutants) and substrates for egg attachment (*e.g*., submerged tree roots and branches and emergent vegetation).	59 FR 65256 65279, 1994
	Larval and Juvenile Transport—Sacramento and San Joaquin Rivers and their tributary channels must be protected from physical disturbance and flow disruption. Adequate river flow is necessary to transport larvae from upstream spawning areas to rearing habitat in Suisun Bay. Suitable water quality must be provided so that maturation is not impaired by pollutant concentrations.	
	Rearing Habitat—Maintenance of the 2 ppt isohaline and suitable water quality (low concentrations of pollutants) within the Estuary is necessary to provide delta smelt larvae and juveniles a shallow protective, food-rich environment in which to mature to adulthood.	
	Adult Migration— Unrestricted access to suitable spawning habitat in a period that may extend from December to July. Adequate flow and suitable water quality may need to be maintained to attract migrating adults in the Sacramento and San Joaquin River channels and their associated tributaries. These areas also should be protected from physical disturbance and flow disruption during migratory periods.	

[1] These PCEs are in addition to more general requirements for habitat areas that provide essential life cycle needs of the species such as, space for individual and population growth and for normal behavior; food, water, air, light, minerals, or other nutritional or physiological requirements; cover or shelter; sites for breeding, reproduction, rearing (or development) of offspring; and habitats that are protected from disturbance or are representative of the historic geographical and ecological distributions of a species.
[2] PCEs that are abiotic, including, physico-chemical water quality parameters such as salinity, pH, and hardness are not evaluated because these processes are not biologically mediated and, therefore, are not relevant to the endpoints included in this assessment.

More detail on the designated critical habitat applicable to this assessment can be found in **Attachment 1** (for the CRLF) and **Attachment 3** (for the DS). Activities that may destroy or modify critical habitat are those that alter the PCEs and jeopardize the continued existence of the species. Evaluation of actions related to use of alachlor that may affect the PCEs of the designated critical habitat for the CRLF and DS form the basis of the critical habitat impact analysis.

As previously discussed, the Agency believes that the analysis of direct and indirect effects to listed species provides the basis for an analysis of potential effects on the designated critical habitat. Because alachlor is expected to directly impact living organisms within the action area, critical habitat analysis for alachlor is limited in a practical sense to those PCEs of critical habitat that are biological or that can be reasonably linked to biologically mediated processes.

2.7 Action Area

Deriving the geographical extent of the California portion of the action area is based on consideration of the types of effects that alachlor may be expected to have on the environment, the exposure levels to alachlor that are associated with those effects, and the best available information concerning the use of alachlor and its fate and transport within the state of California. Specific measures of ecological effect that define the action area include any direct and indirect toxic effect, including reduction in survival, growth, and fecundity as well as the full suite of sublethal effects available in the effects literature. Therefore, the action area extends to a point where environmental exposures are below any measured lethal or sublethal effect threshold for any biological entity at the whole organism, organ, tissue, and cellular level of organization. In situations where it is not possible to determine the threshold for an observed effect, the action area is not spatially limited and is assumed to be the entire state of California. The registered agricultural and non-agricultural uses relative to potential land cover classes from the National Land Cover Data (NLCD), which represent the current and possible future extent of the use sites, represent the initial area of concern and are illustrated in **Figure 2.5**.

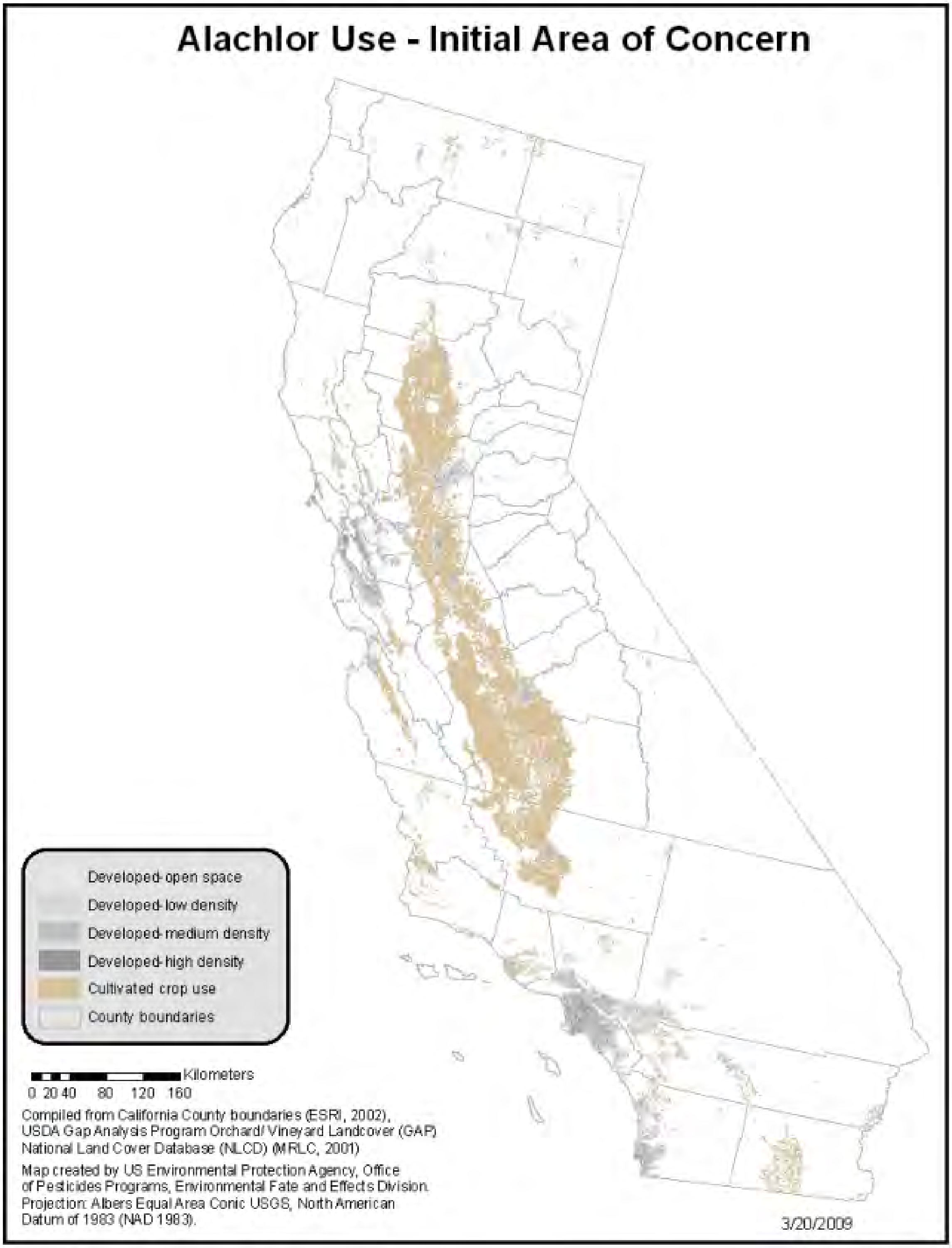

Figure 2.5. Potential Alachlor Use Sites in California Representing the Initial Area of Concern.

A number of alachlor studies have been conducted that have identified some type of biological effect (see **Appendices C, D**, and **E**). Some studies have identified only exposure levels associated with an effect without a corresponding no effect level. For example, the most sensitive available toxicity endpoint for chronic exposure in estuarine/marine invertebrates is a NOAEC of <0.0001 for a copepod (*Tigriopus japonicus*) based on an increase in the generation time for adults in the F_0 and F_1 generations at all of the concentrations tested (Lee *et al*. 2008, E104287). Additionally, the most sensitive available NOAEC for birds based on chronic exposure is <50 mg a.i./kg-diet for mallard ducks (*Anas platyrhynchos*) based on significant treatment-related reductions in hatchling weight at all concentrations tested (MRID 449515-01). Therefore, a threshold for some type of environmental effect has not been identified, and it is not possible to identify an alachlor exposure level that is definitively associated with no environmental effects regardless of the ecological significance of the effect. For this reason, the action area (area where an effect may occur) has been conservatively defined as the entire state of California.

2.8 Assessment Endpoints and Measures of Ecological Effect

Assessment endpoints are defined as "explicit expressions of the actual environmental value that is to be protected."[3] Selection of the assessment endpoints is based on valued entities (*e.g.,* CRLF and DS), organisms important in the life cycle of the valued entities (*i.e*., the assessed species, and the PCEs of their designated critical habitat), the ecosystems potentially at risk (*e.g.,* waterbodies, riparian vegetation, and upland and dispersal habitats), the migration pathways of alachlor (*e.g.,* runoff, spray drift, *etc*.), and the routes by which ecological receptors are exposed to alachlor *(e.g.,* direct contact, *etc.).*

2.8.1. Assessment Endpoints

Assessment endpoints for the CRLF and the DS include direct toxic effects on the survival, reproduction, and growth of individuals, as well as indirect effects, such as reduction of the prey base or effects to its habitat. In addition, potential effects to critical habitat is assessed by evaluating potential effects to PCEs, which are components of the habitat areas that provide essential life cycle needs of the assessed species. Each assessment endpoint requires one or more "measures of ecological effect," defined as changes in the attributes of an assessment endpoint or changes in a surrogate entity or attribute in response to exposure to a pesticide. Specific measures of ecological effect are generally evaluated based on acute and chronic toxicity information from registrant-submitted guideline tests that are performed on a limited number of organisms. Additional ecological effects data from the open literature are also considered. It should be noted that assessment endpoints are limited to direct and indirect effects associated with survival, growth, and fecundity, and do not include the full suite of sublethal effects used to define the action area. According to the Overview Document (USEPA 2004), the Agency relies on acute and chronic effects endpoints that are either direct measures of impairment of survival, growth, or fecundity or endpoints for which there is a scientifically robust, peer reviewed relationship that can quantify the impact of the measured effect endpoint on the assessment endpoints of survival, growth, and fecundity.

[3] From U.S. EPA (1992). *Framework for Ecological Risk Assessment.* EPA/630/R-92/001.

A complete discussion of all the toxicity data available for this risk assessment, including resulting measures of ecological effect selected for each taxonomic group of concern, is included in Section 4 of this document. A summary of the assessment endpoints and measures of ecological effect selected to characterize potential assessed direct and indirect risks for each of the assessed species associated with exposure to alachlor is provided in **Table 2.12**.

As described in the Agency's Overview Document (USEPA, 2004), the most sensitive endpoint for each taxonomic group is used for risk estimation. For this assessment, evaluated taxa include aquatic-phase amphibians, freshwater and saltwater fish, freshwater and saltwater invertebrates, aquatic plants, birds (surrogate for terrestrial-phase amphibians), mammals, terrestrial invertebrates, and terrestrial plants. Acute (short-term) and chronic (long-term) toxicity information is characterized based on registrant-submitted studies and a comprehensive review of the open literature on alachlor.

Table 2.11 identifies the taxa used to assess the potential for direct and indirect effects from the uses of alachlor for each listed species assessed. The specific assessment endpoints used to assess the potential for direct and indirect effects to each listed species are provided in **Table 2.8**.

Table 2.11. Taxa Used in the Analyses of Direct and Indirect Effects for the Assessed Listed Species.

Listed Species	Birds / Terr. Amphibian	Mammals	Terr. Plants	Terr. Inverts.	FW Fish / Amphibian	FW Inverts.	Estuarine /Marine Fish	Estuarine /Marine Inverts.	Aquatic Plants
California red-legged frog	Direct Indirect (prey)	Indirect (prey)	Indirect (habitat)	Indirect (prey)	Direct Indirect (prey)	Indirect (prey)	N/A	N/A	Indirect (food/ habitat)
Delta smelt	N/A	N/A	Indirect (habitat)	N/A	Direct[1]	Indirect (prey)	Direct	Indirect (prey)	Indirect (food/ habitat)

N/A = Not applicable; Terr. = Terrestrial; Invert. = Invertebrate; FW = Freshwater

[1] The most sensitive species across freshwater and saltwater environments was used for the DS.

Table 2.12. Assessment Endpoints Used to Evaluate the Potential for the Use of Alachlor to Result in Direct and Indirect Effects to the CRLF and the DS.

Taxa Used to Assess Direct and/or Indirect Effects to Assessed Species[1]	Assessed Listed Species	Assessment Endpoints	Measures of Ecological Effects
1. Freshwater Fish and Aquatic-phase Amphibians	Direct Effect – -Aquatic-phase CRLF -DS	Survival, growth, and reproduction of individuals via direct effects	1a. Amphibian acute LC_{50} (ECOTOX) or most sensitive fish acute LC_{50} (guideline or ECOTOX) if no suitable amphibian data are available 1b. Amphibian chronic NOAEC (ECOTOX) or most sensitive fish chronic NOAEC (guideline or ECOTOX) 1c. Amphibian early-life stage data (ECOTOX) or most sensitive fish early-life stage NOAEC (guideline or ECOTOX)
	Indirect Effect (prey) -Aquatic-phase CRLF	Survival, growth, and reproduction of individuals via indirect effects on aquatic prey food supply (*i.e.*, fish and aquatic-phase amphibians)	
2. Freshwater Invertebrates	Indirect Effect (prey) -Aquatic-phase CRLF -DS	Survival, growth, and reproduction of individuals via indirect effects on aquatic prey food supply (*i.e.*, freshwater invertebrates)	2a. Most sensitive freshwater invertebrate EC_{50} (guideline or ECOTOX) 2b. Most sensitive freshwater invertebrate chronic NOAEC (guideline or ECOTOX)
3. Estuarine/Marine Fish	Direct Effect -DS	Survival, growth, and reproduction of individuals via direct effects on the DS	3a. Most sensitive estuarine/marine fish LC_{50} (guideline or ECOTOX) 3b. Most sensitive estuarine/marine fish chronic NOAEC (guideline or ECOTOX)
4. Estuarine/Marine Invertebrates	Indirect Effect (prey) -DS	Survival, growth, and reproduction of individuals via indirect effects on aquatic prey food supply (*i.e.*, estuarine/marine invertebrates)	4a. Most sensitive estuarine/marine invertebrate EC_{50} (guideline or ECOTOX) 4b. Most sensitive estuarine/marine invertebrate chronic NOAEC (guideline or ECOTOX)
5. Aquatic Plants (freshwater/marine)	Indirect Effect (food/habitat) -Aquatic-phase CRLF -DS	Survival, growth, and reproduction of individuals via indirect effects on habitat, cover, food supply, and/or primary productivity (*i.e.*, aquatic plant community)	5a. Vascular plant acute EC_{50} (duckweed guideline test or ECOTOX vascular plant) 5b. Non-vascular plant acute EC_{50} (freshwater algae or diatom, or ECOTOX non-vascular)
6. Birds	Direct Effect -Terrestrial-phase CRLF	Survival, growth, and reproduction of individuals via direct effects	6a. Most sensitive bird[b] or terrestrial-phase amphibian acute LC_{50} or LD_{50} (guideline or ECOTOX) 6b. Most sensitive bird[b] or terrestrial-phase amphibian chronic NOAEC (guideline or ECOTOX)
	Indirect Effect (prey) -CRLFs	Survival, growth, and reproduction of individuals via indirect effects on terrestrial prey (surrogate for amphibians)	
7. Mammals	Indirect Effect -Terrestrial-phase CRLF	Survival, growth, and reproduction of individuals via indirect effects on terrestrial prey (mammals)	7a. Most sensitive laboratory rat acute LC_{50} or LD_{50} (guideline or ECOTOX) 7b. Most sensitive laboratory rat chronic NOAEC (guideline or ECOTOX)
8. Terrestrial Invertebrates	Indirect Effect (prey) -Terrestrial-phase CRLF	Survival, growth, and reproduction of individuals via indirect effects on	8a. Most sensitive terrestrial invertebrate acute EC_{50} or LC_{50} (guideline or ECOTOX)[c]

Taxa Used to Assess Direct and/or Indirect Effects to Assessed Species[1]	Assessed Listed Species	Assessment Endpoints	Measures of Ecological Effects
		terrestrial prey (terrestrial invertebrates)	8b. Most sensitive terrestrial invertebrate chronic NOAEC (guideline or ECOTOX)
9. Terrestrial Plants	Indirect Effect (food/habitat) (non-obligate relationship) -Terrestrial- and aquatic-phase CRLF -DS	Survival, growth, and reproduction of individuals via indirect effects on food and habitat (*i.e.*, riparian and upland vegetation)	9a. Distribution of EC_{25} for monocots (seedling emergence, vegetative vigor, or ECOTOX 9b. Distribution of EC_{25} for dicots (seedling emergence, vegetative vigor, or ECOTOX)

[1] For the DS both freshwater and estuarine/marine species are considered since the DS can inhabit both freshwater and estuarine/marine environments.

2.8.2 Assessment Endpoints for Designated Critical Habitat

As previously discussed, designated critical habitat is assessed to evaluate actions related to the use of alachlor that may affect the PCEs of the assessed species' designated critical habitat. PCEs for the assessed species were previously described in Section 2.6. Actions that may modify critical habitat are those that alter the PCEs and jeopardize the continued existence of the assessed species. Therefore, these actions are identified as assessment endpoints. Evaluation of PCEs as assessment endpoints is limited to those of a biological nature (*i.e*., the biological resource requirements for the listed species associated with the critical habitat) and those for which alachlor effects data are available.

Assessment endpoints used to evaluate potential for direct and indirect effects are equivalent to the assessment endpoints used to evaluate potential effects to designated critical habitat. If a potential for direct or indirect effects is found, then there is also a potential for effects to critical habitat. Some components of PCEs are associated with physical abiotic features (*e.g*., presence and/or depth of a water body, or distance between two sites), which are not expected to be measurably altered by use of pesticides.

2.9 Conceptual Model

2.9.1 Risk Hypotheses

Risk hypotheses are specific assumptions about potential adverse effects (*i.e.*, changes in assessment endpoints) and may be based on theory and logic, empirical data, mathematical models, or probability models (USEPA, 1998). For this assessment, the risk is stressor-linked, where the stressor is the release of alachlor to the environment. The following risk hypotheses are presumed for this assessment:

The labeled use of alachlor may:

- … directly affect the CRLF and/or the DS by causing mortality or by adversely affecting growth or fecundity;
- … indirectly affect the CRLF and/or the DS and/or affect their designated critical habitat by reducing or changing the composition of the food supply;

- … indirectly affect the CRLF and/or the DS and/or affect their designated critical habitat by reducing or changing the composition of the aquatic plant community in the species' current range, thus, affecting primary productivity and/or cover;
- … indirectly affect the CRLF and/or the DS and/or affect their designated critical habitat by reducing or changing the composition of the terrestrial plant community in the species' current range;
- … indirectly affect the CRLF and/or the DS and/or affect their designated critical habitat by reducing or changing aquatic habitat in their current range (via modification of water quality parameters, habitat morphology, and/or sedimentation).

2.9.2 Diagram

The conceptual model is a graphic representation of the structure of the risk assessment. It specifies the alachlor release mechanisms, biological receptor types, and effects endpoints of potential concern. The conceptual models for aquatic and terrestrial phases of the CRLF and the DS and the conceptual models for the aquatic and terrestrial PCE components of critical habitat are shown in **Figures 2.6** and **2.7**. Although the conceptual models for direct/indirect effects and effects to designated critical habitat PCEs are shown on the same diagrams, the potential for direct/indirect effects and effects to PCEs will be evaluated separately in this assessment. Exposure routes shown in dashed lines are not quantitatively considered because the contribution of those potential exposure routes to potential risks to the CRLF and the DS and effects to designated critical habitat is expected to be negligible.

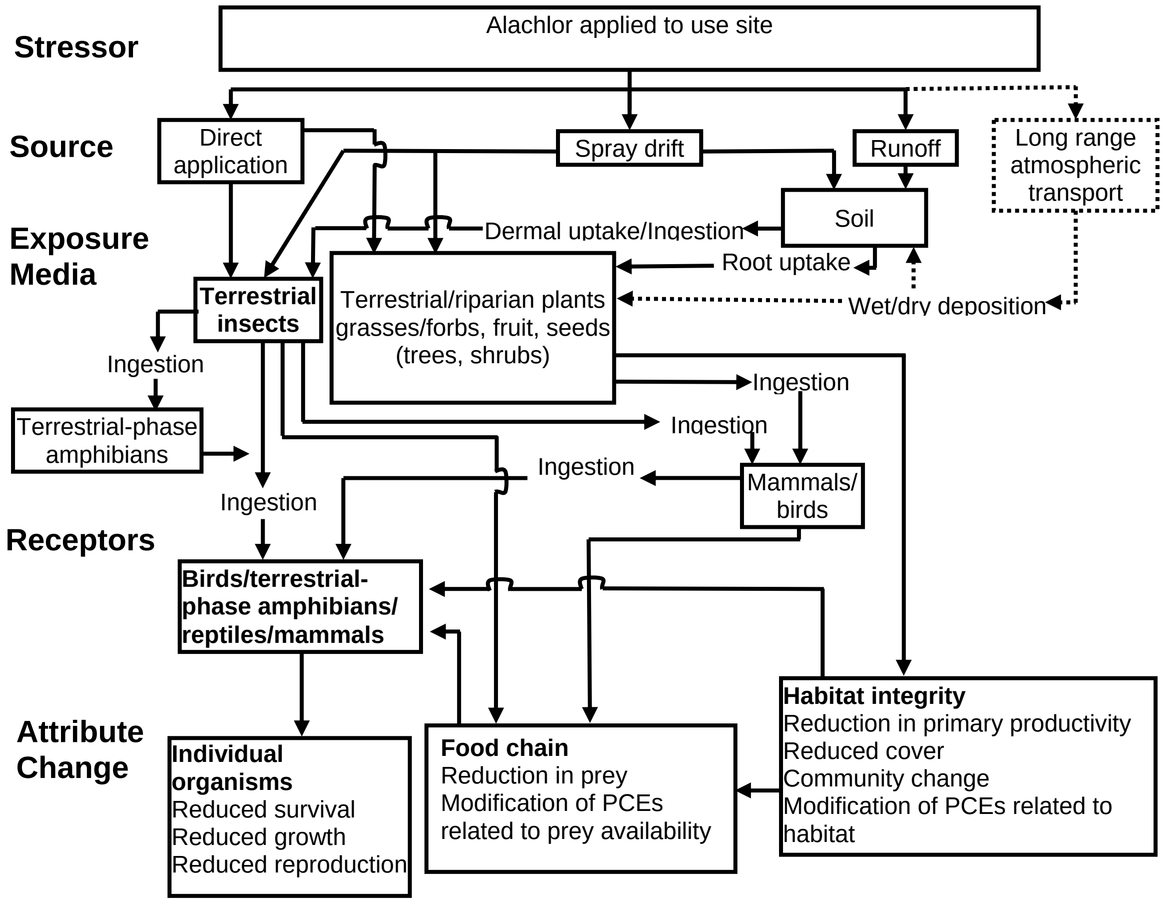

Figure 2.6. Conceptual Model for Risks to Terrestrial-Phase CRLF from Alachlor Use.

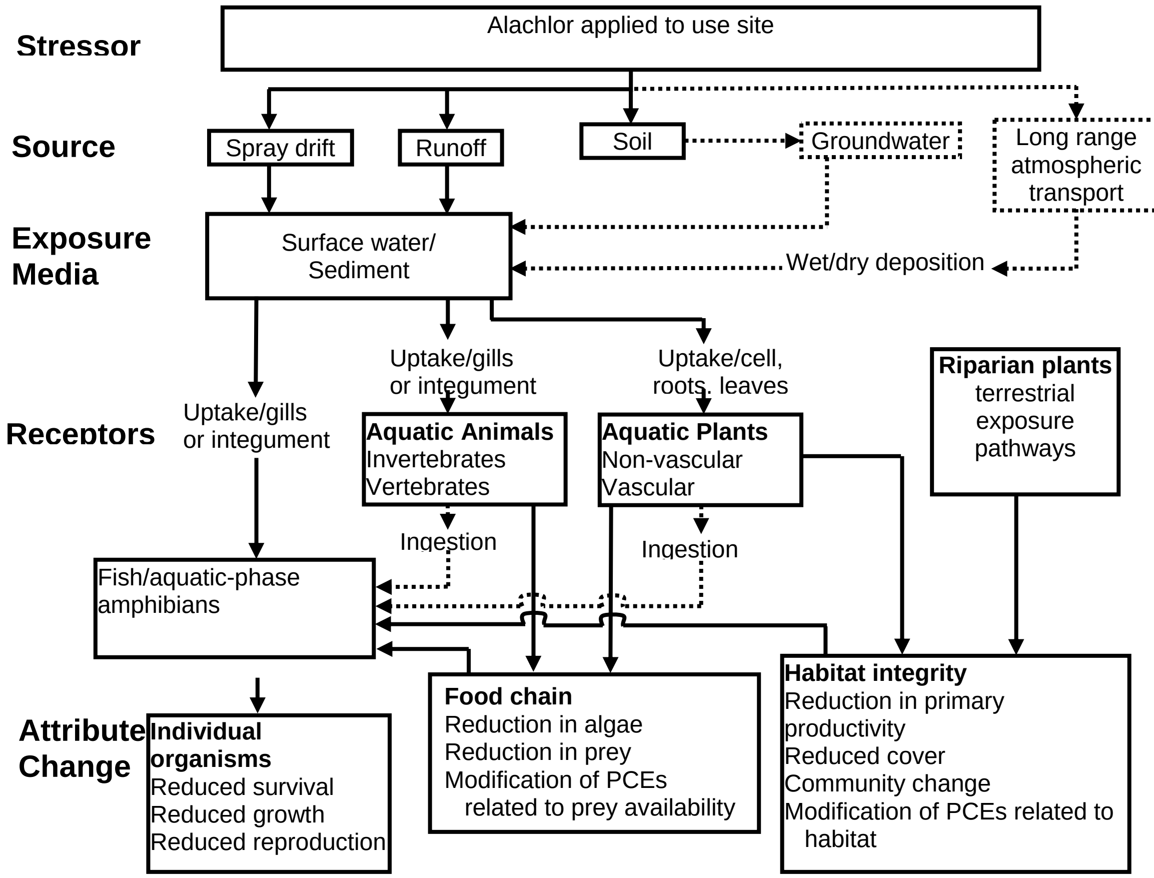

Figure 2.7. Conceptual Model for Risks to Aquatic-Phase CRLF and the DS from Use of Alachlor.

2.10. Analysis Plan

In order to address the risk hypothesis, the potential for direct and indirect effects to the CRLF and the DS, prey items, and habitat is estimated based on a taxon-level approach. In the following sections, the use, environmental fate, and ecological effects of alachlor are characterized and integrated to assess the risks. This is accomplished using a risk quotient (ratio of exposure concentration to effects concentration) approach. Although risk is often defined as the likelihood and magnitude of adverse ecological effects, the risk quotient-based approach does not provide a quantitative estimate of likelihood and/or magnitude of an adverse effect. However, as outlined in the Overview Document (USEPA, 2004), the likelihood of effects to individual organisms from particular uses of alachlor is estimated using the probit dose-response slope and either the level of concern (discussed below) or actual calculated risk quotient value.

2.10.1 Measures of Exposure

Measures of exposure are based on aquatic and terrestrial models that predict estimated environmental concentrations (EECs) of alachlor using maximum labeled application rates and methods of application. The models used to predict aquatic EECs are the Pesticide Root Zone Model coupled with the Exposure Analysis Model System (PRZM/EXAMS). The model used to predict terrestrial EECs on food items is T-REX. The model used to derive EECs relevant to terrestrial and wetland plants is TerrPlant. These models are parameterized using relevant reviewed registrant-submitted environmental fate data.

PRZM (v3.12.2, May 2005) and EXAMS (v2.98.4.6, April 2005) are screening simulation models coupled with the input shell pe5.pl (August, 2007) to generate daily exposures and 1-in-10 year EECs of alachlor that may occur in surface water bodies adjacent to application sites receiving alachlor through runoff and spray drift. PRZM simulates pesticide application, movement and transformation on an agricultural field and the resultant pesticide loadings to a receiving water body via runoff, erosion and spray drift. EXAMS simulates the fate of the pesticide and resulting concentrations in the water body. The standard scenario used for ecological pesticide assessments assumes application to a 10-hectare agricultural field that drains into an adjacent 1-hectare water body, 2-meters deep (20,000 m^3 volume) with no outlet. PRZM/EXAMS was used to estimate screening-level exposure of aquatic organisms to alachlor. The measure of exposure for aquatic species is the 1-in-10 year return peak or rolling mean concentration. The 1-in-10-year 60-day mean is used for assessing chronic exposure to fish; the 1-in-10-year 21-day mean is used for assessing chronic exposure for aquatic invertebrates. Degradates of the parent alachlor were modeled in PRZM/EXAMS using the Total Residues method, where this modeling strategy requires an assumption that all residues of concern have similar physical, chemical, and partitioning characteristics. Application rates for the parent pesticide (alachlor) are used to represent the total mass loading of pesticide and its degradation product(s). Degradation half-lives are calculated based on cumulative residues of concern and parent alachlor (USEPA, 2008).

Exposure estimates for the terrestrial animals assumed to be in the target area or in an area exposed to spray drift are derived using the T-REX model (version 1.4.1, 10/2008). This model incorporates the Kenega nomograph, as modified by Fletcher *et al.* (1994), which is based on a large set of actual field residue data. The upper limit values from the nomograph represented the 95th percentile of residue values from actual field measurements (Hoerger and Kenega, 1972). The model is parameterized considering relevant, reviewed registrant-submitted and open literature fate data. The terrestrial exposure estimates are based on parent alachlor alone.

For the post-emergence (corn only), ground-crack surface (peanuts only), and burndown flowable applications, residues on potential food items (foliage and/or terrestrial invertebrates) on the field of application will be estimated. For the remaining types of flowable applications, estimated residues for terrestrial invertebrates will also be made for the target site of application using T-REX (since invertebrates could be on the field of application during application). However, foliar residues are not expected on the site of application for soil incorporated (pre-plant only) or soil surface (pre-plant and pre-emergence) applications to bare soil. Therefore, for the soil surface applications (pre-plant and pre-emergence) estimated residues on potential herbaceous food items will be bound using estimates from the site of application (to model situations when grasses/weeds might be on the field of application) and areas immediately adjacent to the field of application (to model applications to bare soil).

To estimate the highest potential exposure from foliage (immediately adjacent to the site of application) for the bare-soil surface applications, the spray drift model AgDRIFT will be used to estimate the amount of chemical expected 1 ft off the field of application. The estimated amount of chemical found 1 ft off the site of application (in lb a.i./acre) will then be used as an application rate in T-REX to estimate the foliar residues expected immediately adjacent to the site of application.

For modeling purposes, direct exposures of the CRLF to alachlor through contaminated food are estimated using the EECs for a small bird (20 g) that consumes small insects. Dietary-based and dose-based exposures of potential prey (small mammals) are assessed using the small mammal (15 g) which consumes short grass. The small bird (20g) consuming small insects and the small mammal (15g) consuming short grass are used because these categories result in the largest RQs for the size/dietary categories in T-REX that are appropriate surrogates for the CRLF. Estimated exposures of terrestrial insects to alachlor are bound by using the dietary-based EECs for small insects and large insects.

For the alachlor applications using impregnated dry bulk fertilizer, the bulk fertilizer will be treated as a granular formulation for modeling purposes. Terrestrial exposures from impregnated fertilizer applications will be estimated using T-REX assuming none of the alachlor-impregnated fertilizer is incorporated into the soil. Risk to terrestrial animals from ingesting the fertilizer will be based on LD_{50}/ft^2 values. The LD_{50}/ft^2 values are calculated using an avian toxicity value and the EEC (mg a.i./ft^2) and are directly compared with Agency's levels of concern (LOCs) for risk characterization purposes.

Birds are currently used as surrogates for terrestrial-phase amphibians and reptiles. However, amphibians and reptiles are poikilotherms (body temperature varies with environmental temperature) while birds are homeotherms (temperature is regulated, constant, and largely independent of environmental temperatures). Therefore, amphibians and reptiles tend to have much lower metabolic rates and lower caloric intake requirements than birds or mammals. As a consequence, birds are likely to consume more food than amphibians and reptiles on a daily dietary intake basis, assuming similar caloric content of the food items. Therefore, the use of avian food intake allometric equation as a surrogate to amphibians and reptiles is likely to result in an over-estimation of exposure and risk for reptiles and terrestrial-phase amphibians. For this reason, food intake equations more specific to terrestrial phase amphibians were used to refine the potential dietary exposures to terrestrial phase CRLF. These food intake equations were incorporated into T-REX to form an exposure model called T-HERPS (v. 1.0), which allows for an estimation of food intake for poikilotherms using the same basic procedure as T-REX uses to estimate avian food intake.

EECs for terrestrial plants inhabiting dry and wetland areas are derived using TerrPlant (version 1.2.2, 12/26/2006). This model uses estimates of pesticides in runoff and in spray drift to calculate EECs. EECs are based upon solubility, application rate and minimum incorporation depth.

The spray drift model, AgDRIFT is used to assess exposures of terrestrial animals to alachlor deposited on terrestrial and aquatic habitats by spray drift. In addition to the buffered area from the spray drift analysis, the downstream extent of alachlor that exceeds the LOC for the areas of potential effect is also considered.

2.10.2 Measures of Effect

Data identified in Section 2.8 are used as measures of effect for direct and indirect effects to the CRLF and the DS. Data were obtained from registrant submitted studies or from literature studies identified by ECOTOX. The ECOTOXicology database (ECOTOX) was searched in order to provide more ecological effects data and in an attempt to bridge existing data gaps. ECOTOX is a source for locating single chemical toxicity data for aquatic life, terrestrial plants, and wildlife. ECOTOX was created and is maintained by the USEPA, Office of Research and Development, and the National Health and Environmental Effects Research Laboratory's Mid-Continent Ecology Division.

The assessment of risk for direct effects to the terrestrial-phase CRLF makes the assumption that toxicity of alachlor to birds is similar to the toxicity to terrestrial-phase amphibians and reptiles (this also applies to potential prey items). The same assumption is made for fish and aquatic-phase CRLF.

The acute measures of effect used for animals in the screening-level portion of this assessment are the LD_{50}, LC_{50} and EC_{50}. LD stands for "Lethal Dose", and LD_{50} is the amount of a material, given all at once, that is estimated to cause the death of 50% of the test organisms. LC stands for "Lethal Concentration" and LC_{50} is the concentration of a chemical that is estimated to kill 50% of the test organisms. EC stands for "Effective Concentration" and the EC_{50} is the concentration of a chemical that is estimated to produce a specific effect in 50% of the test organisms. Endpoints for chronic measures of exposure for listed and non-listed animals are the NOAEL/NOAEC and NOEC. NOAEL stands for "No Observed-Adverse-Effect-Level" and refers to the highest tested dose of a substance that has been reported to have no harmful (adverse) effects on test organisms. The NOAEC (*i.e.,* "No-Observed-Adverse-Effect-Concentration") is the highest test concentration at which none of the observed effects were statistically different from the control. The NOEC is the No-Observed-Effects-Concentration. For non-listed plants, only acute exposures are assessed (*i.e.*, EC_{25} for terrestrial plants and EC_{50} for aquatic plants).

The measures of effect for direct and indirect effects to the assessed species and their designated critical habitat are associated with impacts to survival, growth, and fecundity, and do not include the full suite of sublethal effects used to define the action area. According the Overview Document (USEPA, 2004), the Agency relies on effects endpoints that are either direct measures of impairment of survival, growth, or fecundity or endpoints for which there is a scientifically robust, peer reviewed relationship that can quantify the impact of the measured effect endpoint on the assessment endpoints of survival, growth, and fecundity.

2.10.3 Measures of Risk

Risk characterization is the integration of exposure and ecological effects characterization to determine the potential ecological risk from agricultural and non-agricultural uses of alachlor, and the likelihood of direct and indirect effects to CRLF and the DS in aquatic and terrestrial habitats. The exposure and toxicity effects data are integrated in order to evaluate the risks of adverse ecological effects on non-target species. For the assessment of alachlor risks, the risk quotient (RQ) method is used to compare exposure and measured toxicity values. EECs are divided by acute and chronic toxicity values. The resulting RQs are then compared to the Agency's levels of concern (LOCs) (USEPA, 2004) (see **Appendix F**).

For this endangered species assessment, listed species LOCs are used for comparing RQ values for acute and chronic exposures of alachlor directly to the CRLF and the DS. If estimated exposures directly to the assessed species of alachlor resulting from a particular use are sufficient to exceed the listed species LOC, then the effects determination for that use is "may affect". When considering indirect effects to the assessed species due to effects to prey, the listed species LOCs are also used. If estimated exposures to the prey of the assessed species of alachlor resulting from a particular use are sufficient to exceed the listed species LOC, then the effects determination for that use is a "may affect." If the RQ being considered also exceeds the non-listed species acute risk LOC, then the effects determination is a LAA. If the acute RQ is between the listed species LOC and the non-listed acute risk species LOC, then further lines of evidence (*i.e*. probability of individual effects, species sensitivity distributions) are considered in distinguishing between a determination of NLAA and a LAA. If the RQ being considered for a particular use exceeds the non-listed species LOC for plants, the effects determination is "may affect". Further information on LOCs is provided in **Appendix F**.

3.0. Exposure Assessment

3.1 Aquatic Exposure Assessment

The assessment of exposure within the action area is dependent upon a combination of modeling and monitoring data. In accordance with the Overview Document (USEPA, 2004), screening-level exposures are based on modeling which assumes a static water body. Aquatic exposures are quantitatively estimated for all of assessed uses using scenarios that represent high exposure sites for alachlor use. Each of these sites represents a 10-hectare field that drains into a 1-hectare pond that is 2 meters deep and has no outlet. Exposure estimates generated using the standard pond are intended to represent a wide variety of vulnerable water bodies that occur at the top of watersheds including prairie pot holes, playa lakes, wetlands, vernal pools, man-made and natural ponds, and intermittent and first-order streams. As a group, there are factors that make these water bodies more or less vulnerable than the standard surrogate pond. Static water bodies that have larger ratios of drainage area to water body volume would be expected to have higher peak EECs than the standard pond. These water bodies will be either shallower or have large drainage areas (or both). Shallow water bodies tend to have limited additional storage capacity, and, thus, tend to overflow and carry pesticide in the discharge whereas the standard pond has no discharge. As watershed size increases beyond 10 hectares, at some point, it becomes unlikely that the entire watershed is planted to a single crop, which is all treated with the pesticide. Headwater streams can also have peak concentrations higher than the standard pond, but they tend to persist for only short periods of time and are then carried downstream. More details on the uncertainties associated with the various exposure assessments and modeling scenarios specifically may be found in the Uncertainty Section (Section 6.1).

Specific management practices for all of the assessed uses of alachlor were used for modeling, including application rates, number of applications per year, application intervals, and the first application date for each use. Incorporated and broadcast applications were modeled for all uses to provide a range of expected EECs that are representative of actual management practices. The broadcast application is expected to result in the highest EECs, because alachlor will be contained to the upper horizons of the soil profile and can be easily transported to aquatic resources via runoff. The general conceptual model of exposure for this assessment is that the highest exposures are expected to occur in headwater streams adjacent to agricultural fields and non-agricultural use sites (woody ornamentals). Many of the streams and rivers within the action area defined for this assessment are in close proximity to both agricultural and non-agricultural uses sites (for this assessment the action area represents the entire state of California).

Available usage data (USEPA, 2009) suggest that the heaviest usage of alachlor relative to the action area is likely to be in the Central Valley, although these use rates are much less than the use of alachlor in the Midwestern corn/sorghum belt. All existing PRZM scenarios were evaluated, and a subset was selected for use in this assessment. The scenarios were selected to provide a spatial context to predicted exposures.

Currently a suite of 28 PRZM California scenarios are available for use in ecological risk assessments representing predominantly agricultural uses. Of these, 16 were developed specifically for the CRLF assessments, 3 were developed for the Organophosphate (OP) cumulative assessment (USEPA, 2006b), and 9 are standard scenarios. Each scenario is intended to represent a high-end exposure setting for a particular use site. Scenario locations are selected based on various factors including crop acreage, runoff and erosion potential, climate, and agronomic practices. Once a location is selected, a scenario is developed using locally specific soil, climatic, and agronomic data. Each PRZM scenario is assigned a specific climatic weather station providing 30 years of daily weather values.

Specific scenarios were selected for use in this assessment using two criteria. First, an evaluation of all available PRZM scenarios was conducted, and those scenarios that represent alachlor uses (*e.g.,* CA corn) were selected for modeling. Weather information was assigned to these scenarios at development. Second, additional scenarios (CA Nursery and CA Residential) were identified to represent the use of alachlor on woody ornamentals (juniper and yew) for which a scenario within the action area is not available. These scenarios rely on climatic data from San Diego (23188) and San Fransico (23234), respectively. Alachlor use on woody ornamentals was modeled using both the nursery scenario and the residential scenario because CDPR PUR data indicate that alachlor is used for landscapeing purposes, therefore residential use cannot be eliminated from this assessment.

Residential use is a potentially important exposure pathway evaluated in this assessment. The amount of impervious surfaces associated with the urban environment provides a potential direct conduit in which alachlor-contaminated runoff can easily reach surface water resources. Estimating the aquatic exposure from the use of alachor on woody ornamentals (juniper and yew) for residential purposes involves the use of two scenarios, one for California residential turf and one for California impervious surfaces. EECs are derived for both scenarios, and then combined by assuming that 50% of the watershed is lawn (a fraction of which is actually planted with woody ornamentals) and the remainder is impervious surface. It is also assumed that 1.68% of the impervious surface gets over-sprayed during treatment of the ornamental plants. A detailed description of the rationale for these values is provided in **Appendix G**. Information from a number of sources concluded that the usage on juniper and yew are mainly restricted to decorative landscaping, shade, privacy (natural fencing), or foundation protection (Gillman *et al*., 2001; Starbuck, 2003). Therefore as a reasonable estimate for the percent lot treated was approximately 1,638.4ft^2 (0.038 acre, or 15% of a typical lot). This was derived assuming that the entire perimeter of the 0.25 acre lot (104.4 ft length) of potentially treatable area was planted with juniper or yew having a row width of approximately 4 ft based on plant phenology; for modeling purposes, 100% of this 0.04 acre area was assumed to be treated with a broadcast spray application.

Further description (metadata) and copies of the existing PRZM scenarios may be found at the following websites.

http://www.epa.gov/oppefed1/models/water/index.htm#przmexamsshell

http://www.epa.gov/oppefed1/models/water/przmenvironmentdisclaim.htm

A summary of all the modeled scenarios along with associated weather information is included in **Table 3.1**. Both the agricultural and non-agricultural scenarios were used within the standard framework of PRZM/EXAMS modeling using the standard graphical user interface (GUI) shell, PE5.pl. The models and GUI used in this assessment may be found at the following website:

http://www.epa.gov/oppefed1/models/water/index.htm

Table 3.1. Summary of PRZM Scenarios.

Use	Scenario	First Application	Min. Application Interval	Weather Station (WBAN #)
Corn	CAcornOP	March 1 (preplant)	NS	Sacramento (23232)
Sweet Corn	CAcornOP	May 1 (30 d post emergence)	NS	Sacramento (23232)
Sorghum	CAwheatRLF	January 2 (preplant)	NS	Fresno (93193)
Legume Vegetables (Soybeans, dry beans, succulent beans, lima beans)	CARowCropRLF_V2	Nov 20 (preplant)*	NS	San Francisco (23234)
Woody ornamentals (Junipers and Yews)	CANurserySTD_V2 CA ResidentialRLF	Dec 1** (post transplant)	21 days	San Diego (23188) San Francisco (23234)
Cotton	CAcotton_WirrgSTD	March 1 (preplant) 60d prior to emergence	NS	Fresno (93193)
Sunflowers	CAcornOP	March 1 (preplant)	NS	Sacramento (23232)
Peanuts	CARowCropRLF_V2	Nov 20 (preplant)*	NS	San Francisco (23234)

NS = Not specified on the federal label.
*Preplant application was modeled, assuming an initial application 6-8 weeks prior to emergence based on data from USDA-NASS and the phenology of legumes (3 weeks from planting to emergence, and application was assume approximately 3 weeks prior to planting). Initial application date was modeled on Nov. 20.
**Initial application date for woody ornamental use was modeled on December 1, a conservative estimate for timing of the initial application. Timing of application was determined based on historical precipitation trends, modeling the time where precipitation is greatest (Oct – Apr, peak Dec – end of Jan).

3.1.1. Model Inputs

The estimated concentrations from surface water sources were calculated using Tier II PRZM (Pesticide Root Zone Model) and EXAMS (Exposure Analysis Modeling System). PRZM is used to simulate pesticide transport as a result of runoff and erosion from a standardized watershed, and EXAMS estimates environmental fate and transport of pesticides in surface waters. The linkage program shell (PE5.pl) that incorporates the site-specific scenarios was used to run these models.

Scenarios used in this assessment consist of four California-specific scenarios developed for uses being assessed (corn, sorghum, legumes, cotton, and sunflowers), and two California-specific scenarios as surrogate crops for an alachlor use (woody ornamentals). All scenarios were modeled using local weather data selected to represent the highest rainfall potential in a region as described above. Linked site-specific use scenarios and meteorological data were used to estimate exposure as a result of specific use for each modeling scenario. The PRZM/EXAMS model was used to calculate concentrations using the standard ecological water body scenario in

EXAMS. Weather and agricultural practices were simulated over 30 years so that the 1-in-10 year exceedance probability at the site was estimated for the standard ecological water body.

The date of initial application was developed based on several sources of information including data provided by BEAD and Crop Profiles maintained by the USDA (http://www.ipmcenters.org/cropprofiles/ and http://usda.mannlib.cornell.edu/reports/nassr/field/planting/uph97.html). In general, the date of initial application was selected to represent the most vulnerable window of exposure (*e.g.*, timed with highest expected precipitation). The application dates for alachlor in California from 2004 and 2005 (used as a representative sample) show that the majority of applications occur during May, but applications can occur as early as January and as late as September (**Figure 3.1**).

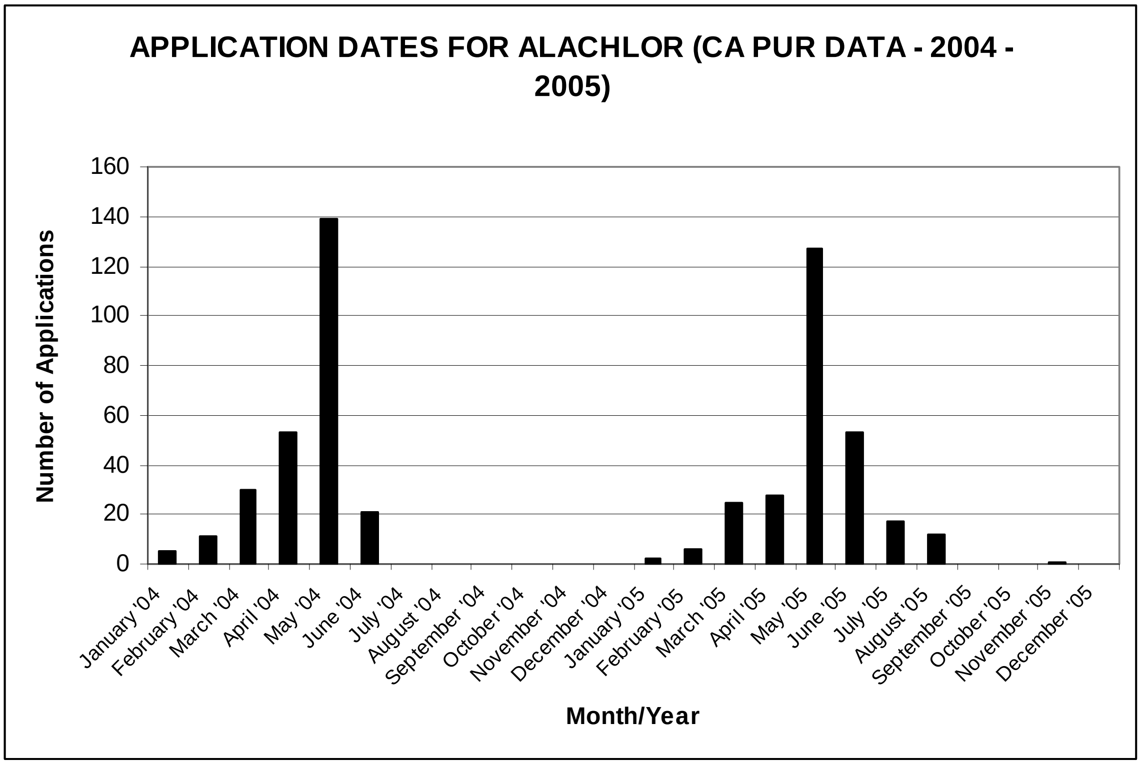

Figure 3.1 Application Dates for Alachlor (CDPR PUR Data; 2004 – 2005)

The appropriate PRZM input parameters were selected from the environmental fate data submitted by the registrant and in accordance with USEPA-OPP EFED water model parameter selection guidelines, Guidance for Selecting Input Parameters in Modeling the Environmental Fate and Transport of Pesticides, Version 2.3, February 28, 2002 (USEPA, 2002). These parameters are consistent with those used in the 1998 RED (USEPA, 1998b) and subsequent risk assessments (USEPA, 2006a) and are summarized in **Table 3.2**. More detail on these assessments may be found at:

http://www.epa.gov/oppsrrd1/REDs/0063.pdf http://www.epa.gov/pesticides/cumulative/common_mech_groups.htm#chloro

Crop specific management practices that were used as inputs for PRZM/EXAMS are summarized in **Table 3.3**, and all chemical properties and fate input parameters are summarized in **Table 3.4**. All PRZM/EXAMS input and output files are included in **Appendix H**.

Table 3.2. Summary of Environmental Fate Data for Alachlor.

Fate Property	Value	MRID[1] (or source)
Molecular Weight	269.77 g/mol	MRID 146114
Vapor Pressure	2.2×10^{-5} torr	Beestman and Deming, 1974
Henry's Law Constant	3.3×10^{-8} atm*m^3/mol	Calculated
Solubility in Water	240 mg/L (24ºC)	Beestman and Deming, 1974
Photolysis in Water	Stable	MRID 23012
Aerobic Soil Metabolism Half-lives	29.7 d (silt loam) 34.0 d (loamy sand) 25.8 d (silt)	MRID 134327
Hydrolysis (25 ºC)	pH 5 – stable pH 7 – stable pH 9 - stable	MRID 134327
Aerobic Aquatic Metabolism (water column)	84 d	Represents 2x the high-end confidence bound on the mean TTR aerobic soil metabolism half-life.
Anaerobic Aquatic Metabolism (benthic)	Stable	Default
Soil-water distribution coefficient (K_d)	0.33	MRID 152209 Represents lowest reported non-sand K_d
[1] Master Record Identification (MRID) is record tracking system used within OPP to manage data submissions to the Agency. Each data submission if given a unique MRID number for tracking purposes.		

Table 3.3. Summary of Management Practices for PRZM/EXAM Modeling Input Parameters.

	Application rate lbs ai/A (kg ai/ha)	Max No. Applications per year	Max. Annual Application Rate lbs ai/A	Application Interval	Application method	Broadcast / Incorporation depth	CAM Input	Spray Drift Efficiency	App. Efficiency
Corn	4.0	2	4.0	NA	Ground	Broadcast & incorporated (10cm)	1, 4	0.01	0.99
Sweet corn	4.0	1	4.0	NA	Ground	Broadcast & incorporated (10cm)	2, 4	0.01	0.99
Sorghum	4.0	2	4.0	NA	Ground	Broadcast & incorporated (10cm)	1, 4	0.01	0.99
Legumes	3.0	1	3.0	NA	Ground	Broadcast & incorporated (10cm)	1, 4	0.01	0.99
Woody Ornamentals	4.0	2	4.0	21	Ground	Broadcast	2	0.01	0.99
Cotton	2.0	1	2.0	NA	Ground	Broadcast & incorporated (4cm)	1, 4	0.01	0.99
Sunflowers	4.0	4	4.0	NA	Ground	Broadcast & incorporated (4cm)	1, 4	0.01	0.99
Peanuts	4.0	2	4.0	NA	Ground	Broadcast & incorporated (10cm)	1, 4	0.01	0.99

NA = Not applicable, a single annual application was modeled.

Table 3.4. Summary of PRZM/EXAMS Chemcial Input Parameters for Alachlor

Input Parameter	Value	Source
Molecular Mass (g/mol)	269.77	MRID 146114
Vapor Pressure at 24°C (torr)	2.2×10^{-5}	Beestman and Deming, 1974
Henry's Law Constant	3.3×10^{-8}	Calculated
Solubility in Water at 24°C (mg/L)	2400	Represents 10x the measured water solubility value (Beestman and Deming, 1974)
Soil-water distribution coefficient (K_d)	0.33	Represents the lowest reported non-sand K_d (MRID 152209)
Aerobic Soil Metabolism Half-life (days)	34.3	Represents the high-end confidence bound on the mean (MRID 134327)
	42	Represents the high-end confidence bound on the mean **total toxic residues half-life** (MRID 134327)
Aerobic Aquatic Metabolism Half-life (days)	84	Represents 2x the high-end confidence bound on the mean aerobic soil metabolism half-life (MRID 134327). (USEPA, 2002)
Anaerobic Aquatic Metabolism Half-life (days)	0 (Stable)	None
Hydrolysis Half-lives (days)	0 (Stable)	Alachlor is stable to hydrolysis at pH 5, 7, and 9 (MRID 134327) therefore assumed to be 0 (US EPA, 2002)
Aqueous Photolysis Half-life (days)	0 (Stable)	MRID 23012

* Post processing of residential scenario output assuming 5% of lot treated, 1% overspray as per guidance

3.1.2. Results

In general, these EECs show a pattern of exposure for all durations that is influenced by the persistence of the compound and the lack of flow through the static water body. Predicted alachlor concentrations, though high across durations of exposure for a single year, do not increase across the 30-year time series; therefore, accumulation is not a concern. The resulting EECs are summarized in **Table 3.5**. PRZM/EXAMS output files are included in **Appendix H**.

Table 3.5. Aquatic Total Toxic Residue EECs (µg/L) for Alachlor Uses in California.

Use Site (application method)	Application Rate (lbs a.i./acre)	No. of Applications	1-in-10 year Peak EEC	1-in-10 year 21-day average EEC	1-in-10 year 60-day average EEC
Corn (broadcast)	4.0	1	44.8	43.7	41.1
Corn (incorporated)	4.0	1	12.6	12.3	11.5
Sweet Corn (broadcast)	4.0	1	11.3	10.7	9.8
Sweet Corn (incorporated)	4.0	1	3.2	3.1	2.8
Sorghum (broadcast)	4.0	1	46.7	45.6	42.7
Sorghum (incorporated)	4.0	1	12.8	12.5	11.7
Soybeans (broadcast)	3.0	1	32.9	31.9	27.1
Soybeans, dry beans, lima beans (incorporated)	3.0	1	9.3	9.0	7.7
Woody ornamentals (Juniper and Yew)	4.0 (nursery use)	1	56.0	54.3	43.0
	4.0 (residential use)	1	6.3	5.5	4.6
Cotton (broadcast)	2.0	1	25.2	24.4	22.8
Cotton (incorporated)	2.0	1	15.3	14.8	13.8
Sunflowers (broadcast)	4.0	1	44.8	43.7	41.1
Sunflowers (incorporated)	4.0	1	27.3	26.6	25.0
Peasnuts (broadcast)	4.0	1	43.9	42.5	36.2
Peanuts (incorporated)	4.0	1	12.4	12.0	10.2

3.1.3. Existing Monitoring Data

A critical step in the process of characterizing EECs is comparing the modeled estimates with available surface water monitoring data. Included in this assessment are alachlor data from the USGS National Water Quality Assessment (NAWQA) program (http://water.usgs.gov/nawqa) focusing on the parent alachlor and three degradate products (alachlor-ESA, alachlor oxanilic acid, and alachlor sulfynilacetic acid), and data from the CADPR that focused on the parent and two degradates (alachlor-ESA and alachlor oxanilic acid). In addition, atmospheric monitoring data for alachlor from the open literature are summarized below.

3.1.3.1. USGS NAWQA Surface Water Data

Data from the USGS NAWQA website for alachlor occurrence in surface water in California were obtained on February 6, 2009. A total of 2,122 surface water samples were analyzed for alachlor spanning a period from 1992 to 2007. Of these, a total of 96 samples detected alachlor (frequency of detection of 4.5%). Detections ranged from 0.0025 to 0.86 µg L^{-1} (MDL ranged from 0.002 to 0.005 µg L^{-1}). Surface water detections generally occurred in the spring and early summer months (March through July), which correlates to the maximum use period, according to CA PUR data. The maximum concentration detected was 0.86 ppb from Stanislaus County in 1992. The three degradate products included in NAWQA (alachlor-ESA, alachlor oxanilic acid, and alachlor sulfynilacetic acid) were infrequently detected, eight, once, and once, respectively. These detections were not well correlated with detections of the parent alachlor. Detections of alachlor-ESA ranged from 0.05-0.07 µg L^{-1} (MDL = 0.02 µg L^{-1}); the detection of alachlor-OXA was 0.06 µg L^{-1} (MDL = 0.02 µg L^{-1}), and the detection of alachlor sulfynilacetic acid was 0.03 µg L^{-1} (MDL = 0.02 µg L^{-1}). These degradates were included in previous assessments and were found to be less toxic than the parent, therefore, they are not included in this assessment.

3.1.3.2. USGS NAWQA Groundwater Data

Data from the USGS NAWQA website for alachlor occurrence in groundwater in California were obtained on February 6, 2009. A total of 747 groundwater samples were analyzed for alachlor spanning a period from 1993 to 2006; there were no reported detections of alachlor (MDL ranged from 0.002 to 0.005 µg L^{-1}).

3.1.3.3. California Department of Pesticide Regulation (CPR) Surface Water Data

Data from the CDPR surface water monitoring database website for the occurrence of alachlor and two major degradates (alachlor-ESA (ethane sulfonic acid), and alachlor OXA (oxanilic acid)) were obtained on March 26, 2009. A total of 2,786 surface water samples were analyzed for alachlor spanning a period from 1992 to 2006. Of these, a total of 69 samples detected alachlor (detection frequency of 2.5%). Concentrations ranged from 0.003 to 0.86 µg L^{-1} (LOQa ranged from 0.002 to 1 µg L^{-1}). Consistent with NAWQA results, detections generally occurred in the spring and early summer months (March through July), which correlates to the maximum use period, according to CA PUR data. However, the majority of these data are included in the NAWQA database; it is presented here to include state-level water quality monitoring. The maximum concentration detected was 0.86 ppb in Stanislaus County in 1992. A total of 56 samples were analyzed for alachlor-ESA and alachlor-OXA, and only one detection was reported for alachlor-ESA at 0.064 µg L^{-1} in Stantinslaus County.

3.1.3.4. Atmospheric Monitoring Data

Available monitoring data for alachlor in air and rainfall were evaluated from the open literature to provide contextual information for the evaluation of the extent of the action area and estimated concentrations in surface water. Alachlor may enter the atmosphere via volatilization and spray drift and is subsequently aerially deposited, typically via wet deposition, or precipitated out on sorbed particles. Based on the available information (Scheyer *et al.*, 2007; Kuang, *et al*, 2003; Majewski *et al.*, 2000; Foreman, *et al*, 1999; USGS, 1998; Goolsby *et al.*, 1997; Gish *et al.*, 1995; Majewski and Capel, 1995; Capel *et al.*, 1994), alachlor has been detected in rainwater and air samples across the United States and in France at variable frequency of detections. Often these studies are non-targeted to alachlor use and there is a lack of ancillary data in these studies to determine whether these detections are due to spray drift or longer-range transport due to volatilization. Alachlor has a relatively low Henry's law constant ($3.3x10^{-8}$) and high water solubility (240 mg/L, Beestman and Deming, 1974). These physical properties have been shown to favor enrichment in rainwater and preferentially fall out in the particle phase, thus, alachlor is more efficiently deposited in precipitation (Scheyer *et al.*, 2007). Scheyer *et al.* (2007) found that there is a distinct seasonality attributed to the concentrations observed in rainwater at medium to long distances from use sites (1-1000 km). This study was conducted as a targeted monitoring study of volatilization of alachlor, and other compounds, from corn fields in eastern France over a two-year period. The concentrations detected in the reviewed studies (generally low, and studies were not conducted in California) suggest that atmospheric transport of alachlor will yield exposures well below those predicted by modeling described above, but transport via atmospheric processes may be an additional route of exposure for the CRLF and DS.

Specifically, alachlor concentrations in rainfall have been measured up to 6 µg/L in France (Scheyer *et al*., 1997). In 1990-1991, the 95th and 99th percentile alachlor levels in rainfall in the mid-west were reported to be 0.26 and 0.95 µg/L, respectively, with a maximum concentration of 3.2 µg/L (USGS, 1998; Goolsby *et al.*, 1997). Goolsby *et al.* (1997) reported detections of alachlor in approximately 20% of the rainwater samples at concentrations up to 3.2 µg/L. Capel *et al*. (1994) reported the frequency of detections and pesticide levels in rainfall from 1991 to 1993 in Minnesota; in 1991, alachlor was detected in 15 % of the samples with a maximum concentration of 3.6 µg/L, in 1992 it was 16 percent and 2.2 µg/L, and in 1993 it was 74 % and 12 µg/L. Subsequent 1994 monitoring data from 6 Minnesota sites around the state found detections in 87% of the samples (range: 82 - 100%) and a maximum level of 1.15 µg/L (range of maximum levels: 0.57 – 1.15 µg/L). Further Gish *et al*. (1995) showed that herbicide volatilization (alachlor and atrazine) was greater under mulched conditions (impregnated bulk fertilizer application), but decreased dramatically after the first irrigation.

The data indicate that alachlor can enter the atmosphere via volatilization and spray drift. The data also suggest that alachlor is frequently found in rain samples and tends to be seasonal, related to application timing. Finally, the data suggest that although frequently detected, alachor concentrations measured in rain samples are less than those seen in the open literature surface water monitoring data (see below, Section 3.1.3.5). The modeling conducted as part of this assessment support the contention that runoff and spray drift are the principal routes of exposure.

3.1.3.5. Summary of Open Literature Sources of Surface Water Monitoring Data for Alachlor

Extensive reviews of both groundwater and surface water monitoring data have been previously reported in the open literature. The 1998 Alachlor RED (USEPA, 1998b) and the Cumulative Risk Assessment for the Chloroacetanilides (USEPA, 2006d) evaluated much of the monitoring data on alachlor at a national scale through 2001 (summarized in **Table 3.6**). Due to label revisions that occurred prior to RED issuance, maximum agricultural application rates were reduced to 4 lbs a.i./acre/yr. Alachlor use has also decreased as alternatives have become available. Therefore, average annual alachlor concentrations recently monitored in surface water and groundwater are not expected to exceed those measured before the label revisions took effect in the late 1990s. Currently, few data are available after the mid-1990's, except NAWQA, CDPR, and ARP which reported only very low concentrations for this time period. NAWQA and CDPR surface water databases are generally non-targeted studies, and, therefore, do not provide confirmatory data for label revisions. Also, the Acetechlor Reregistration Partnership (ARP) study probably does not accurately capture the effect of label revisions on monitoring concentration. Therefore, confirmatory data from targeted studies are not available. No trend can be predicted for peak alachlor concentrations monitored in surface water and groundwater, however, as monitoring data are not representative of peak exposure values. These monitoring studies did not include study sites in California, therefore, it is difficult to compare the reported results to expected exposure levels in California surface waters. However, these data provide important contextual information on the occurrence of alachlor in surface waters, as many of these studies are targeted studies examining the occurrence of alachlor in water resources in relatively close proximity to use sites (*e.g.*, ARP Surface Drinking water Supply Study).

Table 3.6. Summary of Alachlor Detections in Surface Water by Study as Included in the 1998 Alachlor RED and 2006 Chloroacetanilide Cumulative Risk Assessment.

Study	Number of Sites	Maximum Peak (µg/L)	Maximum TWMC[1] (µg/L)
ARP Surface Drinking Water Supply Study 1995-2001	152-175	4.65	0.590
USGS Midwestern Reservoir Reconnaissance 1992	76	~ 5 to 10	Not reported
USGS Mississippi River Basin Study 1991-1992	8	3.6	0.43
USGS Midwestern Stream Reconnaissance 1989	48	51.3	11.6
State of Illinois 1986-1988	30	18	0.81
Lake Erie Basin Case Study 1983-1987	7	91.47	1.74
Monsanto Finished Surface Water Study 1986	30	9.5	1.1
Monsanto Finished Surface Water Study 1985	30	12	1.5
Ohio Tributaries to Lake Erie 1982-1985	8	76	3.3[2]
USGS Cedar River Basin Study 1984	6	23	1.7

[1] TWMC means time weighted mean concentrations, annual unless otherwise noted.
[2] Time weighted mean concentration calculated over a 4 month period of the study; Apr. 15 to Aug. 15.

One of the values from the available monitoring studies (*i.e.*, 91.47 µg/L, see **Table 3.6**) is higher than the highest 1-in-10-year peak EEC value from PRZM/EXAMS (see Section 3.1.2). This value was detected in runoff coming from a small watershed, compared to other watersheds in the study that was primarily dominated by agriculture. The study with the high value was conducted before the maximum application rates for alachlor were reduced from 6 lb a.i./acre to 4 lb a.i./acre in the 1990's. An application rate of 6 lbs a.i./acre was modeled for comparison purposes using the CAnursery scenario. Modeling output showed peak concentrations that are within a reasonable margin of error to the peak monitoring data (84 µg/L compared to 91.5 µg/L). Therefore, due to the reduced application rate the concentration cannot be used to reflect potential concentrations from current use practices and is not quantitatively used in this risk assessment.

3.1.4 Impact of Typical Usage Information on Exposure Estimates

A final piece of the exposure characterization includes an evaluation of usage information. Label application information was provided by EPA's Biological and Economic Analysis Division and was previously summarized in **Table 2.8**. This information suggests that alachlor use on corn and beans (dry and succulent, the two highest uses in the CDPR PUR data) is (at least sometimes) applied near the maximum label rate of 4.0 lbs a.i./acre in California based on CDPR PUR data. This shows that the modeling conducted for this assessment provides reasonable exposure estimates, based on alachlor use patterns.

3.2. Terrestrial Animal Exposure Assessment

T-REX (Version 1.4.1) is used to calculate dietary and dose-based EECs of alachlor for birds (surrogate for reptiles and terrestrial-phase amphibians), mammals, and terrestrial invertebrates. T-REX simulates a 1-year time period. For this assessment, spray and impregnated dry bulk fertilizer applications of alachlor are considered, as discussed below. Terrestrial EECs were derived for the uses previously summarized in **Table 2.7**. Unlike aquatic exposure estimates that represent total residues (parent plus degradates), terrestrial exposure estimates generated using T-REX are for parent alone.

Upper-bound Kenaga nomogram values reported by T-REX are used for derivation of dietary EECs for the terrestrial phase CRLF and their potential prey. When data are absent, as in this case, EFED assumes a 35-day foliar dissipation half life, based on the work of Willis and McDowell (1987). Because, for all of the alachlor uses modeled, the maximum single application rate and the maximum yearly application rate are the same for each use, only a single application (at the maximum application rate) was modeled, since this would result in the highest EECs (as opposed to modeling two applications at lower application rates). Potential direct acute and chronic effects of alachlor to the terrestrial-phase CRLF are initially derived by considering oral exposures modeled in T-REX for a small bird (20g) consuming small invertebrates. Potential impacts to mammalian prey base were evaluated in T-REX for a small mammal (15 g) consuming short grass. Resulting dietary-based EECs (mg/kg-food) and dose-adjusted EECs (mg/kg-bw) are summarized in **Table 3.7.**

Table 3.7. Upper-bound Kenega Nomogram EECs for Dietary- and Dose-based Exposures of the CRLF and its Prey to Alachlor.

Use(s)	Application Rate (lb a.i./acre)	EECs for CRLF (small birds used as a surrogate)		EECs for Prey (small mammals)	
		Dietary-based EEC (ppm)	Dose-based EEC (mg/kg-bw)	Dietary-based EEC (ppm)	Dose-based EEC (mg/kg-bw)
Corn Sweet corn Grain sorghum Peanuts Woody ornamentals Sunflowers	4	540	615	960	915
Soybeans Dry beans~ Lima beans (green)	3	405	461	720	686
Cotton	2	270	308	480	458

The impregnated bulk fertilizer applications (corn, sorghum, and soybeans) of alachlor are treated as granular formulations for modeling purposes. Therefore, an LD50/ft^2 analysis was performed to evaluate potential risks to birds and mammals (for use in risk characterization). The exposure used in this analysis is the mass of alachlor applied to a square foot area (mg/ft^2). Based on an application rate of 4 lbs a.i./acre (maximum bulk fertilizer application rate), the exposure value used in the LD50/ft^2 analysis is 42 mg/ft^2.

3.2.1. Potential Exposure to Terrestrial Invertebrates

T-REX is also used to calculate EECs for terrestrial invertebrates exposed to alachlor. Dietary-based EECs calculated by T-REX for small and large insects (units of a.i./g) are used to bound an estimate of exposure to honey bees (*Apis mellifera*) (used as a surrogate for terrestrial invertebrates) (**Table 3.8**). Available acute contact toxicity data for bees exposed to alachlor (in units of µg a.i./bee), are converted to µg a.i./g (of bee) by multiplying by 1 bee/0.128 g. The EECs are compared to the acute contact toxicity data for bees in order to derive RQs.

Table 3.8. EECs (ppm) for Indirect Effects to the Terrestrial-Phase CRLF via Effects to Terrestrial Invertebrate Prey Items.

Use	Application Rate (lb a.i./acre)	Small Insect	Large Insect
Corn	4	540	60
Sweet corn			
Grain sorghum			
Peanuts			
Woody ornamentals			
Sunflowers			
Soybeans	3	405	45
Dry beans~			
Lima beans			
Cotton	2	270	30

3.3 Terrestrial Plant Exposure Assessment

Terrestrial plants in riparian areas may be exposed to alachlor residues carried from application sites via surface water runoff or spray drift. Exposures can occur directly to seedlings breaking through the soil surface and through root uptake or direct deposition onto foliage to more mature plants. Riparian vegetation is important to the water and stream quality of the assessed species because it serves as a buffer and filters out sediment, nutrients, and contaminants before they enter the watersheds associated with the assessed species' habitat. Riparian vegetation has been shown to be essential in the maintenance of a stable stream (Rosgen, 1996). Destabilization of the stream can have an adverse effect on habitat quality by increasing sedimentation within the watershed.

Concentrations of alachlor on the riparian vegetation were estimated using OPP's TerrPlant model (USEPA, 2006e; Version 1.2.2). The TerrPlant model evaluates exposure to plants via runoff and spray drift and is EFED's standard tool for estimating exposure to non-target plants. The runoff loading of TerrPlant is estimated based on the solubility of the chemical and assumptions about the drainage and receiving areas.

Parameter values for application rate, drift assumption, and incorporation depth are based upon the use and related application method (**Table 3.9**). A runoff value of 0.05 is utilized based on alachlor's solubility, which is classified by TerrPlant as >100 mg/L. For ground flowable application methods, drift is assumed to be 1% (aerial applications were not modeled due to label restrictions for CA). For modeling purposes, the bulk fertilizer applications are treated as granular applications (*i.e.*, no drift is assumed). EECs relevant to terrestrial plants consider pesticide concentrations in drift and in runoff. These EECs are listed in **Table 3.9**. An example output from TerrPlant v.1.2.2 is available in **Appendix I**.

Table 3.9. Screening-Level Exposure Estimates for Terrestrial Plants to Alachlor.

Use	Application rate (lbs a.i./A)	Spray drift EEC (lbs a.i./A)	Dry area EEC (lbs a.i./A)	Semi-aquatic area EEC (lbs a.i./A)
Ground, Surface Applications[1]				
Corn	4	0.04	0.24	2.04
Sweet corn				
Grain sorghum				
Sunflowers				
Peanuts				
Woody ornamentals				
Soybeans	3	0.03	0.18	1.53
Cotton	2	0.02	0.12	1.02
Ground, Soil Incorporated (2 inches) Applications[2]				
Corn	4	0.04	0.14	1.04
Sweet corn				
Grain sorghum				
Peanuts				
Sunflowers				
Soybeans	3	0.03	0.105	0.78
Dry beans				
Lima beans				
Cotton	2	0.02	0.07	0.52
Bulk Fertilizer Applications[3]				
Corn	4	0	0.1	1
Sorghum				
Soybeans	3	0	0.075	0.75

[1] Surface applications are not allowed for the dry beans and lima beans uses.
[2] Soil incorporated applications are not allowed for the woody ornamentals use
[3] The bulk fertilizer applications are treated as granular applications for modeling purposes

For ground applications of alachlor, the highest off-target loadings of alachlor predicted by TerrPlant are approximately 50% of the application rate for semi-aquatic areas adjacent to application sites. As expected, resulting exposure estimates for terrestrial plants are higher for surface than for soil incorporated applications.

4.0. Effects Assessment

This assessment evaluates the potential for alachlor to directly or indirectly affect the CRLF and/or the DS or affect their designated critical habitat. As discussed in Section 2, assessment endpoints for the assessed species include direct toxic effects on survival, reproduction, and growth, as well as indirect effects, such as reduction of the prey base and/or effects to its habitat. In addition, potential effects to critical habitat are assessed by evaluating potential effects to the PCEs, which are components of the critical habitat areas that provide essential needs to the species, such as water quality and food base (see Section 2.4).

Acute (short-term) and chronic (long-term) toxicity information is characterized based on registrant-submitted studies and a comprehensive review of the open literature on alachlor, consistent with the Overview Document (USEPA, 2004). Potential direct and indirect effects to the CRLF and the DS and potential effects to critical habitat are evaluated in accordance with the methods (both screening and species-specific refinements) described in the Agency's Overview Document (USEPA, 2004).

Other sources of information, including use of the acute probit dose response relationships to establish the probability of an individual effect and reviews of the Ecological Incident Information System (EIIS), are conducted to further refine the characterization of potential ecological effects associated with exposure to alachlor.

A summary of the available aquatic and terrestrial organism ecotoxicity information, use of the probit dose-response relationship, and the incident information for alachlor are provided in the following sections. A summary of the available data directly used in this assessment is presented. A more comprehensive list of the available toxicity data is included in **Appendix C** of this assessment.

4.1. Ecotoxicity Study Data Sources

Toxicity endpoints are established based on data generated from guideline studies submitted by the registrant and from open literature studies that meet the criteria for inclusion into the ECOTOX database maintained by EPA/Office of Research and Development (ORD) (USEPA, 2004). Open literature data presented in this assessment were obtained from the ecological assessment for the sunflower and cotton uses (USEPA, 2006), as well as ECOTOX information obtained in a query from December 2008. In order to be included in the ECOTOX database, papers must meet the following minimum criteria:

- the toxic effects are related to single chemical exposure;
- the toxic effects are on an aquatic or terrestrial plant or animal species;
- there is a biological effect on live, whole organisms;
- a concurrent environmental chemical concentration/dose or application rate is reported; and
- there is an explicit duration of exposure.

Meeting the minimum criteria for inclusion in ECOTOX does not necessarily mean that the data are suitable for use in risk estimation. Data that pass the ECOTOX screen are evaluated along with the registrant-submitted data, and may be incorporated qualitatively or quantitatively into this endangered species risk assessment. In general, only effects data in the open literature that are more conservative than the registrant-submitted data are considered. The degree to which open literature data are quantitatively or qualitatively characterized is dependent on whether the information is relevant to the assessment endpoints (*i.e.*, maintenance of survival, reproduction, and growth; alteration of PCEs in the critical habitat impact analysis) identified in the problem formulation. For example, endpoints such as biochemical modifications are not likely to be used to calculate risk quotients unless it is possible to quantitatively link these endpoints with reduction in survival, reproduction, or growth (*e.g.*, the magnitude of effect on the biochemical endpoint needed to result in effects on survival, growth, or reproduction is known). A summary of all accepted open literature and a bibliography of all open literature considered as part of this assessment regardless of whether the data were accepted or rejected by ECOTOX is included in **Appendix E**.

As described in the Agency's Overview Document (USEPA, 2004), the most sensitive endpoint for each taxon is used for RQ calculation. **Tables 4.3** (aquatic organisms) and **4.4** (terrestrial organisms) summarizes the most sensitive ecological toxicity endpoints for the CRLF and the DS and their designated critical habitat based on an evaluation of both the submitted studies and the open literature. Toxicity information used in this assessment is further described in the following sections. Additional information on the available submitted and open literature toxicity studies is provided in **Appendices C** and **D**.

4.2. Toxicity Categories

Toxicity to fish, aquatic invertebrates, birds, and mammals is categorized using the system shown in **Table 4.1** (USEPA, 2004). For non-target terrestrial insects, chemicals with LD_{50} values of <2, 2 – 11, and >11 µg/bee are classified as highly toxic, moderately toxic, and practically nontoxic, respectively. Toxicity categories for terrestrial and aquatic plants have not been defined.

Table 4.1. Categories of Acute Toxicity for Terrestrial and Aquatic Animals.

Toxicity Category	Aquatic Animals [LC_{50}/EC_{50} (mg/L)]	Birds and Mammals [LD_{50} (mg/kg-bw)]	Birds [LC_{50} (mg/kg-diet)]
Very highly toxic	< 0.1	<10	<50
Highly toxic	0.1 - 1	10 – 50	50 – 500
Moderately toxic	> 1 - 10	51 – 500	501 – 1000
Slightly toxic	> 10 - 100	501 – 2000	1001 – 5000
Practically nontoxic	> 100	>2000	>5000

4.3. Toxicity of Chemical Mixtures

As previously discussed in the problem formulation, the available toxicity data show that other pesticides may combine with alachlor to produce synergistic, additive, and/or antagonistic toxic interactions. The results of available toxicity data for mixtures of alachlor with other pesticides are presented in **Appendix A**. If alachlor is present in the environment in combination with other chemicals, the toxicity of the mixture may be increased relative to the toxicity of each individual chemical, offset by other environmental factors, or even reduced by the presence of antagonistic contaminants if they are also present in the mixture. The variety of chemical interactions presented in the available data set suggest that the toxic effect of alachlor, in combination with other pesticides used in the environment, can be a function of many factors including but not necessarily limited to (1) the exposed species, (2) the co-contaminants in the mixture, (3) the ratio of alachlor and co-contaminant concentrations, (4) differences in the pattern and duration of exposure among contaminants, and (5) the differential effects of other physical/chemical characteristics of the receiving waters (*e.g.* organic matter present in sediment and suspended water). Quantitatively predicting the combined effects of all these variables on mixture toxicity to any given taxon with confidence is beyond the capabilities of the available data.

4.4 Toxicity of Alachlor to Aquatic Organisms

Table 4.2 summarizes the most sensitive aquatic toxicity endpoints based on an evaluation of both the submitted studies and the open literature, as previously discussed. A brief summary of submitted and open literature data considered relevant to this ecological risk assessment for the CRLF and DS is presented below. Additional information is provided in **Appendix C.**

Table 4.2. Aquatic Toxicity Profile for Alachlor.

Assessment Endpoint	Acute/ Chronic	Species	Toxicity Value Used in Risk Assessment (mg a.i./L)	Slope (95% C.I.)	MRID/ ECOTOX Ref. No.	Comment
Freshwater fish (can be used as a surrogate for aquatic-phase amphibians)	Acute	Rainbow trout (*Oncorhynchus mykiss*)	96-hr LC_{50} = 1.8	4.5[1]	00023616	The study was conducted using TGAI[2]; this study is classified as 'supplemental' (some study parameters were not reported) and adequate for use in RQ calculations
	Chronic	Rainbow trout (*Oncorhynchus mykiss*)	NOAEC = 0.187	Not Applicable (N/A)	438626-01	The study is classified as 'acceptable' and was conducted using TGAI; the endpoints are based on reduced growth (length and wet weight); LOAEC = 0.388 mg a.i./L
Aquatic-phase amphibian	Acute	African clawed frog (*Xenopus laevis*)	96-hr LC_{50} = 6.1	4.5[1]	E66376 (Osano *et al.*, 2002)	The study was conducted using TGAI ; the study was non-guideline (no guidelines currently exist for an amphibian acute toxicity test) but scientifically sound
	Chronic	African clawed frog (*Xenopus laevis*)	NOAEC = 0.64	N/A	N/A	This endpoint is based on an ACR using acute and chronic data from *Oncorhynchus mykiss* and acute data from *Xenopus laevis*; the *Oncorhynchus mykiss* NOAEC was based on reduced growth (see text for details)
Freshwater invertebrates	Acute	Chironomid (*Chironomus plumosus*)	48-hr EC_{50} = 2.5	4.5[1]	40098001	The study was conducted using TGAI; this study is classified as 'supplemental' because the raw data were not available for review
	Chronic	Chironomid (*Chironomus plumosus*)	NOAEC = 0.036	N/A	N/A	This endpoint is based on an ACR using acute and chronic data from *Daphnia magna* and acute data from *Chironomus plumosus*; the *Daphnia* NOAEC was based on reduced adult length (see text

Assessment Endpoint	Acute/ Chronic	Species	Toxicity Value Used in Risk Assessment (mg a.i./L)	Slope (95% C.I.)	MRID/ ECOTOX Ref. No.	Comment
						for details)
Estuarine/ marine fish	Acute	Sheepshead minnow (*Cyprinodon variegates*)	96-hr LC_{50} = 3.9	4.5[1]	445243-01	The study is classified as 'acceptable' and was conducted using TGAI
	Chronic	Sheepshead minnow (*Cyprinodon variegates*)	NOAEC = 0.41	N/A	N/A	This endpoint is based on an ACR using acute and chronic data from *Oncorhynchus mykiss* and acute data from *Cyprinodon variegates*; the *Oncorhynchus mykiss* NOAEC was based on reduced growth (see text for details)
Estuarine/ marine invertebrates	Acute	Mysid (*Americamysis bahia*)	96-hr LC_{50} = 2.4	8.7 (5.1 – 12.4)	445243-02	The study is classified as 'acceptable' and was conducted using TGAI
	Acute	Eastern oyster (*Crassostrea virginica*)	96-hr shell deposition EC_{50} = 1.6	4.5[1]	445243-03	The study is classified as 'acceptable' and was conducted using TGAI
	Chronic	Copepod (*Tigriopus japonicus*)	NOAEC < 0.0001	N/A	E104287 (Lee *et al*. 2008)	The study was conducted using TGAI and is classified as 'supplemental'; a definitive endpoint could not be established because effects (increase in the generation time for adults in the F_0 and F_1 generations) were seen at all of the conc. tested; a non-native species was used in the study.
Aquatic plants	N/A	Aquatic plant (nonvascular) (*Selenastrum capricornutum*)	NOAEC = 0.00035 EC_{50} = 0.00164	N/A	427638-01	The study is classified as 'acceptable' and was conducted using TGAI; the NOAEC was based on reduced cell density
	N/A	Aquatic plant (vascular) (*Lemna gibba*)	NOAEL = 0.000339 IC_{50} = 0.0023	N/A	446497-02	The study is classified as 'acceptable' and was conducted using TGAI; the NOAEC was based on % inhibition

[1] A slope was not determined in the study, therefore, the default slope is used.
[2] TGAI = Technical grade active ingredient

4.4.1 Toxicity to Fish

Fish toxicity data were used to evaluate potential direct effects to aquatic-phase CRLF and the DS and indirect effects to the CRLF. A summary of acute and chronic fish and aquatic-phase amphibian data, including data from the open literature, is provided in the following sections. Additional information is included in **Appendix C**.

4.4.1.1. Acute Exposure (Mortality) Studies

Acceptable alachlor toxicity data are only available for a few fish species [rainbow trout (*Oncorhynchus mykiss*), bluegill sunfish (*Lepomis macrochirus*), channel catfish (*Ictalurus punctatus*) and sheepshead minnow (*Cyprinodon variegates*)]. LC_{50} values are similar across these species and range from 1.8 (rainbow trout) to 16.7 (channel catfish) mg a.i./L (see **Appendix C** for additional details on these studies). Therefore, alachlor is classified as slightly to moderately toxic to fish on an acute exposure basis.

For the CRLF, the most sensitive freshwater fish species is used as a surrogate to help characterize the potential risk to aquatic-phase CRLF. For the DS, the most sensitive species among the freshwater and estuarine/marine fish species tested is used to calculate risk quotients regardless of the salinity environment because the DS enters both freshwater and saltwater environments. More sensitive acceptable acute LC_{50} values for fish were not located in the open literature. Therefore, the lowest LC_{50} of 1.8 mg a.i./L reported for rainbow trout (MRID 00023616), is used for risk quotient calculations for the CRLF and DS. The LC_{50} value from the only available acute toxicity study with an estuarine/marine fish [*i.e.*, 3.9 mg a.i./L for sheepshead minnow (MRID 445243-01)] will be used to help characterize the risk of alachlor use to DS in saltwater environments.

4.4.1.2. Chronic Exposure (Growth/Reproduction) Studies

Chronic fish toxicity studies are used to assess potential direct effects to the DS (freshwater and estuarine/marine fish) and aquatic-phase CRLFs (freshwater fish) via potential effects to growth and reproduction. For fish, considering both registrant-submitted and open literature studies, there is only one acceptable chronic study available for alachlor [an early life-stage study with rainbow trout) (MRID 438626-01)]. In this study, length and wet weight were reduced by 3% and 11%, respectively, at the 0.388 mg a.i./L concentration. Additionally, there was reduced posthatch survival, increased rates of exopthalmia (abnormal bulging of the eyes), and a 3-day delay in time to swim-up of larvae at the 1.63 mg a.i./L concentration. The corresponding NOAEC for this study is 0.187 mg a.i./L.

Chronic data for estuarine/marine fish and alachlor are not available. To help characterize the risk to DS in saltwater environments, an acute to chronic ratio (ACR) using acute and chronic data from rainbow trout and acute data from sheepshead minnow is used to estimate a chronic endpoint for estuarine/marine fish. This results in a sheepshead minnow NOAEC of 0.41 mg a.i./L (rainbow trout LC_{50} = 1.8 mg a.i./L; rainbow trout NOAEC = 0.187 mg a.i./L; sheepshead minnow LC_{50} = 3.9 mg a.i./L; rainbow trout ACR = 1.8/0.187 = 9.6; 3.9/9.6 = 0.406).

4.4.2. Toxicity to Amphibians

Acute toxicity data for alachlor and aquatic-phase amphibians are available from two studies in the open literature. In the first study, the 96-hr LC_{50} value for African clawed frog (*Xenopus larvis*) embryos (midblastula to early gastrula stages) exposed to technical grade alachlor was 6.1 mg a.i./L (E66376, Osano *et al*., 2002). Sublethal effects, including edema, axial flexure, and gut and eye abnormalities, were also reported in this study (96-hr EC_{50} = 3.6 mg a.i./L).

In an additional study involving fire-bellied toad (*Bombina orientalis*) embryos (newly fertilized), a 96-hr LC_{50} value was not determined (E81388, Kang *et al*., 2005). After a 96-hr exposure to technical grade alachlor, however, there was 52.7% mortality of the embryos at a concentration of 2.7 mg a.i./L. Additionally, various embryonic abnormalities (including abnormalities associated with the neural plate, tail bud, muscle repose, tail fin circulation, operculum, and bastula) occurred at 1.4 mg a.i./L and/or higher concentrations.

In a study using leopard frog (*Rana pipiens*) larvae, the effects of alachlor alone (at a concentration of 0.1 µg a.i./L) and in a mixture of nine chemicals (0.1 µg a.i./L of each chemical and 10 µg a.i./L of each chemical) was investigated (E85815, Hayes *et al*., 2006). The nine pesticide mixture (4 herbicides: atrazine, metolachlor, alachlor, and nicosulfuron; 3 insecticides: cyfluthrin, cyhalothrin, and tebupirimphos; and 2 fungicides: methalaxyl and propiconizole) was meant to represent a potential environmentally relevant mixture. At the concentrations tested, alachlor alone had no impact on the measured endpoints. However, the mixtures containing alachlor (0.1 µg a.i./L of each chemical) did impact some of the measured endpoints (*e.g*., mortality, time to metamorphosis, and size at metamorphosis). All of the animals exposed to the 9-compound mixture at 10 ppb died after the first day of exposure.

Because the mortality endpoint for freshwater fish is lower than the mortality endpoints in the amphibian studies, the acute freshwater fish endpoint will be used to calculate risk quotients for aquatic-phase CRLF and the amphibian data will be used to help characterize risks to CRLF from acute exposure to alachlor. No chronic toxicity data are currently available for amphibians and alachlor, therefore, the chronic endpoint for freshwater fish will be used to calculate risk quotients for aquatic-phase CRLF.

Although chronic data are not available for frogs, an ACR (9.6) using acute and chronic data from rainbow trout and acute data from the African clawed frog is used to estimate a chronic endpoint for amphibians to help characterize risk to aquatic-phase CRLF. This results in an African clawed frog NOAEC of 0.64 mg a.i./L (rainbow trout LC_{50} = 1.8 mg a.i./L; rainbow trout NOAEC = 0.187 mg a.i./L; African clawed frog LC_{50} = 6.1 mg a.i./L; rainbow trout ACR = 1.8/0.187 = 9.6; 6.1/9.6 = 0.64).

4.4.3 Toxicity to Aquatic Invertebrates

Aquatic invertebrate toxicity studies are used to assess potential indirect effects to the DS and the CRLF. A summary of acute and chronic freshwater invertebrate data, including data published in the open literature, is provided below in the following Sections.

4.4.3.1. Acute Studies

Aquatic invertebrate toxicity data are used to evaluate potential indirect effects to the CRLF and the DS because each assessed species depends on aquatic invertebrates for food. For the indirect effects assessment, the most sensitive aquatic invertebrate species is initially used for risk estimation, which is consistent with USEPA (2004). The most sensitive aquatic invertebrate tested is the Eastern oyster (*Crassostrea virginica*) (96-hr shell deposition EC_{50} = 1.6 mg a.i./L) (MRID 445243-03). Other freshwater and estuarine/marine invertebrates have similar sensitivity to alachlor when compared to the Eastern oyster. The 48-hr LC_{50} for the freshwater midge (*Chironomus plumosus*) is 2.5 mg a.i./L (MRID 40098001) while the 96-hr LC_{50} for the estauarine/marine mysid (*Americamysis bahia*) is 2.4 mg a.i./L (MRID 445243-02). Therefore, alachlor is classified as moderately toxic to aquatic invertebrates.

The most important food organism for all sizes of the Delta smelt has been reported to be the copepod *Eurytemora affinis* (USFWS, 1995 and 2004), which is a marine copepod. Supplemental toxicity data are available from the open literature for a non-native copepod, *Tigriopus japonicus* (Lee *et al.*, 2008). In this study, conducted with technical grade alachlor, the 96-hr LC_{50} value was 7.3 mg a.i./L.

4.4.3.2. Chronic Exposure Studies

Toxicity data from chronic exposure to alachlor are available for one freshwater [daphnid (*Daphnia magna*)] and one estuarine/marine invertebrate species [copepod (*Tigriopus japonicus*)]. The daphnid study, conducted with technical grade alachlor, resulted in a NOAEC of 0.11 mg a.i./L based on reduced adult length (MRID 437747-07). There was also reduced egg production and reduced adult survival at concentrations of 0.45 mg a.i./L and 1.7 mg a.i./L, respectively. The LOAEC for this study was 0.23 mg a.i./L.

A definitive NOAEC or LOAEC could not be determined in the copepod study because reproductive effects (*i.e.*, an increase in the generation time for adults in the F_0 and F_1 generations) were seen in all of the concentrations tested (E104287, Lee *et al.*, 2008). The lowest concentration tested in the study was 0.0001 mg a.i./L (0.1 µg a.i./L). Therefore, the resulting NOAEC and LOAEC values were <0.0001 mg a.i./L.

4.4.4 Toxicity to Aquatic Plants

Aquatic plant toxicity studies are used as one of the measures of effect to evaluate whether alachlor may affect primary production. Aquatic plants may also serve as dietary items of aquatic-phase CRLFs. In addition, freshwater vascular and non-vascular plant data are used to evaluate a number of the PCEs associated with the critical habitat impact analysis.

Alachlor is toxic to the freshwater green alga (*Pseudokirchneriella subcapitata*, formerly *Selenastrum capricornutum)*, with a 120-hr EC_{50} of 0.00164 mg a.i./L (1.64 µg ai/L) and a NOAEC of 0.00035 mg a.i./L (0.35 µg ai/L), based on reduced cell density (MRID 427638-01). The aquatic vascular plant tested, duckweed (*Lemna gibba*), is almost as sensitive to alachlor as the freshwater green alga [*i.e.*, EC_{50} = 0.0023 mg a.i./L (2.3 µg ai/L); NOAEC = 0.000339 mg a.i./L (0.339 µg ai/L); based on percent inhibition] (MRID 446497-02). The other aquatic plants tested are less sensitive to alachlor when compared to *P. subcapitata* and duckweed [*i.e.*, the freshwater diatom, *Navicula pelliculosa*, has an EC_{50} value of 2.63 mg ai/L (MRID 446497-04); the marine diatom, *Skeletonema costatum,* has an EC_{50} value of 0.21 mg a.i./L (MRID 446497-03); and the cyanobacteria, *Anabaena flos-aquae*, has an EC_{50} value of >19 mg ai/L (446497-01)].

4.5 Toxicity of Alachlor to Terrestrial Organisms

Table 4.3 summarizes the most sensitive terrestrial toxicity endpoints based on an evaluation of both the submitted studies and the open literature. A brief summary of submitted and open literature data considered relevant to this ecological risk assessment for the CRLF and DS is presented below. Additional information is provided in **Appendix C.**

Table 4.3. Terrestrial Toxicity Profile for Alachlor.

Assessment Endpoint	Acute/ Chronic	Species	Toxicity Value Used in Risk Assessment	Slope (95% C.I.)	MRID/ ECOTOX Ref. No.	Comment
Bird (used as a surrogate for terrestrial-phase amphibians)	Acute	Bobwhite quail (*Colinus virginianus*)	LD_{50} = 1499 mg a.i./kg-bw	4.5[1]	00079523	The study is classified as 'acceptable' and was conducted using TGAI
	Sub-chronic	Bobwhite quail (*Colinus virginianus*)	LC_{50} = >5620 mg a.i./kg-diet	Not Applicable (N/A)	430871-01	The study is classified as 'acceptable' and was conducted using TGAI; there were no mortalities during the study
	Chronic	Mallard duck (*Anas platyrhynchos*)	NOAEC = <50 mg a.i./kg-diet	N/A	449515-01	The study is classified as 'supplemental' and was conducted using TGAI; there were significant reductions in hatchling weight at all concentrations tested
Mammal	Acute	Norway rat (*Rattus norvegicus*)	LD_{50} = 930 mg/kg-bw	4.5[1]	00139383	The study is classified as 'acceptable' and was conducted using TGAI
	Chronic	Sprague Dawley rat	NOAEC = 30 mg/kg-diet	N/A	00075062	The study is classified as 'acceptable' and was conducted using TGAI; there were no reproductive effects at the highest dose tested (30 mg/kg-diet); LOAEL = >30 mg/kg-diet
Terrestrial invertebrate	Acute (Contact)	Honey bee (*Apis mellifera*)	LD_{50} >36.3 μg a.i. /bee	4.5[1]	00028772 (Atkins *et al.* 1973)	The study is classified as 'supplemental' since it was conducted using a formulation (Lasso®, 45% a.i.) rather than TGAI; there was only 0.41% mortality at the highest treatment level
Terrestrial plants	N/A (Vegetative vigor)	Monocot (Rye grass – endpoint based on reduced dry weight)	EC_{25} = 0.068 lb a.i./acre NOAEL = 0.037 lb a.i./acre	N/A	424686-01	The study is classified as 'supplemental'; no solvent control was included in the study; TGAI was used instead of a TEP
		Dicot (Cucumber– endpoint based on reduced dry weight)	EC_{25} = 1.4 lb a.i./acre NOAEL = 0.67 lb a.i./acre	N/A		
	N/A (Seedling emergence)	Monocot (Rye grass - endpoint based on reduced dry weight)	EC_{25} = 0.0067 lb a.i./acre NOAEL – 0.0023 lb a.i./acre	N/A	424687-01	The study is classified as 'supplemental'; no solvent control was included in the study; TGAI was used instead of a TEP
		Dicot (Lettuce – endpoint based on phytotoxicity)	EC_{25} = 0.034 lb a.i./acre NOAEL – 0.019 lb a.i./acre	N/A		

[1] A slope was not determined in the study, therefore, the default slope is used.

4.5.1 Toxicity to Birds and Terrestrial Phase Amphibians

As specified in the Overview Document, the Agency uses birds as a surrogate for terrestrial-phase amphibians when sufficient toxicity data for each specific taxonomic group are not available (USEPA, 2004).

4.5.1.1. Birds: Acute and Subacute Exposure (Mortality) Studies

The available data indicate that alachlor is slightly to practically nontoxic to avian species on an acute oral exposure basis; a study with bobwhite quail (*Colinus virginianus*) resulted in an LD_{50} value of 1,499 mg ai/kg-bw (MRID 00079523). Alachlor is also practically nontoxic to bobwhite quail and mallard duck (*Anas platyrhynchos*) on a subacute dietary exposure basis (LC_{50}>5620 mg ai/kg-diet; MRIDs 43087001, 43087101). There were no mortalities attributed to treatment in the subacute dietary studies.

4.5.1.2. Birds: Chronic Exposure (Growth, Reproduction) Studies

An avian reproduction study with the bobwhite quail (MRID 449515-02) resulted in a NOAEC of 50 mg a.i./kg-diet, based on decreased mean hatchling weight (9% reduction at the 150 mg a.i./kg-diet concentration). Mean hatchling weight was also reduced in a reproduction study with mallard ducks. In the mallard duck study (MRID 449515-01), mean hatchling weight was decreased at all of the concentrations tested (range = 5.5% to 19% reduction) (NOAEC<50 mg a.i./kg-diet); therefore, a NOAEC has not been established for avian species or species for which they are surrogates. In the mallard duck study there was also a statistically significant reduction in egg production, embryo viability and hatchability at the 1,000 mg a.i./kg-diet concentration.

4.5.1.3. Toxicity to Reptiles and Terrestrial Phase Amphibians

No data are currently available for the effects of alachlor on reptiles or terrestrial-phase amphibians.

4.5.2 Toxicity to Mammals

4.5.2.1. Mammals: Acute Exposure (Mortality) Studies

Alachlor is slightly toxic to mammals on an acute oral exposure basis. An acute oral study with Norway rats (*Rattus norvegicus*) resulted in an LD_{50} value of 930 mg a.i./kg-bw (MRID 00139383). Additional information can be found in **Appendix J**.

4.5.2.2. Reproduction Toxicity in Mammals

In a 3-generation reproduction study (MRID 00075062) technical grade alachlor was administered to Charles River Sprague-Dawley rats (*R. norvegicus*) in the diet at concentrations of 0, 3, 10, and 30 mg a.i./kg-diet. Each generation was mated twice during the study. No effects on reproductive parameters were observed. Therefore, the reproductive toxicity NOAEC is 30 mg/kg-diet and a LOAEC for reproductive effects was not established in the study. There were some systemic effects at 30 mg/kg-diet [kidney discoloration and decreased kidney weights and lower ovary weights in females of each parental generation and the F_3 females (maximal decrease of 17%)]. Therefore, the systemic toxicity LOAEC in this study was 30 mg/kg-diet.

4.5.3 Toxicity to Terrestrial Invertebrates

Terrestrial invertebrate toxicity data are used to evaluate potential indirect effects to the CRLF and to adversely modify designated critical habitat. A summary of the available terrestrial insect data is provided below. Additional details on the data are included in **Appendix C**.

Alachlor is considered practically nontoxic to honey bees (*Apis mellifera*) on an acute contact exposure basis (MRID 00028772). In this study, adult bees were exposed to a formulated product (Lasso®, 45% active ingredient) at concentrations up to 36.3 µg a.i./bee. At the highest concentration tested, 0.41% of the bees died. This study is classified as supplemental because it was not conducted using technical grade alachlor. Therefore, the LD_{50} value for alachlor and honey bees is >36.3 µg a.i./bee.

4.5.4 Toxicity to Terrestrial Plants

Terrestrial plant toxicity data are used to evaluate the potential for alachlor to affect the riparian zone of occupied water bodies and critical habitat. Riparian zone effects could impact habitat and stream water quality as discussed in detail in Section 5.2.

Plant toxicity data from both registrant-submitted studies and studies in the scientific literature were reviewed for this assessment. Registrant-submitted studies are conducted under conditions and with species defined in EPA toxicity test guidelines. Sub-lethal endpoints such as plant growth, dry weight, and biomass are evaluated for both monocots and dicots, and evaluate effects at both seedling emergence and vegetative life stages. A guideline study generally evaluates toxicity to ten crop species. A drawback to these studies is that they are conducted on herbaceous agricultural crop species only, and extrapolation of effects to other species, such as woody shrubs, trees, and wild herbaceous species contributes uncertainty to risk conclusions.

Commercial crop species have been selectively bred, and may be more or less resistant to particular stressors than wild herbs and forbs. The direction of this uncertainty for specific plants and stressors, including alachlor, is largely unknown. Homogenous test plant seed lots also lack the genetic variation that occurs in natural populations, so the range of effects seen from tests is likely to be smaller than would be expected from wild populations.

Two terrestrial plant studies with alachlor have been submitted to the Agency: a seedling emergence study (MRID 424687-01) and a vegetative vigor study (MRID 424686-01). Both studies are scientifically sound but do not meet the requirements for Tier 2 seedling emergence or vegetative vigor tests using non-target plants. The report did not state if the control pots were treated with a 75% acetone, 25% deionized water solution for the emergence test or a 1% acetone/deionized water solution for the vegetative vigor test. Additionally, only one parameter was monitored during the vegetative vigor study and the NOAELs for height and weight for cabbage were not determined for the emergence study. The EC_{25} values for onion height and tomato dry weight were not determined for the emergence study. These studies are classified as 'supplemental'.

Based on the results of the submitted terrestrial plant toxicity studies, alachlor is phytotoxic to many plant species. The herbicide reduces plant height, weight and survival. Annual rye grass (*Lolium perenne*), a monocotyledonous species, was the most sensitive species in both the vegetative vigor and seedling emergence studies. In the seedling emergence study, the EC_{25} was 0.0067 lb a.i./acre (NOAEL= 0.0023 lb a.i./acre) based on biomass reduction. In the vegetative vigor study, the EC_{25} was 0.068 lb a.i./acre (NOAEL=0.037 lb a.i./acre) based on biomass reduction. The most sensitive dicot in the seedling emergence study was lettuce (*Lactuca sativa*) based on phytotoxicity, with an EC_{25} of 0.034 lbs a.i./acre and a NOAEC of 0.019 lbs a.i./acre. The most sensitive dicot in the vegetative vigor study was cucumber (*Cucumis sativus*) based on biomass reduction, with an EC_{25} of 1.4 lbs a.i./acre and a NOAEC of 0.67 lbs a.i./acre.

Based on the results of the submitted terrestrial plant toxicity tests, it appears that emerged seedlings are more sensitive to alachlor via soil/root uptake exposure than emerged plants via foliar routes of exposure. However, all tested plants, with the exception of soybeans (*Glycine max*) in the seedling emergence study, exhibited adverse effects following exposure to alachlor.

4.6. Use of Probit Slope Response Relationship to Provide Information on the Endangered Species Levels of Concern

The Agency uses the probit dose-response relationship as a tool for providing additional information on the potential for acute direct effects to individual listed species and aquatic animals that may indirectly affect the listed species of concern (USEPA, 2004). As part of the risk characterization, an interpretation of acute RQs for listed species is discussed. This interpretation is presented in terms of the chance of an individual event (*i.e.*, mortality or immobilization) should exposure at the EEC actually occur for a species with sensitivity to alachlor on par with the acute toxicity endpoint selected for RQ calculation. To accomplish this interpretation, the Agency uses the slope of the dose response relationship available from the toxicity study used to establish the acute toxicity measures of effect for each taxonomic group that is relevant to this assessment. The individual effects probability associated with the acute RQ is based on the mean estimate of the slope and an assumption of a probit dose response relationship. In addition to a single effects probability estimate based on the mean, upper and lower estimates of the effects probability are also provided to account for variance in the slope, if available.

Individual effect probabilities are calculated based on an Excel spreadsheet tool IECV1.1 (Individual Effect Chance Model Version 1.1) developed by the USEPA, OPP, Environmental Fate and Effects Division (June 22, 2004). The model allows for such calculations by entering the mean slope estimate (and the 95% confidence bounds of that estimate) as the slope parameter for the spreadsheet. In addition, the acute RQ is entered as the desired threshold.

For the acute toxicity endpoints used in this assessment, the only study that allowed for the determination of a slope was the estauarine/marine invertebrate (mysid) study (MRID 445243-02). This study resulted in a slope of 8.7 (95% C.I. = 5.1 – 12.4). For the remaining taxa, the default slope is used to estimate individual effect probabilities [*i.e*., slope = 4.5 (95% C.I. = 2 – 9)].

4.7 Incident Database Review

A review of the EIIS database for ecological incidents involving alachlor was completed in January 2009. Based on the EIIS database, there have been a total of 43 reported ecological incidents potentially involving alachlor (9 involving aquatic animals and 34 involving terrestrial plants). These incidents are summarized below. A more complete list of the incidents involving alachlor is included as **Appendix K**.

The nine reported alachlor aquatic animal incidents occurred between 1983 and 1995 and involved from an ‘unknown’ number of dead freshwater fish to ‘thousands’. The legality of use was undetermined in four of the incidents; involved a misuse in three incidents (two intentional and one accidental); and involved a registered use in two incidents. Other chemicals besides alachlor, including other herbicides and/or insecticides, were involved in all but one of the reported incidents involving aquatic animals. The certainty that alachlor was responsible for the fish deaths ranged from ‘unlikely’ (two incidents) to ‘possible’ (four incidents) to ‘highly probable’ (three incidents).

The 34 reported alachlor terrestrial plant incidents occurred between 1991 and 2002 and involved from an ‘unknown’ number of impacted acres to 1,792 acres. The legality of use was undetermined in seven of the incidents; involved an accidental misuse in four incidents; and involved a registered alachlor use in 23 incidents. Other herbicides besides alachlor were involved in all but 11 of the reported incidents involving terrestrial plants (*i.e.*, 11 of the incidents involved only alachlor and no other chemicals). The certainty that alachlor was responsible for the plant damage ranged from ‘possible’ (28 incidents) to ‘probable’ (six incidents).

5.0 Risk Characterization

Risk characterization is the integration of the exposure and effects characterizations. Risk characterization is used to determine the potential for direct and/or indirect effects to the CRLF and the DS or modification to their designated critical habitat from the use of alachlor. The risk characterization provides an estimation (Section 5.1) and a description (Section 5.2) of the likelihood of adverse effects; articulates risk assessment assumptions, limitations, and uncertainties; and synthesizes an overall conclusion regarding the likelihood of adverse effects to the assessed species or their designated critical habitat (*i.e*., “no effect,” “likely to adversely affect,” or “may affect, but not likely to adversely affect”).

5.1 Risk Estimation

Risk is estimated by calculating the ratio of the estimated environmental concentration (EEC) (from PRZM/EXAMS for aquatic organisms, T-REX for terrestrial animals, and TerrPlant for terrestrial plants) (Section 3) and the appropriate toxicity endpoint (Section 4). This ratio is the risk quotient (RQ), which is then compared to pre-established acute and chronic levels of concern (LOCs) for each category evaluated (**Appendix F**).

In cases where the baseline RQ exceeds one or more LOC (*i.e.*, "may affect"), additional factors, including the life history characteristics of the assessed species, refinement of the baseline EECs using site-specific information, and available monitoring data are considered and used to characterize the potential for alachlor to adversely affect the assessed species and/or their designated critical habitat. Risk quotients used to evaluate potential direct and indirect effects to the CRLF and DS and to designated critical habitat are in Sections 5.1.1 (direct effects) and 5.1.2 (indirect effects). RQs are described and interpreted in the context of an effects determination in Section 5.2 (risk description).

5.1.1 Direct Effects RQs

The species considered in this risk assessment include a frog and a fish species. Direct effects to the DS are evaluated using the lowest acute and chronic toxicity values across freshwater and saltwater fish species. Direct effects to the aquatic phase CRLF are evaluated using the lowest freshwater acute and chronic toxicity values across fish and amphibian toxicity studies. However, fish were consistently shown to be more sensitive than aquatic-phase amphibians and the available amphibian studies are classified as 'supplemental'; therefore, fish acute and chronic toxicity values are used to calculate RQs for aquatic-phase amphibians. Direct effects to terrestrial-phase CRLFs are evaluated using the lowest acute and chronic toxicity values for birds exposed to alachlor (since no terrestrial-phase amphibian toxicity data were available for alachlor). Toxicity values used to calculate RQs are discussed in Section 4, and exposure values are discussed in Section 3. RQs used to estimate acute and chronic direct effects are in **Tables 5.1** (DS and aquatic-phase CRLF) and **5.2** (terrestrial-phase CRLF).

Table 5.1. Summary of Aquatic RQs Used to Estimate Direct Effects to Aquatic-Phase CRLF and the DS[1]

Use Site	Exposure Type	EEC	RQ
Corn (broadcast)	Acute	Peak = 44.8 µg/L	0.02
	Chronic	60-day = 41.1 µg/L	0.22
Corn (incorporated)	Acute	Peak = 12.6 µg/L	0.01
	Chronic	60-day = 11.5 µg/L	0.06
Sweet corn (broadcast)	Acute	Peak = 11.3 µg/L	0.01
	Chronic	60-day = 9.8 µg/L	0.05
Sweet corn (incorporated)	Acute	Peak = 3.2 µg/L	0.002
	Chronic	60-day = 2.8 µg/L	0.01
Sorghum (broadcast)	Acute	Peak = 46.7µg/L	0.03
	Chronic	60-day = 42.7 µg/L	0.23
Sorghum (incorporated)	Acute	Peak = 12.8 µg/L	0.01
	Chronic	60-day = 11.7 µg/L	0.06
Soybeans (broadcast)	Acute	Peak = 32.9 µg/L	0.02
	Chronic	60-day = 27.1 µg/L	0.14
Soybeans, dry beans, lima beans (incorporated)	Acute	Peak = 9.3 µg/L	0.01
	Chronic	60-day = 7.7 µg/L	0.04
Woody ornamentals (nursery-use)	Acute	Peak = 56.0 µg/L	0.03
	Chronic	60-day = 43.0 µg/L	0.23
Woody ornamentals (residential use)	Acute	Peak = 6.3 µg/L	0.004
	Chronic	60-day = 4.6 µg/L	0.02
Cotton (broadcast)	Acute	Peak = 25.2 µg/L	0.01
	Chronic	60-day = 22.8 µg/L	0.12
Cotton (incorporated)	Acute	Peak = 15.3 µg/L	0.01
	Chronic	60-day = 13.8 µg/L	0.07
Sunflowers (broadcast)	Acute	Peak = 44.8 µg/L	0.02
	Chronic	60-day = 41.1 µg/L	0.22
Sunflowers (incorporated)	Acute	Peak = 27.3 µg/L	0.02
	Chronic	60-day = 25.0 µg/L	0.13
Peanuts (broadcast)	Acute	Peak = 43.9 µg/L	0.02
	Chronic	60-day = 36.2 µg/L	0.19
Peanuts (incorporated)	Acute	Peak = 12.4 µg/L	0.01
	Chronic	60-day = 10.2 µg/L	0.05

[1] Based on an LC_{50} of 1,800 µg a.i./L (Rainbow Trout) and a NOAEC of 187 µg a.i./L (Rainbow Trout).

None of the RQs for any use exceed the Agency's acute or chronic risk LOC for listed fish. Fish are used as a surrogate species for aquatic-phase CRLFs. These RQs are further characterized in the context of the effects determination in Section 5.2.

Table 5.2. Summary of RQs Used to Estimate Direct Effects to Terrestrial-Phase CRLFs (Upper Bound Kenaga Values, Dose-Based for 20g Bird that Eats Small Insects)[1].

Use Site	Appl. Rate (lb a.i./acre)	Exposure Type	EEC (ppm)	RQ
Flowable Soil Applications				
Corn Sorghum Peanuts Woody ornamentals Sunflowers	4	Acute	615	**0.57**
		Chronic		>**10.8**
Soybeans Dry beans Lima beans	3	Acute	461	**0.43**
		Chronic		>**8.1**
Cotton	2	Acute	308	**0.28**
		Chronic		>**5.4**
Impregnated Bulk Fertilizer Surface Applications[2]				
Corn Sorghum	4	Acute	42 mg a.i./ft^2	**1.93**
Soybeans	3	Acute	31 mg a.i./ft^2	**1.45**

[1] Based on an LD_{50} of 1,499 mg/kg-bw (Bobwhite Quail) and a NOAEC of <50 mg/kg-diet (Mallard Duck)

[2] Treated as a granular formulation for modeling purposes

- Bolded RQs exceed the acute or chronic listed species LOC for birds (0.1 and 1, respectively)

Avian RQs exceed the endangered species LOC of 0.1 for acute risk and 1.0 for chronic risk for all of the alachlor uses modeled (both flowable and the impregnated bulk fertilizer applications). Birds are used as surrogate species for terrestrial-phase CRLFs. These RQs are further characterized in the context of the effects determination in Section 5.2.

5.1.2 Indirect Effects

This section presents RQs used to evaluate the potential for alachlor to induce indirect effects. Pesticides have the potential to exert indirect effects upon listed species by inducing changes in structural or functional characteristics of affected communities. Perturbation of forage or prey availability and alteration of the extent and nature of habitat are examples of indirect effects. A number of these indirect effects are also considered as part of the critical habitat modification evaluation. In conducting a screen for indirect effects, direct effects LOCs for each taxonomic group (*e.g.*, freshwater fish, invertebrates, aquatic plants, and terrestrial plants) are employed to make inferences concerning the potential for indirect effects upon listed species that rely upon non-listed organisms in these taxonomic groups as resources critical to its life cycle (USEPA, 2004). This approach used to evaluate indirect effects to listed species is endorsed by the Services (USFWS/NMFS, 2004). If no direct risk to listed species LOCs are exceeded for organisms on which the assessed species depends for survival or reproduction, indirect effects are not expected to occur.

If LOCs are exceeded for organisms on which the assessed species depends for survival or reproduction, dose-response analysis is used to estimate the potential magnitude of effect associated with an exposure equivalent to the EEC. The greater the probability that exposures will produce effects on a taxa, the greater the concern for potential indirect effects for listed species dependant upon that taxa (USEPA, 2004).

As an herbicide, indirect effects to the assessed species from potential effects on primary productivity of aquatic plants are a principle concern. If plant RQs fall between the risk to endangered species and non-endangered species LOCs, a no effect determination is made for listed species that rely on multiple plant species to successfully complete their life cycle (termed plant dependent species). If plant RQs are above risk to non-endangered species LOCs, this could be indicative of a potential for adverse effects to those listed species that rely either on a specific plant species (plant species obligate) or multiple plant species (plant dependant) for some important aspect of their life cycle (USEPA, 2004). Based on the information provided in Section 2.3, the assessed species do not have any known obligate relationship with a specific species of aquatic plant.

Direct effects to riparian zone vegetation may also indirectly affect the assessed species by reducing water quality and available spawning habitat via increased sedimentation. Direct impacts to the terrestrial plant community (*i.e*., riparian habitat) are evaluated using submitted terrestrial plant toxicity data. If terrestrial plant RQs exceed the Agency's LOC for direct risk to non-endangered plant species, based on EECs derived using EFED's Terrplant model (Version 1.2.1), a conclusion that alachlor may affect the CRLF and DS via potential indirect effects to the riparian habitat (and resulting impacts to habitat due to increased sedimentation) is made. Further analysis of the potential for alachlor to affect the CRLF and the DS via reduction in riparian habitat includes a description of the importance of riparian vegetation to the assessed species and types of riparian vegetation that may potentially be impacted by alachlor use within the action area.

RQs used to evaluate the potential for alachlor to induce indirect effects to the assessed species are presented in Sections 5.1.2.1 to 5.1.2.4. These RQs suggest that potential indirect effects could occur by potentially impacting food availability and primary productivity as indicated by LOC exceedances. These RQs were based on the most sensitive surrogate species tested across aquatic invertebrate, fish, and aquatic plant species tested. Discussion of these RQs in the context of this effects determination is presented in Section 5.2.

5.1.2.1. Aquatic Invertebrates

Aquatic invertebrate RQs are summarized in **Table 5.3** and are used to evaluate the potential for alachlor to affect the CRLF and the DS by potentially impacting the food supply. Both the CRLF and the DS consume aquatic invertebrates as part of their diet. Acute risk quotients for invertebrates were based on peak EECs in the standard pond and the lowest acute toxicity value for freshwater and saltwater invertebrates. Chronic risk was based on 21-day EECs and the lowest chronic toxicity value for freshwater and saltwater invertebrates.

Table 5.3. Summary of Acute and Chronic RQs for Aquatic Invertebrates Used to Evaluate Potential Indirect Effects to the CRLF and the DS Resulting from Potential Impacts to Food Supply.

Use Site	Exposure Type	EEC	RQ (Freshwater Invertebrates)[1]	RQ (Estuarine/ Marine Invertebrates)[2]
Corn (broadcast)	Acute	Peak = 44.8 µg/L	0.02	0.02
	Chronic	21-day = 43.7 µg/L	**1.2**	**>437**
Corn (incorporated)	Acute	Peak = 12.6 µg/L	0.01	0.01
	Chronic	21-day = 12.3 µg/L	0.34	**>123**
Sweet corn (broadcast)	Acute	Peak = 11.3 µg/L	0.005	0.005
	Chronic	21-day = 10.7 µg/L	0.30	**>107**
Sweet corn (incorporated)	Acute	Peak = 3.2 µg/L	0.001	0.001
	Chronic	21-day = 3.1 µg/L	0.09	**>31**
Sorghum (broadcast)	Acute	Peak = 46.7µg/L	0.02	0.02
	Chronic	21-day = 45.6 µg/L	**1.3**	**>456**
Sorghum (incorporated)	Acute	Peak = 12.8 µg/L	0.01	0.01
	Chronic	21-day = 12.5 µg/L	0.35	**>125**
Soybeans (broadcast)	Acute	Peak = 32.9 µg/L	0.01	0.01
	Chronic	21-day = 31.9 µg/L	0.89	**>319**
Soybeans, dry beans, lima beans (incorporated)	Acute	Peak = 9.3 µg/L	0.004	0.004
	Chronic	21-day = 9.0 µg/L	0.25	**>90**
Woody ornamentals (nursery-use)	Acute	Peak = 56.0 µg/L	0.02	0.02
	Chronic	21-day = 54.3 µg/L	**1.5**	**>543**
Woody ornamentals (residential use)	Acute	Peak = 6.3 µg/L	0.003	0.003
	Chronic	21-day = 5.5 µg/L	0.15	**>55**
Cotton (broadcast)	Acute	Peak = 25.2 µg/L	0.01	0.01
	Chronic	21-day = 24.4 µg/L	0.68	**>244**
Cotton (incorporated)	Acute	Peak = 15.3 µg/L	0.01	0.01
	Chronic	21-day = 14.8 µg/L	0.41	**>148**
Sunflowers (broadcast)	Acute	Peak = 44.8 µg/L	0.02	0.02
	Chronic	21-day = 43.7 µg/L	**1.2**	**>437**
Sunflowers (incorporated)	Acute	Peak = 27.3 µg/L	0.01	0.01
	Chronic	21-day = 26.6 µg/L	0.74	**>266**
Peanuts (broadcast)	Acute	Peak = 43.9 µg/L	0.02	0.02
	Chronic	21-day = 42.5 µg/L	**1.2**	**>425**
Peanuts (incorporated)	Acute	Peak = 12.4 µg/L	0.005	0.01
	Chronic	21-day = 12.0 µg/L	0.33	**>120**

1 Based on *Chironomus plumosus* endpoints (EC_{50} = 2,500 µg a.i./L; NOAEC = 36 µg a.i./L).
2 Based on LC_{50} = 2,400 µg a.i./L (*Americamysis bahia*) and NOAEC < 0.1 µg a.i./L (*Tigriopus japonicus*)
Bolded numbers exceed the Agency's listed species LOCs (RQ > acute endangered species LOC of 0.05 and the chronic LOC of 1.0).

None of the acute RQs for freshwater or estuarine/marine invertebrates exceed the risk to endangered species LOC (0.05). The freshwater invertebrate RQs exceed the chronic risk LOC of 1.0 for the corn (broadcast) (RQ = 1.2), sorghum (broadcast) (RQ = 1.3), woody ornamentals (nursery-use) (RQ = 1.5), sunflower (broadcast) (RQ = 1.2), and the peanut (broadcast) (RQ = 1.2) The Agency's chronic risk LOC is exceeded for saltwater invertebrates for all uses modeled [range : >31 (sweet corn, incorporated use) - >543 (woody ornamentals)]. These RQs were based on the most sensitive surrogate species across aquatic invertebrate species tested. Discussion of these RQs in the context of this effects determination is presented in Section 5.2.

5.1.2.2. Terrestrial Invertebrates

Terrestrial invertebrate RQs are used to evaluate the potential for alachlor to affect the CRLF by potentially impacting their food supply. Terrestrial invertebrate RQs are presented in **Table 5.4**.

Table 5.4. Summary of Acute RQs for Terrestrial Invertebrates on the Site of Application Used to Evaluate Potential Indirect Effects to the CRLF Resulting from Potential Impacts to the Food Supply.

Use	Application Rate (lb a.i./acre)	Size Class	EEC (ppm)	RQ[1]
Corn	4	Small insect	540	<**1.9**
Sorghum				
Peanuts				
Woody ornamentals		Large insect	60	<**0.21**
Sunflowers				
Soybeans	3	Small insect	405	<**1.4**
Dry beans				
Lima beans		Large insect	45	<**0.16**
Cotton	2	Small insect	270	<**0.95**
		Large insect	30	<**0.11**

[1] Available acute contact toxicity data for bees exposed to alachlor (in units of µg a.i./bee) are converted to µg a.i./g (of bee) by multiplying by 1 bee/0.128 g (LD_{50} = >36.3 µg a.i./bee = >283.6 µg/g).
- Bolded RQs potentially exceed the interim risk to listed species LOC for terrestrial invertebrates (0.05).

Because the available toxicity data for honey bees resulted in only 0.41% mortality at the highest concentration tested (36.3 µg a.i./bee), a definitive LD_{50} value was not determined. Therefore, although the estimated RQs for all alachlor uses are potentially above the Agency’s interim risk to listed species LOC for terrestrial invertebrates (0.05), the actual RQs would likely be lower than those reported in **Table 5.4**. However, it is not clear if the actual RQs would be above or below the Agency’s LOC without definitive data, therefore, risks cannot be precluded at this time. Discussion of these RQs in the context of this effects determination is presented in Section 5.2.

5.1.2.3. Mammals and Amphibians

Potential risks to mammals are derived using T-REX and acute and chronic rat toxicity data. RQs are typically derived for various sizes of mammals (15 g, 35 g, and 1000 g); however, RQs are not presented for 1000 g mammals because it is improbable that even the largest CRLF would consume a mammal of that size. Therefore, the evaluation for potential indirect effects to the CRLF resulting from potential reductions in mammal abundance as food is based on the 15 g size class, which results in higher RQs than the 35 g mammal. The California mouse (*Peromyscus californicus*) is a particular species known to be consumed by the CRLF. The California mouse is omnivorous and consumes grasses, fruits, flowers, and invertebrates (USC, 2005; http://wotan.cse.sc.edu/perobase/systematics/p_calif.htm). Therefore, the short grass food item was used to determine if mammals could be impacted; however, RQs based on EECs on other food items were also derived for characterization purposes. A range of RQs for mammals is presented in **Table 5.5** (acute) and **Table 5.6** (chronic) (see also **Appendix L**).

Table 5.5. Summary of Acute RQs for 15 g Mammals (LD_{50} = 930 mg/kg-bw) Used to Evaluate Potential Indirect Effects to the CRLF Resulting from Potential Impacts to the Food Supply.

Use Site	Appl. Rate (lb a.i./acre)	Dietary Category	EEC (ppm)	RQ
Flowable Post-Plant, Pre-Plant, Pre-Emergence, and Burndown Applications [1]				
Corn Sorghum Peanuts Woody ornamentals Sunflowers	4	Short grass	915	**0.45**
		Tall grass	420	**0.21**
		Broadleaf plants/small insects	515	**0.25**
		Fruits/pods/large insects	57	0.03
		Seeds	13	0.01
Soybeans Dry beans Lima beans	3	Short grass	686	**0.34**
		Tall grass	315	**0.15**
		Broadleaf plants/small insects	386	**0.19**
		Fruits/pods/large insects	43	0.02
		Seeds	10	0.00
Cotton	2	Short grass	458	**0.22**
		Tall grass	210	**0.10**
		Broadleaf plants/small insects	257	**0.13**
		Fruits/pods/large insects	29	0.01
		Seeds	6	0
Flowable Pre-Plant and Pre-Emergence Bare Soil Applications [2]				
Corn Sorghum Peanuts Woody ornamentals Sunflowers	4 (3.42, it would be lower for sunflowers)[3]	Short grass	783	**0.38**
		Tall grass	359	**0.18**
		Broadleaf plants	440	**0.22**
		Fruits/pods	49	0.02
		Seeds	11	0.01
		Small insects	515	**0.25**
		Large insects	57	0.03
Soybeans Dry beans Lima beans	3 (2.57)[3]	Short grass	588	**0.29**
		Tall grass	270	**0.13**
		Broadleaf plants	331	**0.16**
		Fruits/pods	37	0.02
		Seeds	8	0
		Small insects	386	**0.19**
		Large insects	43	0.02
Cotton	2 (0.53)[3]	Short grass	121	0.06
		Tall grass	56	0.03
		Broadleaf plants	68	0.03
		Fruits/pods	8	0
		Seeds	2	0
		Small insects	257	**0.13**
		Large insects	29	0.01
Impregnated Bulk Fertilizer Surface Applications [4]				
Corn Sorghum	4	Not Applicable	42 mg a.i./ft^2	**1.36**
Soybeans	3	Not Applicable	31 mg a.i./ft^2	**1.02**

[1] Estimated residiues for potential food items found on the site of application.
[2] Estimated residiues for potential food items found immediately adjacent to the site of application (except for small and large insects which may be exposed to alachlor on the site of application).
[3] AgDRIFT was run, to estimate an 'application rate' 1 ft off the site of application, using the default settings in Tier 1. Except for the sunflower and cotton uses that are only on the label that has spray drift restrictions (ASAE medium-coarse droplet size distribution; ≤10 mph wind speed; maximum 4-ft boom height). Only ground applications were modeled for all uses.
[4] Treated as a granular formulation for modeling purposes
- Bolded RQs exceed the acute risk LOC for listed mammals (0.1)

On the site of application, the RQs for 15 g mammals that eat short grass, tall grass, and broadleaf plants/small insects exceed the Agency's acute risk LOC for listed mammals for all alachlor uses (flowable and bulk fertilizer applications). Immediately adjacent to application sites, the RQs exceed the Agency's acute risk LOC for listed mammals that eat short grass, tall grass, and broadleaf plants/small insects for all uses except for cotton.

The only RQs that exceed the Agency's acute risk LOC for non-listed mammals are for the impregnated bulk fertilizer applications (corn, sorghum, and soybeans). The acute risk LOC for non-listed species is not exceeded for any other alachlor use/application combination.

Table 5.6. Summary of Chronic RQs for 15 g Mammals (NOAEC = 30 mg/kg-diet) Used to Evaluate Potential Indirect Effects to the CRLF Resulting from Potential Impacts to the Food Supply.

Use Site	Appl. Rate (lb a.i./acre)	Dietary Category	Chronic Dose-Based RQ	Chronic dietary-Based RQ
Flowable Post-Plant, Pre-Plant, Pre-Emergence, and Burndown Applications[1]				
Corn Sorghum Peanuts Woody ornamentals Sunflowers	4	Short grass	**278**	**32**
		Tall grass	**127**	**15**
		Broadleaf plants/small insects	**156**	**18**
		Fruits/pods/large insects	**17**	**2**
		Seeds	**3.9**	**2**
Soybeans Dry beans Lima beans	3	Short grass	**208**	**24**
		Tall grass	**95**	**11**
		Broadleaf plants/small insects	**117**	**14**
		Fruits/pods/large insects	**13**	**1.5**
		Seeds	**2.9**	**1.5**
Cotton	2	Short grass	**138**	**16**
		Tall grass	**64**	**7**
		Broadleaf plants/small insects	**78**	**9**
		Fruits/pods/large insects	**9**	**1**
		Seeds	**1.9**	**1**
Flowable Pre-Plant and Pre-Emergence Bare Soil Applications[2]				
Corn Sorghum Peanuts Woody ornamentals Sunflowers	4 (3.42, it would be lower for sunflowers)[2]	Short grass	**237**	**27**
		Tall grass	**109**	**13**
		Broadleaf plants	**134**	**15**
		Fruits/pods	**15**	**2**
		Seeds	**3**	**2**
		Small insects	**156**	**18**
		Large insects	**17**	**2**
Soybeans Dry beans Lima beans	3 (2.57)[2]	Short grass	**178**	**21**
		Tall grass	**82**	**9**
		Broadleaf plants	**100**	**12**
		Fruits/pods	**11**	**1**
		Seeds	**2**	**1**
		Small insects	**117**	**14**
		Large insects	**13**	**1.5**
Cotton	2 (0.53)[2]	Short grass	**37**	**4**
		Tall grass	**17**	**1.9**
		Broadleaf plants	**21**	**2.4**
		Fruits/pods	**2**	0.3
		Seeds	0.5	0.3
		Small insects	**78**	**9**
		Large insects	**9**	**1**

[1] Estimated residiues for potential food items found on the site of application.
[2] Estimated residiues for potential food items found immediately adjacent to the site of application (except for small and large insects which may be exposed to alachlor on the site of application). AgDRIFT was run using the default settings in Tier 1. Except for the sunflower and cotton uses that are only on the label that has spray drift restrictions (ASAE medium-coarse droplet size distribution; ≤10 mph wind speed; maximum 4-ft boom height). Only ground applications were modeled for all uses.
- Bolded RQs exceed (or are near) the chronic listed species LOC for mammals (1)

The chronic RQs (all dietary categories; both dose- and dietary-based) for all of the uses modeled exceed the Agency's chronic risk for listed species LOC of 1, except for the RQs for the cotton use (applications to bare soil) and the fruits/pods (dietary-based) and seeds (dose- and dietary-based) dietary categories.

For potential terrestrial-phase amphibian prey items, birds are used as a surrogate. As discussed above for direct effects to terrestrial-phase CRLF, avian RQs exceed the endangered species LOC of 0.1 for acute risk and 1.0 for chronic risk for all of the alachlor uses modeled (both flowable and the impregnated bulk fertilizer applications).

Based on acute and chronic risk LOC exceedances, there is potential for alachlor to impact mammal and amphibian abundance, which could result in indirect effects to the CRLF. Discussion of these RQs in the context of this effects determination is presented in Section 5.2.

5.1.2.4 Aquatic and Terrestrial Plants

Aquatic plants serve as food supply for both the CRLF and the DS and can impact water quality. Additionally, effects to terrestrial plants can impact terrestrial habitat quality and water quality parameters. Therefore, RQs for vascular and non-vascular aquatic plants are used to evaluate the potential for alachlor to affect the CRLF and/or the DS by potentially impacting the food supply and water quality, and, thus, habitat (**Table 5.7**). RQs for terrestrial plants are used to evaluate the potential for alachlor to impact aquatic habitats (*i.e.*, water quality) (aquatic-phase CRLF and DS) and/or terrestrial habitats (terrestrial-phase CRLF) (**Table 5.8**).

Table 5.7. Summary of Acute RQs for Aquatic Plants Used to Evaluate Potential Indirect Effects to the CRLF and DS.

Use Site	Plant Type	EEC	RQ
Corn (broadcast)	Non-vascular aquatic plant[1]	Peak = 44.8 µg/L	**27.3**
	Vascular aquatic plant[2]		**19.5**
Corn (incorporated)	Non-vascular aquatic plant[1]	Peak = 12.6 µg/L	**7.7**
	Vascular aquatic plant[2]		**5.4**
Sweet corn (broadcast)	Non-vascular aquatic plant[1]	Peak = 11.3 µg/L	**6.9**
	Vascular aquatic plant[2]		**4.9**
Sweet corn (incorporated)	Non-vascular aquatic plant[1]	Peak = 3.2 µg/L	**2.0**
	Vascular aquatic plant[2]		**1.4**
Sorghum (broadcast)	Non-vascular aquatic plant[1]	Peak = 46.7 µg/L	**28.5**
	Vascular aquatic plant[2]		**20.3**
Sorghum (incorporated)	Non-vascular aquatic plant[1]	Peak = 12.8 µg/L	**7.8**
	Vascular aquatic plant[2]		**5.6**
Soybeans (broadcast)	Non-vascular aquatic plant[1]	Peak = 32.9 µg/L	**20.1**
	Vascular aquatic plant[2]		**14.3**
Soybeans, dry beans, lima beans (incorporated)	Non-vascular aquatic plant[1]	Peak = 9.3 µg/L	**5.7**
	Vascular aquatic plant[2]		**4.0**
Woody ornamentals (nursery-use)	Non-vascular aquatic plant[1]	Peak = 56.0 µg/L	**34.1**
	Vascular aquatic plant[2]		**24.3**
Woody ornamentals (residential use)	Non-vascular aquatic plant[1]	Peak = 6.3 µg/L	**3.8**
	Vascular aquatic plant[2]		**2.7**
Cotton (broadcast)	Non-vascular aquatic plant[1]	Peak = 25.2 µg/L	**15.4**
	Vascular aquatic plant[2]		**11.0**
Cotton (incorporated)	Non-vascular aquatic plant[1]	Peak = 15.3 µg/L	**9.3**
	Vascular aquatic plant[2]		**6.7**
Sunflowers (broadcast)	Non-vascular aquatic plant[1]	Peak = 44.8 µg/L	**27.3**
	Vascular aquatic plant[2]		**19.5**
Sunflowers (incorporated)	Non-vascular aquatic plant[1]	Peak = 27.3 µg/L	**16.6**
	Vascular aquatic plant[2]		**11.9**
Peanuts (broadcast)	Non-vascular aquatic plant[1]	Peak = 43.9 µg/L	**26.8**
	Vascular aquatic plant[2]		**19.1**
Peanuts (incorporated)	Non-vascular aquatic plant[1]	Peak = 12.4 µg/L	**7.6**
	Vascular aquatic plant[2]		**5.4**

1 Based on an EC_{50} = 1.64 µg a.i./L (*Selenastrum capricornutum*)
2 Based on an EC_{50} = 2.3 µg a.i./L (*Lemna gibba*)
Bolded numbers exceed the Agency's aquatic plant LOC (RQ < 1.0).

The RQs for non-vascular and vascular aquatic plants exceed the Agency's LOCs for all uses modeled [non-vascular plant RQ range = 2.0 (sweet corn, incorporated use) – 34.1 (woody ornamental use); vascular plant RQ range = 1.4 (sweet corn, incorporated use) – 24.3 (woody ornamental use).

Potential indirect effects resulting from effects on terrestrial vegetation were assessed using RQs from terrestrial plant seedling emergence and vegetative vigor EC_{25} data as a screen. Based on the results of the submitted terrestrial plant toxicity tests, emerging seedlings are more sensitive to alachlor via soil/root uptake than emerged plants via foliar routes of exposure. Therefore, the seedling emergence data were used to estimate terrestrial plant RQs for alachlor use. RQs used to estimate potential indirect effects to the CRLF and/or the DS from potential effects to terrestrial plants within their habitat areas are summarized in **Tables 5.8**.

Table 5.8. Non-Listed Species Terrestrial Plant RQs for Alachlor Use in California[1.]

Use	Application Type	ADJACENT UPLAND		ADJACENT WETLAND		DRIFT ONLY	
		Monocot	Dicot	Monocot	Dicot	Monocot	Dicot
4 lb a.i./acre							
Corn Sorghum Peanuts Woody ornamentals Sunflowers	Soil Surface	**36**	**7**	**304**	**60**	**6**	**1**
Corn Sorghum Peanuts Sunflowers	Soil Incorporated (2-inch incorporation)	**21**	**4**	**155**	**31**	**6**	**1**
Corn Sorghum	Impregnated Bulk Fertilizer[2]	**15**	**3**	**149**	**29**	<0.1	<0.1
3 lb a.i./acre							
Soybeans	Soil Surface	**27**	**5**	**228**	**45**	**4**	0.9
Soybeans Dry beans Lima beans	Soil Incorporated (2-inch incorporation)	**16**	**3**	**116**	**23**	**4**	0.9
Soybeans	Impregnated Bulk Fertilizer[2]	**11**	**2**	**112**	**22**	<0.1	<0.1
2 lb a.i./acre							
Cotton	Soil Surface	**9**	**2**	**76**	**15**	**1**	0.3
	Soil Incorporated (2-inch incorporation)	**5**	**1**	**39**	**8**	**1**	0.3

[1] Based on the following: monocots - EC_{25} = 0.0067 lb a.i./acre (seedling emergence) and EC_{25} = 0.068 lb a.i./acre (vegetative vigor) in annual dry grass (*Lolium perenne*); dicots - EC_{25} = 0.034 lbs a.i./acre [seedling emergence, lettuce (*Lactuca sativa*)] and EC_{25} = 1.4 lbs a.i./acre [vegetative vigor, cucumber (*Cucumis sativus*)].
[2] Impregnated bulk fertilizer applications are treated as granular applications for modeling purposes.
- Bolded RQs exceed the Agency's non-listed species LOC for terrestrial plants (RQ = 1).

Monocots are more sensitive to alachlor than are dicots, based on available data. However, for adjacent upland and wetland plants, terrestrial plant RQs for both monocots and dicots exceed the Agency's risk to non-listed species LOC for all alachlor uses and application types (RQs range from 1 – 60 for dicots; and 5 – 304 for monocots). For the drift only RQs (range = <0.1 – 6), all of the RQs exceed the Agency's LOC except for the impregnated bulk fertilizer applications (all uses) and the dicot RQs for the cotton use (all application types).

Therefore, LOCs were exceeded for both aquatic and terrestrial plants, which could result in indirect effects to the CRLF or the DS. These LOCs and their impact on the effects determination are described in Section 5.2.

5.1.3 Primary Constituent Elements of Designated Critical Habitat

For alachlor use, the assessment endpoints for designated critical habitat PCEs involve the same endpoints as those being assessed relative to the potential for direct and indirect effects to the listed species assessed here. Therefore, the effects determinations for direct and indirect effects presented in Section 5.1 are used as the basis of the effects determination for potential modification to designated critical habitat.

5.2 Risk Description

The risk description synthesizes overall conclusions regarding the likelihood of adverse impacts leading to an effects determination (*i.e.*, "no effect," "may affect, but not likely to adversely affect," or "likely to adversely affect") for the assessed species and the potential for effects to their designated critical habitat.

If the RQs presented in the Risk Estimation (Section 5.1) show no direct or indirect effects for the assessed species, and no effects to PCEs of the designated critical habitat, a "no effect" determination is made, based on alachlor's use in California. However, if LOCs for direct or indirect effect are exceeded or there are effects to the PCEs of the critical habitat, the Agency concludes a preliminary "may affect" determination for the FIFRA regulatory action regarding alachlor.

None of the RQs for alachlor exceed the listed species LOCs (acute and chronic risk) for fish. Fish are used as a surrogate for aquatic phase amphibians. The RQs for 20 g birds that eat short grass (used as a screening-level surrogate for terrestrial-phase CRLF) exceed the risk to endangered species LOCs (acute and chronic) for all of the alachlor uses (both flowable and the impregnated bulk fertilizer applications). Therefore, there is a potential for direct effects to terrestrial-phase CRLF from all alachlor uses.

Regarding the potential for indirect effects to DS and CRLF and effects to their designated critical habitat, at least some of the RQs for mammals, aquatic invertebrates, birds, plants (aquatic and terrestrial) and potentially terrestrial invertebrates exceed the Agency's listed species LOCs. Therefore, there is a potential for indirect effects to DS and CRLF and effects to their critical habitat. Due to the potential for direct and/or indirect effects, alachlor use 'may affect' DS and CRLF and/or their designated critical habitat.

Following a "may affect" determination, additional information is considered to refine the potential for exposure based on the life history characteristics (*i.e.*, habitat range, feeding preferences, *etc*.) of the assessed species. Based on the best available information, the Agency uses the refined evaluation to distinguish those actions that "may affect, but are not likely to adversely affect" from those actions that are "likely to adversely affect" the assessed species and its designated critical habitat.

The criteria used to make determinations that the effects of an action are "not likely to adversely affect" the assessed species or modify its designated critical habitat include the following:

- <u>Significance of Effect</u>: Insignificant effects are those that cannot be meaningfully measured, detected, or evaluated in the context of a level of effect where "take" occurs for even a single individual. "Take" in this context means to harass or harm, defined as the following:

- Harm includes significant habitat modification or degradation that results in death or injury to listed species by significantly impairing behavioral patterns such as breeding, feeding, or sheltering.
- Harass is defined as actions that create the likelihood of injury to listed species to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering.

- Likelihood of the Effect Occurring: Discountable effects are those that are extremely unlikely to occur.
- Adverse Nature of Effect: Effects that are wholly beneficial without any adverse effects are not considered adverse.

A description of the risk and effects determination for each of the established assessment endpoints for the CRLF and DS and their designated critical habitat is provided in the following sections. The effects determination section for each listed species assessed will follow a similar pattern. Each will start with a discussion of the potential for direct effects, followed by a discussion of the potential for indirect effects. The section will end with a discussion on the potential for effects to the critical habitat from the use of alachlor.

5.2.1. Direct Effects

5.2.1.1. DS and Aquatic-Phase CRLFs

None of the RQs for any alachlor use exceed the acute or chronic risk LOCs for listed fish. The listed species LOC of 0.05 is associated with a probability of an individual effect of approximately 1 in 418,000,000 (using a default slope of 4.5).

The available alachlor toxicity data for aquatic-phase amphibians (African clawed frog; LC_{50} = 6.1 mg a.i./L, NOAEC = 640 µg a.i./L) and estuarine/marine fish (sheepshead minnow; LC_{50} = 3.9 mg a.i./L, NOAEC = 410 µg a.i./L) indicate that amphibians and estuarine/marine fish are equally as (or perhaps slightly less) sensitive to alachlor than freshwater fish. For example, using the acute and chronic toxicity endpoints for the African clawed frog and the sheepshead minnow would also result in no LOC exceedances for any alachlor use. The sheepshead minnow results are consistent with a 5-year field study that investigated the effects of three pesticides (including alachlor) on estuaries in North Carolina from runoff from adjacent farm lands (MRID 44105503) (see **Appendix A** for details). The pesticides in this study did not have a measurable impact on the estuarine biological community adjacent to application sites.

Although risks to fish are not expected based on the available toxicity data and exposure models, there are nine reported fish kills associated with alachlor in the EIIS database (they occurred between 1983 and 1995 and involved from an ‘unknown’ number of dead freshwater fish to ‘thousands’) (see **Appendix K**). Therefore, to explore further the potential impact of alachlor use on fish, a more detailed evaluation of the nine reported fish kills associated with alachlor was conducted. In seven of the nine reported fish kill incidents involving alachlor, other chemicals known to be highly toxic to fish were also involved in the incidents and are the more likely cause of the fish kills (*i.e.*, I000636-003, B000164-001, I000799-009, I000038-001, I003826-017, I002793-001, and I000636-012).

In another incident (I005002-008), the co-formulated product Bullet® (alachlor + atrazine) (this product is not registered for use in California) was used on an agricultural field and shortly after it began to rain. Within 3 days, dead fish (bass and bluegill sunfish) were evident in a nearby pond. Ten days following application, alachlor, atrazine, and metolachlor residues were detected in the water from the affected pond. This incident was reported as an 'accidental misuse' (the reason is not provided in the EIIS report).

In the remaining reported incident (I000431-001), no other pesticides besides alachlor were associated with the fish kill. In this incident, a fish kill in a 'fish tank' occurred 3.5 weeks following the application of Micro-Tech® to a 165-acre agricultural field. Both perch and bass, but not catfish, were affected. No water or fish tissues were analyzed for residues. The legality of use for this incident was classified as 'undetermined'. Therefore, in one of the nine reported fish kill incidents, a registered use of alachlor could not be excluded as a potential cause of the incident. Because of the relatively low toxicity of alachlor to fish and the timing of the incident (*i.e.*, it occurred 3.5 weeks after the alachlor use), however, it is unlikely that the fish were directly affected by alachlor. A more likely scenario is that the alachlor impacted the aquatic plant community of the 'fish tank' which in turn affected the water quality parameters in the tank (*e.g.*, dissolved oxygen) and indirectly affected the fish.

Therefore, the weight of evidence based on the currently available data suggests that direct effects to aquatic-phase CRLFs and DS are not expected from the use of alachlor in California. The potential for indirect effects is evaluated in Section 5.2.2.

5.2.1.2. Terrestrial-Phase CRLF

Acute and chronic LOCs are exceeded for birds. Acute RQs range from 0.28 (flowable applications to cotton) to 1.93 (bulk fertilizer applications to corn/sorghum). These RQs exceed the acute risk to endangered species LOC and are associated with a probability of an individual effect of approximately 1 in 1 to 1 in 156, depending on the use being evaluated (**Table 5.9**).

Table 5.9. Probability of Individual Effects to Terrestrial-Phase CRLF Based on Acute Data from Birds.

Use Site	Appl. Rate (lb a.i./acre)	Acute RQ	Slope (95% C.I.)[1]	Chance of Individual Effects
Flowable Soil Applications				
Corn	4	0.57	4.5	~1 in 7.35
Sorghum			2	~1 in 3.2
Peanuts			9	~1 in 71.4
Woody ornamentals				
Sunflowers				
Soybeans	3	0.43	4.5	~1 in 20.2
Dry beans			2	~1 in 4.3
Lima beans			9	~1 in 206
Cotton	2	0.28	4.5	~1 in 156
			2	~1 in 7.4
			9	~1 in 3,070,000
Impregnated Bulk Fertilizer Surface Applications[1]				
Corn	4	1.93	4.5	~1 in 1.1
Sorghum			2	~1 in 1.4
			9	~1 in 1
Soybeans	3	1.45	4.5	~1 in 1.3
			2	~1 in 1.6
			9	~1 in 1.1

[1] Default slopes were used

Birds were used as surrogate species for terrestrial-phase CRLFs. Terrestrial-phase amphibians are poikilotherms, which means that their body temperature varies with environmental temperature, while birds are homeotherms (temperature is regulated, constant, and largely independent of environmental temperatures). As a consequence, the caloric requirements of terrestrial-phase amphibians are markedly lower than birds. Therefore, on a daily dietary intake basis, birds consume more food than terrestrial-phase amphibians. This can be seen when comparing the caloric requirements for free living iguanid lizards (used in this case as a surrogate for terrestrial phase amphibians) to song birds (USEPA, 1993):

$$\text{iguanid FMR (kcal/day)} = 0.0535\ (\text{bw g})^{0.799}$$

$$\text{passerine FMR (kcal/day)} = 2.123\ (\text{bw g})^{0.749}$$

With relatively comparable slopes to the allometric functions, one can see that, given a comparable body weight, the free-living metabolic rate (FMR) of birds can be 40 times higher than reptiles, though the requirement differences narrow with high body weights.

Because the existing risk assessment process is driven by the dietary route of exposure, a finding of safety for birds, with their much higher feeding rates and, therefore, higher potential dietary exposure, is reasoned to be protective of terrestrial-phase amphibians. For this not to be the case, terrestrial-phase amphibians would have to be 40 times more sensitive than birds for the differences in dietary uptake to be negated. However, existing dietary toxicity studies in terrestrial-phase ampibians for alachlor are lacking. To quantify the potential differences in food intake between birds and terrestrial-phase CRLFs, food intake equations for the iguanid lizard were used to replace the food intake equation in T-REX for birds, and additional food items of the CRLF were evaluated. These functions were encompassed in a model called T-HERPS. T-HERPS is available at: http://www.epa.gov/oppefed1/models/terrestrial/index.htm.

For the uses with the highest application rates (4 lb a.i./acre), none of the acute RQs for terrestrial herpetofauna exceed the Agency's listed species LOC for acute exposure (**Table 5.10**). Therefore, none of the registered uses of alachlor are expected to result in acute direct effects to terrestrial-phase CRLF (see also **Appendix M**).

Table 5.10. Upper Bound Kenaga, Acute Terrestrial Herpetofauna Dose-Based Risk Quotients for Alachlor (4 lb a.i./acre, 1 Application).

Size Class (grams)	LD_{50} (mg a.i./kg-bw)	EECs and RQs									
		Broadleaf Plants/ Small Insects		Fruits/Pods/ Seeds/ Large Insects		Small Herbivore Mammals		Small Insectivore Mammal		Small Amphibians	
		EEC	RQ	EEC	RQ	EEC	RQ	EEC	RQ	EEC	RQ
1.4	1499.00	20.98	0.01	2.33	0.00	N/A[1]	N/A	N/A	N/A	N/A	N/A
37	1499.00	20.62	0.01	2.29	0.00	N/A	N/A	N/A	N/A	0.72	0.00
238	1499.00	13.51	0.01	1.50	0.00	93.03	0.06	5.81	0.00	0.47	0.00

[1] N/A = not applicable (a 1.4 or 37 g frog is not expected to be large enough to eat a 35 g mammal).

At the 4 lb a.i./acre application rate (corn, sorghum, peanuts, woody ornamentals, and sunflowers), the sub-acute RQs for terrestrial herpetofauna that eat broadleaf plants/small insects and small herbivorous mammals exceed the Agency's listed species LOC (**Table 5.11**). None of the RQs for any other use exceed the LOC based on sub-chronic exposure.

For chronic exposure, all of the alachlor uses exceed the Agency's chronic listed species LOC of 1 for at least one of the dietary categories using an avian NOAEC of 50 mg a.i./kg-diet (**Table 11**). However, since a definitive NOAEC was not determined for birds (*i.e.*, the NOAEC <50 mg a.i./kg-diet) all of the calculated RQs for chronic exposure are greater-than values. Therefore, risks to terrestrial-phase frogs from chronic exposure cannot be precluded for any of the uses or dietary categories at this time.

Table 5.11. Upper Bound Kenaga, Sub-Acute and Chronic Terrestrial Herpetofauna Dietary-Based Risk Quotients for Alachlor (1 Application).

Appl. Rate (lb a.i./acre)	Endpoint (mg a.i./kg-diet)	EECs and RQs									
		Broadleaf Plants/ Small Insects		Fruits/Pods/ Seeds/ Large Insects		Small Herbivore Mammals		Small Insectivore Mammals		Small Amphibians	
		EEC	RQ	EEC	RQ	EEC	RQ	EEC	RQ	EEC	RQ
Sub-Acute (Dietary)											
4	LC_{50} = 5620	540.00	**0.10**	60.00	0.01	632.59	**0.11**	39.54	0.01	18.74	0.00
3		405.00	0.07	45.00	0.01	474.44	0.08	29.65	0.01	14.06	0.00
2		270.00	0.05	30.00	0.01	316.29	0.06	19.77	0.00	9.37	0.00
Chronic (Dietary)											
4	NOAEC <50	540.00	**>10.80**	60.00	**>1.20**	632.59	**>12.65**	39.54	**>0.79**	18.74	**>0.37**
3		405.00	**>8.10**	45.00	**>0.90**	474.44	**>9.49**	29.65	**>0.59**	14.06	**>0.28**
2		270.00	**>5.40**	30.00	**>0.60**	316.29	**>6.33**	19.77	**>0.40**	9.37	**>0.19**

- Bolded numbers (black) exceed the Agency's listed species LOC.
- Bolded numbers (gray) potentially exceed the Agency's listed species LOC.

These results indicate that the risk of direct adverse effects to terrestrial-phase CRLF from acute oral or sub-acute dietary exposure is likely low. However, the risk (or potential risk) to terrestrial-phase CRLF from chronic dietary exposure cannot be precluded and exists for all dietary classes relevant to the CRLF (for all of the registered alachlor uses).

5.2.2. Indirect Effects, DS and Aquatic-Phase CRLF

As discussed in Section 2, the diet of aquatic-phase CRLF tadpoles and DS larvae is composed primarily of unicellular aquatic plants (*i.e.*, algae and diatoms) and detritus. However, aquatic invertebrates are also consumed by both CRLFs and the DS, and fish are consumed by adult CRLFs. Therefore, potential impacts to each of these potential food items are evaluated.

5.2.2.1. Potential Impacts to Fish (Indirect Effects to CRLF Only)

Fish are food items of the CRLF. None of the RQs exceed the Agency's listed species LOCs for fish. Therefore, indirect effects to CRLF from a decline in potential fish prey are not expected from the use of alachlor in California.

5.2.2.2. Potential Impacts to Aquatic Invertebrates

CRLF

The acute risk to listed and non-listed species LOCs of 0.05 and 0.5 were not exceeded for freshwater invertebrates for any alachlor use based on toxicity values from the most sensitive freshwater species for which data are available. The highest acute RQ is 0.02 [for the woody ornamental (nursery) use]. At this RQ and using the default slope (4.5), the probability of an effect would be approximately 1 in 9.6E+13. Based on chronic exposure, the Agency's chronic risk LOC of 1 is exceeded for the corn (broadcast) (RQ = 1.2), sorghum (broadcast) (RQ = 1.3), woody ornamentals (nursery-use) (RQ = 1.5), sunflower (broadcast) (RQ = 1.2), and the peanut (broadcast) (RQ = 1.2) uses. Based on the CADPR PUR data, from 1999 to 2006 an average of <1 lb of alachlor per year was applied to woody ornamentals in California. Alachlor was not used at all on sunflowers, sorghum, or peanuts in California during the same time period (again, based on the CADPR PUR data). Therefore impacts to wildlife in California from these uses are not expected. Corn, however, is one of the major uses for alachlor in California.

The NOAEC used to calculate the chronic freshwater invertebrate RQs is based on an endpoint of reduced adult length and the EECs are only slightly above the NOAEC (~1.5X for the woody ornamental use and ~1X for the corn, sorghum, sunflower, and peanut uses). What effect reduced adult length would have on aquatic invertebrate populations is unclear, but the effect is not likely to reduce aquatic invertebrate populations to a level that would impact the CRLF (*i.e.*, from loss of potential prey items).

Based on the low anticipated direct impacts to the most sensitive freshwater invertebrates, any potential impact to aquatic-phase CRLFs would likely be immeasurable in the environment and would, therefore, constitute an insignificant effect. Therefore, exceedance of the chronic risk to endangered species LOC suggests that there could be some effect to sensitive freshwater aquatic invertebrates from the woody ornamental and sunflower uses; however, such an effect would likely be insignificant to the CRLF.

DS

The DS eats small zooplankton. They primarily eat planktonic copepods, cladocerans, amphipods, and insect larvae. However, the most important food organism appears to be *Eurytemora affinis*, which is a euryhaline copepod (USFWS, 1995 and 2004). Based on toxicity data from mysid, none of the RQs for any alachlor use exceed the Agency's acute risk LOCs (listed or non-listed species) [acute RQs range from 0.001 (sweet corn) to 0.02 (woody ornamentals)]. At the highest RQ (0.02) and using a slope of 8.7, the probability of an effect would be approximately 1 in 1.03E+49. Therefore, impacts to potential estuarine/marine invertebrate prey are not expected from acute exposure to alachlor.

For chronic risk to estuarine/marine invertebrates, the only species for which data from chronic exposure are available is the copepod (non-native) *Tigriopus japonicus*. Based on the *T. japonicus* endpoint, the chronic RQs exceed the Agency's LOC for all uses [RQs range from >31 (sweet corn) to >543 (woody ornamentals)]. In the *T. japonicus* study, a definitive NOAEC was not established because potential reproductive effects (*i.e.*, increase in the generation time by ~1 day related to the duration of the nauplius phase) were seen at all concentrations tested (the lowest concentration = 0.1 µg a.i./L). No other reproductive traits (*i.e.*, fecundity, sex ratio, or survival rate) showed any significant change after exposure to alachlor, even at the highest concentration tested (100 µg a.i./L). If the 100 µg a.i./L endpoint is used to calculate RQs, none of the RQs for any alachlor use exceed the chronic risk LOC [the highest RQ = 0.54 (woody ornamentals)]. This indicates that effects to copepod fecundity, sex ratio, and survival rate are not expected from alachlor use.

Therefore, the conclusion regarding the potential for indirect effects to the DS from a reduction in prey availability is dependent on the significance of the effect of extending the nauplius phase in copepod prey by roughly one day. Although increased generation time could decrease copepod populations over an extended period of time, the impact from alachlor use to copepod (and other estuarine/marine invertebrate) populations is not expected to be large enough to impact the DS indirectly. This, again, is consistent with the study that investigated the effects of three pesticides (including alachlor) on estuaries in North Carolina from runoff from adjacent farm lands (MRID 44105503) (see **Appendix A** for details). The pesticides in this study did not have a measurable impact on the estuarine biological community adjacent to application sites.

5.2.2.3. Potential Impacts to Aquatic Plants

CRLF tadpoles consume primarily algae, and DS larvae consume phytoplankton. Algal RQs ranged from approximately 2 (sweet corn) to 34 (woody ornamentals), which means that the EECs calculated for alachlor uses are ~2 to 34 times higher than the most sensitive algal EC_{50} of 1.64 µg/L. From spray drift alone, impacts to non-vasular aquatic plants are expected up 216 ft (0.07 km) and 151 ft (0.05 km) from application sites for corn and dry beans (the two most common alachlor uses in California), respectively (see Section 5.2.4 and **Table 5.13**). Based on the downstream dilution analysis, effects to aquatic plants could extend up to 285 km from use sites for the corn and woody ornamental (nursery) uses. Therefore, impacts to aquatic plants found near alachlor use sites are expected.

5.2.3. Indirect Effects, Terrestrial-Phase CRLFs

As discussed in Section 2, the diet of terrestrial-phase CRLFs includes terrestrial invertebrates, small mammals, and amphibians. Potential impacts to each of these potential food items are evaluated below.

5.2.3.1. Terrestrial Invertebrates

When the terrestrial-phase CRLF reaches juvenile and adult stages, its diet is mainly composed of terrestrial invertebrates. Alachlor is classified as practically nontoxic to non-target insects on an acute contact exposure basis. An acute contact LD_{50} for terrestrial invertebrates could not be determined based on available data. For honey bees, a contact concentration of 36.3 µg a.i./bee (equivalent to 284 ppm) resulted in 0.41% mortality of exposed adults. Only one concentration was used in this study; therefore, a definitive LD_{50} value and response slope could not be determined. Using an LD_{50} of >36.3 µg a.i./bee results in RQs less than 1.9 and 0.21 for small and large insects, respectively; however, it is not clear if the actual RQs are above or below the interim LOC of 0.05 for acute risk to endangered terrestrial invertebrates.

The chance of individual effects for terrestrial invertebrates using the IECv1.1.xls spreadsheet, the acute risk to endangered species LOC of 0.05, and default slope of 4.5 (upper and lower bound = 2 and 9) is ~1 in 4.18E+08 (with upper and lower bounds of ~1 in 216, and ~1 in 1.75E+31).

As stated above, in the submitted honey bee study, a concentration of 284 ppm resulted in 0.41% mortality. Based on T-REX, a flowable alachlor application of 4 lb a.i./acre results in EEC values of 60 and 540 ppm for large and small insects, respectively. Therefore, the concentration on the site of application at the maximum allowable application rate is not expected to reach levels high enough to cause 0.41% mortality in large insects.

For small insects, the concentration on the site of application is expected to be 1.9 times the concentration that would result in 0.41% mortality. AgDRIFT (Tier I ground application, Very Fine to Fine ASAE droplet size distribution) was used to model the fraction of applied pesticide that is predicted to be 5 ft (1.5 m) off the field. The resulting fraction of applied pesticide is 45% (*i.e.*, 1.79 lb a.i./A). Inputting an application rate of 1.79 lb a.i./acre into T-REX results in EECs of 242 ppm for small insects, which is below the concentration that resulted in 0.41% mortality in adult bees. Therefore, the concentration of alachlor >5 ft from the site of application is not expected to reach levels high enough to cause even 0.41% mortality in small insects based on the available honey bee data.

Therefore, the Agency concludes that the potential for alachlor use to impact terrestrial invertebrate populations to levels high enough to impact the CRLF is low and discountable.

5.2.3.2. Mammals

Terrestrial-phase CRLFs consume small mammals. This assessment used a 15 g mammal as a potential mammalian prey. Several RQs for mammals exceed the Agency's acute risk to listed species LOC, however, the only RQs that exceed the Agency's acute risk LOC for non-listed mammals are for alachlor applications via impregnated bulk fertilizer [RQs = 1.36 (corn and sorghum) and 1.02 (soybeans)]. Assuming a default probit slope of 4.5, the probability of an individual effect would be approximately 1 in 1.4 and 1 in 2 for the RQs 1.4 and 1.0, respectively. Assuming that probability of an individual effect provides insight into the potential for reductions in a local population of small mammals, a probability of 1 in 1 to 2 could result in a measurable impact to mammal abundance on and/or around alachlor application sites (corn, sorghum, soybeans) and could, therefore, constitute a potentially significant effect.

Regarding the potential for impacts from chronic exposure, almost all of the RQs for chronic exposure exceed the Agency's chronic risk LOC for non-listed mammals [all alachlor use/application combinations, dietary- and dose-based for 15 g mammal (all dietary categories)]. The RQs range from 0.3 (cotton; flowable bare soil application; seed-eating mammal, dietary-based) to 278 (corn, sorghum, peanuts, woody ornamentals, and sunflowers; flowable, non-bare soil application; mammal that eats short grass, dose-based). These RQs are based on a NOAEC of 30 mg/kg-diet (the highest concentration tested) from a 3-generation rat study in which no reproductive effects were observed (*i.e.*, no reproductive LOAEC was determined) (MRID 00075062). Therefore, the actual NOAEC for reproductive effects is likely higher than 30 mg/kg-diet. How much higher is unknown at this time.

In the 3-generation study, there were some systemic effects at 30 mg/kg-diet [kidney discoloration and decreased kidney weights and lower ovary weights in females of each parental generation and the F_3 females (maximal decrease of 17%)]. However, none of these effects were correlated with growth, survival, or reproductive endpoints.

Therefore, the Agency concludes that the potential for alachlor use to impact mammalian prey populations to levels high enough to impact the CRLF is low and discountable.

5.2.3.3. Amphibians

CRLF are known to prey on aquatic-phase amphibians. The potential risk to amphibians based on fish toxicity data is expected to be low based on the RQs (*i.e.*, all RQs < 0.05). Additionally, using the acute toxicity endpoints for the African clawed frog (African clawed frog; LC_{50} = 6.1 mg a.i./L) results in no LOC exceedances for any alachlor use. Therefore, indirect effects to CRLF from a decline in potential aquatic phase amphibian prey are not expected from the use of alachlor in California.

Terrestrial amphibian prey of the CRLF include small amphibians such as tree frogs that do not prey on mammals. Therefore, the mammalian food group is not relevant in the evaluation of potential reductions in amphibian prey abundance. The RQs for acute and sub-acute dietary exposure from T-HERPS range from 0 to 0.01 (acute) and 0.01 to 0.10 (sub-acute) (including only the broadleaf plants/small insects and fruits/pods/seeds/large insect dietary categories) (see above). Therefore, none of the RQs for acute or sub-acute exposure exceed the Agency's acute risk LOC for non-listed terrestrial animals. This indicates that the risk for indirect adverse effects to the CRLF from loss of amphibian prey after acute or sub-acute exposure to alachlor is low.

For chronic exposure, however, all of the alachlor uses exceed the Agency's chronic risk to non-listed species LOC of 1 for at least one of the relevant dietary categories (chronic RQs for the broadleaf plants/small insects and fruits/pods/seeds/large insect dietary categories range from >0.60 to >10.8). However, since a definitive NOAEC was not determined for birds (*i.e.*,effects were seen at all treatment levels) all of the calculated RQs for chronic exposure are greater-than values. Therefore, risks to terrestrial-phase amphibians from chronic exposure cannot be precluded for any of the alachlor uses at this time.

Again, it is difficult to determine if potential effects from chronic exposure to alachlor would impact terrestrial amphibian abundance to an extent that could result in indirect effects to the CRLF; however, such impacts cannot be precluded at this time based upon available information. Therefore, the Agency concludes that there exists the potential for alachlor use to impact terrestrial phase amphibian prey populations to levels high enough to impact the CRLF.

5.2.3.4. Potential Effects to Habitat

Aquatic plants serve several important functions in aquatic ecosystems. Non-vascular aquatic plants are primary producers and provide the autochthonous energy base for aquatic ecosystems. Vascular plants provide structure, rather than energy, to the system, as attachment sites for many aquatic invertebrates, and refugia for juvenile organisms, such as fish and frogs. Emergent plants help reduce sediment loading and provide stability to nearshore areas and lower streambanks. In addition, vascular aquatic plants are important as attachment sites for egg masses of aquatic species. Results of the indirect effects assessment are used as the basis for the habitat modification analysis. From spray drift alone, impacts to non-vasular aquatic plants are expected up to 216 ft (0.07 m) and 151 ft (0.05 m) from application sites for corn and dry beans (the two most common alachlor uses in California), respectively (see Section 5.2.4 and **Table 5.13**). Based on the downstream dilution analysis, effects to aquatic plants could extend up to 285 km from use sites for the corn and woody ornamental (nursery) uses. Therefore, impacts to aquatic plants found near alachlor use sites are expected.

Terrestrial plants serve several important habitat-related functions for the listed assessed species. Among other things, riparian vegetation helps to maintain the integrity of aquatic systems by providing bank and thermal stability, serving as a buffer to filter out sediment, nutrients, and contaminants before they reach the watershed, and serving as an energy source (CRLF and DS). In addition to providing shelter and cover from predators while foraging, upland vegetation, including grassland and woodlands, provides cover during dispersal (CRLF).

Based on the results of the submitted terrestrial plant toxicity studies and the reported terrestrial plant incidents, the herbicide alachlor is phytotoxic to many plant species (seedling emergence endpoints are more sensitive than vegetative vigor endpoints). Additionally, monocots are more sensitive to alachlor than are dicots, based on available data. However, for adjacent upland and wetland plants, terrestrial plant RQs for both monocots and dicots exceed the Agency's risk to non-listed species LOC for all alachlor uses and application types. For the drift only RQs, all of the RQs exceed the Agency's LOC except for the impregnated bulk fertilizer applications (all uses) and the dicot RQs for the cotton use (all application types). Based on the spray drift analysis, effects to terrestrial plants could occur >800 ft (>0.24 km) from alachlor application sites (see Section 5.2.4).

A general conclusion that can be drawn from these data is that the inhibition of new growth may occur in non-target terrestrial plants from registered uses of alachlor. Inhibition of new growth could result in degradation of high quality riparian habitat over time because as older growth dies from natural or anthropogenic causes, plant biomass may be prevented from being replenished in the riparian area. Inhibition of new growth may also slow the recovery of degraded riparian areas that function poorly due to sparse vegetation because alachlor deposition onto bare soil would be expected to inhibit the growth of new vegetation. Additionally, because effects were seen in most species tested in the seedling emergence and vegetative vigor studies, it is likely that many species of herbaceous plants could be potentially affected by exposure to alachlor.

It is difficult to estimate the magnitude of potential impacts of alachlor use on riparian habitat and the magnitude of potential effects on stream water quality from such impacts as they relate to survival, growth, and reproduction of the CRLF and DS. The level of exposure and any resulting magnitude of effect on riparian vegetation are expected to be highly variable and dependent on many factors. The extent of runoff and/or drift into stream corridor areas is affected by the distance the alachlor use site is offset from the stream, local geography, weather conditions, and quality of the riparian buffer itself. The sensitivity of the riparian vegetation is dependent on the susceptibility of the plant species exposed to alachlor and composition of the riparian zone (*e.g.* vegetation density, species richness, height of vegetation, width of riparian area).

In summary, terrestrial and aquatic plant RQs are above plant LOCs for all uses; therefore, labeled use of alachlor has the potential to affect both aquatic and riparian vegetation within CRLF and DS habitats.

5.2.4 Spatial Extent of Potential Effects

Since this assessment defines taxa that are predicted to be exposed through runoff and drift to alachlor at concentrations above the Agency's LOC, analysis of the spatial extent of potential effects requires expansion of the area from the treated site to include all areas where risk to the CRLF and/or the DS exceed LOCs.

To determine this area, the footprint of alachlor's use pattern is identified, using corresponding land cover data. The spatial extent of the effects determination also includes areas beyond the initial area of concern that may be impacted by runoff and/or spray drift (potential use areas + distance down stream or down wind from use sites where organisms relevant to the CRLF and/or DS may be affected). The determination of the buffer distance and downstream dilution for spatial extent of the effects determination is described below.

5.2.4.1. Spray Drift

In order to determine terrestrial and aquatic habitats of concern due to alachlor exposures through spray drift, it is necessary to estimate the distance that spray applications can drift from the treated area and still be present at concentrations that exceed levels of concern. Applications of alachlor via impregnated bulk fertilizer are not expected to result in any spray drift (they are considered similar to granular applications for modeling purposes). For the flowable uses, a quantitative analysis of spray drift distances was completed using AgDRIFT (v. 2.01) using default inputs for ground applications (*i.e*., high boom, ASAE droplet size distribution = Very Fine to Fine, 90^{th} data percentile), except for the analyses for cotton which used labeled restrictions (*i.e*., high boom, ASAE droplet size distribution = Fine to Medium/Coarse, 90^{th} data percentile).

For direct effects to the terrestrial-phase CRLF, the RQs for 20 g birds that eat small insects were used to estimate the fraction of the application rates that would no longer exceed the listed species LOC (*i.e*., fraction of applied = LOC/RQ). This number was used in AgDRIFT to calculate the distance from the field where the amount of alachlor that equaled the 'fraction of applied' would be expected to occur (as spray drift) (**Table 5.12**). For direct effects to aquatic-phase CRLF and DS, the distance from the site of application in which spray drift could reach levels high enough to exceed the acute risk to endangered species LOC, the 'active rate' (*i.e*., the highest maximum labeled rate) and the 'initial average concentration' (*i.e*., LC_{50} value X 0.05) were inputted into AgDRIFT. For this analysis, the farm pond (*i.e.*, a pond with a depth of 2 meters and a downwind width of 63.61 m and flight line width of 157.21 m) was used as a proxy for CRLF and DS habitat. The other AgDRIFT inputs were the same as described above in the terrestrial distance analysis.

Table 5.12. Distance from Alachlor Use Site Needed to Reduce Exposure from Spray Drift to Levels that Do Not Exceed LOCs for Direct Effects.

Use Site	Appl. Rate (lb a.i./acre)	Terrestrial-phase CRLF		Aquatic-phase CRLF and DS
		RQ	Distance from Site of Appl. (in ft)	Distance from Site of Appl. (in ft)
Corn Sweet corn Sorghum Peanuts Woody ornamentals (nursery) Woody ornamentals (residential) Sunflowers	4	0.57	16.4	0
Soybeans Dry beans Lima beans	3	0.43	13.1	0
Cotton	2	0.28	3.3	0

For indirect effects, a spray drift analysis is conducted using endpoints for terrestrial and aquatic plants (the most sensitive taxa to alachlor). The distance from the field for terrestrial plants is based on the most sensitive terrestrial plant non-listed species endpoint (*i.e.*, monocot seedling emergence EC_{25} = 0.0067 lb a.i./acre). This endpoint is used to estimate the fraction of the application rates that would no longer exceed the 25% level of effects for terrestrial plants (*i.e.*, fraction of applied = 0.0067 lb a.i./acre divided by the application rate in lb a.i./acre) (**Table 4.2**).

For aquatic plants, the RQ for the most sensitive aquatic plant (*i.e.*, non-vascular plant EC_{50} = 1.64 µg a.i./L) is used to estimate the initial average concentration that would no longer exceed the LOC for plants (*i.e.*,EC_{50} X 1). This number is used in AgDRIFT to calculate the distance from the field where the amount of alachlor that equals the 'initial average concentration' would be expected to occur (as spray drift) (**Table 5.13**).

Table 5.13. Distance from Alachlor Use Site Needed to Reduce Exposure from Spray Drift to Levels that Do Not Exceed LOCs for Indirect Effects.

Use Site	Appl. Rate (lb a.i./acre)	Terrestrial Plants	Aquatic Plants
		Distance from Site of Appl. (in ft)	Distance from Site of Appl. (in ft)
Corn	4	886	216
Sorghum			
Peanuts			
Woody ornamentals			
Sunflowers			
Soybeans	3	748	151
Dry beans			
Lima beans			
Cotton	2	246	85

Therefore, when only spray drift is considered, potential risks for both direct and indirect effects would be below concern levels at distances from alachlor use sites equal to or greater than 886 ft (0.27 km)

5.2.4.2 Downstream Dilution Analysis

The maximum downstream extent of alachlor exposure in streams and rivers where the EEC can potentially be above levels that would exceed LOCs was estimated to determine the potential areas of effect for CRLF and DS in aquatic environments (see **Appendix N** for details). For potential direct effects to aquatic-phase CRLF and DS, none of the RQs for fish exceed the Agency's listed species LOC. Therefore, the potential for direct effects to aquatic-phase CRLF and DS is low from all uses.

Considering the potential for indirect effects, for the DS (based on the chronic endpoint for estuarine/marine invertebrates) the area of potential effects extends to >43.4 km from residential use sites (woody ornamentals) and >285 km from application sites for corn and woody ornamentals (nursery). For aquatic-phase CRLF (based on toxicity data for non-vascular aquatic plants) the area of potential effects extends up to 5 km from residential use sites (woody ornamentals); 285 km from woody ornamentals (nursery) and corn use sites.

5.2.4.3. Overlap of Potential Areas of Effect and CRLF and DS Habitat

The spray drift and downstream dilution analyses help to identify areas of potential effect to the CRLF and DS from registered uses of alachlor. The potential area of effects for the CRLF and DS from alachlor spray drift extend from the site of application to 0.27 km from the site of application depending on the use. For exposure to runoff and spray drift, the area of potential effects extends up to 285 km downstream from the site of application (again, depending on the use). When these distances are added to the footprint of the initial area of concern (which represents potential alachlor use sites) and compared to CRLF and DS habitat, there are several areas of overlap (**Figures 5.1** and **5.2)**. The overlap between the areas of effect and CRLF and DS habitat, including designated critical habitat, indicates that alachlor use in California has the potential to affect the CRLF and DS.

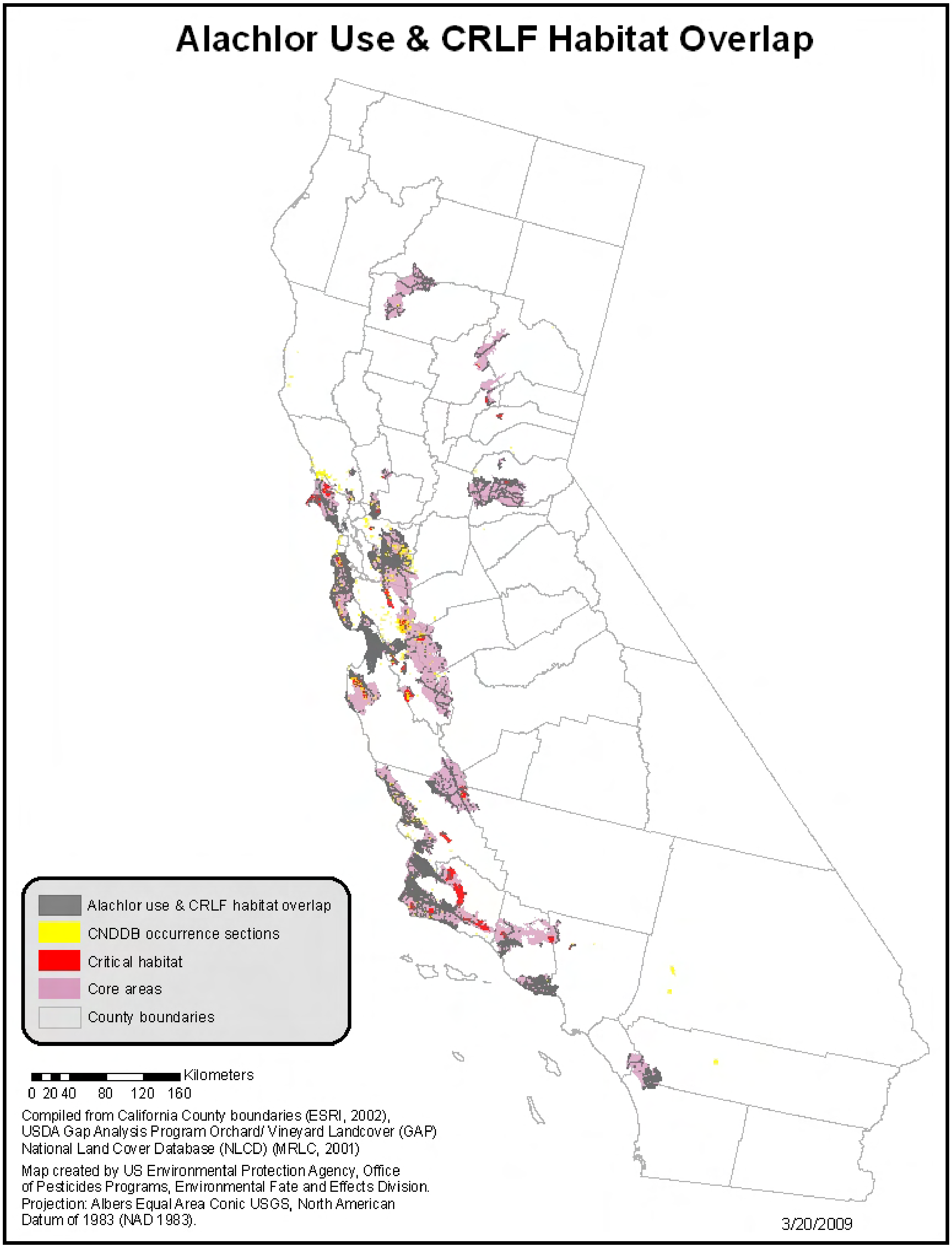

Figure 5.1. Overlap Map: CRLF Habitat and Alachlor Initial Area of Concern.

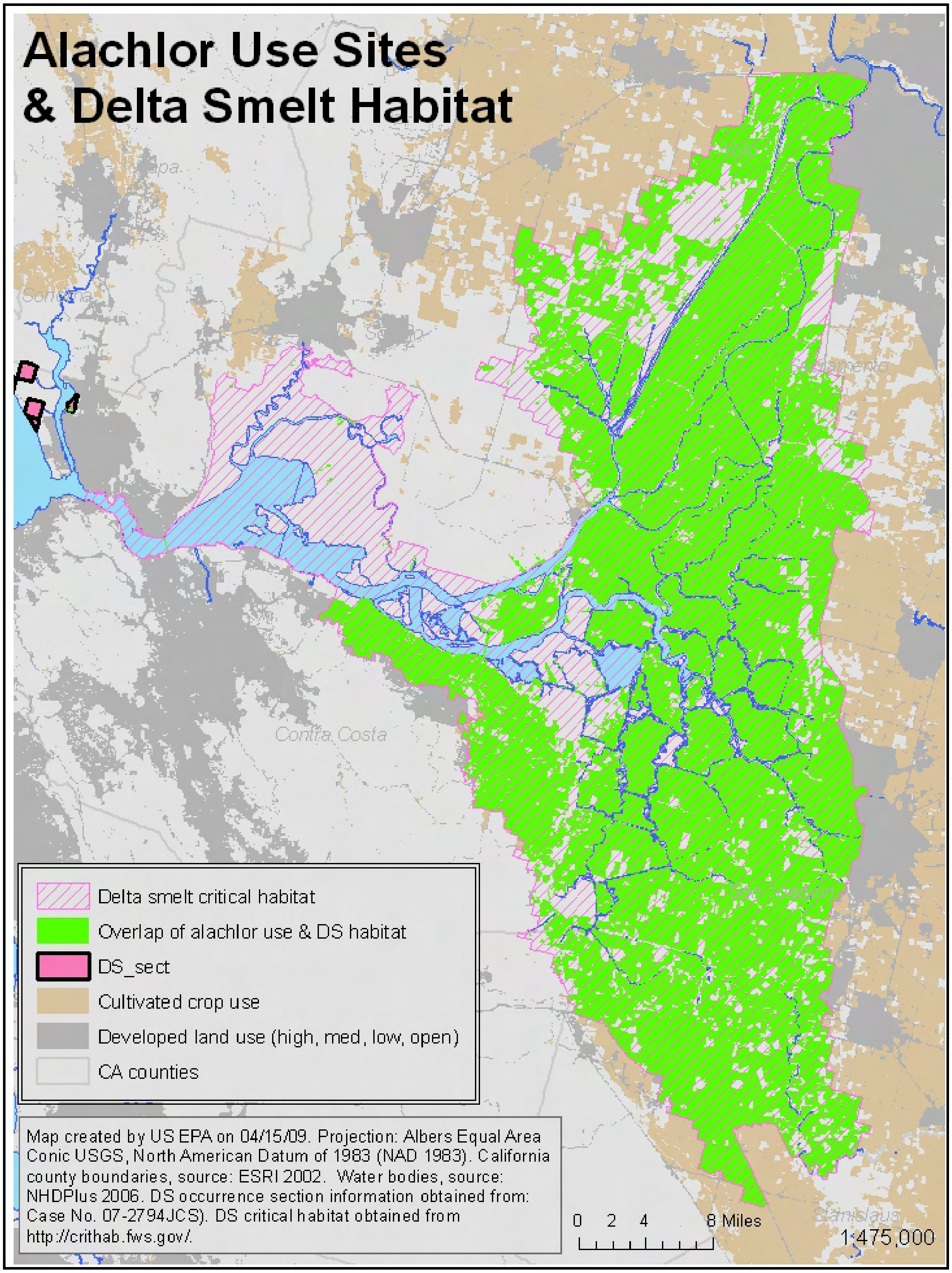

Figure 5.2. Overlap Map: DS Habitat and Alachlor Initial Area of Concern.

5.3. Effects to Designated Critical Habitat

The risk conclusions for the designated critical habitat are based on conclusions described for indirect effects previously described. Potential effects to habitat is described below.

5.3.1. CRLF

5.3.1.1. Aquatic-Phase PCEs

Three of the four assessment endpoints for the aquatic-phase primary constituent elements (PCEs) of designated critical habitat for the CRLF are related to potential effects to aquatic and/or terrestrial plants:

- Alteration of channel/pond morphology or geometry and/or increase in sediment deposition within the stream channel or pond: aquatic habitat (including riparian vegetation) provides for shelter, foraging, predator avoidance, and aquatic dispersal for juvenile and adult CRLFs.
- Alteration in water chemistry/quality including temperature, turbidity, and oxygen content necessary for normal growth and viability of juvenile and adult CRLFs and their food source.
- Reduction and/or modification of aquatic-based food sources for pre-metamorphs (*e.g.,* algae).

Conclusions for potential indirect effects to the CRLF via direct effects to aquatic and terrestrial plants are used to determine whether effects to critical habitat may occur. As previously discussed, alachlor may cause effects to habitat by potentially impacting aquatic plants and terrestrial plants.

The remaining aquatic-phase PCE is "alteration of other chemical characteristics necessary for normal growth and viability of CRLFs and their food source." Alachlor may impact algae as food items for tadpoles. Alachlor may also impact riparian areas that are predominantly grassy or herbaceous, and the potential areas of effect overlap with designated critical habitat for the CRLF (see **Fig. 5.1**). Therefore, there is a potential for effects to habitat by potentially impacting the chemical characteristics of the habitat.

5.3.1.2. Terrestrial-Phase PCEs

Two of the four assessment endpoints for the terrestrial-phase PCEs of designated critical habitat for the CRLF are related to potential effects to terrestrial plants:

- Elimination and/or disturbance of upland habitat; ability of habitat to support food source of CRLFs: Upland areas within 200 ft (0.06 km) of the edge of the riparian vegetation or drip line surrounding aquatic and riparian habitat that are comprised of grasslands, woodlands, and/or wetland/riparian plant species that provides the CRLF shelter, forage, and predator avoidance.

- Elimination and/or disturbance of dispersal habitat: Upland or riparian dispersal habitat within designated units and between occupied locations within 0.7 mi (1.1 km) of each other that allow for movement between sites including both natural and altered sites which do not contain barriers to dispersal.

As an herbicide, alachlor may affect sensitive terrestrial plants; information from the reported terrestrial plant incident data support this. Additionally, risk to terrestrial plant LOCs are exceeded for all uses and the potential areas of effect overlap with designated critical habitat for the CRLF (see **Fig. 5.1**).

The third terrestrial-phase PCE is "reduction and/or modification of food sources for terrestrial-phase juveniles and adults." To assess the impact of alachlor on this PCE, acute and chronic toxicity endpoints for terrestrial invertebrates, mammals, and terrestrial-phase frogs are used as measures of effects. There is a potential for habitat modification based on potential reductions in prey base (mammals and frogs, as previously described), and, again, the areas of potential effect overlap with CRLF critical habitat (**Fig. 5.1**).

The fourth terrestrial-phase PCE is based on alteration of chemical characteristics necessary for normal growth and viability of juvenile and adult CRLFs and their food source. There is a potential for habitat modification based on potential direct (Section 5.2.1) and indirect effects (Sections 5.2.2) to terrestrial-phase CRLFs.

5.3.2. DS

Primary constituent elements (PCEs) of designated critical habitat for the DS include the following:

- Spawning Habitat—shallow, fresh or slightly brackish backwater sloughs and edgewaters to ensure egg hatching and larval viability. Spawning areas also must provide suitable water quality (*i.e.*, low "concentrations of pollutants) and substrates for egg attachment (*e.g.*, submerged tree roots and branches and emergent vegetation).

- Larval and Juvenile Transport—Sacramento and San Joaquin Rivers and their tributary channels must be protected from physical disturbance and flow disruption. Adequate river flow is necessary to transport larvae from upstream spawning areas to rearing habitat in Suisun Bay. Suitable water quality must be provided so that maturation is not impaired by pollutant concentrations.

- Rearing Habitat—Maintenance of the 2 ppt isohaline and suitable water quality (low concentrations of pollutants) within the estuary is necessary to provide Delta smelt larvae and juveniles a shallow protective, food-rich environment in which to mature to adulthood.

- Adult Migration— Unrestricted access to suitable spawning habitat in a period that may extend from December to July. Adequate flow and suitable water quality may need to be maintained to attract migrating adults in the Sacramento and San Joaquin River channels and their associated tributaries. These areas also should be protected from physical disturbance and flow disruption during migratory periods.

- PCEs also include more general requirements for habitat areas that provide essential life cycle needs of the species such as space for individual and population growth and for normal behavior; food, water, air, light, minerals, or other nutritional or physiological requirements; cover or shelter; sites for breeding, reproduction, rearing (or development) of offspring; and habitats that are protected from disturbance or are representative of the historic geographical and ecological distributions of a species.

The potential for direct effects to the DS from alachlor use could not be precluded based on incident data. Furthermore, it was concluded that alachlor is likely to adversely affect the DS by potentially affecting its habitat (aquatic and terrestrial plants) and the potential areas of effect overlap with critical habitat designated for DS (**Fig. 5.2**). Therefore, alachlor may also affect critical habitat of the DS that is located in close proximity to alachlor use sites.

5.4. Effects Determinations

5.4.1 CRLF

The weight of evidence indicates that alachlor use has the potential to directly adversely affect CRLF. The risk to aquatic-phase CRLF is low based on the RQ analyses. Although the risk to terrestrial-phase CRLF from acute or sub-acute dietary exposure is low, the potential risk to terrestrial-phase CRLF from chronic dietary exposure cannot be precluded and exists for all dietary classes relevant to the CRLF (for all of the registered alachlor uses).

Regarding the potential for indirect effects, exceedance of the chronic risk to endangered species LOC indicates that there could be some effect to sensitive aquatic invertebrates from the woody ornamental and sunflower uses; however, such an effect would likely be insignificant to the CRLF. Furthermore, the Agency concludes that the potential for alachlor use to impact terrestrial invertebrate populations to levels high enough to impact the CRLF is low and discountable. Impacts to non-vascular aquatic and terrestrial plants, however, are expected from all of the current alachlor uses. Additionally, the Agency concludes that there exists the potential, which cannot currently be precluded, for alachlor use to impact amphibian prey populations to levels high enough to impact CRLF. Spatial analyses show that potential areas of effect from alachlor use overlap with CRLF habitat and their designated critical habitat. Therefore, the Agency makes a “may affect, and likely to adversely affect” determination for the CRLF and a habitat effects determination for their designated critical habitat from the use of alachlor based on the potential for direct and indirect effects and effects to the PCEs of critical habitat.

5.4.2 DS

The weight of evidence indicates that alachlor use will not directly adversely affect DS. Regarding the potential for indirect effects, the impact from alachlor use to estuarine/marine invertebrate populations is not expected to be large enough to impact the DS indirectly. This, again, is consistent with the study that investigated the effects of three pesticides (including alachlor) on estuaries in North Carolina from runoff from adjacent farm lands which showed no measurable impact on the estuarine biological community adjacent to application sites (MRID 44105503). Impacts to non-vascular aquatic and terrestrial plants, however, are expected from all of the current alachlor uses. Spatial analyses show that potential areas of effect from alachlor use overlap with DS habitat and their designated critical habitat. Therefore, the Agency makes a "may affect, and likely to adversely affect" determination and a a determination of potential effects to PCEs of the designated critical habitat for the DS from the use of alachlor based on the potential for indirect effects and effects to habitat.

The labeled use of alachlor may:

- … directly affect terrestrial-phase CRLF by causing acute mortality or by adversely affecting chronic growth or fecundity;
- … indirectly affect the CRLF and the DS and/or affect their designated critical habitat by reducing or changing the composition of the food supply;
- … indirectly affect the CRLF and the DS and/or affect their designated critical habitat by reducing or changing the composition of the aquatic plant community in the species' current range, thus, affecting primary productivity and/or cover;
- … indirectly affect the CRLF and the DS and affect their designated critical habitat by reducing or changing the composition of the terrestrial plant community in the species' current range;
- … indirectly affect the CRLF and the DS and affect their designated critical habitat by reducing or changing aquatic habitat in their current range (via modification of water quality parameters, habitat morphology, and/or sedimentation).

6.0 Uncertainties

6.1 Exposure Assessment Uncertainties

6.1.1 Maximum Use Scenario

The screening-level risk assessment focuses on characterizing potential ecological risks resulting from a maximum use scenario, which is determined from labeled statements of maximum application rate and number of applications with the shortest time interval between applications. The frequency at which actual uses approach this maximum use scenario may be dependant on pest resistance, timing of applications, cultural practices, and market forces.

6.1.2. Impact of Vegetative Setbacks on Runoff

Unlike spray drift, models are currently not available to evaluate the effectiveness of a vegetative setback on runoff and loadings. The effectiveness of vegetative setbacks is highly dependent on the condition of the vegetative strip. For example, a well-established, healthy vegetative setback can be a very effective means of reducing runoff and erosion from agricultural fields (USDA, NRCS, 2000). Alternatively, a setback of poor vegetative quality or a setback that is channelized can be ineffective at reducing loadings. Until such time as a quantitative method to estimate the effect of vegetative setbacks on various conditions on pesticide loadings becomes available, the aquatic exposure predictions are likely to overestimate exposure where healthy vegetative setbacks exist and underestimate exposure where poorly developed, channelized, or bare setbacks exist.

6.1.3 Aquatic Exposure Modeling of Alachlor

The standard ecological water body scenario (EXAMS pond) used to calculate potential aquatic exposure to pesticides is intended to represent conservative estimates, and to avoid underestimations of the actual exposure. The standard scenario consists of application to a 10-hectare field bordering a 1-hectare, 2-meter deep (20,000 m^3) pond with no outlet. Exposure estimates generated using the EXAMS pond are intended to represent a wide variety of vulnerable water bodies that occur at the top of watersheds including prairie pot holes, playa lakes, wetlands, vernal pools, human-made and natural ponds, and intermittent and lower order streams. As a group, there are factors that make these water bodies more or less vulnerable than the EXAMS pond. Static water bodies that have larger ratios of pesticide-treated drainage area to water body volume would be expected to have higher peak EECs than the EXAMS pond. These water bodies will be either smaller in size or have larger drainage areas. Smaller water bodies have limited storage capacity and thus may overflow and carry pesticide in the discharge, whereas the EXAMS pond has no discharge. As watershed size increases beyond 10-hectares, it becomes increasingly unlikely that the entire watershed is planted with a single crop that is all treated simultaneously with the pesticide. Headwater streams can also have peak concentrations higher than the EXAMS pond, but they likely persist for only short periods of time and are then carried and dissipated downstream.

The Agency acknowledges that there are some unique aquatic habitats that are not accurately captured by this modeling scenario and modeling results may, therefore, under- or over-estimate exposure, depending on a number of variables. For example, some organisms may inhabit water bodies of different size and depth and/or are located adjacent to larger or smaller drainage areas than the EXAMS pond. In addition, the Services agree that the existing EXAMS pond represents the best currently available approach for estimating aquatic exposure to pesticides (USFWS/NMFS 2004).

In general, the linked PRZM/EXAMS model produces estimated aquatic concentrations that are expected to be exceeded once within a ten-year period. The Pesticide Root Zone Model is a process or “simulation” model that calculates what happens to a pesticide in an agricultural field on a day-to-day basis. It considers factors such as rainfall and plant transpiration of water, as well as how and when the pesticide is applied. It has two major components: hydrology and chemical transport. Water movement is simulated by the use of generalized soil parameters, including field capacity, wilting point, and saturation water content. The chemical transport component can simulate pesticide application on the soil or on the plant foliage. Dissolved, adsorbed, and vapor-phase concentrations in the soil are estimated by simultaneously considering the processes of pesticide uptake by plants, surface runoff, erosion, decay, volatilization, foliar wash-off, advection, dispersion, and retardation.

Uncertainties associated with each of these individual components add to the overall uncertainty of the modeled concentrations. Additionally, model inputs from the environmental fate degradation studies are chosen to represent the upper confidence bound on the mean values that are not expected to be exceeded in the environment approximately 90 percent of the time. Mobility input values are chosen to be representative of conditions in the environment. The natural variation in soils adds to the uncertainty of modeled values. Factors such as application date, crop emergence date, and canopy cover can also affect estimated concentrations, adding to the uncertainty of modeled values. Factors within the ambient environment such as soil temperatures, sunlight intensity, antecedent soil moisture, and surface water temperatures can cause actual aquatic concentrations to differ for the modeled values.

The modeling for the residential woody ornamental uses two scenarios in tandem requiring post-processing of the modeled output in order to derive a weighted EEC that represent the contribution of both the pervious residential and the impervious surfaces. The residential scenario assumes that less than 100% of the watershed of a urban/suburban system will be treated (assuming a typical lot equals approximately ¼ of an acre). For alachlor treatments to woody ornamentals (post transplant), it was estimated that approximately 15% of the surface area of the lot could be treated because a typical lawn would not be planted with 100% juniper or yew (section 3). These EECs are based on potential routes of exposure and it is unlikely that every home will be planted with juniper and yew (*i.e.*, there are other planting options). In general, incorporation of impervious surfaces into the exposure assessment results in increasing runoff volume in the watershed, which tends to reduce overall pesticide exposure assuming 1.68% overspray to the impervious surface. Alternative assumptions for percent impervious surfaces, percentage of use site treated, and percentage of overspray should be considered in order to characterize the assumptions presented above in the context of the individual exposure assessment and risk conclusions.

In order to account for uncertainties associated with modeling, available monitoring data were compared to PRZM/EXAMS estimates of peak EECs for the different uses. As discussed above, several data values were available from NAWQA, and other sources (*e.g.*, California DPR) for alachlor concentrations measured in surface waters receiving runoff from agricultural areas. However, there is only a limited dataset relevant to potential exposure in California. For the NAWQA data, the specific use patterns (*e.g.* application rates and timing, crops) associated with the agricultural areas are unknown, however, they are assumed to be representative of potential alachlor use areas. Use information is well correlated with the California DPR surface water data, as the Cal PUR data is of high quality at the county level. Use information for other data sources is unknown.

6.1.4. Uncertainties Regarding Dilution and Chemical Transformations in Estuaries

PRZM-EXAMS modeled EECs were initially calibrated to represent relatively small ponds and low-order streams. Therefore it would seem likely that results from the PRZM-EXAMS model should greatly over-estimate potential concentrations in much larger receiving water bodies such as estuaries, embayments, and coastal marine areas; chemicals in runoff water (or spray drift, *etc*.) should simply be diluted by a much larger volume of water than would be found in the 'typical' EXAMS pond. However, as chemical constituents in water draining from freshwater streams encounter brackishness or other near-marine-associated conditions, there is potential for important chemical transformations to occur. Many chemical compounds can undergo changes in mobility, toxicity, or persistence when changes in pH, conductivity (Eh), salinity, dissolved oxygen (DO) content, or temperature are encountered. For example, desorption and re-mobilization of some chemicals from sediments can occur with changes in salinity (*e.g.*, Means 1995; Swarzenski *et al*. 2003; Jordan *et al*. 2008), changes in pH (*e.g.*, Wood and Baptista 1993; Parikh *et al*. 2004; Fernandez *et al.* 2005), Eh changes (Wood and Baptista 1993; Velde and Church 1999), and other factors. Thus, although chemicals in discharging rivers may be diluted by large volumes of water within receiving estuaries and embayments, the hydrochemistry of the marine-influenced water may negate some of the attenuating impact of the greater water volume; for example, the effect of dilution may be partly counteracted by increased mobility of a chemical in brackish water. In addition, freshwater contributions from discharging streams and rivers do not instantaneously mix with more saline water bodies. In these settings, water will commonly remain highly stratified, with fresh water lying atop denser, heavier saline water – meaning that exposure to concentrations found in discharging stream water may propagate some distance beyond the outflow point of the stream (especially near the water surface).

Therefore, EFED does not automatically assume that discharging water will be rapidly diluted by the entire water volume within an estuary, embayment, or other coastal aquatic environment; PRZM-EXAMS model results should be considered consistent with concentrations that might be found near the head of an estuary unless there is specific information to indicate otherwise. Conditions nearer to the mouth of a bay or estuary, however, may be closer to a marine-type system, and thus more subject to the notable buffering, mixing, and diluting capacities of an open marine environment. Conversely, tidal effects (pressure waves) can propagate much further upstream than the actual estuarine water, so discharging river water may become temporarily partially impounded near the mouth (discharge point) of a channel, and resistant to mixing until tidal forces are reversed.

6.1.4 Ground Water Uncertainties

Although the potential impact of discharging ground water on CRLF populations is not explicitly delineated, it should be noted that, in some areas of the country, ground water could provide a source of pesticide to surface water bodies – especially low-order streams, headwaters, and ground water-fed pools. This is particularly likely if the chemical is persistent and mobile, the pesticide is applied to highly permeable soils overlying shallow unconfined ground water, and rainfall is sufficient to drive the chemical through the soil to ground water. Soluble chemicals that are primarily subject to photolytic degradation will be very likely to persist in ground water, and can be transportable over long distances. Similarly, many chemicals degrade slowly under anaerobic conditions (common in aquifers) and are thus more persistent in ground water. Under the right hydrologic conditions, this ground water may eventually be discharged to the surface – often supporting stream flow in the absence of rainfall. Continuously flowing low-order streams in particular are sustained by ground water discharge, which can constitute 100% of stream flow during baseflow (no runoff) conditions. Thus, it is important to keep in mind that pesticides in ground water may impact surface water quality during base flow conditions with subsequent impact on CRLF habitats. However, many smaller streams in CA are net dischargers of water to ground water that go dry during portions of the year and are not supplied by baseflow from ground water.

Although concentrations in a receiving water body resulting from ground water discharge cannot be explicitly quantified, it should be assumed that significant attenuation and retardation of the chemical will have occurred prior to discharge. Nevertheless, where alachlor is applied to highly permeable soils over shallow ground water where there is a net recharge to adjacent streams, ground water could still be a consistent source of chronic background concentrations in surface water, and may also add to surface runoff during storm events (as a result of enhanced ground water discharge typically characterized by the 'tailing limb' of a storm hydrograph).

6.1.5 Usage Uncertainties

County-level usage data were obtained from CDPR PUR database. Eight years of data (1999 – 2006) were included in this analysis because statistical methodology for identifying outliers, in terms of area treated and pounds applied, was provided by CDPR for these years only. CDPR PUR documentation indicates that errors in the data may include the following: a misplaced decimal; incorrect measures, area treated, or units; and reports of diluted pesticide concentrations. In addition, it is possible that the data may contain reports for pesticide uses that have been cancelled. The CPDR PUR data does not include home owner applied pesticides; therefore, residential uses are not likely to be reported. As with all pesticide usage data, there may be instances of misuse and misreporting. The Agency made use of the most current, verifiable information; in cases where there were discrepancies, the most conservative information was used.

6.1.6 Terrestrial Exposure Modeling of Alachlor

The Agency relies on the work of Fletcher *et al*. (1994) for setting the assumed pesticide residues in wildlife dietary items. These residue assumptions are believed to reflect a realistic upper-bound residue estimate, although the degree to which this assumption reflects a specific percentile estimate is difficult to quantify. It is important to note that the field measurement efforts used to develop the Fletcher *et al*. (1994) estimates of exposure involve highly varied sampling techniques. It is entirely possible that much of this data reflects residues averaged over entire above ground plants in the case of grass and forage sampling.

It was assumed that ingestion of food items in the field occurs at rates commensurate with those in the laboratory. Although the screening assessment process adjusts dry-weight estimates of food intake to reflect the increased mass in fresh-weight wildlife food intake estimates, it does not allow for gross energy differences. Direct comparison of a laboratory dietary concentration-based effects threshold to a fresh-weight pesticide residue estimate would result in an underestimation of field exposure by food consumption by a factor of 1.25 – 2.5 for most food items.

Differences in assimilative efficiency between laboratory and wild diets suggest that current screening assessment methods do not account for a potentially important aspect of food requirements. Depending upon species and dietary matrix, bird assimilation of wild diet energy ranges from 23 – 80%, and mammal's assimilation ranges from 41 – 85% (USEPA, 1993). If it is assumed that laboratory chow is formulated to maximize assimilative efficiency (*e.g.*, a value of 85%), a potential for underestimation of exposure may exist by assuming that consumption of food in the wild is comparable with consumption during laboratory testing. In the screening process, exposure may be underestimated because metabolic rates are not related to food consumption.

For the terrestrial exposure analysis of this risk assessment, a generic bird or mammal was assumed to occupy either the treated field or adjacent areas receiving a treatment rate on the field. Actual habitat requirements of any particular terrestrial species were not considered, and it was assumed that species occupy, exclusively and permanently, the modeled treatment area. Spray drift model predictions suggest that this assumption leads to an overestimation of exposure to species that do not occupy the treated field exclusively and permanently.

6.1.7 Spray Drift Modeling

Factors, including variations in topography, cover, and meteorological conditions over the transport distance are not accounted for by the AgDRIFT model (*i.e*., it models spray drift from aerial and ground applications in a flat area with little to no ground cover and a steady, constant wind speed and direction). Therefore, in most cases, the drift estimates from AgDRIFT may overestimate exposure, especially as the distance increases from the site of application, since the model does not account for potential obstructions (*e.g*., large hills, berms, buildings, trees, *etc*.). Furthermore, conservative assumptions are made regarding the droplet size distributions being modeled (‘ASAE Very Fine to Fine’) and boom height (‘High’) unless spray drift restrictions are specified on the label. Alterations in any of these inputs would decrease the area of potential effect.

6.2 Effects Assessment Uncertainties

6.2.1 Age Class and Sensitivity of Effects Thresholds

It is generally recognized that test organism age may have a significant impact on the observed sensitivity to a toxicant. The acute toxicity data for fish are collected on juvenile fish between 0.1 and 5 grams. Aquatic invertebrate acute testing is performed on recommended immature age classes (*e.g*., first instar for daphnids, second instar for amphipods, stoneflies, mayflies, and third instar for midges).

Testing of juveniles may overestimate toxicity at older age classes for pesticide active ingredients that act directly without metabolic transformation because younger age classes may not have the enzymatic systems associated with detoxifying xenobiotics. In so far as the available toxicity data may provide ranges of sensitivity information with respect to age class, this assessment uses the most sensitive life-stage information as measures of effect for surrogate aquatic animals, and is therefore, considered as protective.

6.2.2 Impact of Multiple Stressors on the Effects Determination

The influence of length of exposure and concurrent environmental stressors to the CRLF and the DS (*i.e.*, construction of dams and locks, fragmentation of habitat, change in flow regimes, increased sedimentation, degradation of quantity and quality of water in the watersheds of the action area, predators, *etc*.) will likely affect the species' response to alachlor. Additional environmental stressors may increase sensitivity to the herbicide, although there is the possibility of additive/synergistic reactions. Timing, peak concentration, and duration of exposure are critical in terms of evaluating effects, and these factors are expected to vary both temporally and spatially within the action area. Overall, the effect of this variability may result in either an overestimation or underestimation of risk. However, as previously discussed, the Agency's LOCs are set to be protective given the wide range of possible uncertainties.

6.2.3. Use of Surrogate Species Effects Data

Freshwater fish are used as surrogate species for aquatic-phase amphibians. Some data are available on alachlor that evaluated its toxicity to amphibians. Overall, these data do not suggest that amphibians are more sensitive than fish to alachlor. Therefore, endpoints based on freshwater fish ecotoxicity data are assumed to be protective of potential direct effects to aquatic-phase amphibians including the CRLF, and extrapolation of the risk conclusions from the most sensitive tested species to the aquatic-phase CRLF is likely to overestimate the potential risks to those species. Efforts are made to select the organisms most likely to be affected by the type of compound and usage pattern; however, there is an inherent uncertainty in extrapolating across phyla. In addition, the Agency's LOCs are intentionally set very low, and conservative estimates are made in the screening level risk assessment to account for these uncertainties.

6.2.5. Sublethal Effects

The assessment endpoints used in ecological risk assessment include potential effects on survival, growth, and reproduction of the CRLF and the DS and organisms on which these species depend for survival and reproduction such as invertebrates. A number of studies were located that evaluated potential sublethal effects to fish from exposure to alachlor. Although many of these studies reported toxicity values that were less sensitive than the submitted studies, they were not considered for use in risk estimation (see **Appendix K**).

EPA is required under the FFDCA, as amended by FQPA, to develop a screening program to determine whether certain substances (including all pesticide active and other ingredients) "may have an effect in humans that is similar to an effect produced by a naturally occurring estrogen, or other such endocrine effects as the Administrator may designate." Following the recommendations of its Endocrine Disruptor Screening and Testing Advisory Committee (EDSTAC), EPA determined that there were scientific bases for including, as part of the program, androgen and thyroid hormone systems, in addition to the estrogen hormone system. EPA also adopted EDSTAC's recommendation that the Program include evaluations of potential effects in wildlife. When the appropriate screening and/or testing protocols being considered under the Agency's Endocrine Disrupter Screening Program (EDSP) have been developed and vetted, alachlor may be subjected to additional screening and/or testing. For further information on the status of the Endocrine Disrupter Screening Program please visit our website: http://www.epa.gov/endo/.

6.2.6. Exposure to Pesticide Mixtures

In accordance with the Overview Document and the Services Evaluation Memorandum (USEPA, 2004; USFWS/NMFS, 2004), this assessment considers the single active ingredient of alachlor. However, the assessed species and its environments may be exposed to multiple pesticides simultaneously. Interactions of other toxic agents with alachlor could result in additive effects, more than additive effects, or less than additive effects. As previously discussed, evaluation of pesticide mixtures is beyond the scope of this assessment because of the myriad of factors that cannot be quantified based on the available data. Those factors include identification of other possible co-contaminants where the CRLF and the DS reside and their concentrations, differences in the pattern and duration of exposure among contaminants, and the differential effects of other physical/chemical characteristics of the receiving waters (*e.g*. organic matter present in sediment and suspended water). Evaluation of factors that could influence additivity/synergism/antagonism is beyond the nature and quality of the available data to allow for an evaluation. However, it is acknowledged that not considering mixtures could over- or under-estimate risks depending on the type of interaction and factors discussed above.

6.3. Uncertainty in the Potential Effect to Riparian Vegetation vs. Water Quality Impacts

Effects to riparian vegetation were evaluated using submitted guideline seedling emergence and vegetative vigor studies. LOCs were exceeded for seedling emergence and vegetative vigor endpoints with the seedling emergence endpoint being considerably more sensitive. Based on LOC exceedances and the lack of readily available information to allow for characterization of riparian areas of the CRLF and the DS, it was concluded that alachlor use is likely to adversely affect these species by potentially impacting grassy/herbaceous riparian vegetation resulting in increased sedimentation. However, soil retention/sediment loading is dependent on a number of factors including land management and tillage practices. Use of herbicides (including alachlor) may be incorporated into a soil conservation plan. Therefore, although this assessment concludes that alachlor is likely to adversely affect the assessed listed species and their designated critical habitat by potentially impacting sensitive herbaceous riparian areas, it is possible that adverse impacts on sediment loading may not occur in areas where soil retention strategies are used.

6.2.4 Location of Wildlife Species

For the terrestrial exposure analysis of this risk assessment, a generic bird or mammal was assumed to occupy either the treated field or adjacent areas receiving a treatment rate on the field. Actual habitat requirements of any particular terrestrial species were not considered, and it was assumed that species occupy, exclusively and permanently, the modeled treatment area. Spray drift model predictions suggest that this assumption leads to an overestimation of exposure to species that do not occupy the treated field exclusively and permanently.

7.0 Risk Conclusions

Based on the best available information, the Agency makes a May Affect, and Likely to Adversely Affect (LAA) determination for the CRLF and the DS from the labeled uses of alachlor as described in **Table 7.1**. The effects determination is based on potential direct and indirect effects to terrestrial-phase CRLF and indirect effects to aquatic-phase CRLF and the DS. The LAA determination applies to all currently registered alachlor uses in California.

Additionally, the Agency has determined that there is the potential for effects to designated critical habitat of the CRLF and the DS from the use of the alachlor. A summary of the risk conclusions and effects determinations for each listed species assessed and their designated critical habitat is presented in **Tables 7.1** and **7.2**. Further information on the results of the effects determination is included as part of the Risk Description in Section 5.2. Given the LAA determination for the CRLF and the DS and potential effects to designated critical habitat for both species, a description of the baseline status and cumulative effects for the CRLF is provided in **Attachment 2** and the baseline status and cumulative effects for the DS is provided in **Attachment 4**.

Table 7.1. Effects Determination Summary for Effects of Alachlor on the CRLF and the DS.

Species	Effects Determination [1]	Basis for Determination
California red-legged frog (*Rana aurora draytonii*)	**LAA[1]**	**Potential for Direct Effects**
		Aquatic-phase (Eggs, Larvae, and Adults): None of the RQs for freshwater fish (used as a surrogate for aquatic-phase amphibians) exceed the Agency's LOCs for any registered alachlor use.
		Terrestrial-phase (Juveniles and Adults): The risk of direct adverse effects to terrestrial-phase CRLF from acute or sub-acute dietary exposure is low. However, the risk (or potential risk) to terrestrial-phase CRLF from chronic dietary exposure cannot be precluded and exists for all dietary classes relevant to the CRLF (for all of the registered alachlor uses).
		Potential for Indirect Effects
		Aquatic prey items, aquatic habitat, cover and/or primary productivity Alachlor could potentially impact terrestrial and aquatic plants to an extent that could result in indirect effects to the CRLF.
		Terrestrial prey items, riparian habitat CRLFs could be affected as a result of potential impacts to grassy/herbaceous vegetation. Potential effects to amphibian food item abundance that may indirectly affect terrestrial phase CRLFs could not be precluded.
Delta Smelt (*Hypomesus transpacificus*)	**LAA[1]**	**Potential for Direct Effects** None of the RQs for freshwater fish exceed the Agency's LOCs for any registered alachlor use.
		Potential for Indirect Effects Labeled uses of alachlor have the potential to adversely affect the DS by reducing available food (aquatic plants), by impacting the riparian habitat of grassy and herbaceous riparian areas, and/or by impacting water quality via effects to aquatic vegetation.

[1] May affect, likely to adversely affect (LAA)

Table 7.2. Effects Determination Summary for Alachlor Use and CRLF and DS Critical Habitat Impact Analysis.

Assessment Endpoint	Effects Determination	Basis for Determination
Modification of aquatic-phase PCEs (DS and CRLF)	**Habitat Effects**	As described in Table 7.1., the effects determination for the potential for alachlor to affect aquatic-phase CRLFs and the DS is LAA. These determinations are based on the potential for alachlor to indirectly affect the DS and aquatic-phase CRLF. Additionally, the potential areas of effect overlap with critical habitat designated for the CRLF and DS. Therefore, potential effects to aquatic plants and terrestrial (riparian) plants identified in this assessment could result in aquatic habitat modification.
Modification of terrestrial-phase PCE (CRLF)		As described in Table 7.1., the effects determination for the potential for alachlor to affect terrestrial-phase CRLFs is LAA. This determination is based on the potential for alachlor to directly affect terrestrial-phase CRLFs and their food supply and habitat. Additionally, the potential areas of effect overlap with critical habitat designated for the CRLF. Therefore, these potential effects could result in modification of critical habitat.

Based on the conclusions of this assessment, a formal consultation with the U. S. Fish and Wildlife Service under Section 7 of the Endangered Species Act should be initiated.

When evaluating the significance of this risk assessment's direct/indirect and adverse habitat modification effects determinations, it is important to note that pesticide exposures and predicted risks to the listed species and its resources (*i.e*., food and habitat) are not expected to be uniform across the action area. In fact, given the assumptions of drift and downstream transport (*i.e*., attenuation with distance), pesticide exposure and associated risks to the species and its resources are expected to decrease with increasing distance away from the treated field or site of application. Evaluation of the implication of this non-uniform distribution of risk to the species would require information and assessment techniques that are not currently available.

When evaluating the significance of this risk assessment's direct/indirect and adverse habitat modification effects determinations, it is important to note that pesticide exposures and predicted risks to the species and its resources (i.e., food and habitat) are not expected to be uniform across the action area. In fact, given the assumptions of drift and downstream transport (i.e., attenuation with distance), pesticide exposure and associated risks to the species and its resources are expected to decrease with increasing distance away from the treated field or site of application. Evaluation of the implication of this non-uniform distribution of risk to the species would require information and assessment techniques that are not currently available. Examples of such information and methodology required for this type of analysis would include the following:

- Enhanced information on the density and distribution of CRLF and the DS life stages within the action area and/or applicable designated critical habitat. This information would allow for quantitative extrapolation of the present risk assessment's predictions of individual effects to the proportion of the population extant within geographical areas where those effects are predicted. Furthermore, such population information would allow for a more comprehensive evaluation of the significance of potential resource impairment to individuals of the assessed species.

- Quantitative information on prey base requirements for the assessed species. While existing information provides a preliminary picture of the types of food sources utilized by the assessed species, it does not establish minimal requirements to sustain healthy individuals at varying life stages. Such information could be used to establish biologically relevant thresholds of effects on the prey base, and ultimately establish geographical limits to those effects. This information could be used together with the density data discussed above to characterize the likelihood of adverse effects to individuals.
- Information on population responses of prey base organisms to the pesticide. Currently, methodologies are limited to predicting exposures and likely levels of direct mortality, growth or reproductive impairment immediately following exposure to the pesticide. The degree to which repeated exposure events and the inherent demographic characteristics of the prey population play into the extent to which prey resources may recover is not predictable. An enhanced understanding of long-term prey responses to pesticide exposure would allow for a more refined determination of the magnitude and duration of resource impairment, and together with the information described above, a more complete prediction of effects to individual species and potential modification to critical habitat.

8.0 References

Becker, R.L., D. Hergfel, K.R. Ostlie, E.J. Stamm-Katovick. 1989. Pesticides – Surface Runoff, Leaching, and Exposure Concerns. Universtiy of Minnesota Extension Service Publication AG-BU-3911, 32 pp. *As Cited by: Goolsby, D.A., E.M. Thurman, M.L. Pomes, M.T. Meyer, W.A. Battaglin. 1997. Herbicides and their Metabolites in Rainfall: Origin, Transport, and Deposition Patterns across the Midwestern and Northeastern United States, 1990-1991. Environ. Sci. Technol. 31:1325-1333.*

Capel, P., Lin, M., Wotzka, J. 1994. Pesticides in Rain in Minnesota, 1991-1993:An Interim Report (includes addendum dated Sept, 1994). Unpublished studyprepared by University of Minnesota. 33 p.

Doherty, M. A. (1997). Biochemical Toxicology of Herbicide Mixtures on Thalassiosira weisflogii. *Ph.D.Thesis, Univ.of Maryland College Park,MD* 276 p. EcoReference No.: 105925

Environmental Fate and Effects Division. 2006. Memorandum: Standardized Soil Mobility Classification Guidance. U.S. Environmental Protection Agency, Office of Prevention, Pesticides and Toxic Substances. April 21, 2006.

Fernandez, S., C. Santin, J. Marquinez, and M.A. Alvarez. 2005. Changes in soils due to polderization in coastal plain estuaries. Geophysical Research Abstracts 7, 3pp.

Fletcher, J.S., J.E. Nellessen, and T.G. Pfleeger. 1994. Literature review and evaluation of the EPA food-chain (Kenaga) nomogram, and instrument for estimating pesticide residues on plants. Environmental Toxicology and Chemistry 13 (9):1383-1391.

Foreman, W.T., M.S. Majewski, D.A. Goolsby, F.W. Wiebe. 2000. Pesticides in the atmosphere of the Mississippi River Valley, Part I – Air. The Science of the Total Environment, 248:213-216, Gillman, Jeffery, J.P McKinnon, D.L Brown. Choosing Landscpe Evergreens. University of Minnesota, Dept. of Horticultural Science, Extension Service. FO-01430: http://www.extension.umn.edu/distribution/horticulture/DG1430.html.

Gish, T.J., A. Sadeghi, B.J. Wienhold. 1995. Volatilization of Alachlor and Atrazine as Influenced by Surface Litter. Chemosphere, 31(4):2971-2982.

Goolsby, D.A., E.M. Thurman, M.L. Pomes, M.T. Meyer, W.A. Battaglin. 1997. Herbicides and their Metabolites in Rainfall: Origin, Transport, and Deposition Patterns across the Midwestern and Northeastern United States, 1990-1991. Environ. Sci. Technol. 31:1325-1333.

Hansch, C., Leo, A., Hoekman, D. 1995. Exploring QSAR Hydrophobic, Electronic and Steric Constants. ACS, Washington, DC.

Hayes, T. B., Case, P., Chui, S., Chung, D., Haeffele, C., Haston, K., Lee, M., Mai, V. P., Marjuoa, Y, Parker, J., and Tsui, M. 2006. Pesticide Mixtures, Endocrine Disruption, and Amphibian Declines: Are We Underestimating the Impact? *Environ.Health Perspect.* 114: 40-50. EcoReference No.: 85815

Hoerger, F., and E.E. Kenaga. 1972. Pesticide residues on plants: Correlation of representative data as a basis for estimation of their magnitude in the environment. In F. Coulston and F. Korte, *eds*., Environmental Quality and Safety: Chemistry, Toxicology, and Technology, Georg Thieme Publ, Stuttgart, West Germany, pp. 9-28.

Howe, G. E., Gillis, R., and Mowbray, R. C. 1998. Effect of Chemical Synergy and Larval Stage on the Toxicity of Atrazine and Alachlor to Amphibian Larvae. *Environ.Toxicol.Chem.* 17: 519-525. EcoReference No.: 18805

Jordan, T.E., J.C. Cornwell, R.B. Walter, and J.T. Anderson. 2008. Changes in phosphorus biogeochemistry along an estuarine salinity gradient. Limnology and Oceanography 53(1): 172-184.

Kang, H. S., Gye, M. C., and Kim, M. K. 2005. Effects of Alachlor on Survival and Development of Bombina orientalis (Boulenger) Embryos. *Bull.Environ.Contam.Toxicol.* 74: 1199-1206. EcoReference No.: 81388

Kuang, Z. L.L. McConnell, A. Torrents, D. Merrit, S. Tobash. 2003. Atmospheric deposition of pesticides to an agricultural watershed of the Chesapeake Bay. J. Environ. Qual. 32:1611-1622.

Lee, K. W., Raisuddin, S., Hwang, D. S., Park, H. G., Dahms, H. U., Ahn, I. Y., and Lee, J. S. 2008. Two-Generation Toxicity Study on the Copepod Model Species Tigriopus japonicus. *Chemosphere* 72: 1359-1365. EcoReference No.: 104287

Majewski , M.S. and P.D. Capel. 1995. Pesticides in the atmosphere: distribution, trends, and governing factors. Ann Arbor Press, Inc. Chelsea, MI.

Majewski, M.S., W.T. Foreman, D.A. Goolsby. 2000. Pesticides in the atmosphere of the Mississippi River Valley, Part I – Rain. The Science of the Total Environment, 248:201-212.

Means, J.C. 1995. Influence of salinity upon sediment-water partitioning of aromatic hydrocarbons. Marine Chemistry 51(1): 3-16.

Osano, O., Admiraal, W., and Otieno, D. 2002. Developmental Disorders in Embryos of the Frog Xenopus laevis Induced by Chloroacetanilide Herbicides and Their Degradation Products. *Environ.Toxicol.Chem.* 21: 375-379.
EcoReference No.: 66376

Parikh S.J., J. Chorover, and W.D Burgos. 2004. Interaction of phenanthrene and its primary metabolite (1-hydroxy-2-naphthoic acid) with estuarine sediments and humic fractions. Journal of Contaminant Hydrology 72(1-4): 1-22.

Pennington, P. L. 1996. The Toxicity of the Herbicides Atrazine and Alachlor on the Estuarine Phytoplankter Pavlova sp. (Prymnesiophyceae) with an Emphasis on Acute Toxicity Testing of Individual Herbicides, Herbicide Mixtures and Multi-Generational Chronic Bioassays. *M.S.Thesis, Univ.of Charleston, Charleston, SC* 142 p. EcoReference No.: 106637.

Rosgen, D. 1996. *Applied river morphology*. Wildlife Hydrology, Pagosa Springs, CO.

Ross, M.A., C.R. Medlin. 2001. Herbicide Mode-of-Action Categories. Ref. #WS-24-W. Purdue University, Weed Science. Online at: http://www.btny.purdue.edu/Pubs/WS/WS-24-W.pdf Last update March 21, 2001.

Scheyer, A. S. Morville, P. Mirabel, M. Millet. 2007. Pesticides Analysed in Rainwater in Alsace Region (Eastern France): Comparison between Urban and Rural Sites. Atmospheric Environment, 41:7241-7252.

Starbuck, Christopher J. 2003. Selecting Landscape Plants: Needled Evergreens. University of Missouri-Columbia, Outreach & Extension. G 6815: http://extension.missouri.edu/explorepdf/agguides/hort/g06815.pdf

Swarzenski, P.W., D. Porcelli, P.S. Andersson, and J.M Smoak. 2003. The behavior of U- and Th-series nuclides in the estuarine environment. Reviews in Mineralogy and Geochemistry 52(1): 577-606.

USC. 2005. The Peromyscus Database: California Mouse (*Peromyscus californicus*). University of South Carolina , Genetic Stock Center. http://wotan.cse.sc.edu/perobase/systematics/p_calif.htm

USEPA. 1992. Framework for Ecological Risk Assessment. U.S. Environmental Protection Agency, Risk Assessment Forum, Washington, DC, EPA/630/R-92/001, 1992.

USEPA (1993). Wildlife Exposure Factors Handbook. EPA/600/R-93/187. Office of Research and Development. December, 1993.

USEPA. 1998. Guidance for Ecological Risk Assessment. Risk Assessment Forum. EPA/630/R-95/002F, April 1998.

USEPA. 1998a. Alachlor, The Partially Revised HED Chapter of the Reregistration Eligibility Decision Document (RED), Case 0063, Chemical 090501. U.S. Environmental Protection Agency, Office of Pesticide Programs, Health Effects Division, Memorandum. May 18, 1998.

USEPA. 1998b. Reregistration Eligibility Decision (RED) Alachlor. U.S. Environmental Protection Agency, Office of Prevention, Pesticides and Toxic Substances. EPA 738-R-98-020. Dec., 1998. Online at: http://www.epa.gov/oppsrrd1/REDs/0063.pdf.

USEPA. 2002. Guidance for Selecting Input Parameters in Modeling the Environmental Fate and Transport of Pesticides. U.S. Environmental Protection Agency, Office of Prevention, Pesticides and Toxic Substances, Office of Pesticide Programs, Environmental Fate and Effects Division. Feb. 28, 2002. Online at: http://www.epa.gov/oppefed1/models/water/input_guidance2_28_02.htm/.

USEPA. 2004. Overview of the Ecological Risk Assessment Process in the Office of Pesticide Programs. Office of Prevention, Pesticides, and Toxic Substances. Office of Pesticide Programs. Washington, D.C. January 23, 2004.

USEPA. 2006. Alachlor: Determination of Degradates to be Included in Drinking Water Exposure Assessment. U.S. Environmental Protection Agency, Office of Pesticide Programs, Health Effects Division, Memorandum, Nov. 13, 2006.

USEPA. 2006a Alachlor New Uses (Cotton and Sunflower): Tier II Drinking Water Exposure Assessment for Alachlor on Cotton and Sunflower. U.S. Environmental Protection Agency, Office of Pesticide Programs, Environmental Fate and Effects Division, Memorandum, Aug. 31, 2006.

USEPA. 2006b. Organophosphate Cumulative Risk Assessment, 2006 Update. US EPA, Office of Pesticide Programs, 1200 Pennsylvania Ave NW Washington, DC 20460.

USEPA. 2006c. Risks of Atrazine Use to Federally Listed Endangered Barton Springs Salamanders (Eurycea sosorum). Pesticide Effects Determination. Office of Pesticide Programs, Environmental Fate and Effects Division. August 22, 2006.

USEPA. 2006d. Cumulative Drinking Water Exposure Assessment for Chloroacetanilides (Acetochlor and Alachlor). U.S. Environmental Protection Agency, Office of Pesticide Programs, Environmental Fate and Effects Division, Memorandum, Feb. 6, 2006.

USEPA. 2006e. TerrPlant Version 1.2.2 (December 26, 2006) User's Guide. U.S. Environmental Protection Agency, Office of Pesticide Programs, Environmental Fate and Effects Division.

USEPA. 2008. White Paper on Methods for Assessing Ecological Risks of Pesticides with Persistent, Bioaccumulative and Toxic Characteristics. Submitted to the FIFRA Scientific Advisory Panel. October 28-31, 2008. Office of Pesticide Programs, Environmental Fate and Effects Division, Washington, D.C.

USEPA. 2009. County-Level Usage for Propanil, Paraquat Dichloride, Pendimethalin, Myclobutanil, Prometryn, Dicofol, Alachlor, and Endosulfan In California in Support of a Red Legged Frog Endangered Species Assessment. U.S. Environmental Protection Agency, Office of Pesticide Programs, Environmental Fate and Effects Division, Memorandum, February 4, 2009.

U.S. Fish and Wildlife Service (USFWS). 1995. Sacramento-San Joaquin Delta Native Fishes Recovery Plan. U.S. Fish and Wildlife Service, Portland, Oregon. Available online at: http://ecos.fws.gov/docs/recovery_plan/961126.pdf (Accessed on January 28, 2008).

U.S. Fish and Wildlife Service (USFWS). 2004. 5-Year Review. U.S. Fish and Wildlife Service, Sacramento Field Office, Sacramento, California. 50 pp. Available online at:

http://www.fws.gov/sacramento/es/documents/DS%205-yr%20rev%203-31-04.pdf (Accessed on January 28, 2007).

USFWS/NMFS. 1998. Endangered Species Consultation Handbook: Procedures for Conducting Consultation and Conference Activities Under Section 7 of the Endangered Species Act. Final Draft. March 1998

USFWS/NMFS. 2004. 50 CFR Part 402. Joint Counterpart Endangered Species Act Section 7 Consultation Regulations; Final Rule. FR 47732-47762.

USGS. 1998. Herbicides in Rainfall Across the Midwestern and Northeastern United States, 1990-91. USGS Fact Sheet 181-97. http://ks.water.usgs.gov/pubs/fact-sheets/fs.181-97.html.

Velde B. and T. Church. 1999. Rapid clay transformations in Delaware salt marshes. Applied Geochemistry 14(5): 559-568.

Wood, T.M. and A.M. Baptista. 1993. A model for diagnostic analysis of estuarine geochemistry. Water Resources Research 29(1): 51–71.

Submitted Fate Studies

MRID	Reference
40396301	Monsanto Co. (1987) Alachlor Product Chemistry--Response to Agency Review: R.D. No. 831. Unpublished study. 47 p.
152209	Brightwell, B.; Rueppel, M. (1976) Lasso and Ramrod: Physical Properties and Their Relationship to the Environment: Report No. 443. Unpublished study prepared by Monsanto Co. 14 p.
134327	Suba, L.; Pearson, D.; Malik, J. (1979) The Environmental Studies of Alachlor: R.D. #258: Report Nos. MSL-0860, MSL-0861. Final rept. (Unpublished study received Oct 11, 1979 under 524-285; submitted by Monsanto Co., Washington, DC; CDL:241135-A; 241136)
23012	Sutherland, M.L.; Curtis, T.G.; Darlington, W.A.; et al. (1972) Final Report on Lasso and the Environment: Part 4: Photolysis of Lasso on Soil and in Water: Agricultural Research Report No. 262. (Unpublished study received Jun 29, 1973 under 3F1372; submitted by Monsanto Co., Washington, D.C.; CDL:093660-L)
101531	Banduhn, M.; Livingston, C.; Kloek, J.; et al. (1981) Comparative Environmental Fate and Crop Uptake Studies of Encapsulated and Unencapsulated Alachlor: Report No. MSL-2070. (Unpublished study received May 10, 1982 under 524-344; submitted by Mon- santo Co., Washington, DC; CDL:070841-B)
134327	Suba, L.; Pearson, D.; Malik, J. (1979) The Environmental Studies of Alachlor: R.D. #258: Report Nos. MSL-0860, MSL-0861. Final rept. (Unpublished study received Oct 11, 1979 under 524-285; submitted by Monsanto Co., Washington, DC; CDL:241135-A; 241136)
23014	Sutherland, M.L.; Curtis, T.G.; Darlington, W.A.; et al. (1972) Final Report on Lasso and the Environment: Part 6: Soil Dissipa- tion of Lasso: Agricultural Research Report No. 264. (Unpub- lished study received Jun 29, 1973 under 3F1372; submitted by Monsanto Co., Washington, D.C.; CDL:093660-N)

44405301 Blumhorst, M. (1997) Soil Adsorption/Desorption of (carbon 14) Alachlor Sulfonic Acid Metabolite (ESA) by the Batch Equilibrium Method: Lab Project Number: 97-24-M-1: 115S12: MSL-14976. Unpublished study prepared by EPL Bio-Analytical Services, Inc. 106 p.

27139 Weidner, C.W. (1974) Degradation in Groundwater and Mobility of Herbicides. Master's thesis, Univ. of Nebraska, Dept. of Agron- omy. (Unpublished study received Jul 19, 1978 under 201-403; submitted by Shell Chemical Co., Washington, D.C.; CDL:234472-O)

27140 Lavy, T.L. (1974) Mobility and Deactivation of Herbicides in Soil- Water Systems: Project A-024-NEB. (Available from: National Technical Information Service, Springfield, VA: PB-238 632; un- published study received Jul 19, 1978 under 201-403; prepared by Univ. of Nebraska, Water Resources Research Institute, sub- mitted by Shell Chemical Co., Washington, D.C.; CDL:234472-P)

78301 Guth, J.A. (1975) CGA-24705 Leaching Model Study with the Herbicide CGA-24705 in Four Standard Soils: Nr. SPR 3/75. (Unpublished study received Jul 23, 1981 under 100-587; prepared by Ciba- Geigy, Ltd., Switzerland, submitted by Ciba-Geigy Corp., Greens- boro, N.C.; CDL:245628-E)

134327 Suba, L.; Pearson, D.; Malik, J. (1979) The Environmental Studies of Alachlor: R.D. #258: Report Nos. MSL-0860, MSL-0861. Final rept. (Unpublished study received Oct 11, 1979 under 524-285; submitted by Monsanto Co., Washington, DC; CDL:241135-A; 241136)

152209 Brightwell, B.; Rueppel, M. (1976) Lasso and Ramrod: Physical Prop- erties and Their Relationship to the Environment: Report No. 443. Unpublished study prepared by Monsanto Co. 14 p.

42528001 Schott, M.; Schlicher, M. (1990) Terrestrial Field Dissipation of Alachlor at Two Sites in California: Lab Project Number: MSL-10395: RD 1138. Unpublished study prepared by Monsanto Company. 313 p.

42528002 Schlicher, M.; Schott, M. (1991) Terrestrial Field Dissipation of Alachlor Metabolites in Chico, California: Lab Project Number: MSL-10844: 1138. Unpublished study prepared by Monsanto Company. 251 p.

Submitted Ecotoxicity Studies:

79523 Fink, R.; Beavers, J.B.; Brown, R.; et al. (1979) Final Report: Acute Oral LD50--Bobwhite Quail: Project No. 139-179. (Unpub- lished study received Jun 2, 1981 under 524-285; prepared by Wildlife International, Ltd., submitted by Monsanto Co., Wash- ington, D.C.; CDL:245260-A)

43087001 Grimes, J.; Jaber, M. (1986) Alachlor: A Dietary LC 50 Study with the Mallard: Lab Project Number: 139-232: WL-86-205: RD 1218. Unpublished study prepared by Wildlife International Ltd. 31 p.

43087101 Grimes, J.; Jaber, M. (1986) Alachlor: A Dietary LC 50 Study with the Bobwhite: Lab Project Number: RD 1217: 139-231: WL-86-206. Unpublished study prepared by Wildlife International Ltd. 32 p.

44951501 Gallagher, S.; Beavers, J.; Jaber, M. (1999) Alachlor: A Reproduction Study with the Mallard (Anas platyrhynchos): Lab Project Number: R.D.1492: WL-97-144: 139-436. Unpublished study prepared by Wildlife International, Ltd. 183 p.

44951502 Gallagher, S.; Beavers, J.; Jaber, M. (1999) Alachlor: A Reproduction Study with the Northern Bobwhite (Colinus virginianus): Lab Project Number: R.D.NO.1492: WL-97-145: 139-435. Unpublished study prepared by Wildlife International, Ltd. 182 p.

43774704 Bowman, J.; Downing, J.; Hurshman, B. (1995) Acute Toxicity of MON 5775 to Rainbow Trout (Oncorhynchus mykiss): Amended Final Report: Lab Project Number: 41729: AB-94-156: 1321. Unpublished study prepared by ABC Labs, Inc. 26 p.

43774706 Bowman, J.; Hurshman, B. (1995) Acute Toxicity of MON 5760 to Rainbow Trout (Oncorhynchus mykiss): Final Report: Lab Project Number: 41731: AB-94-154: 1321. Unpublished study prepared by ABC Labs, Inc. 27 p.

43774703 Bowman, J.; Hurshman, B. (1994) Acute Toxicity of MON 5775 to Daphnia magna: Final Report: Lab Project Number: 41728: AB-94-155: R.D. 1321. Unpublished study prepared by ABC Labs, Inc. 26 p.

43774705 Bowman, P.; Hurshman, B. (1994) Acute Toxicity of MON 5760 to Dahnia magna: Final Report: Lab Project Number: 41730: AB-94-153: 1321. Unpublished study prepared by ABC Labs, Inc. 26 p.

44524301 Graves, W.; Swigert, J.; Krueger, H. (1998) A 96-Hour Flow-Through Acute Toxicity Test with the Sheepshead Minnow (Cyprinodon variegatus): Alachlor: Final Report: Lab Project Number: 1410: WL-96-189: 139A-195. Unpublished study prepared by Wildlife International Ltd. 43 p.

44524302 Graves, W.; Swigert, J.; Krueger, H. (1998) A 96-Hour Flow-Through Acute Toxicity Test with the Saltwater Mysid (Mysidopsis bahia): Alachlor: Final Report: Lab Project Number: 139A-196: WL-96-190: 1410. Unpublished study prepared by Wildlife International Ltd. 42 p.

44524303 Graves, W.; Swigert, J.; Krueger, H. (1998) A 96-Hour Shell Deposition Test with the Eastern Oyster: Alachlor: Final Report: Lab Project Number: 139A-197: WL-96-191: 1410. Unpublished study prepared by Wildlife International Ltd. 44 p.

23615 Thompson, C.M.; Forbis, A.D.; McAllister, W.A. (1978) Acute Toxic- ity of Technical Alachlor (AB-78-166) to Bluegill Sunfish (?~Lepomis macrochirus~?). (Unpublished study received Aug 16, 1978 under 524-285; prepared by Analytical Biochemistry Labora- tories, Inc., submitted by Monsanto Co., Washington, D.C.; CDL: 234628-C)

23616 Forbis, A.D.; McAllister, W.A. (1978) Acute Toxicity of Technical Alachlor to Rainbow Trout (?~Salmo gairdneri). (Unpublished study received Aug 16, 1978 under 524-285; prepared by Analyt- ical Biochemistry Laboratories, Inc., submitted by Monsanto Co., Washington, D.C.; CDL:234628-D)

28550 Thompson, C.M.; Forbis, A.D. (1978) Acute Toxicity of Lasso (MCB/ Cg) (AB-78-111) to Rainbow Trout (?~Salmo_gairdneri~?). (Unpub- lished study received Dec 28, 1979 under 524-285; prepared by Analytical Bio Chemistry Laboratories, Inc., submitted by Monsanto Co., Washington, D.C.; CDL:241535-B)

28551 Thompson, C.M.; Forbis, A.D. (1978) Acute Toxicity of Lasso (MCB/ Cg) (AB-78-111) to Bluegill Sunfish (?~Lepomis macro~?- ?~chirus~?). (Unpublished study received Dec 28, 1979 under 524-285; prepared by Analytical Bio Chemistry Laboratories, Inc., submitted by Monsanto Co., Washington, D.C.; CDL: 241535-C)

43774707 Forbis, A. (1995) Chronic Toxicity of Alachlor to Daphnia magna Under Flow-Through Test Conditions: Final Report: Lab Project Number: 41517: AB-94-061: 1321. Unpublished study prepared by ABC Labs, Inc. 39 p.

43862601 Rhodes, J.; Muckerman, M. (1995) Early Life-Stage Toxicity of Alachlor to the Rainbow Trout (Oncorhynchus mykiss) Under Flow-Through Conditions: Lab Project Number: 42187: AB-94-280: R.D. 1331. Unpublished study prepared by ABC Labs, Inc. 339 p.

139383 Heenehan, P.R.; Rinehart, W.E.; Braun, W.G. (1979) Acute Oral Toxicity Study in Rats; Compound: Alachlor(Technical): Project No. 4899-77; BD-77-433. Rev. (Unpublished study, including letters dated Aug 6, 1979 from W.E. Rinehart to George Levin- skas; Aug 31, 1979 from F.B. Oleson to W.D. Carpenter, received Oct 16, 1979 under 524-285; prepared by Bio/dynamics, Inc., submitted by Monsanto Co., Washington, D.C.; CDL:241273-N)

75062 Schroeder, R.E.; Hogan, G.K.; Smock, M.E.; et al. (1981) A Three Generation Reproduction Study in Rats with Alachlor: Project No. 77-2066. Final rept. (Unpublished study received Jul 10, 1981 under 524-285; prepared by Bio/dynamics, Inc., submitted by Monsanto Co., Washington, D.C.; CDL:070177-A)

42763801 Blasberg, J.; Hicks, S.; Leak, T. (1993) Acute Toxicity of (carbon 14)-Alachlor to Selenastrum capricornutum Printz: Final Report: Lab Project Number: 40815: AB-93-52: 1171. Unpublished study prepared by ABC Labs, Inc. 56 p.

42468601 Canez, V. (1992) Tier 2 Vegetative Vigor Nontarget Phytotoxicity Study Using Alachlor: Lab Project Number: BL91-480: 0624-91-9. Unpublished study prepared by Pan-Agricultural Laboratories Inc. 226 p.

42468701 Chetram, R. (1992) Tier 2 Seed Germination/Seedling Emergence Nontarget Phytotoxicity Study Using Alachlor: Lab Project Number: BL91-481: 0624-91-8. Unpublished study prepared by Pan-Agricultural Laboratories Inc. 281 p

44649701 Drottar, K.; Kruegar, H. (1998) Alachlor: A 5-Day Toxicity Test with the Freshwater Alga (Anabaena flos-aquae): Final Report: Lab Project Number: 139A-215: WL-97-172: 1435. Unpublished study prepared by Wildlife International Ltd. 43 p.

44649702 Drottar, K.; Kruegar, H. (1998) Alachlor: A 14-Day Toxicity Test with Duckweed (Lemna gibba G3): Final Report: Lab Project Number: 1435: 139A-216: WL-97-174. Unpublished study prepared by Wildlife International LTD. 46 p.

44649703 Drottar, K.; Kruegar, H. (1998) Alachlor: A 5-Day Toxicity Test with the Marine Diatom (Skeletonema costatum): Final Report: Lab Project Number: 1435: 139A-217: WL-97-171. Unpublished study prepared by Wildlife International Ltd. 44 p.

44649704 Drottar, K.; Kruegar, H. (1998) Alachlor: A 5-Day Toxicity Test with the Freshwater Diatom (Navicula pelliculosa): Final Report: Lab Project Number: 1435: 139A-218: WL-97-173. Unpublished study prepared by Wildlife International Ltd. 43 p.

45046001 Sutherland, C.; Kendall, T.; Krueger, H. (2000) MON 5775: A 5-Day Toxicity Test with Freshwater Alga (Selenastrum capricornutum): Lab Project Number: 1999-191: RDNO1499: 139A-24. Unpublished study prepared by Wildlife International, Ltd. 50 p.

45046002 Sutherland, C.; Kendall, T.; Krueger, H. (2000) MON 5775: A 5-Day Toxicity Test with the Freshwater Alga (Anabena flos-aquae): Lab Project Number: RDNO1499: 1997-197: 139A-244. Unpublished study prepared by Wildlife International, Ltd. 49 p.

45046003 Sutherland, C.; Kendall, T.; Krueger, H. (2000) MON 5775: A 5-Day Toxicity Test with the Freshwater Diatom (Navicula pelliculosa): Lab Project Number: RDNO1499: WL1999-193: 139A-246. Unpublished study prepared by Wildlife International, Ltd. 49 p.

45046004 Sutherland, C.; Kendall, T.; Krueger, H. (2000) MON 5775: A 5-Day Toxicity Test with the Marine Diatom (Skeletonema costatum): Lab Project Number: RDNO1499: WL-199-192: 139A-248A. Unpublished study prepared by Wildlife International, Ltd. 49 p.

45046005 Sutherland, C.; Kendall, T.; Krueger, H. (2000) MON 5775: A 5-Day Toxicity Test with Duckweed (Lemna gibba G3): Lab Project Number: RDNO1499: WL-1999-196: 139A-245A. Unpublished study prepared by Wildlife International, Ltd. 45 p.

40098001 Mayer, F.; Ellersieck, M. (1986) Manual of Acute Toxicity: Interpretation and Data Base 410 Chemicals and 66 Species of Fresh-Water Animals. US Fish & Wildlife Service; Resource Publica tion (160): 579 p.

00028772 Atkins, E.L., E.A. Greywood, R.L. Macdonald (1973). Toxicity of Pesticides and Other Agricultural Chemicals to Honey Bees: Laboratory Studies. Rev. by University of California – Riverside, Dept. of Entomology. Riverside, CA: UC Agricultural Extension Service.

00028549 McAllister, W.A., A.D. Forbis (1978). Acute Toxicity of Technical Alachlor (AB-78-200) to Daphnia magna. (Unplublished study received Dec. 28, 1979 under 524-285; prepared by Analytical Bio Chemistry Laboratories, Inc., submitted by Monsanto Co., Washington, DC; CDL:241535-A)

00028552 McAllister, W.A., A.D. Forbis (1978). Acute Toxicity of Lasso (MCB/Cg) (AB-78-111) to Daphnia magna. (Unplublished study received Dec. 28, 1979 under 524-285; prepared by Analytical Bio Chemistry Laboratories, Inc., submitted by Monsanto Co., Washington, DC; CDL:241535-D)

00087855b McAllister, W.A. (1979). Residue accumulation study in channel catfish (Ictalurus punctatus) with alachlor-ul-phenyl-^{14}C under static conditions. Accumulation, distribution, and elimination. Report No. 22764. Unpublished study performed and submitted by Analytical BioChemistry Laboratories, Inc. Columbia, MO.

44105503 Kirby-Smith, W.W., S.J. Eisenreich, J.T. Howe, R.A. Luetich, Jr. (1993). The Effects in Estuaries of Pesticide Runoff from Adjacent Farm Lands. Duke University Marine Laboratory, Laboratory Report ID – CF 813415.