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Animal and Plant Health Inspection Service

National Marine Fisheries Services Biological Assessment for the APHIS Rangeland Grasshopper and Mormon Cricket Suppression Program

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Executive Summary

The U.S. Department of Agriculture (USDA), Animal and Plant Health Inspection Service (APHIS) evaluated the potential for chemical-related impacts to 28 evolutionarily significant units (ESUs) of federally listed Pacific salmonids and their critical habitat within the proposed action area for the Rangeland Grasshopper and Mormon Cricket Suppression Program. The intent of the suppression program is to reduce populations of various species of grasshoppers and Mormon crickets on rangeland in 17 Western States. Chemical treatments include a seasonal one-time treatment of diflubenzuron, carbaryl or malathion which can be made by ground or air. All three chemicals are applied at reduced rates, compared to their recommended label use, and are applied over an entire treatment area/spray block, or in alternating swaths within a treatment area/spray block. Conducting grasshopper treatments is based on many factors including the number of grasshoppers present in the area, grasshopper and plant species composition, life-cycle stage of the grasshoppers, range condition, the economic significance of the infestation, and whether it is economically and logistically feasible to conduct an effective program.

In this biological assessment, USDA–APHIS uses a risk assessment approach to evaluate response data to characterize the potential hazard/risk of the use of these chemicals to salmonids and their habitat. Toxicity data related to potential direct and indirect effects to salmonids were compared to exposure estimates for each of the chemicals to characterize risk to listed salmonids and their designated critical habitat. APHIS reviewed the ecology of the listed salmonids (their distribution throughout the program action area) to determine whether a listed entity is found within the program treatment areas and, thus, would likely be exposed to any of the program chemicals.

Based on this review, APHIS identified eight ESUs that potentially co-occur in the program area, and then used results from the risk characterization for the three chemicals to develop program application buffers and other mitigation measures to avoid and/or minimize the potential for adverse impacts to listed species and their critical habitat (table ES-1). In cases where no listed species or designated critical habitat was found in the program treatment area, a no effect determination was made (table ES-2).

Table ES–1. Proposed application buffers to protect listed salmonids

Insecticide Treatment	Method of Application	Application Buffer (feet)
Carbaryl	Aerial Ultra Low Volume (ULV)	3500
	Aerial Bait	1000
	Ground ULV	350
	Ground Bait	200
Diflubenzuron	Aerial ULV	1500
	Ground ULV	150
Malathion	Aerial ULV	3500
	Ground ULV	500

Table ES–2. Effects Determination for Various Salmonid ESUs

Species	Location	Effects Determination
Sockeye Salmon (*Oncorrynchus nerka*)	Snake River	May affect- Not likely to adversely affect
	Ozette Lake	No effect
Chinook Salmon (*Oncorrynchus tshawytscha*)	Sacramento River Winter Run	May affect- Not likely to adversely affect
	Upper Columbia River Spring Run	May affect- Not likely to adversely affect
	Snake River Spring/Summer Run	May affect- Not likely to adversely affect
	Snake River Fall Run	May affect- Not likely to adversely affect
	Puget Sound	No effect
	Upper Willamette	No effect
	Lower Columbia River	No effect
	California Coastal	No effect
	Central Valley Spring Run	No effect
Coho Salmon (*Oncorrynchus kisutch*)	Central California Coastal	No effect
	South Oregon/N. California	No effect
	Lower Columbia River	No effect
	Oregon Coast	No effect
Chum Salmon (*Oncorrynchus keta*)	Hood Canal Summer Run	No effect
	Columbia River	No effect
Steelhead (*Oncorrynchus mykiss*)	Southern California	No effect
	Upper Columbia River	May affect- Not likely to adversely affect
	Central Coastal California	No effect
	South Central California Coast	No effect
	Snake River Basin	May affect- Not likely to adversely affect
	Lower Columbia River	No effect
	California Central Valley	No effect
	Upper Willamette River	No effect
	Middle Columbia River	May affect- Not likely to adversely affect
	Northern California	No effect
	Puget Sound	No effect

In addition to the chemical specific application buffers, the following operational restrictions will apply to all proposed treatment methods to further reduce insecticide exposure to listed salmonids.

- Avoid applications when winds speeds exceed 10 mph
- Avoid applications when wind direction is blowing towards salmonid critical habitat
- Use reduced area agent treatments (RAAT) adjacent to salmonid critical habitat

- Avoid applications under conditions where a temperature inversion is possible or when a storm event is imminent.

In aggregate, the incorporation and use of buffers and other operational procedures including the restrictions listed above, APHIS anticipates that any impacts associated with the use and fate of program pesticides will be insignificant and discountable to listed salmonids and their habitats. Based on our assessment of the potential exposure, response and subsequent risk characterization of program operations, APHIS concludes that the proposed action is not likely to adversely affect the Sacramento River winter, Upper Columbia River spring, Snake River spring/summer and fall chinook, the Snake River sockeye, or the Upper Columbia, Snake River Basin, and Middle Columbia steelhead and will not result in the destruction and/or adverse modification of critical habitat.

I. Introduction and Program Description

A. Introduction

This biological assessment (BA) is intended to initiate informal consultation with the U.S. Department of the Commerce, National Marine and Fisheries Service (NMFS), to assess the potential impacts of the U.S. Department of Agriculture (USDA), Animal and Plant Health Inspection Service (APHIS) Rangeland Grasshopper and Mormon Cricket Suppression Program (program) to threatened and endangered salmonids and their habitat designated or proposed as critical under the Endangered Species Act of 1973 (ESA) in areas of the 17 Western States where APHIS grasshopper or Mormon cricket activities could occur.

The goal of this Biological Assessment (BA) is to evaluate the potential effects of APHIS program activities on listed salmonids and their designated and proposed critical habitat, and to determine whether they are likely to be adversely affected by the proposed action. This will determine whether formal consultation or a conference is necessary (50 Code of Federal Regulation 402.12(a)).

This BA is divided into five sections—(1) Description of the Proposed Program; (2) Exposure Analysis; (3) Response Analysis; (4) Risk Characterization, and (5) Program Assessment and Effects Determination. The first section contains information regarding the program or action that APHIS has been delegated under the Plant Protection Act. The next three sections of the BA describe the risk assessment that was developed to provide support for effects determinations and the development of mitigation measures described in this assessment. The risk assessment is divided into three primary sections with section two of the BA focused on the environmental fate and residue estimation, or exposure analysis, for each pesticide proposed for use in the program. The third section of the document discusses the effects analysis for all three pesticides. The fourth section integrates the effects and exposure analysis by providing a risk characterization for direct and indirect impacts to listed salmonids. The final section provides the effects determination for species and critical habitat considered in this BA based on results from the risk characterization section. It also discusses areas of uncertainty related to the ecological risk assessment.

APHIS Authority

APHIS has authority under the Plant Protection Act of 2000 (PPA) (7 United States Code (U.S.C.) § 7701) to take actions to control and minimize the economic, ecological, and human health impacts that harmful plant pests can cause. APHIS uses this authority to protect U.S. agriculture, forests, and other natural resources from harmful pest species.

Section 417 of the PPA (7 U.S.C. § 7717) authorizes APHIS' efforts to minimize the economic impacts of grasshoppers. Section 417(a) states that subject to the availability of funds, the Secretary "shall carry out a program to control grasshoppers and Mormon crickets on all Federal lands to protect rangeland."

Section 417(c)(1) states that "Subject to the availability of funds pursuant to this section, on request of the administering agency or the agriculture department of an affected State, the Secretary, to protect rangeland, shall immediately treat Federal, State, or private lands that are infested with grasshoppers or Mormon crickets at levels of economic infestation, unless the Secretary determines that delaying treatment will not cause greater economic damage to adjacent owners of rangeland." Section 417(c)(2) states, "In carrying out this section, the Secretary shall work in conjunction with other Federal, State, and private prevention, control, or suppression efforts to protect rangeland."

APHIS has the authority to implement Section 417 of the PPA through the Rangeland Grasshopper and Mormon Cricket Suppression Program. The priorities of the APHIS program are:

- to conduct surveys for grasshopper and Mormon cricket populations on rangelands in the western United States,

- to provide technical assistance on grasshopper management to land owners/managers, and

- subject to the availability of funds, to suppress grasshoppers and Mormon crickets on rangeland when direct intervention is requested by the land owner/manager.

Additional information regarding technical assistance and other aspects of the program can be obtained from the USDA Agricultural Research Service site at http://www.sidney.ars.usda.gov/grasshopper/index.htm.

B. Project Description—APHIS Rangeland Grasshopper and Mormon Cricket Suppression Program

Grasshoppers are important natural components of rangeland ecosystems, serving as a food source for wildlife and playing an important role in nutrient cycling. Rangeland is also an important agricultural resource used for livestock production. Under certain conditions, grasshopper populations can reach economic threshold levels resulting in significant damage to forage and rangeland habitat.

Grasshopper infestations often occur over extensive areas so that individual land managers alone cannot manage infestations. Therefore, a rapid and effective response may be needed to minimize the destruction of rangeland vegetation or, in some cases, to prevent grasshopper migration to adjacent cropland. The migratory patterns of grasshoppers make coordination of management efforts across State boundaries essential. In cooperation with other Federal agencies, State departments of agriculture, and private individuals/ranchers, APHIS provides direct supervision and leadership for grasshopper management programs.

In cases where an insecticide treatment is considered, the decision to conduct a grasshopper treatment is based on many factors including the number of grasshoppers present in the area,

grasshopper and plant species composition, life-cycle stage of the grasshoppers, range condition, the economic significance of the infestation, and whether it is economically and logistically feasible to conduct an effective program (figure 1-1). When both State and private lands are involved, the land owner/manager must cost-share from 33 to 67% of the total treatment costs; therefore, they are not likely to request APHIS apply the treatment unless they are reasonably certain their investment is worthwhile.

In some cases, an APHIS rangeland treatment protects not only the rangeland, but reduces the likelihood that grasshoppers will move from rangeland onto crops and other lands that border rangeland. There are also situations where crops may be growing within a large rangeland block that requires treatment. If those crops comprise no more than 10% of the area to be treated, APHIS has the option to treat the crops with a properly labeled insecticide at the crop owner's expense in order to maintain the continuity of a spray block.

1. APHIS Activities—Surveys

The survey of immature grasshoppers (i.e., nymphal survey) is conducted in the spring and early summer of the treatment year. Program personnel conduct surveys of the grasshopper nymphal populations by counting the number of grasshopper nymphs present in a given area. Grasshopper nymphal survey sampling stations occur in sufficient numbers to provide current information about various factors, including the stage of grasshopper development; location of sensitive areas such as bee yards and aquatic resources; the condition of the rangeland in relation to grasshopper numbers; and the extent of the infestation. These field data are recorded in an electronic database, from which grasshopper density geographic information system (GIS) maps are generated and verified within 7 days. These near real-time survey maps are valuable projections used for planning large-scale treatment programs and fiscal tracking, and for local decisions on treatments within a State. The data are then considered with other available information (e.g., weather forecasts) to assist the cooperating group in a decision whether to initiate a control program within that crop year.

The survey of adult grasshoppers begins soon after nymphal grasshoppers have dispersed and reached the adult stage. This survey, conducted during the late summer and early fall, is timed to coincide with peak populations, and is completed before the grasshopper populations decline. These survey data are recorded and mapped, similar to the nymphal survey. In contrast to the nymphal survey, the adult survey data are useful in predicting if and where potential grasshopper problems are likely to occur in the spring and early summer of the next growing season.

2. APHIS Activities—Technical Assistance

The survey data collected by the program is used by the agency and land managers/owners to assess whether treatments are warranted. Treatments must be requested from a Federal land management agency or a State agriculture department (on behalf of a State or local government, or private group or individual) that has jurisdiction over the land before APHIS can begin a treatment. Upon request, APHIS personnel conduct a site visit to determine whether APHIS action is warranted. Relevant factors influencing this decision may include, but are not limited to, the pest species, timing of treatment relative to the biological stage of the pest species, costs

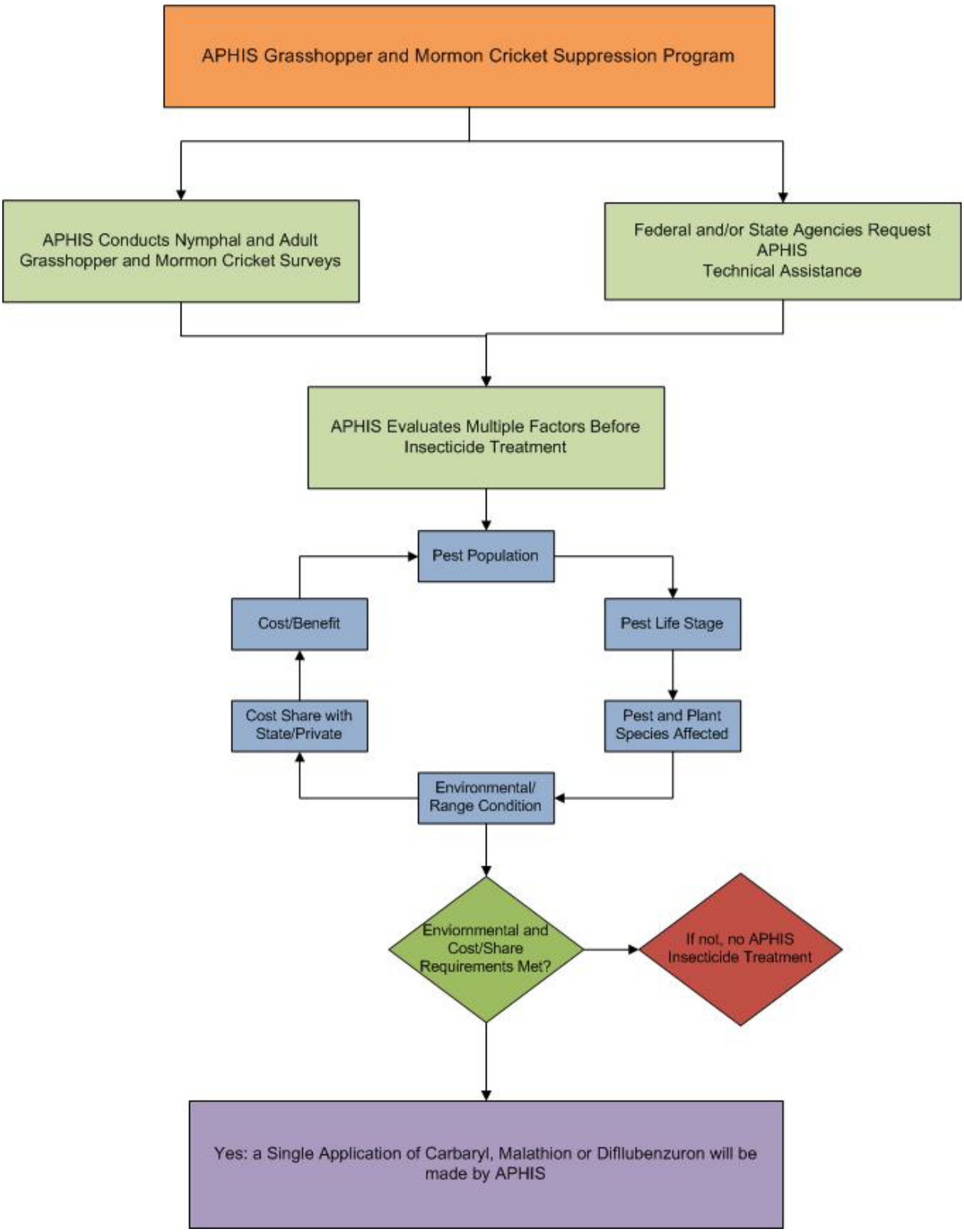

Figure 1-1. USDA APHIS activities related to the Grasshopper/Mormon Cricket Suppression Program

and benefits of conducting the action, and ecological impacts. Based on survey results conducted during the growing season, APHIS is better able to predict the potential for grasshopper populations and to respond quickly before extensive loss occurs to rangeland. Thus, State and Federal officials may initiate early coordination of local programs and request APHIS' assistance in a timely and effective cooperative effort.

3. Cooperator Role

Federal and State land management agencies, State agriculture departments, and private groups or individuals may carry out a variety of activities that may reduce the potential for grasshopper outbreaks. Some of these activities are grazing management practices, cultural and mechanical methods, preventative integrated pest management (IPM), and prescribed burning of rangeland areas. These techniques have been tried with varying success in rangeland management, and some have been associated with the prevention, control, or suppression of harmful grasshopper populations on rangeland. However, some periodic outbreaks occur which require insecticide applications.

Landowners can, and often do, conduct grasshopper treatment activities independent from APHIS. These treatments may include the use of insecticides at label rates and frequencies higher than those used by the program, or landowners may apply labeled insecticides that the program does not use which would result in increased risk to non-target organisms.

4. APHIS Activities—Chemical Treatment

a. Insecticides Used By APHIS

When direct intervention is requested by land managers, APHIS' role in the suppression of grasshoppers is achieved through a single application of an insecticide—carbaryl, diflubenzuron, or malathion. All three insecticides are labeled by the U.S. Environmental Protection Agency, Office of Pesticide Programs (EPA–OPP) for rangeland use in the control of grasshoppers, including Mormon crickets.

APHIS has chosen and approved the use of these insecticides based on (1) effective performance against grasshoppers on rangeland, and (2) minimal or negligible impact on the environment and nontarget species (Reuter and Foster, 1996). A number of other products and insecticides are labeled for use against grasshoppers on rangeland, but are not considered by APHIS for use because of efficacy or environmental concerns. The most widely used insecticide in the program is diflubenzuron which is typically used at a reduced agent area treatment (RAAT) rate.

b. APHIS Insecticide Application Methods

Program insecticides used for grasshopper suppression can be applied in one of two different forms: liquid ultra-low-volume (ULV) sprays, or solid-based baits (table 1–1). Depending upon the area requiring treatment, both forms have advantages and disadvantages. Habitat diversity, topographical features, meteorological conditions, economic concerns, and environmental considerations all have important roles in choosing the best form of treatment (Foster and

Table 1–1. Characteristics of Insecticides Used by the APHIS Rangeland Grasshopper and Mormon Cricket Suppression Program*

Insecticide	Mode of Action	Application Rates (lb a.i./acre)	Total Volume Applied	Program Use	Comments
Carbaryl ULV spray	AChE Inhibitor	0.37 5–0.50 conventional 0.25–0.375 RAATs	Conventional: 32 fl. oz. per acre (carbaryl and water in 1:1 ratio) RAATs – ½ conventional	Effective against grasshoppers and crickets season-long; can be used in wet and cool conditions.	Currently used less often than dimilin, but more often than malathion; has longer residual than malathion.
Carbaryl bait (solid formulation)	AChE Inhibitor	0.50 conventional (5% formulation) 0.20 RAATs (2% formulation)	Conventional and RAATs: 10 lb/acre of bran flakes, apple pumice.	Effective for crickets, but not consumed by all grasshoppers; can be used season-long.	Little drift when applied; used mostly for crickets who consume bait almost immediately.
Difluenzuron ULV spray	Insect growth regulator	0.016 conventional 0.012 RAATs	Conventional: 31 fl. oz. per acre (1 part dimilin, 20 parts water, 10 parts vegetable oil)	Effective only against immature grasshoppers and crickets; early-season use only.	Most commonly used spray—slow acting, takes a week or longer to notice effects
Malathion ULV spray	AChE Inhibitor	0.62 conventional 0.31 RAATs	Conventional: 8 fl. oz. per acre RAATs – ½ conventional	Effective against grasshoppers and crickets season-long; favorable for dry and hot conditions.	Historically was insecticide most commonly used, but not used much in recent years; used when a fast-acting result is needed; very little residual.

(*Note: only one of these insecticides would be applied, and only one application would be made in any given year.)

Onsager, 1996a). Both ULV sprays and baits can be distributed by aerial or ground applications. Aerial applications are typical for treatments over large areas. Some grasshopper outbreak locations are economically or logistically accessible only by aircraft, while other locations may be best treated by ground applications. Ground applications are most likely to be made when treating localized grasshopper outbreaks, or for treatments where the most precise placement of insecticide is desired.

(1) Baits

Baits have been used for grasshopper control since the late 1800s (Foster, 1996). The most common form of bait used today is wheat bran. A small amount of additives also may be mixed with bait to extend the product shelf life or assist in applying the product evenly. Other bait formulations include rolled whole grain and pelleted products that are impregnated with carbaryl. The pellet product is commonly used for Mormon cricket control, and is being used more often for grasshopper treatments. The carbaryl bait used for grasshopper suppression is prepared by mixing the appropriate amount of SEVIN® XLR PLUS carbaryl insecticide with a cereal grain substrate, as recommended on the current Section 3 label.

In general, baits have environmental advantages over liquid insecticide applications. Compared to sprays, baits are easier to direct toward the target area, are much more specific toward grasshoppers, act primarily through ingestion, and affect fewer nontarget organisms than sprays (Peach et al., 1994, Foster, 1996).

(2) Ultra-low-volume (ULV) Application

ULV applications are defined as any application of 0.5 gallon, or less, per acre of insecticide in liquid form. Liquid sprays, especially when applied at ULV rates, have several desirable characteristics when considering grasshopper suppression. For example, ULV applications typically produce a quicker, higher, and more predictable grasshopper mortality rate than bait applications (Fuller et al., 1996). Generally, contract costs are substantially lower for applying sprays compared to bait applications (Foster and Onsager, 1996b).

When applying ULV treatments, it is vital to control spray distribution to avoid drift and minimize off-target movement of material (Sanderson and Huddleston, 1996). Drift can become a critical factor in protecting environmentally sensitive areas. Drift is also unsatisfactory from a program standpoint because drift results in less insecticide landing in the treatment area, which reduces program efficacy.

Various carriers and adjuvants may be required under the label for different uses for each product; however, the carrier most often used in this program is either natural or synthetic oils. One adjuvant that may be used with insecticides considered for use by APHIS is canola oil. The maximum rate that oil would be applied for any grasshopper suppression application is 10 ounces of oil per acre. The risk of effects from oil at this rate when considering the proposed mitigation measures is considered to be low.

(3) Reduced Agent Area Treatments (RAATs)

This strategy uses insecticides at low rates combined with a reduction in the area treated for grasshopper suppression. The reduced agent area treatments (RAATs) strategy relies on the effects of an insecticide to suppress grasshoppers within treated swaths, and the conservation of grasshopper predators and parasites in swaths not directly treated (untreated). (Figure 1–2 illustrates the RAATs strategy.)

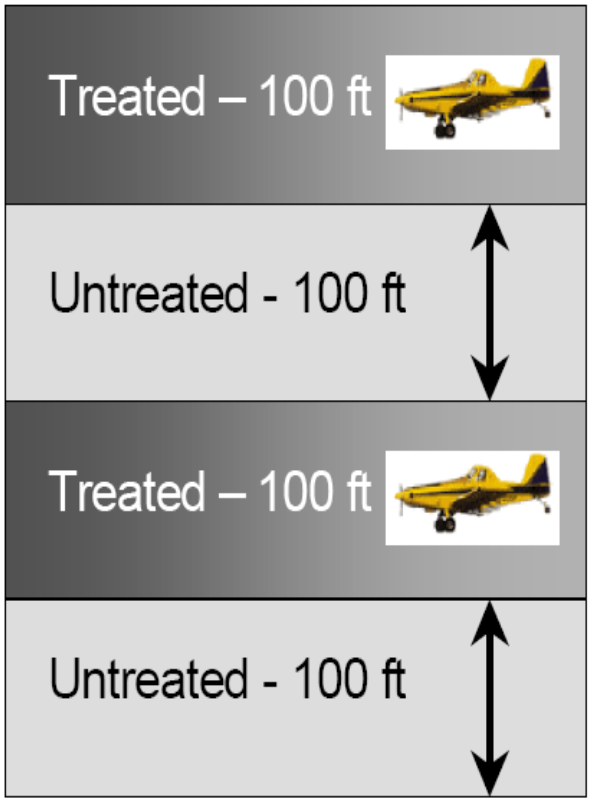

Figure 1–2. Diagram of a reduced agent area treatment (RAATs) showing treated swaths alternating with untreated swaths. In this example, the amount of the area that is treated is reduced by 50%, and the rate of the insecticide would also be reduced from a conventional rate.

For more than 20 years, various studies by APHIS have suggested that reduced rates of insecticides could provide acceptable levels of grasshopper suppression (Foster et al. 1979, 1989; Reuter et al., 1993; Reuter and Foster, 1996), although none of these findings were implemented in the field. The concept of reducing the area of coverage while also applying less insecticide per treated acre was developed in 1995, with the first field tests of RAATs in Wyoming

(Lockwood and Schell, 1997). The potential economic advantages of this method were proposed by Larsen and Foster (1996), and empirically demonstrated by Lockwood and Schell (1997). Widespread efforts to communicate the advantages of RAATs across the Western States were undertaken in 1998, and have continued on an annual basis. The viability of this method at operational scales was initially demonstrated by Lockwood et al. (2000), and subsequently confirmed by Foster et al. (2000). The first government agencies to adopt RAATs in their grasshopper suppression programs were the Platte and Goshen County Weed and Pest Districts in Wyoming; they also funded research at the University of Wyoming to support the initial studies in 1995. This method is now commonly used by government agencies and private landowners in States where grasshopper control is required.

The insecticides for use in the program under this alternative are carbaryl, diflubenzuron, and malathion. All these insecticides are currently registered for use and labeled by EPA for rangeland control of grasshoppers, have been demonstrated to be effective, and would be used by APHIS personnel in strict adherence to label instructions. It has been demonstrated that an acceptable level of grasshopper control can be achieved by reducing application rates to typically one-half the rates used in conventional control programs (Lockwood et al., 2000), and applying the insecticides to only a portion of the land.

An important part of the RAATs strategy is the amount of area that is not directly treated (i.e., untreated). The concept of leaving intermittent swaths untreated is designed to both reduce cost and conserve nontarget biological resources, including predators and parasites of grasshoppers that are present in untreated areas. There is no standardized percentage of area that is left untreated. The proportion of land treated in a RAATs approach is a complex function of the rate of grasshopper movement, which is a function of developmental stage, population density, and weather (Narisu et al, 1999, 2000), as well as the properties of the insecticide (insecticides with longer residuals allow wider spacings between treated swaths). Foster et al. (2000) left 20 to 50% of their study plots untreated, while Lockwood et al. (2000) left 20 to 67% of their treatment areas untreated.

The goal of grasshopper suppression using the RAATs strategy is to economically and environmentally suppress grasshopper populations to a desired level, rather than to reduce those populations to the greatest possible extent. The efficacy of a RAATs strategy in reducing grasshoppers is, therefore, less than conventional treatments and more variable. Foster et al. (2000) reported that grasshopper mortality using RAATs was reduced 2 to 15% from conventional treatments, depending on the insecticide, while Lockwood et al. (2000) reported 0 to 26% difference in mortality between conventional and RAATs-treated areas.

c. Insecticide Application Rates

All APHIS grasshopper treatments using carbaryl, diflubenzuron, and malathion are conducted in adherence with EPA-approved label directions. Maximum application rates for each product, for uses other than grasshopper control, can range 25% higher for malathion, to greater than 95% higher for diflubenzuron when compared to maximum labeled use rates for grasshopper control. Maximum rates for grasshopper control that may be utilized by private landowners, as an example, are also much higher than those proposed for use by APHIS (table 1–2).

Table 1–2. Labeled Rates (fl. oz./acre) for Grasshopper and Mormon Cricket Control

	Maximum Labeled Grasshopper Rate	APHIS Proposed Full Rate	APHIS Proposed RAATs Rate
Carbaryl	1.0	0.5	0.375
Diflubenzuron	2.0	1.0	0.75
Malathion	12.0	8.0	4.0

d. Location of Potential APHIS Grasshopper Treatment Areas

APHIS may conduct a treatment to suppress an economically damaging grasshopper population on rangeland in the 17 Western States (Arizona, California, Colorado, Idaho, Kansas, Montana, Nebraska, Nevada, New Mexico, North Dakota, Oklahoma, Oregon, South Dakota, Texas, Utah, Washington, and Wyoming) (figure 1–3). However, rangeland does not occur over the entire area of most of those States.

Historically, APHIS has conducted treatments in each of the 17 Western States. Should grasshopper populations reach economically damaging levels, the land manager will request a treatment and, pending available funds, it is possible that APHIS could conduct a treatment on rangeland in any of the 17 Western States. Historically, applications in salmonid habitats have been infrequent. A summary of the available county-level program insecticide application data for California, Idaho, Washington and Oregon between 2003 and 2009 show few applications occurred in designated salmonid critical habitat (figure 1-3). RAAT applications of carbaryl bait at a rate of 10 lb/acre were the most common treatment method followed by ground or aerial applications of diflubenzuron using RAATs. APHIS has made no malathion applications in any of the four States in the past 4 years. The amount of acreage treated varied, with Idaho treating sites ranging in size from approximately 50 to 12,800 acres. Oregon and Washington had fewer numbers of treatments and acreage, with the size of the treatment acreage ranging from 13 to 2,428 acres. A majority of the applications were on Federal lands.

APHIS cannot accurately predict what areas will be treated in the future; however, a model is in development that will be a predictive tool for determining where future outbreaks and potential treatment areas could occur. This model will facilitate early cooperation between APHIS and NMFS in the identification of potential treatment areas, and whether they occur at the same location as listed salmonids and their critical habitat.

e. Operational Procedures for APHIS Grasshopper Treatments

There are a number of operational procedures the program must follow when applying insecticide treatments. These procedures have been put in place to assure that a treatment is efficacious, economical, and conducted to ensure the safety of workers and the environment. All APHIS grasshopper treatments must follow all applicable Federal, State, tribal, and local laws and regulations regarding pesticide use. In addition, APHIS personnel and contractors, must strictly follow all EPA- and State-approved label instructions for pesticide use.

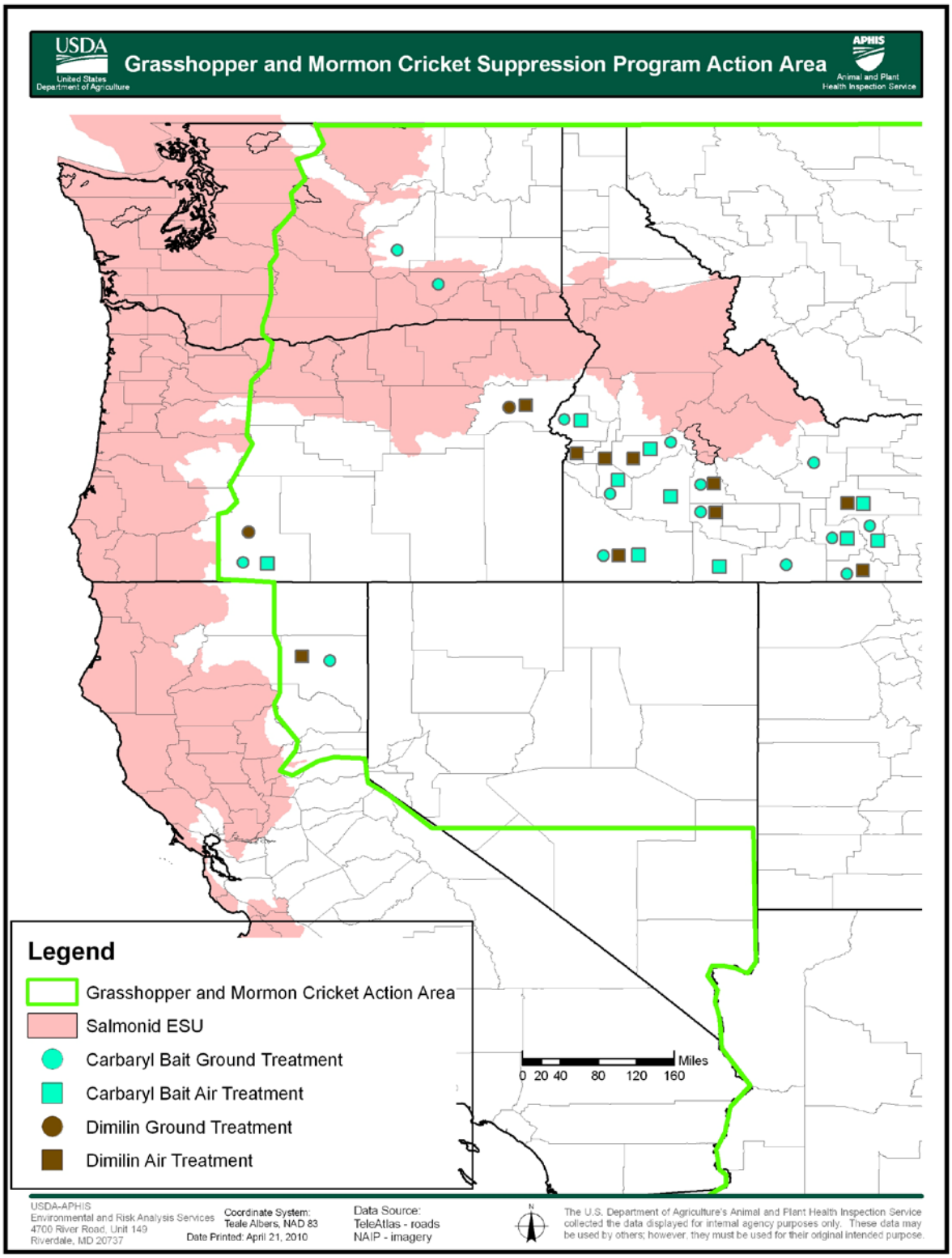

Figure 1-3. County-level Program insecticide applications from 2003-2009 in States where salmonid critical habitat is present

APHIS has also implemented several measures that go beyond label instructions in order to protect workers and the environment. For example, all planes must have a positive on/off system that will prevent leaks from the nozzles. A positive emergency shutoff valve between the tank and the pump is also required, as well as avoidance of aerial ferrying and turnaround routes over water bodies and sensitive habitats (USDA–APHIS, 2006a). These requirements reduce the risk of accidental release of insecticides into aquatic habitats and other sensitive habitats.

Operational procedures are also in place to assure, as much as possible, that insecticide application will be limited to the treatment area. In the use of reduced rates, the accurate deposition of the insecticide is essential if grasshopper populations are to be suppressed efficaciously. Weather plays an important role in aerial application. Winds may displace the insecticide, and high air temperatures combined with low humidity may cause fine droplets to evaporate and drift without reaching the target. During applications APHIS personnel constantly monitor wind conditions, as well as ground and air temperatures. Should wind speed in the treatment area exceed 10 miles per hour, or a change in wind direction is noted towards sensitive habitat, or should a temperature inversion be detected, spray programs end until conditions are again favorable. Temperature inversions are characterized by stable air and increasing temperatures with height above the ground.

The program has also established treatment restriction buffers around waterbodies to protect those features from insecticide drift and runoff. The labels for all the insecticides APHIS uses preclude a direct application to water (defined as reservoirs, lakes, ponds, pools left by seasonal streams, springs, wetlands, and perennial streams and rivers). APHIS maintains the following additional buffers for water bodies that are not designated critical habitat for salmonids: 500-foot buffer for aerial sprays, 200-foot buffer for aerially applied bait, and a 50-foot buffer for all ground applications.

f. Program Application Buffers to Protect Listed Species and Critical Habitat

Application buffers as well as additional mitigation measures to protect listed salmonids and their critical habitat have also been established for those areas where program activities and listed salmonids and their designated critical habitat are present (table 1-3). These buffers are variable, based on the method of application and the potential for direct and indirect risks of each insecticide.

In addition to the chemical-specific application buffers, additional measures have been incorporated into the program to reduce the potential exposure of salmonids to program insecticide treatment:

- Avoid applications when wind speeds exceed 10 mph
- Avoid applications when wind direction is blowing towards salmonid critical habitat
- Use reduced area agent treatments (RAATs) adjacent to salmonid critical habitat
- Avoid applications under conditions where a temperature inversion is possible or when a storm event is imminent.

Table 1–3. Insecticide Application Buffers from Salmonid Critical Habitat

Insecticide Treatment	Method of Application	Application Buffer (ft)
Carbaryl	Aerial Ultra Low Volume (ULV)	3,500
	Aerial Bait	1000
	Ground ULV	350
	Ground Bait	200
Diflubenzuron	Aerial ULV	1,500
	Ground ULV	150
Malathion	Aerial ULV	3,500
	Ground ULV	500

Temperature inversions are characterized by stable air and increasing temperatures with height above the ground. The avoidance of applications during storm events is required to reduce the probability of off-site transport of program insecticides via runoff. Variability in weather patterns, in particular rainfall, even within small geographic areas requires a site specific evaluation by program personnel of conditions prior to application to determine if a rainfall event could occur and whether a storm event would result in conditions where runoff could occur to aquatic habitats given site conditions and the proposed application buffers.

C. Grasshopper Biology

1. General Description

Grasshoppers and Mormon crickets are closely related insects, both belonging to the order Orthoptera. Mormon crickets are a flightless species of long-horned grasshopper. Grasshoppers occur throughout the North American continent and around the world; however, Mormon crickets are mostly found in the Great Basin and other areas of the western United States. Nearly 400 species of grasshoppers are known to inhabit the 17 Western States. Of these species, approximately 40 or more species commonly occur at population levels that cause damage to rangeland, grasses, and surrounding crops. Most of the economically damaging species are relatively small or intermediate in size. Although as many as 15 to 45 grasshopper species may be found in a particular area, only a few species are responsible for economic damage. It is important to remember that a single species usually does not cause significant damage; however, species in combination may cause extensive damage. Because the program is limited to treatments on rangeland, only those species that are economically damaging to rangeland are considered in the program.

Grasshopper species vary in density and species composition. These factors depend on the weather, soil, vegetation, topography, and use of the rangeland habitat. They are generally grouped into grass feeders, forb feeders, or mixed feeders. While some grasshoppers eat most vegetation, others may discriminate by choosing young green leaves over yellowing, older leaves (Pfadt, 1994). Grasshopper habitats may change because of the differential effects of weather,

parasites, disease, or insecticidal treatments. Increases in the abundance of food and suitable habitat or decreases in natural enemies are also likely to trigger population explosions (Capinera and Horton, 1989).

2. Grasshopper Outbreaks

Grasshoppers have the potential for sudden and explosive population increases that may make them a threat for the following season. Although survey techniques are used by the program for guidance in understanding outbreaks, presently there are no simple ecological explanations to accurately predict grasshopper outbreaks. Reference to a certain density of grasshoppers, such as $13/yd^2$, does not adequately define all outbreaks. Instead, the significance of grasshopper density is only understood within the specific ecological context. For example, when the forage base is sparse and has low productivity, grasshopper densities as small as $8/yd^2$ may cause considerable damage. In other areas with high plant abundance and/or high productivity, grasshopper densities may need to exceed $13/yd^2$ before causing extensive damage (Belovsky et al., 1996). Typically, grasshopper populations are most severe in a hot and dry growing season that follows an unseasonably mild winter (i.e., high survival of grasshoppers).

Grasshopper infestation levels have been associated with weather-related factors and geographical regions. Generally, States located in the northern shortgrass prairie region (such as Wyoming and Montana), experience a positive relationship between infestation levels and average summer temperatures. Data indicate that dry, warm summer conditions are more conducive for outbreaks in northern areas (Capinera and Horton, 1989). Although generalization may not be appropriate for all areas, this positive relationship holds for the U.S. northern shortgrass prairie regions where densities are highest during or after warm, dry weather or drought (Capinera and Horton, 1989). In southern areas or desert areas, food may limit grasshopper populations, thus, outbreaks may occur after periods with greater than normal rainfall (Fielding and Brusven, 1990).

As mentioned before, grasshopper outbreaks are responsible for economic damage from direct and indirect impacts to rangeland. Therefore, an outbreak requiring control is determined by an economic threshold. The economic threshold is site-specific where the damage caused by the pest must be at least as great as the cost of treatment (Davis and Skold, 1996). Some economic factors that are considered in determining outbreaks include ranch type, rangeland productivity, cost of alternative sources of forage for livestock, and nontreatment options available to the rancher (Davis and Skold, 1996).

3. Life History and Ecology of Grasshoppers and Mormon Crickets

Grasshoppers and Mormon crickets belong to the order Orthoptera, and go through three development stages: egg, nymph, and adult. Each species may possess a unique set of ecological and physiological adaptations that allow them to grow, survive, and reproduce in their environment. Habitat plays an important role in providing nutritive food plants, adequate living space, satisfactory soil conditions for the eggs, and favorable biotic relationships for all life stages. Generally, there is only one generation per year. However, in northern regions, eggs may occasionally require as many as 2 years to develop, depending upon species and climatic

conditions. In warmer areas, south of Kansas, *Melanoplus sanguinipes* may produce a smaller second generation each year.

The egg-laying habits of grasshopper species also may differ. After mating with a male, the female digs a small hole in the soil with her ovipositor and deposits the first group of eggs. Once egg laying begins, the female continues to mate and deposit eggs throughout her life. The number of eggs laid may range from 3 pods per week to 1 pod every 1 to 2 weeks, and each pod may contain as many as 15 to 100 eggs. Grasshopper egg pods vary not only in the number of eggs but also in egg size, shape, structure, and location deposited. Incubation of eggs may begin immediately after being deposited in the soil, depending upon climatic temperatures. Newly-hatched grasshoppers are active, and begin feeding on green and nutritious host plants. A young grasshopper must shed (molt) its exoskeleton to grow and mature to an adult stage. As the grasshoppers grow and develop, they molt at intervals, changing their structures and form. Depending on species and sex, grasshoppers typically molt four to six times during their nymphal or immature life stage, and require 30 to 40 days to complete development, depending on weather conditions. Mormon crickets vary from grasshoppers in that they pass through seven nymphal instars, and immature development may take 60 to 90 days. The insect stage between molts is referred to as an instar. When the last instar molts, the exoskeleton hardens, the insect becomes an adult, and is ready to mate and reproduce (Pfadt, 1994).

4. Damage

Damage caused by grasshoppers goes beyond actual consumption of forage. Although each species alone may not cause significant damage, a combination of species in an area may cause extensive damage to rangeland. Vegetation damage during serious grasshopper outbreaks may be so severe that all grasses and forbs are destroyed; thus, plant growth is impaired for several years. Some consequences of grasshopper outbreaks include reduced grazing for livestock; loss of food and habitat for plants and wildlife, including endangered and threatened species; and soil erosion, sometimes resulting in decreased water quality. Damage to native plant communities from grasshopper outbreaks may reduce biological diversity, as well as create opportunities for the expansion of aggressive and exotic weeds (Lockwood and Latchininsky, 2000).

The economic damage resulting from high grasshopper density and the resulting defoliation may reach an economic threshold. Economic threshold is defined as the point where the damage caused by grasshoppers exceeds the cost of controlling the grasshoppers. At this point, rangeland managers save money by treating the grasshoppers to prevent further damage. This threshold is determined by density surveys conducted by the program and the value of the rangeland's plant resource. The economic threshold is an important tool in grasshopper management as a way of determining economic costs and benefits. Hewitt and Onsager (1983) found that the average dollar value of annual forage lost to grasshoppers on 262 million hectares reached nearly $400 million a year, comprising 23% of the total forage value.

Rangeland plants protect soil from erosion and maintain watersheds for rivers and streams. Grasshopper consumption produces different levels of impact on rangeland plants, depending on which part of the grass the grasshopper feeds on. Those feeding on the growing parts of the grass are highly destructive; whereas, other species of grasshoppers are less destructive. For example, some grasshopper species cut off seed stalks, thus eliminating seed production,

reducing coverage of plants across the ground, and making soil erosion more likely to occur in denuded areas. Grasshopper-induced changes may lead to soil degradation, interruption of nutrient cycles, and loss of important plant species or seed production leading to reduced diversity of rangeland habitats. When grasshoppers consume plant cover, soil is more susceptible to the drying effects of the sun, making plant roots less capable of holding soil in place. Soil damage results in erosion and disruption of nutrient cycling, water infiltration, seed germination, and other ecological processes which are important components of rangeland ecosystems.

II. Exposure Analysis for Grasshopper Insecticides—Environmental Fate and Transport Modeling

This section provides information about the environmental fate and transport modeling of insecticides (carbaryl, diflubenzuron, and malathion) applied at program rates. This information is integrated into sections four and five of this BA in the discussion regarding the potential exposure to program insecticides and the risk to listed salmonids and their critical habitat.

A. Environmental Fate of Program Insecticides

1. Carbaryl

a. Soil

Overall, carbaryl is not persistent in soil due to multiple degradation pathways including hydrolysis, photolysis, and microbial metabolism. Microbes play a significant role in the degradation of carbaryl in soil (Xu, 2003). Chapalamadugu and Chaudhry (1991) revealed that two soil bacteria, *Pseudomonas* spp., can metabolize carbaryl or its primary metabolite, 1-naphthol to CO_2 within 36 hours.

In aerobic soil, carbaryl quickly degraded with an approximate half-life of 4 days (Miller, 1993a). A significant amount of CO_2 was produced, ranging from 0.1% at day 1 to 59.7% at day 14. Carbaryl degrades more slowly in anaerobic aquatic soil, with an estimated half-life of 72 days (Miller, 1993b). 1-naphthol is the major degradate with minor compounds of 1,4-naphthoquinone, 5-hydroxy-1-naphthyl methylcarbamate and 1-naphthyl-(hydroxymethyl) carbamate. The degradate 1-naphthol may represent up to 67% of applied carbaryl. None of the minor degradates account for more than 2.5% of the total applied dose. Degradation of 1-naphthol in soil is rapid, with levels below detection after 14 days (EPA, 2003).

The adsorption coefficient values (K_{oc}) of carbaryl range from 100 to 1054 (Jana and Das, 1997; EPA, 2003a, US FS, 2008a), indicating carbaryl moderately binds to soil. Carbaryl sorption to soil has been shown to increase with increasing percent organic carbon (Shareef and Shaw, 2008). Sorption experiments using two types of soils, Red Bay (AB) and Astatula (AS), were further separated into two layers—topsoil (0-30 cm) and subsoil (31-60 cm) (Nkedi-Kizza and

Brown, 1998). The properties of individual soil are AB top (pH 6.3), AB sub (pH 5.3), AS top (pH 5.6) and AS sub (pH 4.8). The sorption coefficient values (K_{oc}) of carbaryl in soils were 338, 144, 590, and 671 mg/kg on AB topsoil, AB subsoil, AS topsoil, and AS subsoil, respectively. The half-lives of carbaryl on the four soils ranged from 8 to 18 days. Given the same soil, carbaryl degraded much faster in topsoil than in subsoil.

Terrestrial field dissipation studies were conducted at two locations, one in California and one in North Carolina (Norris, 1991). Data showed that most residues remain in the first 0-0.15 meters (m) of soil, with only one finding in the layer of 0.3-0.45 m. The dissipation half-lives of carbaryl were estimated to be from 0.76 to 10.9 days. In a forestry dissipation study, half-lives ranged from 21 days on foliage to 75 days in leaf litter (US FS, 2008a).

Carbaryl bait, due to its application method, will exhibit reduced soil effects relative to spray applications (USDA, APHIS, 1987). Little transport of carbaryl through runoff or leaching to groundwater is expected due to the low water solubility, moderate sorption, and rapid degradation in soils. There are no reports of carbaryl detection in groundwater, and less than 1% of carbaryl applied to a sloping plot was detected in runoff (Caro et al., 1974).

b. Water

Hydrolysis is the primary degradation pathway for carbaryl at pH 7 and above. The compound degrades rapidly at pH 7 and 9 at 25 ^{o}C, with half-lives of approximately 10 to 17 hours and 3 hours, respectively (Aly and El-Dib, 1971; EPA, 2003a). Studies to support the registration of carbaryl in the United States show a similar effect of pH on hydrolysis rates with a half-life of 12 days at a pH of 7 and 3.2 hours at a pH of 9 (EPA, 2003b). Carbaryl is assumed to be hydrolytically stable at a pH of 5 (EPA, 2003b). The identified degradation products are 1-naphthol, methylamine and CO_2 (Aly and El-Dib, 1971; Larkin and Day, 1986). In natural water, carbaryl is expected to degrade faster due to the presence of microorganisms. The half-lives of carbaryl in streams, rivers, and brooks, as a result of forest spraying, are 25, 28, and 23 hours, respectively (Stanley and Trial, 1980). Bonderenko et al. (2004) reported aqueous half-lives of carbaryl in natural waters from California and Washington state ranging from 0.3 to 4.7 days. Degradation in the study was temperature dependent with shorter half-lives at higher temperatures. Armbrust and Cosby (1991) reported hydrolysis half-lives of carbaryl in filtered and sterilized seawater at pH 7.9 and 8.2 at 24 ^{o}C were 24 and 23 hours, respectively, and the major degradation product was 1-naphthol. Naphthol was not degraded in dark sterile seawater, but was undetected within 96 hours in raw seawater. When exposed to artificial sunlight, carbaryl had a half-life of 5 hours and naphthol was completely degraded in 2 hours. Carbaryl has a reported solubility range of 23 to 120 mg/L (EPA, 2003b, US FS, 2008a).

The aqueous photolysis of carbaryl was determined to be 21 days in sterile distilled water under artificial sunlight at a concentration of 10.1 ppm and pH 5 (Das, 1990). The intensity of artificial light was comparable to that of natural sunlight, at 510.5 and 548.8 watts/m^2, respectively. Other reported aqueous photolysis half-lives are much shorter than that obtained from sterile water. Wolfe et al. (1978) reported a photolysis half-life for carbaryl as 6.6 days, and Zepp et al. (1976) as 50 hours near the water surface. The aqueous photolysis rates increase as intensity of sunlight increases; therefore, the rate of hydrolysis is much faster in summer than in winter. Wolfe et al. (1976) calculated aqueous photolysis half-lives of carbaryl in surface water (in <10 cm water) at

latitude 40 degrees North in different seasons—64 hours in spring, 52 hours in summer, 102 hours in fall, and 200 hours in winter. The major photolysis product is 1-naphthol, which will further photooxidize rapidly to 2-hydroxyl-1,4-naphtho-quinone in basic conditions (Wauchope and Haque, 1973).

Suspended particulates in natural water may remove some carbaryl from the aqueous phase. Karinen et al. (1967) reported that 50% of initial carbaryl dissipated from estuarine water after 38 days at 8 ^{o}C in the absence of mud; in the presence of mud, 90% of initial applied carbaryl was withdrawn from the water after 10 days at the same temperature due to significant removal of carbaryl by mud.

Microbial degradation under oxic conditions in combination with other degradation pathways results in a relatively short half-life for carbaryl in water. Aerobic aquatic metabolism is much quicker with a reported half-life range of 4.9 to 8.3 days compared to anaerobic aquatic metabolism range of 15.3 to 72 days (Thomson and Strachan, 1981; EPA, 2003a).

c. Air

Carbaryl has a half-life in air of 1 to 4 months. The low vapor pressure and Henry's law constant of carbaryl makes it unlikely that there will be significant volatilization from soil, water, or treated surfaces (Dobroski et al., 1985). Carbaryl may be found in the atmosphere associated with air-borne particulates or as spray drift and can react with hydroxyl radicals in the ambient atmosphere (Kao, 1994).

d. Vegetation

Carbaryl has a short residual half life on plant surfaces. Insecticidal properties are retained for 3 to 10 days (EPA, OPTS, 1985). The major metabolite is 1-naphthol. Although carbaryl is a polar compound, bioconcentration in plants is not of concern due to limited plant uptake related to low water solubility and rapid degradation (Nash, 1974).

Based on forestry field dissipation studies, foliar half-lives of 21 days have been reported with a leaf litter half-life of 75 days (EPA, 2003a).

e. Fish

Carbaryl is not subject to significant bioaccumulation due to its low water solubility and low octanol-water partition coefficient (Dobroski et al., 1985; EPA, 2003a). Uptake of carbaryl in fish has been detected with 95% excreted within 8 hours (Tompkins, 1966). Bioconcentration factors (BCF) in fish and invertebrates are low with values less than 15 (US FS, 2008a).

2. Diflubenzuron

a. Soil

Mobility and leachability of diflubenzuron in soils is low, and residues are usually not detectable after 7 days (Eisler, 2000). Diflubenzuron has been shown to bind readily with organic matter in

soils, and is relatively immobile in the environment (EPA, 1997). Adsorption values vary depending on soil type (40, 40, 20, 25, 130, 110, 150 and 3500 for a sand clay, silty clay loam, silt loam, sand loam, sand clay loam, clay, a clay hydrosol, and a peat hydrosoil, respectively) and indicate preferential adsorption to soil over remaining in solution due to low solubility (Sundaram, et al., 1997; EPA, 1997). Soil adsorption coefficients ranging from 8700 to 10000 have also been reported in the literature (US FS, 2004).

The persistence of diflubenzuron in soils is microbe dependent. The half-life of diflubenzuron under field conditions ranges from 7 days to about 19 days (Nigg et al., 1986). In standardized laboratory studies, the aerobic soil metabolism ranged from 2 to 14 days. The major metabolite was 4-chlorophenyl urea which composed approximately one-third of the radioactivity 7 to 14 days after treatment. The other major metabolite was CO_2, along with three other metabolites (2,6-difluorobenoic acid, 2,6-difluorobenzamide, 4-chloroaniline) that consisted of less than 10% of the total radioactivity. The same half-life range was observed in the anaerobic soil metabolism study with the same approximate distribution of metabolites (EPA, 1997). Field dissipation studies, in general, support the laboratory half-life of diflubenzuron with orchard and bare ground dissipation half-lives of 5.8 to 13.2 days. However, field dissipation studies in California citrus and Oregon apple orchards reported half-live values of 68.2 to 78 days.

b. Water

Diflubenzuron is stable to hydrolysis at pH values of 5 and 7, with a reported hydrolysis half-life at pH 9 of 32 days (Ivie et al., 1980; EPA, 1997). Degradation half-lives in the presence of oxygen are slightly shorter ($T_{½}$ = 0.42 days) compared to degradation in the absence of oxygen ($T_{½}$ = 0.97 days) (Anton et al., 1993). Due to its low solubility (0.2 mg/L) and preferential binding to organic matter, diflubenzuron seldom persists more than a few days in water (Schaefer and Dupras, 1977; Schaefer et al., 1980). Persistence in water is typically short with a dissipation half-life of 3.3 to 8.2 days, based on field studies in littoral enclosures (Knuth and Heinis, 1995, Boyle et al., 1996). Half-life values in sediment were similar to those in water, with reported half-life values ranging from 6.2 to 10.4 days. Sundaram et al. (1991) reported maximum DT_{50} and DT_{90} values of 1.3 and 4.2 days, respectively in pond water and 0.2 and 1.0 days in streams. Under anaerobic conditions, the metabolic half-life for diflubenzuron is reported as 34 days (EPA, 1997).

c. Air

The vapor pressure of diflubenzuron is relatively low (0.00012 mPa) and volatilization from water is not expected, based on the reported low Henry's Law Constant value (1.8×10^{-9} atm*m^3/mol) (Wauchope et al., 1992, EPA, 1997). Based on the low use rate and fate characteristics for diflubenzuron, exposure from volatilization is expected to be minimal.

d. Vegetation

Diflubenzuron is not systemic in plants and does not translocate to either pollen or nectar (Crompton Crop Protection 2005). Diflubenzuron applied to foliage remains adsorbed to leaf surfaces for several weeks with little or no absorption or translocation from plant surfaces (Eisler 1992, 2000).

e. Fish

Diflubenzuron is not expected to bioconcentrate in fish, based on results from a bioconcentration study using the bluegill. During a 28-day exposure, levels reached steady state in the tissue and viscera, and greater than 99% of the test material was excreted during the 14-day depuration period (EPA, 1997).

3. Malathion

a. Soil

The persistence of malathion in soils depends primarily on microorganism activity, pH, and organic matter content. Persistence is decreased with microbial activity, moisture, and high pH. The half-life of malathion in natural soil varies from 2 hours (Miles and Takashima, 1991) to 11 days (Neary, 1985; EPA, 2006). The primary route of degradation of malathion in surface soils appears to be microbially mediated soil metabolism (half-life <1-2.5 days) and hydrolysis (pH 7 half-life 6.21 days and pH 9 half-life 12 hours) (EPA, OPPTS, 2000b). Known degradates include malathion monoester, ethyl hydrogen fumarate, diethyl succinate, malathion mono- and dicarboxylic acids, demethyl mono- and dicarboxylic acids, and CO_2 (EPA, OPPTS, 2000b). The principal degradation products are monocarboxylic and dicarboxylic acids (Walker and Stojanovic, 1973).

Malathion and associated degradates, in general, are soluble, and do not adsorb strongly to soils (EPA, OPPTS, 2000b). Malathion K_{oc} values range from 151 to 183 (US FS, 2008b).

Inorganic degradation of malathion may be more important in soils that are relatively dry, alkaline, and low in organic content, such as those that predominate in the western program areas. Malathion is subject to hydrolysis under neutral and alkaline conditions, but is more stable under acidic conditions. It does not penetrate much beyond the soil surface and does not adsorb tightly to inorganic soil particles, although it binds tightly with organic matter (Jenkins et al., 1978). Demethyl and carboxylic acid degradates are expected to be highly mobile, especially in alkaline soil (EPA, OPPTS, 2000b). Adsorption to organic matter and rapid degradation make it unlikely that detectable quantities of malathion would leach to groundwater (LaFleur, 1979).

Malathion degradation products also have short half-lives. Malaoxon, the major malathion degradation product of toxicological concern, has half-lives of 3 and 7 days in soils of pH 7.2 and 8.2, respectively (Paschal and Neville, 1976). This longer half-life relative to that of malathion is proposed to be a result of malaoxon's biocidal effect on soil microbes which contribute to malathion's degradation.

b. Water

Degradation of malathion in water is mostly by photolysis and microbial degradation under acidic conditions, and chemical transformation under alkaline conditions (Wolfe et al., 1976). The half-life of malathion ranges from 0.67 (natural river water) to 42 days (distilled water) (Howard, 1991). Guerrant et al. (1970) found the malathion half-life in pond, lake, river, and

other natural waters varied from 0.5 days to 10 days, depending on pH. Malathion is likely to have longer persistence in acidic aquatic environments. The half-life of malathion was calculated from program monitoring data for natural waters during the 1997 Medfly Cooperative Eradication Program in Florida to be 8 hours in a retention pond, and 32 hours in the Hillsborough River (USDA, APHIS, 1997). Half-life in seawater at pH 8 was 2.6 days (Horvath, 1982). Aerobic and anaerobic aquatic metabolism studies submitted for registration show half-lives to be short in water and sediment under alkaline conditions. Reported water and sediment half-lives, in an aerobic aquatic metabolism study, were reported as 1.09 and 2.05 days respectively, at a pH of 7.08. The reported half-life in water and sediment, for the anaerobic aquatic metabolism study, was 2.5 days at a range of pH values from 7.8 to 8.7 (EPA, 2006).

c. Air

Volatility is not expected to be a major pathway of exposure based on the low vapor pressure and Henry's Law constant that have been reported for malathion (HSDB, 2009). The atmospheric vapor phase half-life of malathion is 5 hours (HSDB, 2009).

d. Vegetation

The half-life of malathion on foliage has been shown to range from 1 to 6 days (Matsumara, 1985; Nigg et al., 1981; El-Refai and Hopkins, 1972; US FS, 2008b).

e. Fish

Bioconcentration factors are low for fish, ranging from 7.36 in lake trout to 34.4 in willow shiners (HSDB, 2009; Tsuda et al., 1989). The concentration in fish tissue decreases readily and consistently with a decrease of malathion in water.

B. Aquatic Residue Estimations for Aerial and Ground Applications of Program Insecticides

Pesticide deposition to salmonid critical habitat from program applications can occur through various transport processes including volatility, drift and runoff. Based on the reported low vapor pressure values measured for each program insecticide, low application rates, and in some cases short atmospheric half-lives, volatility was not considered to be a significant transport pathway. Drift and runoff were also considered and are discussed in more detail below. Drift was considered the primary mechanism of potential off-site transport to salmonid habitat based on the proposed application buffers and other mitigation measures.

1. Aerial and Ground Drift Assumptions

The method of calculating aquatic exposure concentrations and effective buffer zones for the program is through the use of two aerial drift deposition models. The models (AgDrift and AgDisp) allow for specific application information to be used as input into the model, and then determine the amount of drift that would occur at a user-defined distance from the spray block.

The difference between deposition at the edge of a field and a selected buffer zone can be used as a means to reduce the total amount of insecticide that would be expected at a certain distance from the spray block. Buffer zones, in addition to the previously mentioned mitigation measures, for listed salmonids can then be set, based on the reduction in exposure to levels that would not be expected to result in direct or indirect effects to individuals, populations or species as a whole.

AgDrift and AgDisp are pesticide drift deposition models that provide the user with the ability to provide site- and application-specific information as input to determine application efficiency and off-site drift residues. AgDisp is a model which was developed by the USDA Forest Service beginning in the early 1980's, and served as the platform for the development of the AgDrift model which has become a regulatory tool for the EPA-OPP in the registration of pesticides (Hewitt et al., 2002; Teske and Curbishley, 2003). Both models have a tiered approach that allows the user to choose default values or provide more specific data, based on the available information. Both models have been validated under various application scenarios in the literature (Duan et al., 1992a; Duan et al., 1992b; Teske and Thistle, 2004; Teske et al., 2000). In general, aerial application predictions slightly underestimate drift within the first 80 m, but overpredict at increasing distances by a factor of two to four at distances up to approximately 300 m (Teske and Thistle, 2003; Duan et al., 1992a.b; Bird et al., 2002; Thistle et al, 2008).

For this risk assessment the AgDrift model was used to simulate all ground applications, while AgDisp was used to simulate all aerial ULV and bait applications. The AgDisp model was used in the aerial applications to assess buffer distances and application heights that are beyond those that have been validated using AgDrift (Teske and Thistle, 2004). Input data for the AgDrift and AgDisp models were based on pesticide labels for each product and specific application information available in the APHIS workplan for the program (APHIS, 2006a). While several types of aircraft are available for application in the program, the quantitative differences in drift are minimal at the buffer zones being assessed. Therefore, the focus of the modeling work was to emphasize those parameters that have the greatest influence on drift. Multiple factors can influence pesticide drift; however, release height, wind speed and direction, and nozzle atomization are the primary factors influencing drift (Bird et al., 1996; Teske et al., 2000).

Unless otherwise specified, release height was set at 75 feet (ft) with a maximum allowed sustained windspeed of 10 miles per hour, and the American Society of Agricultural and Biological Engineers (ASABE) droplet size distribution of fine to very fine (table 2–1). ASABE has developed standardized parameters for different droplet size spectra that can be selected in both drift models. The very fine to fine droplet size spectrum selected for all of the air and ground ULV simulations is consistent with an application recommended for use in the program.

The intent of the program is to make applications as close as possible to the spray block. However, in some cases where rapid elevation changes are likely to occur, applications must be made at a height that will ensure pilot safety and the appropriate swath width. All applications were simulated on an area where the buffer was on a zero grade and there were no upslope or downslope between the spray block and sensitive habitat. In addition, the maximum height of vegetation between the spray block and habitat was no greater than 0.1 meters high. This provides a conservative estimate regarding the ability of plants and terrain to intercept drift between the spray block and sensitive areas.

Table 2–1. AgDisp Aerial Modelling Input Parameters

Parameters	Carbaryl	Diflubenzuron	Malathion
Wind Speed (mph)	10	10	10
Wind Direction	-90	-90	-90
Relative Humidty (%)	36	36	36
Temperature (°F)	90	90	90
Release height (ft)	75	75	75
Spray Volume Rate (fl oz/acre)	16	31	4
Active Rate (ULV/Bait) (lb ai/ac)	0.375/0.20	0.012	0.31
Aircraft Type	AT 502	AT 502	AT 502
Droplet Size Dist.:			
ULV	ASABE Very Fine to Fine	ASABE Very Fine to Fine	ASABE Very Fine to Fine
Bait	ASABE Coarse to Extremely Coarse		
Swath Width ULV (ft)	150	150	150
Bait	100		
Swath Displacement (ft)	150/100	150	150
Canopy type	Grass 0.02 to 0.1 m high	Grass 0.02 to 0.1 m high	Grass 0.02 to 0.1 m high

A sustained 10-mile-per-hour windspeed was used as a representative maximum that is allowed in program applications in all simulations. The wind direction was assumed to be at -90^{o} directly towards the sensitive habitat for the entire length of all swaths with no reduced area of application occurring over the spray block (figure 2–1).

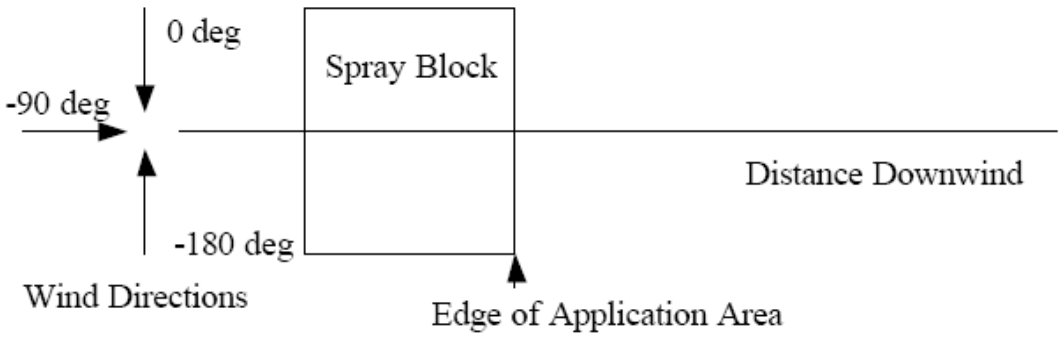

Figure 2–1. Wind direction relative to the spray block and the distance downwind (Teske and Curbishley, 2003).

Other parameters that influence drift are meteorological conditions. In addition to wind speed, both drift models allow the user to input temperature and humidity. Temperature and humidity values for this exercise were selected from all geographically representative areas where the program could potentially make applications. Meteorological data was obtained from the AgDisp model which allows the user to view a 30-year compendium of meterological data from 239 sites in the United States (1961–1990 National Solar Radiation Data Base, Version 1.0, Solar and Meterological Surface Observational Network (SAMSON)) (Teske and Curbishley, 2003).

The 25th percentile humidity value and the 75th percentile highest temperature were selected based on weather data from Lubbock, Texas, which reported a temperature value of (90 ^{o}F) with a humidity value of 36%. Bismarck, North Dakota, and Pocatello, Idaho, were also evaluated, and based on combination of maximum temperature and minimum humidity values for those areas, all three had similar application efficiencies and drift fractions based on their respective worst-case temperature and humidity values. Therefore, the temperature and humidity value from Lubbock, Texas, was used since it would maximize the potential for pesticide drift.

Aerial drift output from AgDisp at varying distances was used to determine the impact of buffer zone distance on drift reduction (figure 2-2). Typical of drift curves is a rapid decrease in deposition as you move away from the spray block. Differences in input variables can have a significant impact on the reduction in drift. However, as you move further away from the spray block, the influence of these parameters becomes less, with atmospheric stability becoming more of a factor in aerial transport. For all three products the rate of drift transport decreased more so in the first 2000 feet of buffer compared to the reduction between 2000 and 4000 feet. The rate of deposition decreases less beyond 4000 feet and will not reach a value of zero based on limitations and assumptions in the model.

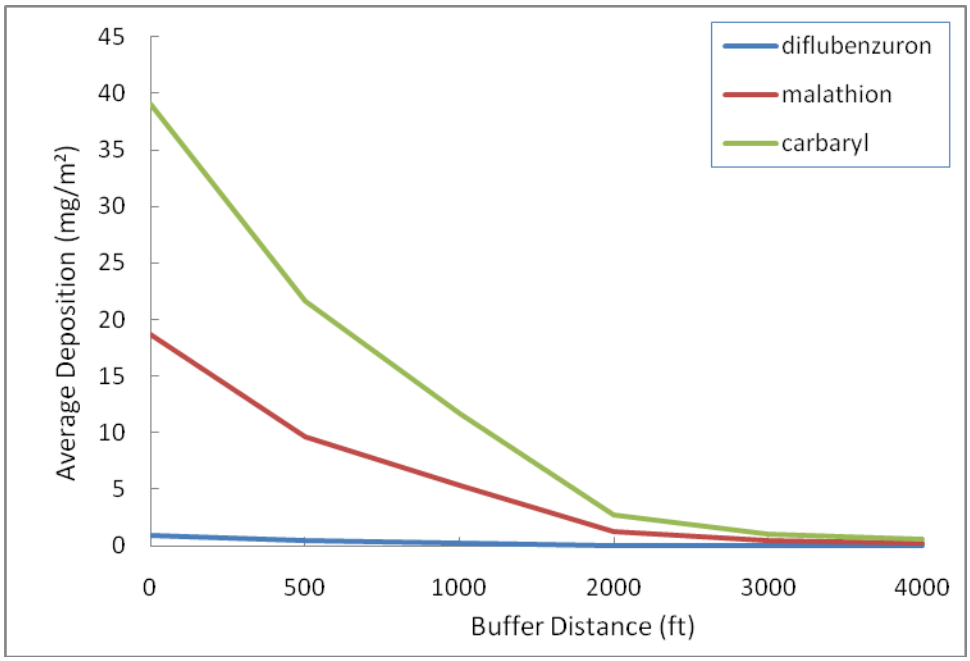

Figure 2-2. Aerial deposition of program insecticide applications at varying buffer distances from the edge of the spray block

Input parameters for ground ULV applications require fewer variables when compared to those used for aerial applications. The same maximum RAAT application rate and droplet size distribution for aerial applications were used; however, release height was based on a high boom height scenario (table 2–2). Estimated environmental concentrations are 90th percentile values. Carbaryl bait ground applications were modeled using the ASABE fine to medium coarse droplet size distribution, which has a median droplet size of 340.87 µm. AgDrift and AgDisp do not allow for ground applications using large diameter dry bait formulations. Therefore, the application was assumed to be from a liquid application. Modeling results with these assumptions substantially overestimate drift from ground bait applications. Baits are applied as a dry material with a median bait size of typically 1500 µm, which would result in very little off-site transport of carbaryl from drift or volatilization. Another limitation with AgDrift ground application scenarios is the lack of ability to model treating alternating swaths as proposed in the RAATs. The lack of ability to model alternating application swaths is another factor that results in modeled values above those expected from ULV or bait ground treatments.

Table 2–2. AgDrift Ground Modeling Parameters for RAAT ULV Applications

Parameters	Carbaryl	Diflubenzuron	Malathion
Application rate (lb a.i./.ac)	0.375	0.012	0.31
Boom Height (in)	50	50	50
Droplet Size Distribution	ASABE Very Fine to Fine	ASABE Very Fine to Fine	ASABE Very Fine to Fine
Data Percentile	90	90	90
Swath Width (ft)	45	45	45

Similar to AgDisp, output from the AgDrift model can be used to assess the impacts of buffer distance on reductions in drift. The influence of buffers is greater in the ground applications compared to the aerial applications primarily due to the much lower application height for ground applications. The first 50 feet of buffer result in greater than 96% reductions in drift for all three program insecticides (figure 2-3). Limitations in the model do not allow for the estimate of zero residues.

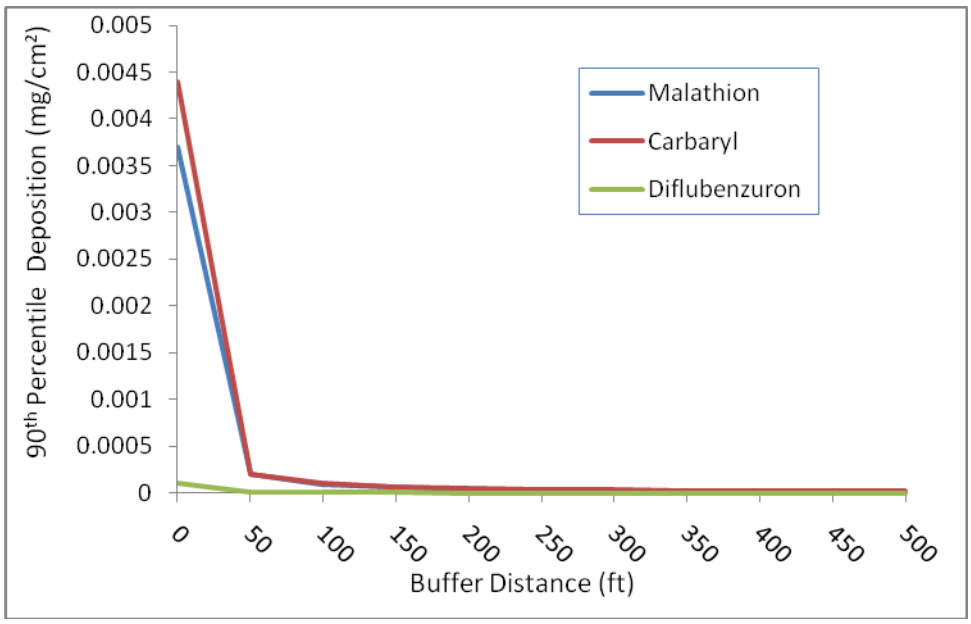

Figure 2-3. Ground deposition of program insecticide applications at varying buffer distances from the edge of the spray block

2. Observed Residue Values from Program Applications

Available monitoring data was compared to the modeled data as a means to determine the predictability of modeling data to residues from actual field applications. USDA–APHIS conducts environmental monitoring as part of its activities in several programs. This data is collected to determine if applications are being made properly and, in cases where sensitive habitats are present, to confirm that the proposed application buffers are protective (USDA APHIS, 2003; 2004; 2005; 2006b; 2007, 2009, 2010). Grab samples from the tank mix, water, sediment, and vegetation as as drift card samples are collected as part of the environmental monitoring program. Monitoring data from 2003–2009 was reviewed to determine if the assumptions in the drift modeling were conservative and representative of the type of drift expected from program activities. Data from previous years was not available because the program was not making applications for several years prior to 2003 due to lack of funding. Emphasis in this review will be on the drift card data because it provides a standardized unit of measurement (mg/m^2) that can be compared to output from the terrestrial deposition estimate in AgDrift/AgDisp. Aquatic residue data will also be summarized; however, because the size and type of the waterbodies and subsequent volumes are not quantified, comparisons to output from AgDrift/AgDisp are not applicable. As previously stated, program pesticide applications are primarily made using diflubenzuron, and the monitoring data reflects this preference.

Environmental samples were collected from several States (California, Idaho, Washington, Montana, Oregon, Nebraska, Nevada, South Dakota, and Utah) under a variety of environmental conditions. In 2003, approximately 38 dye cards and 10 water samples were collected from various treatments. All post-treatment water (n= 10) and sediment (n = 2) samples were below the level of detection (LOD) for diflubenzuron (1.4 µg/L and 45 µg/L, respectively). All dye

card samples were either below the detection (0.093 mg/m^2) or quantification (0.3 mg/m^2) limit with the exception of four samples collected from Utah with values that ranged from 0.32 to 0.93 mg/m^2. No buffer is listed for these samples, so it is unclear where they were collected in relation to the spray block.

In 2004, 11 dye card samples were collected from two sites in Nevada. All samples were below the LOD (0.093 mg/m^2) with the exception of two samples. One sample collected 30 ft downwind of the spray block measured 0.61 mg/m^2, while the other sample collected 300 ft downwind of the spray block measured 0.92 mg/m^2. The remaining dye cards that were below the LOD had buffers ranging from 130 to 450 feet from the application block. Water (n=10) and sediment (n=15) samples were below the LOD for all samples in both matrices.

In 2005, nine dye card samples from Montana were all below the LOD for diflubenzuron (0.093 mg/m^2). Four sediment samples from an application in Idaho and two water samples from applications in South Dakota had diflubenzuron residues below the LOD for sediment (45 µg/L) and water (1.4 µg/L). Application buffers were not reported for the drift card, sediment or water data.

In 2006, 29 dye card samples were collected from different applications in Utah, Nevada and Idaho with 5 samples below the LOD (0.015 mg/m^2), two samples containing trace amounts (0.015 – 0.051 mg/m^2), and 22 samples containing quantifiable amounts of diflubenzuron. The average concentration of diflubenzuron was 0.10 mg/m^2, with a maximum value of 0.42 mg/m^2. Distances from the treatment block to the sensitive sites or whether the samples were collected within the spray block were not provided; therefore, the samples have limited use in quantifying drift reductions from the application of buffers. The total number of water samples collected during 2006 was 29 with 25 samples having residues below the LOD, one sample containing trace amounts, and 3 samples containing levels ranging from 4.8 to 13 µg/L. The highest test concentration resulted from a direct application to a small ditch (1 ft wide by 0.5 ft deep) while the other residues were collected from a farm pond. Four sediment samples were below the LOD for diflubenzuron (45 µg/L).

Limited water monitoring (n=6), from Nevada in 2007, revealed that all samples were below the level of detection or quantification for diflubenzuron with the exception of two samples that contained 1.4 and 4.7 µg/L diflubenzuron. Dye card results from monitoring in 2007 revealed that of 15 samples collected at 500 ft from the spray block, 7 were below the LOD (0.015 mg/m^2) and 8 contained quantifiable levels ranging from 0.05 to 0.15 mg/m^2 for an average of 0.05 mg/m^2. A comparison of the quantifiable range of concentrations to the within spray block range of concentrations (1.3-3.5 mg/ m^2) demonstrates that drift reduction at 500 ft ranged from approximately 88 to 99%. Drift reduction would be greater when factoring in dye cards with diflubenzuron levels that were below the LOD.

Dye card sampling for aerial ULV applications of diflubenzuron and carbaryl applications near bodies of water were collected in 2008. Six dye card samples collected approximately 500 ft from an application site in Montana revealed diflubenzuron concentrations below the LOD (0.015 mg/m^2) in three samples, and below the level of quantification (LOQ) for the other three samples (0.069 mg/m^2). The average dye card concentration was 0.021 mg/m^2 using 0.5 the LOD and LOQ. Twelve drift card samples from various aquatic habitats were collected after a

carbaryl ULV application in Montana. Six of the samples had concentrations below the LOD (0.078 mg/m^2), while the other samples had a range of 0.10 to 0.88 mg/m^2 for an average concentration of 0.25 mg/m^2.

Dye card and water samples were collected in 2009 for diflubenzuron and carbaryl applications in Montana, Nevada, Oregon and Utah. Approximately 39 diflubenzuron dye card samples were collected at varying distances outside of the spray block. The average dye card concentration was 0.023 mg/m^2 with a range of < 0.015 to 0.46 mg/m^2. Six post treatment diflubenzuron water samples revealed concentrations below the level of detection (1.4 µg/L). Environmental samples related to carbaryl liquid and bait applications were also collected in 2009 from applications in Montana. In one case carbaryl bait applications were made to a field where an agreement was in place not to irrigate the treated field. A violation of the agreement by the landowner occurred and a discharge of irrigation water to the Little Bitterroot occurred. Water samples were collected upstream of the discharge as well as 5.5 and 8.0 miles below the area where the discharge occurred. The residue value upstream of the discharge was 1.2 µg/L while residue values at 5.5 and 8.0 miles below the discharge were 2.0 and 1.6 µg/L, respectively. There is uncertainty regarding whether these values represent any contribution from APHIS applications. The residues measured downstream are only slightly above the value measured upstream. The samples were collected approximately one week after irrigation was applied to the field and predates the actual application of the bait to the field by approximately two weeks. In another case, water sampling was conducted when aerial applications of liquid carbaryl were made well inside the standard 500 foot application buffer from aquatic resources. A drift card sample collected 20 feet from the Little Bitterroot River had a measured carbaryl residue of 0.71 mg/m^2 while an adjacent water sample that was collected from the river had a measured residue of 3.8 µg/L. Residues collected 1.6 and 3.9 miles downstream at the same time had carbaryl residues of 8.6 and 1.1 µg/L, respectively. No upstream samples were collected and it is unclear whether other carbaryl applications were taking place in the area so there is uncertainty regarding the potential contribution of program applications to carbaryl loading into the Little Bitterroot River.

For those dye card samples where the buffer was at, or less than 500 feet from the spray block during a RAAT treatment, the average drift card residue value was 0.032 mg/ m^2 with a range of <0.015 to 0.46 mg/ m^2 based on available data. Between 2003 and 2009 approximately 63 post treatment water samples and 25 sediment samples were collected to determine diflubenzuron residues. Greater than 92% of water samples that were collected were below the level of detection for diflubenzuron while 100% of all sediment samples were below the level of detection. Five of the water samples contained residues ranging from 1.4 to 13.0 µg/L. The detection of diflubenzuron in some water samples are a result of applications that occurred without proper adherence to program restrictions regarding the 500 foot buffer from aquatic sites. The same is true for the water sampling data for carbaryl collected during 2009.

3. Drift Simulations

AgDisp and AgDrift were used to estimate residues in an aquatic environment under all application scenarios. Estimated environmental concentrations varied based on the type of application and insecticide used for each treatment (table 2–3). The aerial bait associated residue values are highly conservative since the size of the bait is underrepresented in the modeling scenario. The average median diameter for modeling bait was based on an ASABE coarse to

very coarse particle size, which is well below the size of the pellet bait currently being used. The average size of the pellet is 2 to 5 millimeters, which is substantially larger than the median diameter value (527 µm) that was assumed in this assessment.

Table 2–3. Estimated Aquatic Environmental Concentrations from the Application of Program Insecticides

Insecticide	Type of Application	Application Boom Height	Application Buffer (ft)	Instantaneous Concentration (µg/L)
Carbaryl	Aerial ULV	75 ft	3500	2.30
	Aerial Bait	75 ft	1000	0.54
	Ground ULV	50 in	350	0.82
	Ground Bait	50 in	200	0.28
Diflubenzuron	Aerial ULV	75 ft	1,500	0.69
	Ground ULV	50 in	150	0.07
	Aerial ULV	75 ft	3,500	1.50
Malathion	Ground ULV	50 in	500	0.43

The water body that is simulated downwind of the application site is 25-ft wide with an average depth of 1.0 ft with no inflow or outflow and the assumption of uniform insecticide mixing. The selection of the water body size was designed to capture sensitive off-channel salmonid habitat that is critical for salmonid reproduction (NMFS, 2008, 2009). The assumption is that if these habitats are protected, then other larger flowing waterbodies that serve as salmonid habitat would also be protected.

The aquatic residue values calculated using the AgDisp and AgDrift model do not represent residues that would be expected to occur during program applications in salmonid habitat. The intent of this exposure assessment is to generate a residue based on multiple conservative assumptions that could then be integrated with the effects analysis to generate a risk characterization demonstrating that the risk to listed salmonids and their habitat is insignificant and discountable from program insecticide applications. The multiple conservative, and in some cases unrealistic, assumptions in the exposure analysis are an attempt to account for some of the uncertainty in the risk characterization discussed in section four of this biological assessment. Conservative input parameters used in AgDisp regarding temperature, humidity, application height, wind direction and wind speed occurring simultaneously results in application conditions that would be unlikely to occur during an application event. Also, the spray block was assumed to occur parallel to the length of the waterbody, with drift occuring directly toward the waterbody for the entire length of each swath. In addition, changes in elevation between the spray block and sensitive habitat were not considered and interception by terrestrial and riparian plants within the buffer zone was minimized. The output from the AgDrisp model was compared to the available dye card data for diflubenzuron that has been collected from previous monitoring efforts and summarized in this biological assessment. The average residue value on dye cards collected at, or within, the 500 foot application buffer was 0.037 mg/ m^2 based on the available data from 2007 through 2009. The average drift value that was estimated using

AgDisp at 500 feet was 0.87 mg/ m^2 which is approximately 23 times greater than the value that was observed in the field. The difference between measured and observed dye card results is likely due to the multiple assumptions in the input parameters that were assumed to occur simultaneously. These would not be expected to occur for any of the program insecticide applications since it would result in a low probability of a successful suppression treatment. Application of a ten fold reduction in estimated insecticide residues based on the measured data at 500 feet results in aerial ULV application residues of 0.15 µg/L for malathion, 0.004 µg/L for diflubenzuron, and 0.23 µg/L for carbaryl applications. The ten fold reduction is less than half of what was observed as a difference between measured and observed dye card data and would still be viewed as an overestimate of drift that would be expected to occur to salmonid habitat. These values will be used in the risk characterization to assess potential direct and indirect risk of program applications to salmonids. Monitoring drift card data for ground applications was unavailable and therefore aquatic residues were not reduced from model estimates. Other mitigation measures proposed in the program such as reduced area applications and wind direction restrictions were not considered in the aerial or ground drift modeling. These factors and the models tendency to overpredict residues at the aerial buffer distances and application heights modeled in this assessment would also result in lower residue drift values to aquatic habitats.

In addition to the input parameters assumed in the drift modeling, another important assumption in the exposure estimate was that concentrations were considered instantaneous, with no degradation during aerial transport, or any degradation or dissipation occurring in the simulated waterbody after deposition. AgDisp and AgDrift were used to estimate off-site transport of spray drift related to program applications but were not used to determine the environmental fate and degradation of each insecticide. Also, no pesticide interception from emergent aquatic plant vegetation that would be present in a shallow static water system was considered.

An aspect of both models that may not be considered conservative is the integrated deposition of residues over the surface of the water body and uniform mixing. This effect is minimized in this assessment since a relatively narrow body of water is used and that at the distances being modeled the influence of integrating concentrations over an area compared to a point deposition becomes less. This is because the influence of droplet size, application height, and other factors diminish with increasing buffer width compared to atmospheric stability which becomes a more significant influence.

4. Runoff Simulations

AgDisp and AgDrift provide estimates of off-site residues related to drift in terrestrial and aquatic environments. However, they do not provide an estimate of the amount of runoff that could occur into aquatic habitats. Several aquatic fate models exist to estimate environmental loading into aquatic habitats. EPA–OPP has developed a tiered approach for the use of aquatic fate models that allow the user to estimate aquatic concentrations based on default "reasonable worst-case conditions," or to calculate estimated aquatic concentrations based on crop-specific soil and weather conditions (EPA, 2004). None of the available models allow the user to calculate the effects of application buffers in reducing pesticide runoff.
The runoff contribution from aerial ULV applications in the program is considered minimal due to the large application buffers that are being applied adjacent to aquatic environments. The

effectiveness in the use of application buffers to reduce runoff can vary based on site conditions, the type of vegetation present in the buffer, and the fate of the insecticide; however, the products used in the program and the large buffers ensure that runoff will not be a significant contribution of off-site pesticide movement when products are applied according to label specifications and APHIS policy.

A majority of the published data regarding the effects of buffer zones on reducing pesticide runoff is based on pesticides that have a high runoff potential due to persistence and a low binding affinity to organic matter. Hatfield et al. (1995) demonstrated that grassed filter strips ranging from 40 to 60 ft removed 10 to 40% of the herbicides atrazine, cyanazine, and metolachlor. Arora et al. (1996) found that a 66-ft-wide riparian buffer on a 3% slope removed anywhere from 8 to 100% of the herbicides atrazine, metolachlor, and cyanazine during storm events. The variability in pesticide retention within the buffer zone was related to the amount of runoff during storm events. In a review by Neary et al (1993), buffers of approximately 50 ft, or larger, were effective in greatly reducing pesticide runoff to water bodies. Syverson and Bechmann (2004) demonstrated that with an approximate 15-ft-wide buffer, sediment-bound residues of glyphosate, fenpropimorph, and propiconazole were reduced 39, 71, and 63%, respectively. Removal efficiency of soluble fractions of each product was 24 to 70% for glyphosate, 32 to 78% for propiconazole, and 61 to 73% for fenpropimorph. These types of removal efficiencies have been observed for other pesticides as well, such as 2,4-D and trifluralin (Lacas et al., 2005). Asmussen et al. (1977) documented 70% reductions in 2,4-D levels, while Rhode et al. (1980) demonstrated a 94% reduction in the herbicide trifluralin, which has a relative higher binding affinity, using grassed buffers of 24.4 m. Equivalent buffer distances have been established for trapping sediment, which would suggest that pesticides that sorb to sediment would also be reduced with similar sized buffer zones (Wenger, 1999; Gril et al., 1997).

To illustrate the impact of buffer length on runoff, the malathion rate of 0.31 lb ai/ac was used to estimate the total amount of insecticide that would result in an application to an area that results in a 25:1 ratio of a uniform application draining into a body of water that is one foot deep. Ten percent of the total amount applied was assumed to occur into the shallow waterbody. Based on the available literature for other pesticides, a range of potential reduction rates was assumed to occur every 25 feet from the body of water. At a buffer distance of 500 feet malathion residues were 0.88 µg/L assuming 25% removal and 0.26 ng/L assuming 50% removal (figure 2-4). The results estimated in this exercise are not designed to be comparable to the residues estimated from drift estimates but to illustrate that residues from runoff are not expected to significantly contribute to loading into aquatic habitats under actual application conditions.

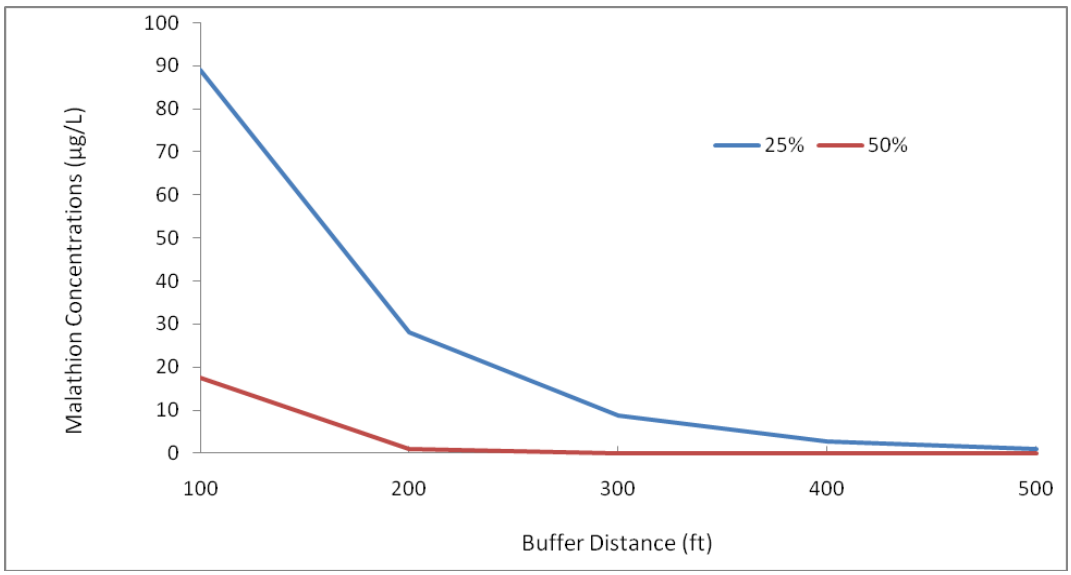

Figure 2-4. Runoff related malathion residues based on a range of removal efficiencies

Larger ratios of applied area to aquatic habitat are possible; however, this also results in larger areas that are treated farther from the aquatic habitat and require greater amounts of rainfall to reach an aquatic habitat. Estimates of initial residues in this exercise are considered conservative since degradation and dissipation were not considered and, most significantly, the volume of water was static. A runoff event would result in increased volume in the aquatic habitat of interest from direct deposition of rainfall as well as the water required in the runoff to remove the pesticide from the spray block.

APHIS realizes the results from the literature do not represent all of the site conditions that exist throughout the potential areas of application; however, they do provide supporting evidence that relatively small buffer zones can substantially reduce pesticide loading from surface runoff to aquatic environments. Based on the significantly larger application buffers proposed in this program, a situation where significant runoff could contribute to aquatic environmental loading from program applications would require such a significant rainfall event and channelized water flow that the pesticide would be substantially diluted and/or sorbed to sediment by the time it could potentially reach listed salmonid habitat. In addition, it's the policy of the program to not make applications when rainfall is expected, which would further reduce the potential for significant runoff from program applications.

III. Response Analysis for Grasshoppper Insecticides

The response analysis for all three insecticides is based on available peer-reviewed and grey literature as well as on-line database searches using multiple sources. All three products have been registered for some time so a large amount of data exists, in particular acute toxicity data, regarding effects. The goal in this section of the BA is not to discuss all the available studies that

have been published but to gather response data that characterizes the range of sensitivities to multiple taxa based on acute and chronic exposures. The focus of this section is to discuss laboratory-related response data for all representative taxa. Available acute and chronic fish response data is summarized for each insecticide as well as impacts to aquatic invertebrates and plants. The inclusion of aquatic invertebrate and plant response data to program insecticides is meant to provide response information on indirect effects to listed salmonids such as prey availability and sheltering. In cases where multiple acute toxicity values were available for a single species, the lowest toxicity value was selected in the analysis. This was considered to be a more conservative approach in the response analysis, compared to calculating geometric mean toxicity values, which are typically used in ecological risk assessments. Studies that were evaluated for sub-lethal response data and not used in the effects analysis are listed in the appendices of this assessment. The use of field-collected data will be discussed in the risk characterization section in context to the available laboratory toxicity data and exposure modeling results.

A. Carbaryl

1. Mode of Toxic Action

Carbaryl is a carbamate insecticide (figure 3-1):

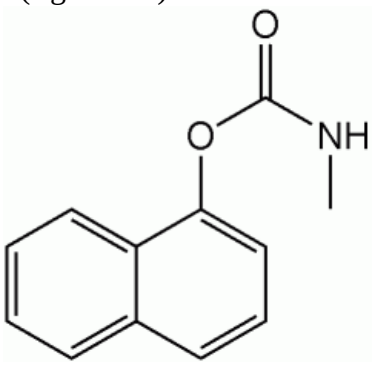

Figure 3–1. Chemical structure of carbaryl

The mode of action of carbamates occurs primarily through acetylcholinesterase (AChE) inhibition (Smith, 1987; Klaassen et al., 1986). The AChE enzyme is responsible for the breakdown (hydrolysis) of acetylcholine, a neurotransmitter that permits transmission of nerve impulses across the nerve synapse. Carbamates exhibit a reversible pesticide-enzyme binding reaction (carbamylation), which results in gradual decreases in binding as their concentration decreases through metabolism and excretion.

2. Fish

Acute carbaryl toxicity to fish ranges from slightly to highly toxic. The 96-hour median lethal concentration of carbaryl ranges from 0.25 mg/L for the Atlantic salmon, *Salmo salar* to 20 mg/L for *Amelurus melas* (Mayer and Ellersieck, 1986). Species of catfish and minnow are generally 10 times more tolerant than salmonids (figure 3–2; appendix A).

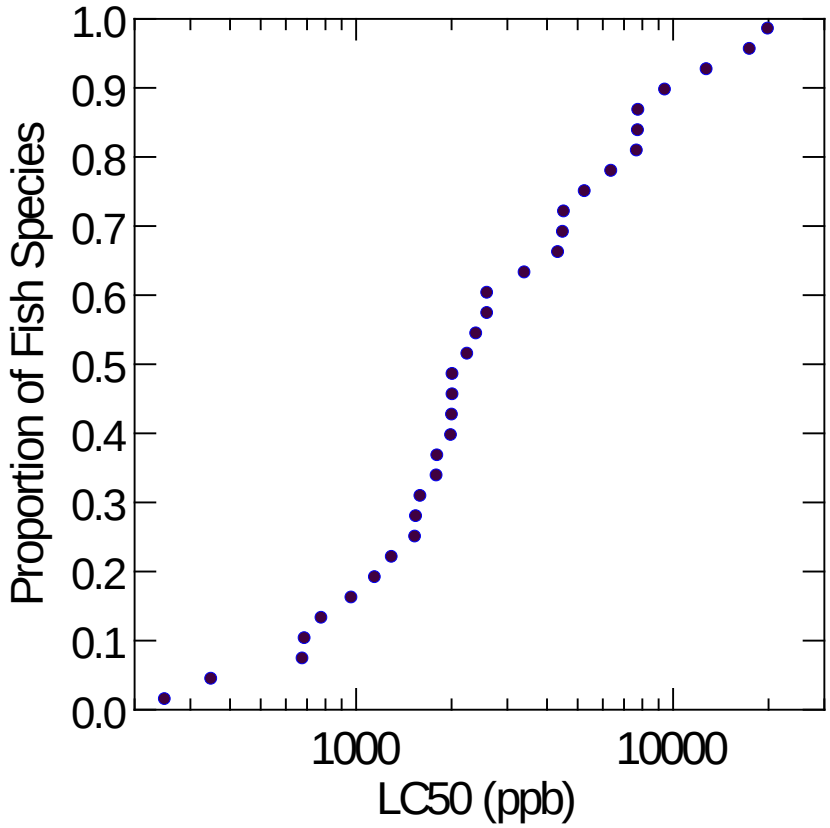

Figure 3–2. Cumulative distribution of acute fish toxicity values for carbaryl

Acute sublethal effect levels related to carbaryl can vary depending on the endpoint and test species. Little et al. (1990) noted several carbaryl-related behavioral effects after a 96-hour exposure period using the rainbow trout. Effects on swimming capacity, swimming activity, and the number of *Daphnia* consumed were statistically significant at a concentration of 1.0 mg/L carbaryl. No effects on the above endpoints were noted at the next concentration, 0.1 mg/L, which represents the no observed effect concentration (NOEC). In a 6-hour exposure of carbaryl to the cutthroat trout, *O. clarki*, the concentrations where no effects were seen on predator avoidance or swimming performance was 200 and 500 µg/L, respectively (Labenia et al., 2007). In a 7-day exposure using the fathead minnow at different age classes, the NOEC value ranged from <250 µg/L to 500 µg/L based on growth. The NOEC value that was less than 250 µg/L was repeated, and the second test demonstrated a NOEC value of 1.0 mg/L based on growth (Pickering et al., 1996).

In addition to behavioral responses, the inhibition of brain cholinesterase (BChe) and the regulation of the muscarinic cholinergic receptors (MChR) after carbaryl exposure have been evaluated for several species in short-term exposures (Zinkl et al., 1987; Jones et al., 1998; Beauvais et al., 2001; Ferrari et al., 2004a; Ferrari et al., 2004b; Beyers and Sikoski, 1994).

Ferrari et al. (2004a, 2004b) determined the BChe inhibition concentration (IC_{50}) for larval rainbow trout and the goldfish to be 19 µg/L to 2.62 mg/L, respectively. The IC_{50} value for trout (19 µg/L) was calculated using non-linear regression with 95% confidence intervals of 15 and 23 µg/L. The lowest concentration that appears to have been tested, (~ 6 µg/L) resulted in approximately 35% inhibition. Beauvais et al. (2001) documented a statistically significant effect on brain cholinesterase at carbaryl concentrations of 188 µg/L. No other concentrations were tested and, thus, a NOEC could not be established. Beyers and Sikoski (1994) determined the 24-hour NOEC for cholinesterase inhibition to be 30 µg/L for the Colorado squawfish. Jones et al. (1998) measured MChR in several cold and warmwater fish species. MChR was affected in rainbow trout 2.2 mg/L and higher but not at doses of 0.5, 0.8 and 1.3 mg/L. No effects on MChR were observed for the Lahonton or Apache trout at the highest concentration tested, 2.2 and 1.3 mg/L, respectively. For the four warmwater species tested in the study, fathead minnow, razorback sucker, bonytail chub and Colorado squawfish there was a species dependent effect on MChR however no impacts were observed for any species at or below a concentration of 1.3 mg/L.

In longer-term studies, chronic NOEC concentrations have been established for the fathead minnow, *G. elegans*, and *Ptychocheilus lucius*. In studies ranging from 32- to 35-day exposures, a NOEC value of 210, 650, and 445 µg/L was calculated for the fathead minnow *P. promelas*, bonytail chub, *Gilea elegans*, and Colorado pikeminnow, *P. lucius*, respectively. Both *G. elegans* and *P. lucius* are currently listed species (Beyers et al., 1994). Carlson (1972) reports a NOEC of 210 µg/L for the fathead minnow in a fish full-life cycle study.

3. Aquatic Plants

Aquatic plant toxicity testing is not typically required for insecticides under EPA regulatory requirements. However, studies have been submitted testing the effects to the freshwater green algae *Pseudokirchneriella subcapitata*, with a reported effective concentration (EC_{50}) and NOEC of 1.27 and 0.29 mg/L, respectively, for the technical active ingredient (US FS, 2008). In another study the effects of carbaryl on four algal species, seven cyanobacteria species, and the aquatic macrophyte, *Lemna minor* found statistically significant effects at the one dose used in the study (3.7 mg/L)(Peterson et al., 1994). Boonyawanich et al. (2001) reported 96-hour EC_{50} values of 0.996, 0.785, and 0.334 g/L for the three aquatic plants, *Ipomoea aquatica*, *Pistia stratiotes*, and *Hydrocharis dubia*.

4. Aquatic Invertebrates

Carbaryl is very highly toxic to all aquatic insects, and highly to very highly toxic to most aquatic crustaceans. The toxicity from 96-hour static tests ranged from 1.5 µg/L in the shrimp, *Paneaus aztecus*, to 22.7 mg/L in the mussel, *Mytilus edulis* (Mayer, 1987; EPA, 2003a) (figure 3–3; appendix A). Peterson et al. (1994) evaluated EC/LC_{50} values for crustacea ranging from 5 to 9 µg/L (cladoceran, mysid), 8 to 25 µg/L (scud), and 500 to 2500 µg/L (crayfish). Aquatic insects have a similar range of sensitivity.

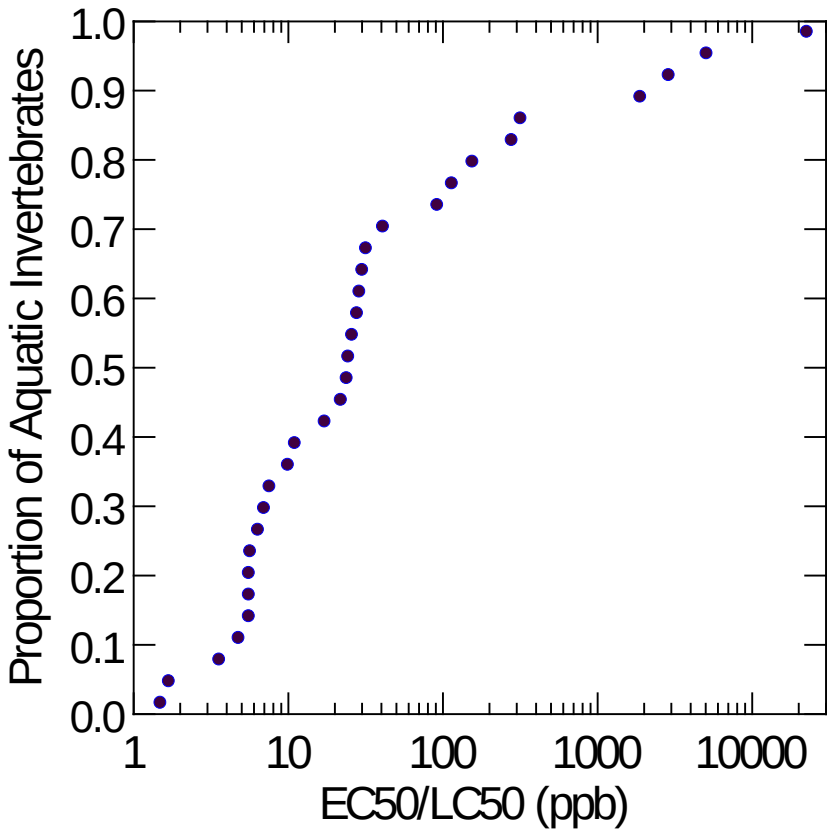

Figure 3–3. Distribution of acute aquatic invertebrate toxicity values for carbaryl

Chronic toxicity of carbaryl to aquatic invertebrates is variable depending on the test species and endpoint measured. Reproductive and growth related NOECs ranging from 1.0 to 15 µg/L have been reported for cladocerans while a NOEC of 500 µg/L was reported for the midge based on impacts on emergence (Hanazato, 1991; EPA, 2003a; US FS, 2008a).

5. Aquatic Toxicity of Formulations and Metabolites

Based on the available toxicity data for various formulations of carbaryl, the toxicity appears to be comparable to the range of sensitivities discussed for technical carbaryl effects to fish and aquatic invertebrates (table 3–1). Data for the proposed formulation used in this program is limited to two acute fish and one algal study. The formulation proposed for use in this program, Sevin® XLR Plus, contains approximately 44 percent carbaryl and an unknown quantity of 1,2-propanediol according to the available material safety data sheet. Other inerts that have been noted in the Sevin® XLR Plus formulation are an unknown sticker material and fine particulates (US FS, 2008a).

Table 3–1. Formulation Aquatic Toxicity Data for Carbaryl

Test Organism	Endpoint/Length	% AI	Toxicity Value	Reference
Onchorrynchus mykiss	96-hour LC_{50}	44	1.4 mg/L	EPA, 2003a
Onchorrynchus mykiss	96-hour LC_{50}	81.5	3.3 mg/L	EPA, 2003a
Onchorrynchus mykiss	96-hour LC_{50}	95	1.35 mg/L	Katz, 1961
Onchorrynchus kisutch	96-hour LC_{50}	95	0.99 mg/L	Katz, 1961
Onchorrynchus mykiss	96-hour LC_{50}	50	3.45 mg/L	EPA, 2003a
Onchorrynchus clarki	96-hour LC_{50}	49	6.7 mg/L	Woodward and Mauck, 1980
Cyprinus carpio	96-hour LC_{50}	50	3.30 mg/L	Kaur and Dhawan, 1993
Gambusia affinis	96-hour LC_{50}	5	204 mg/L	Naqvi and Hawkins 1988
Lepomis macrochirus	96-hour LC_{50}	44	9.8 mg/L	EPA, 2003a
Lepomis macrochirus	96-hour LC_{50}	30	49.0 mg/L	EPA, 2003a
Lepomis macrochirus	96-hour LC_{50}	50	22.0 mg/L	EPA, 2003a
Daphnia magna	48-hour EC_{50}	47.3	6.66 µg/L	EPA, 2003a
Daphnia magna	48-hour EC_{50}	81.5	7.2 µg/L	EPA, 2003a
Pseudokirchneriella subcapitata	96-hour EC_{50} 96-hour NOEC	XLR Plus	3.2 mg/L 1.8 mg/L	EPA, 2003a; US FS, 2008a

Available toxicity data for the primary metabolite of carbaryl, 1-naphthol, was compiled and compared to toxicity data for the parent compound (table 3–2). Available acute and chronic fish data for 1-naphthol is within the range of known EC_{50}/LC_{50} and NOEC values for carbaryl and fish. The same also holds true when comparing available aquatic invertebrate data for carbaryl and 1-naphthol. However, in studies where comparisons were made between technical carbaryl and 1-naphthol, the metabolite appears to be more toxic. Rao et al. (1984) reported that the 96-hr LC_{50} for technical grade carbaryl was 5.9 mg/L while the comparative value for 1-naphthol was 1.46 when using the fish *Cirrhinus mrigala*. Tilak et al. (1981) demonstrated that the acute fish toxicity of formulated carbaryl was less toxic than the metabolite 1-naphthol. Calculated 96-hr LC_{50} of carbaryl for *Catla catla, Anabas testudinens, Mystus casius* and *Mystus vittatus* were 6.4, 6.6, 4.6 and 2.4 mg/L, respectively, compared to 1-naphthol toxicity values which were 4.3, 3, 0.33 and 1.0 mg/L, respectively. Shea and Berry (1983) also reported higher comparative toxicity of 1-napthol to technical grade carbaryl; however, no toxicity values were reported.

Table 3–2. 1-Naphthol Laboratory Acute and Chronic Aquatic Toxicity Values

Test Organism	Endpoint/Length	Toxicity Value	Reference
Lepomis macrochirus	96-hour LC_{50}	0.75 mg/L	EPA 2003a
Cyprinodon variegates	96-hour LC_{50}	1.2 mg/L	EPA 2003a
Oncorynchus mykiss	96-hour LC_{50}	1.4 mg/L	EPA 2003a
Daphnia magna	48-hour EC_{50}	0.73 mg/L	EPA 2003a
Mysidopsis bahia	96-hour LC_{50}	0.21 mg/L	EPA 2003a
Crassostrea virginica	48-hour LC_{50}	2.1 mg/L	EPA 2003a
Pimepheles pomalis	32-days NOEC	0.10 mg/L	EPA 2003a

B. Diflubenzuron

1. Mode of Toxic Action

Diflubenzuron is classified as an insect growth regulator.

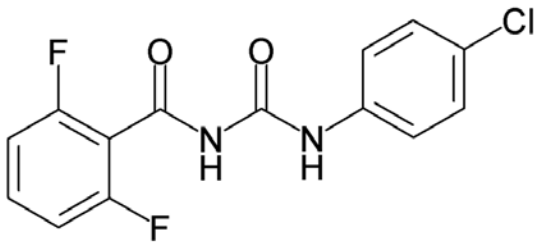

Figure 3–4. Chemical structure of diflubenzuron

The mode of action for diflubenzuron is through inhibition of chitin synthesis (interference with the formation of the insect's exoskeleton). The likely mechanism is through blockage of chitin synthetase, the ultimate enzyme in the biosynthesis pathway of chitin (Cohen, 1993; EPA, 1997). Exposure of insect life stages to diflubenzuron can result in larvicidal and ovicidal effects. The larvae are unable to molt properly due to a lack of chitin in the new cuticle. Exposure of larvae may occur through dermal contact, but the primary route of intoxication is ingestion. Ovicidal effects may occur through direct contact of eggs or through exposure of gravid females by ingestion or dermal routes. The larva develops fully in the egg, but is either unable to hatch or dies soon after hatching due to chitin deficiency in the cuticle. This inhibition of chitin synthesis affects primarily immature insects, but can also affect other arthropods and some fungi.

2. Fish

Toxicity of diflubenzuron to aquatic organisms varies by taxa. On an acute basis, diflubenzuron is considered slightly to practically nontoxic to fish. The median lethal concentration of diflubenzuron in water ranges from 10 mg/L for smallmouth bass to 660 mg/L in bluegill sunfish (Willcox and Coffey, 1978; Julin and Sanders, 1978; US FS, 2004; EPA, 1997). In several cases, LC_{50} values are above the highest test concentration used in the study. Toxicity values and references are summarized in appendix A. Sublethal acute effects have also been observed in exposures ranging from 6 to 96 hours. Maduenho and Martinez (2008) observed several sublethal impacts including reductions in erythrocytes and hemoglobin content as well as the induction of glutathione-s-transferase (GST) in *Prochilodus lineatus* after exposure to 25 mg/L of diflubenzuron. Granett et al. (1978) measured swimming behavior response in male Atlantic salmon parr in repeated 10 minute exposure trials to a granular formulation of diflubenzuron at a nominal concentration of 10 µg/L. The time spent in dosed plumes as well as the choice of flumes was found to be statistically significant when compared to controls.

In a subacute 30-day study using steelhead trout, fathead minnow, and guppies, the NOEC was determined to be greater than the highest test concentration of 45 µg/L based on survival and growth endpoints in early life stages (Hansen and Garton, 1982a). Julin and Sanders (1978)

exposed rainbow trout eyed eggs and fingerlings continuously for thirty days to diflubenzuron and found no effects at concentrations ranging from 0.029 to 0.30 mg/L.

A life-cycle study with the fathead minnow was conducted to support registration of diflubenzuron. These multi-generation studies are required by EPA–OPP when the pesticide meets certain criteria regarding toxicity and availability in aquatic ecosystems. In the 10-month continuous exposure life-cycle study, the low observed effect concentration (LOEC) and maximum acceptable toxicant concentration (MATC) values were found to be greater than 100 µg/L, with a NOEC value of 100 µg/L (EPA, 1997). The NOEC value does not indicate that concentrations above this level caused an adverse effect, but that it is an artifact of the study design where the highest test concentration was 100 µg/L. In another long term exposure study using the mummichog the reproductive NOEC was reported as approximately 50 µg/L (US FS, 2004).

Diflubenzuron does not bioconcentrate to significant levels, based on bioconcentration studies that were conducted using the bluegill sunfish and white crappie (Eisler, 2000).

3. Aquatic Plants

The lowest aquatic plant toxicity value is the NOEC for duckweed (*L. minor*) (190 µg/L) (Thompson and Swigert, 1993). The EC_{50} for the green algae *Selenastrum capricornutum* is 200 µg/L (EPA, 1997). Chitinous algae (diatoms) are not adversely affected by diflubenzuron (Antia et al., 1985).

4. Aquatic Invertebrates

The acute and chronic toxicity of diflubenzuron to aquatic invertebrates is variable and dependent on the group of aquatic organism being tested (figure 3–5; appendix A). The acute median lethal concentration of diflubenzuron in water to crustaceans ranges from 0.75 µg/L in *Daphnia magna* (US FS, 2004) to 2.95 µg/L in the grass shrimp *Palaemonetes pugio* (Wilson and Costlow, 1986). The median lethal concentration of diflubenzuron in water to immature stages of aquatic insects ranges from 0.5 µg/L in the mosquito *Aedes nigromaculatum* (Miura and Takahashi, 1974) to 57 mg/L in the perlodid stonefly, *Skwala* sp. (Mayer and Ellersieck, 1986). The median lethal concentration of diflubenzuron in water to the snail *Physa* sp. is greater than 125 mg/L (Willcox and Coffey, 1978).

Based on the available sublethal data for diflubenzuron, cladocerans and copepods appear to be the more sensitive group with an acute NOEC range of 0.3 to 1.0 µg/L and a chronic NOEC range of 0.04 to 0.25 µg/L (EPA, 2007; US FS, 2004). Adverse effects on growth, survival, reproduction and behavior occur between 0.062 and 2 µg/L (Testler and Costlow, 1981; Nebeker et al., 1983; Eisler, 2000).

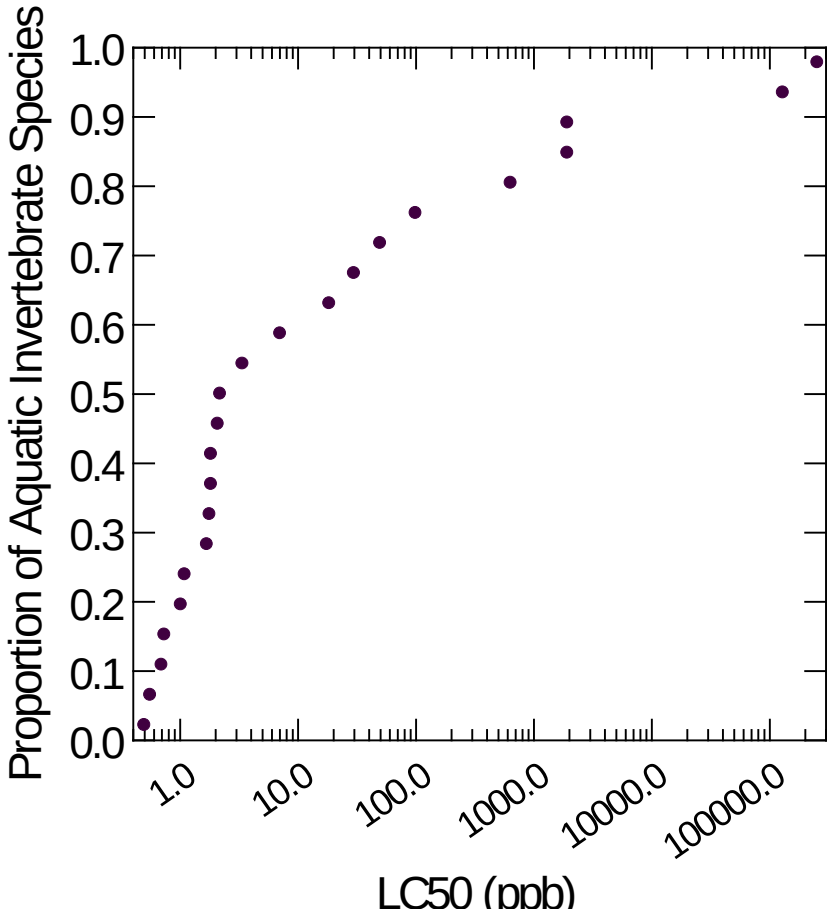

Figure 3–5. Distribution of aquatic invertebrate toxicity values for diflubenzuron

5. Aquatic Toxicity of Formulations and Metabolites

Environmental degradation of diflubenzuron can result in four primary metabolites including CO_2. The other three metabolites are 4-chlorophenyl urea, 2-6 difluorobenzoic acid and 4-chloroaniline. Under aerobic and anaerobic conditions in soil the metabolite 4-chlorophenyl urea, can compose 37% of the applied radioactivity in soil metabolism studies while 2-6 difluorobenzoic acid may compose 23% under anaerobic conditions (EPA, 2007). The metabolite, 4-chlorophenyl urea, can further degrade to 4-chloroaniline. Available data for 4-chloroaniline demonstrates that acute fish toxicity is greater for the metabolite when compared to the parent for the bluegill, rainbow trout, and fathead minnow (table 3-3). Diflubenzuron acute toxicity to these fish species is greater than 100 mg/L. Acute toxicity of 4-chloroaniline to *Daphnia magna* and *C. plumosus* is approximately 100 times lower than comparable values for diflubenzuron.

The other two environmental metabolites, 2-6 difluorobenzoic acid and 4-chlorophenyl urea, are considered less toxic, or comparable in toxicity, to diflubenzuron based on available response data for aquatic organisms. Acute lethality studies for the channel catfish, bluegill, fathead

minnow and midge, *C. plumosus* report EC_{50}/LC_{50} values greater than 100 mg/L while the reported 96-hr LC_{50} for the rainbow trout was 72 mg/L after exposure to 4-chlorophenyl urea (Julin and Sanders, 1978). Fish toxicity from exposure to 2-6 difluorobenzoic acid is similar to the parent and 4-chlorophenyl urea based on acute lethality values for the fathead minnow, rainbow trout and bluegill. The fathead minnow LC_{50} was estimated to be approximately 69 mg/L while median lethal concentrations were greater than 100 mg/L for the rainbow trout and bluegill (Vitozzi and De Angelis, 1991). The 48-hr EC_{50} for *C. plumosus* after exposure to 2-6 difluorobenzoic acid was greater than 100 mg/L (Julin and Sanders, 1978).

Table 3–3. Diflubenzuron Metabolite (4-chloroaniline) Toxicity to Aquatic Organisms.

Test Organism	Endpoint/Length	Toxicity Value	Reference
Bluegill	96-hour LC_{50}	2.4 mg/L	Julin and Sanders, 1978
Rainbow trout	96-hour LC_{50}	14 mg/L	Julin and Sanders, 1978
Fathead minnow	96-hour LC_{50}	12 mg/L	Julin and Sanders, 1978
Channel catfish	96-hour LC_{50}	23 mg/L	Julin and Sanders, 1978
Zebra fish *Brachydanio rerio*	35-day NOEC (Reproduction)	0.2 mg/L	Bresch et al., 1990
	21-day NOEC	1.8 mg/L	US FS, 2004
Japanese Medaka	96-hour LC_{50}	37.7 mg/L	Holcombe et al, 1995
	28-day NOEC (Weight)	<2.25 mg/L	Holcombe et al, 1995
Chironomus plumosus	48-hour EC_{50}	43 mg/L	Julin and Sanders, 1978
Daphnia magna	48-hour EC_{50}	0.31 mg/L	US FS, 2004
	21-day NOEC	0.01 mg/L	Kuhn et al., 1989

Several aquatic studies have been conducted testing the toxicity of formulations of diflubenzuron (table 3-4). Toxicity studies using the 25% wettable powder formulation are the most common based on available data. The below table represents the range of sensitivities of taxa to some of the formulation data however additional toxicity data with wettable powder formulation is available (Fischer and Hall, 1993). The material safety data sheet for Dimilin® 2L, which is the formulation proposed for use in this program, contains 22% diflubenzuron as well as a surfactant at less than 2% by weight and "non-hazardous inert ingredients" at less than 78% by weight. Some of the known inerts in the below formulations are also present in the formulation proposed in this program, so the data provide some limited information regarding how acute toxicity may differ between formulated and technical material. Based on the available acute fish and aquatic invertebrate toxicity data for formulated diflubenzuron, the range of sensitivities is similar to those for the technical active ingredient (table 3–4).

Table 3–4. Diflubenzuron Aquatic Toxicity Values for the Typical End Use Product (TEP)

Test Organism	Endpoint/Length	% AI	Toxicity Value	Reference
Rainbow Trout *Oncorhynchus mykiss*	96-hour LC_{50}	25	240 mg/L	Julin and Sanders, 1978
Channel Catfish *Ictalurus punctatus*	96-hour LC_{50}	25	370 mg/L	Julin and Sanders, 1978
Fathead minnow *Pimephales promelas*	96-hour LC_{50}	25	430 mg/L	Julin and Sanders, 1978
Bluegill sunfish *Lepomis macrochirus*	96-hour LC_{50}	25	660 mg/L	Julin and Sanders, 1978
Coho salmon *Oncorhynchus kisutch*	96-hour LC_{50}	25	>100 mg/L	McKague and Pridmore, 1978
Mummichog *Fundulus heteroclitus*	96-hour LC_{50}	25	32.99 mg/L	Lee and Scott, 1989
Fathead minnow *Pimephales promelas*	30-day NOEC	NR	>36 µg/L	Nebeker et al., 1983
Fairy shrimp *Streptocephalus sudanicus*	48-hour EC_{50}	NR	0.74 µg/L	Lahr et al., 2001
Grass shrimp *Paleomenetes pugio*	96-hour LC_{50}	25	1.39 µg/L	Wilson and Costlow, 1986
Daphnia magna	48-hour EC_{50}	25	15 µg/L	EPA, 1997
Gammarus pseudolimnaeus	96-hour EC_{50}	25	25 µg/L	EPA, 1997
Chironomus plumosus	48-hour EC_{50}	25	0.56 mg/L	Julin and Sanders, 1978
Backswimmer *Anisops sardeus*	48-hour EC_{50}	NR	1.93 µg/L	Lahr et al., 2001
Eastern Oyster *Crassostrea virginica*	96-hour EC_{50}	25	130 mg/L	EPA, 1997

NR = Active ingredient not reported but used a Dimilin formulation.

C. Malathion

1. Mode of Toxic Action

Malathion is an organophosphate insecticide whose mode of toxic action is primarily through acetylcholinesterase (AChE) inhibition (figure 3-6) (Smith, 1987; Klaassen et al., 1986). Detoxification typically occurs rapidly for malathion and its oxon, malaoxon, via ester hydrolysis, demethylation, and phoshphorothiolate ester hydrolysis.

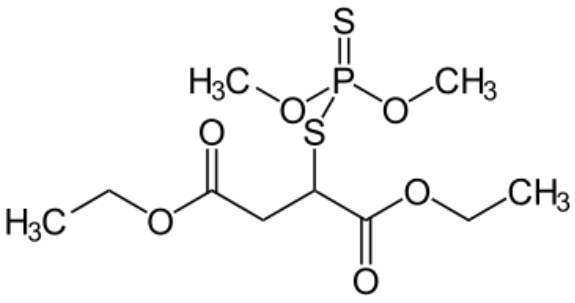

Figure 3–6. Chemical structure of malathion

2. Fish

The acute toxicity of malathion varies from moderately toxic to some species of fish to very highly toxic to other species, with an LC_{50} of 4 µg/L in rainbow trout to 15,300 µg/L for the federally listed bonytail chub, *Gila elegans* (Mayer and Ellersieck, 1986; Beyers et al., 1994, US FS, 2008b) (figure 3–7; appendix A).

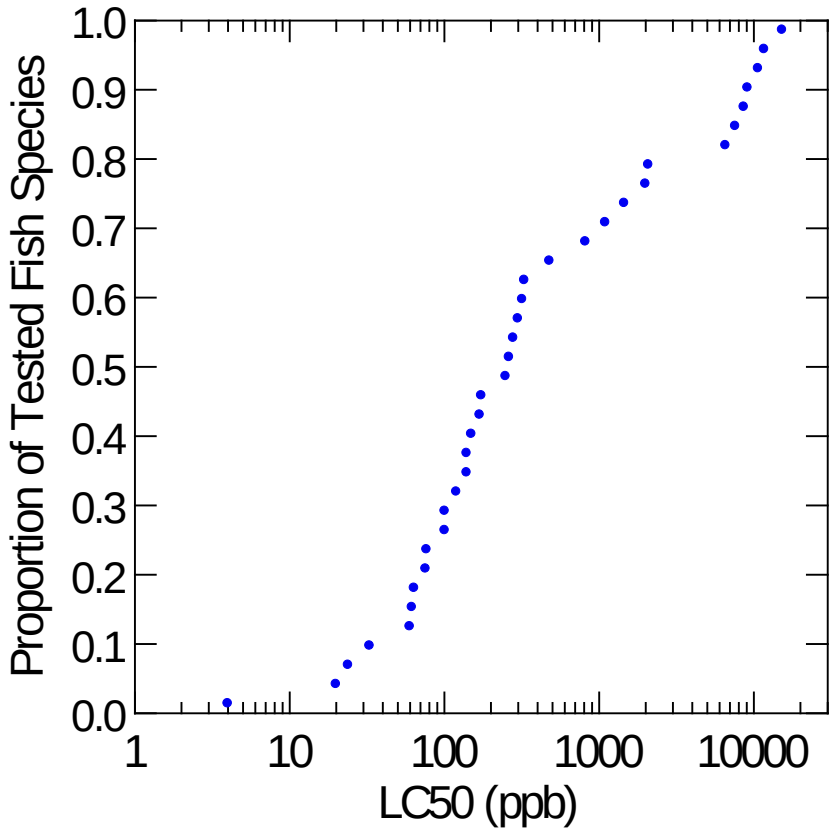

Figure 3-7. Acute toxicity of malathion to freshwater and saltwater fish species

An analysis of the relative toxicity of malathion to taxonomic families (Macek and McAllister, 1970) determined that the least susceptible families include catfish and minnows, and the more susceptible families include trout, salmon, perch, and sunfish.

Several acute sublethal and chronic laboratory toxicity studies are available for malathion using freshwater and saltwater fish species.

Beyers and Sikoski (1994) determined a cholinesterase inhibition based NOEC of 371 µg/L during a 24-hour exposure to the federally listed Colorado squawfish, (*Ptychocheilus lucius*). In another study, Beyers et al. (1994) determined the acute 96-hour NOEC for *P. lucius* and *G. elegans* for growth to be 1680 µg/L and 990 µg/L, respectively, for each species. Beauvais et al. (2000) noted changes in four measured swimming responses of rainbow trout after exposure to 20 and 40 µg/L malathion during 24- and 96-hour exposures. Lower test concentrations were not tested; therefore, no NOEC could be determined. These effects were correlated with cholinesterase inhibition that was detected during the study. Richmonds and Dutta (1992) measured cholinesterase activity in bluegill during a 24-hour exposure and determined the NOEC and LOEC to be 8.0 and 16 µg/L, respectively, based on a statistically significant inhibition of brain cholinesterase activity. In another acute sublethal exposure study, Cook et al. (2005) exposed zebrafish embryos for 120 hours to a range of malathion concentrations (0.5–3.0 mg/L) and measured survival, hatching, body length, and eye diameter. Concentrations where each response was not statistically significant were 2.0, 2.0, 1.5, and 0.5 mg/L for survival, hatching, body length, and eye diameter, respectively. Eye diameter effects were also noted in the solvent control.

In a 97-day continuous exposure study using the rainbow trout, the NOEC was determined to be 21 µg/L, while the LOEC was 44 µg/L (EPA, 2006). In another chronic study, the flagfish was exposed during a 110-day period with a resulting NOEC value of 8.6 µg/L (EPA, 2006). In a review of reproductive and behavioral studies conducted with malathion, EPA reported a reproductive NOEC of 20 µg/L for the bluegill after an 8-week exposure, based on effects to adult survival and egg production. Spinal deformations were also observed at several concentrations with a reported MATC of 3.6 to 7.4 µg/L. In another study review by EPA, sheepshead minnow embryos were exposed to a range of malathion concentrations to determine the potential for abnormal swimming behavior associated with skeletal malformations. Effects were seen at 3 mg/L and 10 mg/L, with a resulting NOEC of 1.0 ppm (EPA, 2006).

3. Aquatic Plants

Based on a review of the literature and available databases, such as ECOTOX, the green algae *Pseudokirchneriella subcapitata* is the most sensitive aquatic plant with a reported EC_{50} of 2040 ppb and a corresponding NOEC of 500 ppb (Yeh and Chin, 2006). The most tolerant species is the blue green algae *Nostoc calcicola*, with a NOEC of 200,000 ppb and no reported EC_{50} value (Piri and Ordog, 1999). Premazzi (1984) provides summaries of two studies where phytoplankton dosed at 1 mg/L of malathion had a 7% decrease on C^{14} fixation; however, no other effects were reported, and it is unknown whether the decrease was statistically significant. Moore (1970) reported a NOEC of 1.45 mg/L based on percent inhibition of growth in *Euglena gracilis*. Studies with malathion and the aquatic macrophyte: *Spirodela polyrhiza* (large duckweed) report a NOEC of 24,065 ppb (Whothley and Schott, 1973 cited in US FS, 2008b). Tagatz et al. (1974)

reported no effects to *Juncus* spp. after applications of ULV malathion at 57 g/ha applied three times biweekly. Based on the lack of toxicity to terrestrial plants at rates much higher than those proposed in the program, toxic effects to aquatic plants would not be expected to occur from program applications of malathion.

4. Aquatic Invertebrates

Malathion is moderately to very highly toxic to most aquatic invertebrates on an acute basis, depending on the sensitivity of the species. The median lethal concentration of malathion ranges from 0.5 µg/L in the scud (Mayer and Ellersieck, 1986) to greater than 130 mg/L in freshwater snails and mussels (Keller and Ruessler, 1997; Tchounwou et al., 1991) (figure 3–8, appendix A). Amphipods and cladocerans are the most sensitive group of aquatic invertebrates. Aquatic insect toxicty ranges from 0.69 µg/L for the stonefly nymph, to 385 µg/L in snipe fly larvae (Mayer and Ellersieck, 1986).

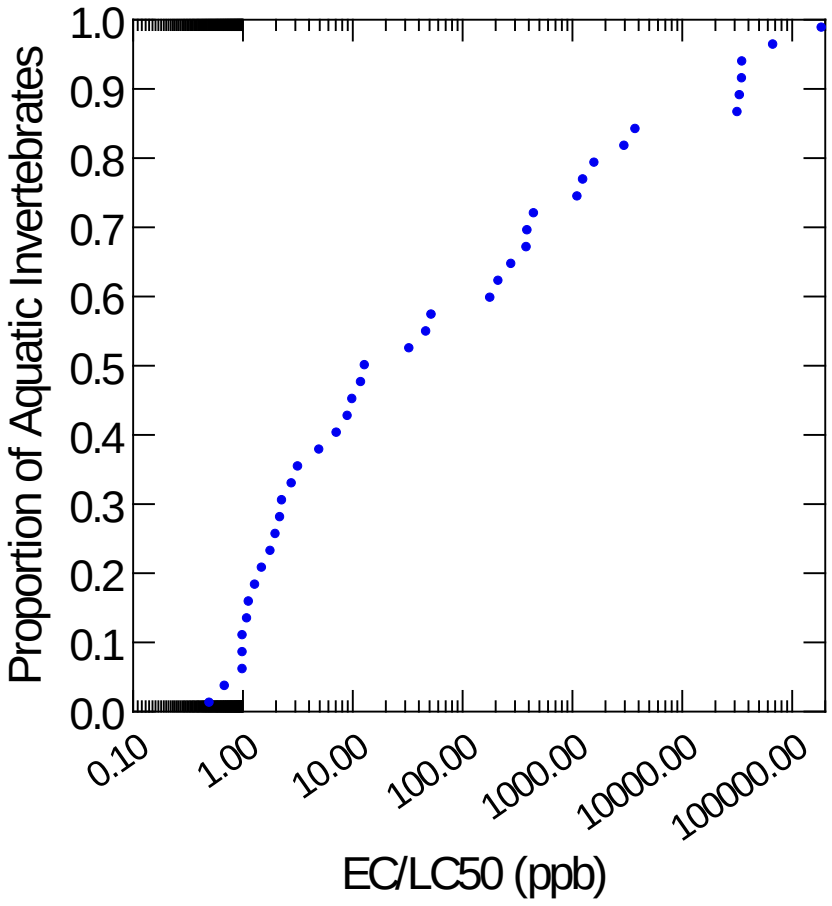

Figure 3–8. Acute aquatic invertebrate toxicity distribution for malathion

Snell and Persoone (1989) reported 24-hour NOEC values of 11.4 and 22.9 mg/L for the rotifers, *B. plicatilis* and *B. rubens*, respectively. Desi et al. (1976) showed reduced shell closing activity for a freshwater mussel, *Andonta cygnea*, during 48-hour exposure to malathion at 10,000 µg/L,

and no change was noted at 1000 µg/L or less. In a 7-day static test using *D. magna*, the reported NOEC was 1.0 µg/L (Desi et al., 1976). Reported NOEC values for *C. tentans*, based on mortality and AChE activity, are 320 and 0.26 µg/L, based on 9-day and 24-hour exposures. Relyea (2005) reported NOEC values of 320 µg/L, based on effects on dragonfly and giant water bug populations after dosing with malathion. In a 21-day continuous exposure study using *D. magna* the reported NOEC was 0.06 µg/L, while the reported LOEC was 0.10 µg/L (EPA, 2006).

5. Aquatic Toxicity of Formulations and Metabolites

Several formulation-related studies have been conducted using malathion; however, little data appears to be available for the formulation proposed for use in suppressing grasshoppers (table 3–5). Limited data for Fyfanon shows acute toxicity to aquatic invertebrates to be within the range of acute toxicity values for the technical active ingredient. Formulation-related differences in toxicity when compared to the technical active ingredient are expected to be minor since the formulation proposed in the program, Fyfanon ULV, is composed of 96.5% malathion and contains a relatively minor quantity of other ingredients. Information from the material safety data sheet does not list what the remaining ingredients are in the formulation. Based on the available data for other formulations of malathion, sublethal and lethal acute toxicity appears to be within the range reported for aquatic studies conducted using the technical material.

Table 3–5. Malathion Aquatic Toxicity Values for the Typical End Use Product (TEP)

Test Organism	Length/Endpoint	% AI	Toxicity Value	Reference
Rainbow trout	7–10 d 25% reduction in Brain AChE	55 EC	175 µg/L	Post and Leasure, 1974
Brook trout	7–10 d 25% reduction in Brain AChE	55 EC	120 µg/L	Post and Leasure, 1974
Coho salmon	7–10 d 25% reduction in Brain AChE	55 EC	300 µg/L	Post and Leasure, 1974
Chinook salmon	96-hour LC_{50}	500 EC	120 µg/L	Parkhurst and Johnson, 1955
Sheepshead minnow	96-hour LC_{50}	57 EC	55 µg/L	EPA, 2006
Bluegill sunfish	96-hour LC_{50}	57 EC	25 µg/L	Pickering et al., 1962
Fathead minnow	96-hour LC_{50}	57 EC	190 µg/L	Pickering et al., 1962
Mummichog *Fundulus heteroclitus*	96-hour EC_{50}	50 EC	22.51 µg/L	Trim, 1987
Daphnia magna	48-hour EC_{50}	25 NR	3.0 µg/L	Rassoulzadegan and Akyurtlakli, 2002
Daphnia magna	48-hour EC_{50}	57 EC	2.2 µg/L	EPA, 2006
Culex fatigens	48-hour EC_{50}	57 EC	450 µg/L	Azmi et al., 1998
Eastern Oyster *Crassostrea virginica*	96-hour EC_{50}	57 EC	2960 µg/L	EPA, 2006
Anisops sardeus	48-hour LC_{50}	NR^{+}	42.2 µg/L	Lahr et al., 2001
Fairy shrimp *Streptocephalus sudanicus*	48-hour LC_{50}	NR^{+}	67,750 µg/L	Lahr et al., 2001

EC = Emulsifiable Concentrate; NR = Not reported; NR^{+} = Percent ai not reported but a Fyfanon formulation was tested.

Several metabolites of malathion can occur in aquatic environments. EPA (2006) provides a summary of a study where the fathead minnow was used to determine the relative toxicity of several known and proposed hydrolytic metabolites of malathion. Using the fathead minnow 96-hour LC_{50} (8.65 mg/L), this value was compared to the threshold level value (TLm) for each of the metabolites (table 3–6).

Table 3–6. Toxicity of Hydrolytic Metabolites of Malathion to the Fathead Minnow

Metabolite	96-hour TLm (mg/L)
Dimethylphosphorodithioic acid	23.5
Diethyl fumarate	4.5
2-mecaptodiethyl succinate	35.0
Dimethylphosphorothionic acid	42.5
Maleic acid	5.0
Diethyl maleate	18.0
Dimethyl phosphate	18.0
Thioglycolic acid	30.0
Dimethyl phosphate	225.0
Diethyl succinate	140.0
Diethyl dl-tartarate	650.0
Bis(hydroxymethyl) phosphinic acid	29.0
Ethylene phosphate	34.0

With the exception of diethyl fumarate and maleic acid, all metabolites were less toxic to the fathead minnow when compared to malathion. Confidence intervals were not presented but, based on the similarity of the malathion, diethyl fumarate, and maleic acid values, they are not expected to be statistically significant from the parent toxicity value. Bender and Westman (1978) conducted 96-hour LC_{50} studies using the eastern mudminnow, *Umbra pygmaea*, to test the acute toxicity of malathion, diethyl fumarate, dimethyl-phosphorodithioic acid, 2-mercaptodiethyl succinate, and dimethylphosphorothionic acid. Results from the study demonstrated the parent compound to be the most toxic with reported LC_{50} values of 0.24, 8.50, 17.00, 47.00, and 26.04 mg/L, respectively.

Another metabolite that can form in aquatic systems is malaoxon. Available aquatic toxicity data show that malaoxon is approximately 1.5 to 6 times more toxic to fish and 1.8 to 93 times more toxic to amphibians (table 3–7). The conversion of malathion to malaoxon in aquatic environments can range from approximately 1.8 to 10% (Bavcon et al., 2005; EPA, 2007; CDPR, 1993). Little data appears to exist for malaoxon toxicity to aquatic invertebrates. The estimated 24-hour EC_{50} maloxon value for *C. tentans* is 5.4 µg/L. Comparable values using *Chironomus sp.* and malathion provide a range of results with values ranging from 1.9 to 4.12 µg/L, suggesting similar or slightly less toxicity than the parent (EPA, 2007). This comparison has a great deal of uncertainty because it is based on one test species and multiple studies where the exact methods are unknown. It is assumed that malaoxon is most likely more toxic to aquatic invertebrates than the parent; however, due to its low percentage of occurrence in aquatic

systems and its rapid breakdown, it is not anticipated to pose a greater aquatic risk compared to malathion.

Table 3–7. Malaoxon Toxicity to Aquatic Organisms

Test Organism	Endpoint/ Length	Toxicity Value	Malathion Value	Reference
Common carp *Cyprinus carpio*	48-hour LC_{50}	1600 µg/L	2,100 µg/L	EPA, 2007
Killifish *Oryzias latipes*	48-hour LC_{50}	280 µg/L	1,800 µg/L	Tsuda et al., 1997
African clawed frog *Xenopus laevis*	96-hour EC_{50}	180 µg/L	330 µg/L	Snawder and Chambers, 1989
Foothill yellow-legged frog *Rana boylii*	96-hour LC_{50}	2.3 µg/L	2,137 µg/L	Sparling and Fellers, 2007
Chironomus riparius	24-hour EC_{50}	5.4 µg/L	NA	EPA, 2007

IV. Environmental Risk Assessment for Rangeland Grasshopper Suppression Program—Insecticides

A. Insecticide Risk Assessment Methodology

The goal of this section is to discuss the relationship between the chemical response data discussed in section III with the exposure concentrations that were estimated for all program insecticides and application methods in section II. The integration of the exposure and response analysis for each chemical characterizes the potential risk that could occur to listed salmonids and their designated critical habitat. In cases where the range of response data for each insecticide does not fall within the range of potential exposure values APHIS concludes that potential impacts to individuals and populations of listed salmonid and their critical habitat is negligible and can not be meaningfully measured. Further evaluation of the assumptions used in the risk characterization is required to refine the risk where residues exceed the response data. For this assessment, direct risk to listed salmonids is defined as acute or chronic toxicity to fish based on the available effects data. Indirect risk is defined as any impacts to salmonid prey items, and vegetation that may serve as habitat or provide a food source for invertebrates that serve as prey for salmonids. Impacts to listed salmonids and primary constituent elements (PCE) of critical habitat are characterized below and are based on available response data that could have the potential to impact breeding, feeding and sheltering of listed salmonids.

B. Risk Characterization

1. Carbaryl

a. Direct and Indirect Risk to Fish

Comparison of the distribution of acute and sublethal and chronic effects data for fish to the residues estimated under different applications demonstrates that the range of residues are below the range of response data that was discussed in section III of this BA (figure 4-1). The lack of overlap between the response data for fish and the residues that were estimated using the available drift models suggests that direct acute and sublethal risk to fish in small, static waterbodies is not expected. Effects from consumption of contaminated prey are also not expected to be a significant pathway of exposure, based on the low residues and low BCF values reported for carbaryl.

Indirect risk to listed fish species can occur through the loss of habitat or reduction in prey base. To determine potential habitat loss from carbaryl applications, the most sensitive aquatic plant endpoint was used as a benchmark to compare to estimated aquatic residues that would be expected from aerial ULV and bait applications as well as ground applications. Several aquatic plant toxicity values are available for carbaryl; however, the most sensitive species was the green algae *Pseudokirchneriella subcapitata* that had a reported NOEC value of 0.37 mg/L. Comparing the NOEC to the aquatic residue range estimated for all application methods of carbaryl (0.23- 0.82 µg/L) resulted in residues that are greater than 451 to 1608 times below the

threshold NOEC value. This suggests that carbaryl risk to aquatic plants that may serve as habitat or food for fish and aquatic invertebrates is very low.

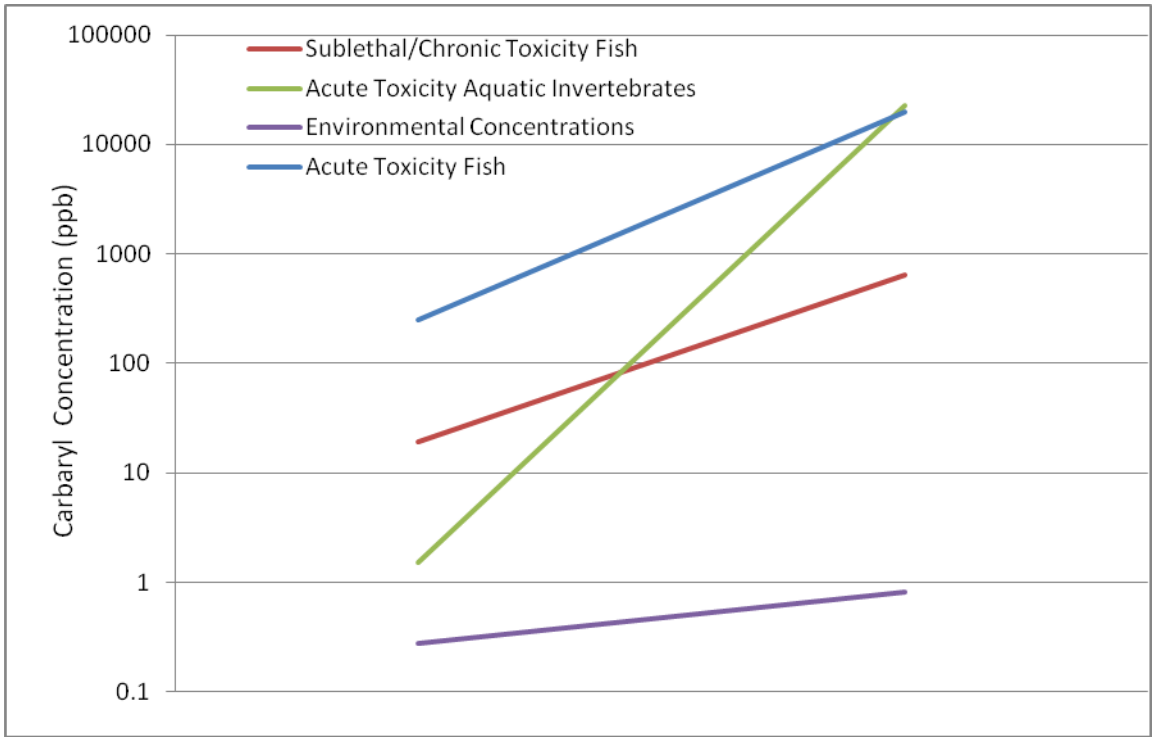

Figure 4-1. Carbaryl risk characterization for direct and indirect effects to listed salmonids

Indirect effects to fish can also occur in situations where there is a reduction in prey base, primarily through impacts to aquatic invertebrates and fish. Based on the distribution of available fish and aquatic invertebrate toxicity data for carbaryl and the estimated residues, risk to prey items for listed salmonids is not expected under the different application scenarios described in this BA. Based on a similar distribution of senstitivities, NMFS calculated a probability distribution plot for carbaryl and aquatic invertebrates and determined EC_{50} concentrations at the 50^{th}, 10^{th}, and 5^{th} percentiles (NMFS, 2009). The values from this exercise, when incorporating all studies, was 45.23, 2.29 and 0.98 µg/L for the 50^{th}, 10^{th}, and 5^{th} percentiles, respectively. Values were slightly higher when using geometric means 50^{th}, 10^{th}, and 5^{th} percentile values of 69.53, 4.33 and 1.97 µg/L. Residue values estimated from the various application methods of carbaryl proposed in the program are below the range of concentrations at each percentile suggesting that indirect impacts to listed salmonids from the loss of prey items is not expected.

b. Aquatic Field Studies Regarding Fish and Aquatic Invertebrates

Several aquatic field studies have been published and summarized using carbaryl to determine impacts to aquatic invertebrates and fish (US FS, 2008a; Relyea and Diecks, 2008; NMFS, 2009). The value of these studies in providing insight into aquatic community impacts from carbaryl applications is limited since all studies had dosing levels and/or frequencies much higher than what would occur from activities in this program. Select studies and their results are summarized below.

In a field study related to the grasshopper control program, applications of carbaryl were made in proximity to the Little Missouri River over a two-year period and impacts to fish and aquatic invertebrates assessed (Beyers et al., 1995). Measured carbaryl concentrations were 85.1 ppb in a drought year and 12.0 ppb in a nondrought year 1 hour after application. Brain cholinesterase was measured in the fathead chub (*Platygobio gracilis*) in a drought and non-drought year after applications of Sevin-4-Oil® for the control of rangeland grasshoppers. No effects were seen on brain cholinesterase activity for either season when compared to chubs from the reference site. Invertebrate sampling resulted in an increase in the coefficient of variation in invertebrate drift 3-hr after treatment at a measured concentration of 12.3 µg/L 4-hr post treatment. The increase in variability was not observed after that sampling event, and concentrations of carbaryl decreased to 0.100 µg/L 96-hr post treatment. No impacts in invertebrate drift were noted in the second year of application where carbaryl concentrations of 12.6 µg/L were measured 2-hr post treatment. It should be noted that the residues measured in this study are not based on current methods of carbaryl liquid applications and do not incorporate current rates and program application restrictions.

Courtemanch and Gibbs (1980) reported similar impacts on invertebrate drift in field studies after direct application of Sevin-4-Oil® to streams. Residues were not measured; however, correlations to other studies in the manuscript suggest aquatic residues of 26 to 42 µg/L caused the increase in drift, which is well above residues predicted from program applications. In another field study that assessed brain cholinesterase levels after carbaryl treatment, Haines (1981) noted a depression in brook trout cholinesterase activity when Sevin-4-Oil® was applied at 1 lb a.i./ac in a forestry application in Maine. Similar results have been seen in other field studies, with brook trout AChE depression following 1 lb/acre treatments. Due to the rapid reversibility associated with carbaryl, AChE levels returned to normal within 48 hours (Hurlbert, 1978). In another field study a split application of Sevin-2-Oil®, at 280 g/ha for each application was used to evaluate impacts to brook trout and slimy sculpin as well as aquatic invertebrates (Holmes et al, 1981). Maximum measured residues were 313.7 and 122.6 µg/L after each application and declined to less than 1 µg/L after 10 days. Invertebrate drift was impacted however overall impacts to aquatic invertebrates was reported as negligible and stomach contents from both fish species demonstrated no reductions in food availability.

The effects measured in the above studies are difficult to extrapolate and apply to conditions in the current program. While sublethal effects have been noted in fish with depressed AChE as well as some impacts to invertebrates in the field due to carbaryl, the application rates and measured aquatic residues where it was observed in these studies is well above values that would be expected from current program operations.

2. Diflubenzuron

a. Direct and Indirect Risk to Fish

Characterization of risk to listed salmonids from diflubenzuron applications were made by comparing the residue values in the exposure analysis from ground and aerial applications to the distribution of available acute and chronic fish toxicity data (figure 4-2). Residue values were below the distribution of acute and chronic response data, suggesting that direct risk to listed salmonids is not expected from diflubenzuron applications as proposed in this assessment.

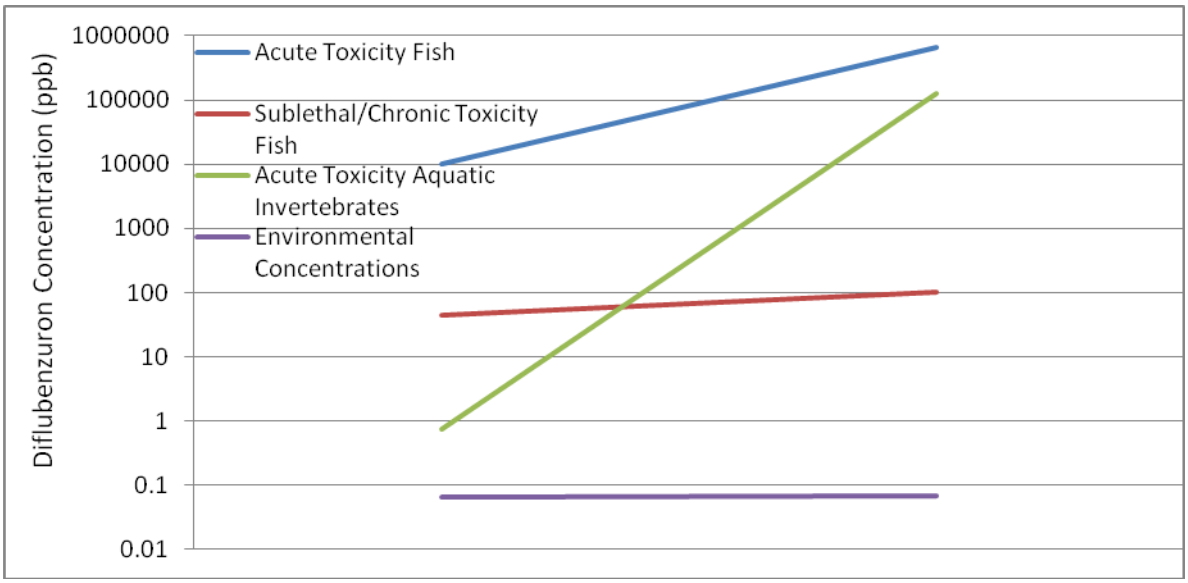

Figure 4-2. Diflubenzuron risk characterization for direct and indirect effects to listed salmonids

Indirect risk to fish species can be defined as a loss of habitat or prey base that provides food and shelter for fish populations. To determine the potential impacts of diflubenzuron applications on habitat loss through effects to aquatic plants, the most sensitive plant species (*Lemna minor*) was used as a benchmark endpoint for protection of aquatic habitat. The NOEC concentration in the 5-day exposure study was 190 µg/L. Residues from ground and aerial applications are greater than 2000 times below the NOEC concentration for aquatic plants, suggesting that risk to aquatic plants that serve as habitat or as a food source to salmonid prey is not expected.

Indirect impacts to listed salmonids through the loss of prey items are also not expected based on the available fish and invertebrate toxicity data. As previously mentioned, the fish toxicity data is well above the estimated residues from the drift analysis, and the distribution of aquatic invertebrate toxicity data is also above the residues estimated from ground and aerial applications of diflubenzuron. Risk from the consumption of contaminated prey is not expected based on the low BCF values that have been reported for diflubenzuron.

b. Aquatic Field Studies Regarding Fish and Aquatic Invertebrates

The laboratory variability in sensitivities to diflubenzuron is supported by several field studies that have assessed the impacts of diflubenzuron in different aquatic habitats. A review of several aquatic field studies demonstrated that when effects were observed it was at diflubenzuron levels not expected from program activities (Fischer and Hall, 1993; Eisler, 2000; US FS, 2004; EPA, 1997). While these studies may have limited use due to the study design and relevance to the program, they can provide support to laboratory results and insight into ecosystem level impacts that would not be observed in standard laboratory toxicity studies.

Ali and Mulla (1978b) tested a formulation of diflubenzuron and found that crustaceans, such as cladocerans and copepods, were the most sensitive taxa after two applications to a lake at a rate of 156 g a.i./ha. In addition, mayfly nymphs were severely reduced which supports other ecosystem-type exposure studies testing the effects of diflubenzuron. Mayfly nymphs were reduced after continous applications of diflubenzuron in laboratory streams over a 5-month period (Hansen and Garton, 1982b). Mayfly nymphs within the genera *Baetis*, *Rithrogena*, *Paralepthophlebia*, and *Ephemerella* were the most sensitive. Coleoptera (family Elmidae), Oligochaeata, and Gastropoda numbers were not affected at the highest test concentration (10 µg/L). The same trend was also observed in other flowing water ecosystems where diflubenzuron application rates of 0.4 to 0.8 oz a.i./acre reduced numbers of dipterans, as well as cladocerans, copepods, mayfly nymphs, corixids, and springtails (Eisler, 1992). Cladocerans and certain aquatic hemipterans have also been shown to be the most sensitive organisms in dosing studies in ephemeral pools (Lahr, 1998). In freshwater lakes, ponds, and marshes, the types of invertebrates most susceptible to diflubenzuron are amphipods (scuds), cladocerans, some midges, caddisflies, and mayflies (Ali and Mulla, 1978a, b; Apperson et al., 1978; Fischer and Hall, 1992; Hansen and Garton, 1982b; Sundaram et al., 1991). In particular, cladocerans (*Daphnia* sp.) and caddisflies (*Clistoronia* sp.) are at high risk of adverse effects from full coverage applications of diflubenzuron. Mayflies (*Callibaetis* sp.), amphipods (*Gammarus* sp.), and some midges (*Tanytarsus* sp.) are at moderate risk. Dragonfly larvae, stonefly larvae, aquatic beetles, crayfish, bivalves, chironomid midges, and snails are at low risk. Recovery of invertebrate taxa affected by diflubenzuron at a dose of 10 µg/L has been observed in outdoor pond studies during the duration of the study while other taxa may take longer (Ali and Kok Yokomi, 1989).

Several studies are available which assessed the direct effects of diflubenzuron to invertebrates, while comparatively few exist which assess effects to fish. Tanner and Moffett (1995) noted effects on fish growth at diflubenzuron levels as low as 2.5 10 µg/L, while ponds directly treated with diflubenzuron at a concentration of 5 or 13 µg/L did not show any effects on fish growth (Apperson et al., 1978; Colwell and Schaefer, 1980). A shift in diet was noted by Colwell and Schaefer (1980); however, this did not translate into an effect on growth in fish. Boyle et al. (1996) noted diflubenzuron-related impacts to some aquatic invertebrates indirectly resulting in increased algal biomass in outdoor micorocosm dosed bi-weekly or monthly at 10 µg/L. These reductions did not result in indirect impacts to bluegill and largemouth bass.

3. Malathion

a. Direct and Indirect Risk to Fish

Available acute and chronic effects data for malathion and fish were above the estimated aquatic concentrations for ground and aerial applications (figure 4-3). Examples of endpoints evaluated in both short- and long- term studies consisted of reproductive parameters, cholinesterase inhibition, swimming behavior, skeletal malformations, and eye diameter. The range of available toxicity data above the estimated exposure values suggests that direct acute and chronic risk to listed salmonids from malathion is not expected. Consumption of contaminated prey is not expected to be a significant pathway of exposure for salmonids based on expected residues and the low BCF.

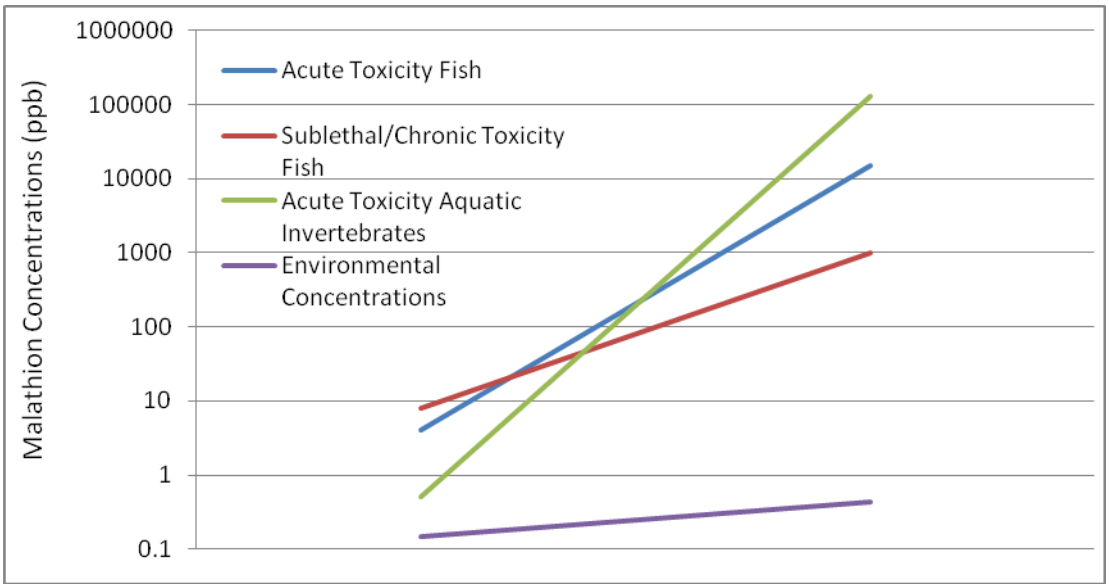

Figure 4-3. Malathion risk characterization for direct and indirect effects to listed salmonids.

To address indirect risk of malathion applications to fish habitat, estimated residues were compared to the lowest available aquatic plant toxicity value. Toxicity to plants, including algae, could result in indirect effects to habitat and food for fish and aquatic invertebrates. Using the lowest reported laboratory NOEC value, the benchmark effects level for aquatic plants was 500 µg/L, which is well above the estimated environmental concentration from aerial and ground applications of malathion. Estimated residues were greater than 1,000 times below the aquatic plant NOEC for aerial and ground applications, respectively. Indirect risk to aquatic plants from the proposed malathion applications is not expected.

The other area of potential indirect risk is the impact of malathion on prey items used by listed salmonids. Comparison of available acute fish and aquatic invertebrate toxicity distribution data

to the residues estimated from ground and aerial malathion applications demonstrates that estimated residues are not expected to result in impacts to aquatic prey items for salmonids.

b. Aquatic Field Studies Regarding Fish and Aquatic Invertebrates

EPA (2006) provides a review of two field studies in which multiple malathion applications were made over water for mosquito control, and effects to fish were monitored in estuarine environments. Mortality and AChE inhibition were noted in both studies; however, these results have limited use in assessing risk from program-related malathion applications. In another EPA study review, four malathion applications were made to freshwater ponds containing bluegill over an 11-week period. Reduction in bluegill populations were attributed to a loss of aquatic invertebrates at 0.02 and 0.002 mg/L, which is above levels predicted from program activities. In another review, malathion applications were made within 25 ft of a creek in Alabama and monitored for aquatic invertebrate and fish effects over a three-year period. A slight reduction in AChE was noted in fish collected at the area of application; however, there were no effects on the population during the study. There were some differences in the abundance of invertebrate taxa, but the authors could not attribute the differences to malathion applications. Relyea and Diecks (2008) observed sublethal impacts to amphibians due to the loss of aquatic invertebrates in an outdoor field microcosm study. Dosing occurred weekly for 7 weeks at 10 µg/L, with additional doses of 50 and 250 µg/L in some cases. Dosing levels and frequency of dosing exceed those expected from proposed applications in this program.

V. Effects Determination for Listed Salmonids

The purpose of this section of the BA is to integrate the exposure, response, and risk characterization phases of the document to support the effects determination for each ESU. The synthesis of this information with the ecology of the species allows APHIS to make effects determinations for listed species under the jurisdiction of NMFS within the action area where the program may make an insecticide application. Due to the fact that APHIS cannot at this time predict with reasonable certainty where an application may occur within the action area, APHIS assumes that there could be the likelihood of insecticide application at the same location as a listed salmonid. While this assumption may be more likely for some listed salmonids than others based on their life histories, it allows effects determinations to be developed for all listed salmonids, and where appropriate mitigation measures have been developed that are integrated into the local, site-specific treatments and consultations. Currently, APHIS staff at the State level contact the local NMFS office for informal consultation. Variability occurs from State to State regarding the consultation process for this program. Thus, it is the intent of this BA to provide a scientifically sound and legally defensible basis for the use of mitigation measures to ensure that listed salmonids and their designated critical habitat are conserved in the program area. This BA also provides consistency and uniformity for future consultations by establishing program buffers to protect listed salmonids and critical habitat. APHIS will continue to work closely with NMFS staff to determine any change in status for currently listed salmonids, as well as to assess program impacts to any recently listed species.

A. Species Descriptions

For those species where program activities and a salmonid ESU occur at the same location, the information below provides a brief summary of each species, with general information regarding life history and other species information is provided. Maps of each ESU are available at the NMFS Northwest Regional Office Web site at http://www.nwr.noaa.gov/.

1. Chinook Salmon Life History and Description

Chinook salmon belong to the family Salmonidae and are 1 of 8 species of Pacific salmonids in the genus *Oncorhynchus*. Chinook salmon are easily the largest of any salmon with adults often exceeding 40 lb; some fish have been reported to exceed 120 lb. Chinook salmon are very similar to coho salmon in appearance while at sea (blue-green back with silver flanks), except for their large size, small black spots on both lobes of the tail, and black pigment along the base of the teeth. Chinook salmon are anadromous and semelparous.

Adult female chinook will prepare a spawning bed, called a redd, in a stream area with suitable gravel composition, water depth, and velocity. The adult female chinook may deposit eggs in four to five nesting pockets within a single redd. After laying eggs in a redd, adult chinook will guard the redd from 4 to 25 days before dying. Eggs are deposited at a time to ensure that young salmon fry emerge during the following spring when the river or estuary productivity is sufficient for juvenile survival and growth. Juvenile chinook may spend from 3 months to 2 years in freshwater after emergence and before migrating to estuarine areas as smolts, and then into the ocean to feed and mature.

Juveniles feed on plankton initially, then later on insects and small fish. Coastwide, chinook salmon remain at sea for 1 to 6 years (more commonly 2 to 4 years), with the exception of a small proportion of yearling males (called jack salmon) which mature in freshwater or return after 2 or 3 months in saltwater. Adults do not feed during the freshwater spawning migration.

a. Upper Columbia River Spring-run Chinook Salmon, *Oncorhynchus shawytscha*

Status: The Upper Columbia River spring-run chinook salmon was listed as an endangered species on March 24, 1999 (64 *Federal Register* (FR) 14308). Critical habitat was designated on February 16, 2000 (65 FR 7764).

Pertinent Species Information: This ESU includes stream-type chinook salmon spawning above Rock Island Dam—that is, those in the Wenatchee, Entiat, and Methow Rivers, as well as the Columbia River and estuary. Designated habitat includes all river reaches accessible to chinook salmon in Columbia River tributaries, upstream of the Rock Island Dam and downstream of Chief Joseph Dam in Washington State, excluding the Okanogan River. Counties in Washington include Chelan, Douglas, Okanogan, Grant, Kittitas, Benton, Franklin, Yakima, Klickitat, Walla Walla, Skamania, Clark, Cowlitz, and Wahkiakum. In Oregon, critical habitat is found in the counties of Gilliam, Morrow, Sherman, Umatilla, Hood River, Wasco, Multnomah, Clatsop, and Columbia.

Factors contributing to the decline of the Upper Columbia River spring-run chinook salmon include Columbia River hydroelectric development. This resulted in a major disruption of migration corridors, and affected flow regimes and estuarine habitat. Some populations in this ESU must migrate through nine mainstem dams. Access to a substantial portion of historical habitat was blocked by Chief Joseph and Grand Coulee Dams. There are local habitat problems related to irrigation diversions and hydroelectric development, as well as degraded riparian and instream habitat from urbanization and livestock grazing.

b. Snake River Fall Chinook Salmon, *Oncorhynchus tshawytscha*

Status: The Snake River fall Chinook salmon was listed as a threatened species on April 22, 1992 (57 FR 14653). This status was reclassified to endangered by an emergency interim rule on August 18, 1994 (59 FR 42529). Critical habitat for the Snake River fall Chinook salmon was designated on December 28, 1993 (58 FR 68543).

Pertinent Species Information: From the Pacific Ocean, Snake River fall salmon enter the Columbia River and travel upstream about 324 miles (520 kilometers (km)) to the Snake River. The majority of spawning is in the mainstream Snake River, from the upper extent of Lower Granite Dam pool to Hells Canyon Dam. Spawning also occurs in the lower reaches of the Imnaha, Grande Ronde, Clearwater, and Tucannon Rivers.

Snake River fall Chinook salmon have declined to low numbers of fish that are thinly spread over a large and complex river system. Hydropower development, water withdrawal and diversions, water storage, harvest, inadequate regulatory mechanisms, and artificial propagation are factors contributing to the decline, and represent a continued threat to the Snake River fall Chinook salmon.

c. Snake River Spring/Summer Chinook Salmon, *Oncorhychus tahawytscha*

Status: The Snake River spring/summer Chinook salmon was listed as a threatened species on April 22, 1992 (57 FR 14653). This status was reclassified to endangered by an emergency interim rule on August 18, 1994 (59 FR 14653). Critical habitat for the Snake River spring/summer Chinook salmon was designated on December 28, 1993 (58 FR 68543).

Pertinent Species Information: From the Pacific Ocean, Snake River Spring/summer Chinook salmon enter the Columbia River and travel upstream about 324 miles (520 km) to the Snake River. The Snake River contains five principal subbasins that currently produce spring and/or summer-run Chinook. Three of the five subbasins, the Clearwater, Grande Ronde, and Salmon Rivers, are large, complex systems; the other two, the Tucannon and Imnaha Rivers, are small systems in which the majority of salmon production is in the mainstream rivers. The Asotin, Granite, and Sheep Creeks are small streams that enter the Snake River and provide small spawning and rearing areas.

Snake River spring/summer Chinook salmon declined to low numbers thinly spread over a large and complex river system. Hydropower development, water withdrawal and diversions, water storage, harvest, inadequate regulatory mechanisms, and artificial propagation are factors

contributing to their decline and represent a continued threat to the Snake River spring/summer Chinook salmon's existence.

d. Columbia River Chum Salmon, *Oncorhynchus keta*

Status: The Columbia River chum salmon was listed as a threatened species on March 25, 1999 (64 FR 14508). Critical habitat was designated on February 16, 2000 (65 FR 7764).

Pertinent Species Information: Chum salmon belong to the family Salmonidae and are one of eight species of Pacific salmonids in the genus *Oncorhynchus*. Chum salmon are anadromous, semelparous, and spawn primarily in freshwater. They have the widest natural geographic and spawning distribution of any Pacific salmonid, primarily because its range extends farther along the shores of the Arctic Ocean than that of the other salmonids. Juvenile chum salmon are distinguished by parr marks of relatively regular height that are smaller than the vertical diameter of the eye, and that are faint or absent below the lateral. Adult chum salmon have greenish to dusky mottling on the sides, with males exhibiting distinctive reddish-purple vertical barring. Adult chum salmon in Washington State range in size from 17 to 38 in, with an average weight of 9 to 11 lb. Chum salmon spawn in the lowermost reaches of rivers and streams, typically within 100 km of the ocean.

This ESU includes all naturally spawned populations of chum salmon in the Columbia River and its tributaries in Washington and Oregon. Critical habitat for the Columbia River chum salmon is designated to include all river reaches accessible to listed chum salmon (including estuarine areas and tributaries) in the Columbia River downstream from Bonneville Dam, excluding Oregon tributaries upstream of Milton Creek at river km 144 near the town of St. Helens. Also included are adjacent riparian zones. The following counties lie partially or wholly within these basins (or contain migration habitat for the species): Oregon—Clatsop, Columbia, Multnomah, and Washington; Washington—Clark, Cowlitz, Lewis, Pacific, Skamania, and Wahkiakum.

The decline of the Columbia River chum salmon is due to habitat loss and overfishing. Habitat loss is due to channel excavations, dewatering, channelization, flood control, major water diversions, poor forestry practices, and bulkheading of nearshore marine habitats.

e. Snake River Sockeye Salmon, *Oncorhynchus nerka*

Status: The Snake River sockeye salmon was listed as an endangered species on November 20, 1991 (56 FR 58619). Critical habitat was designated for the Snake River sockeye salmon on December 28, 1993 (58 FR 68543).

Pertinent Species Information: The Snake River sockeye salmon is a member of the trout family (*Salmonidae*). These Pacific salmon are anadromous, spending their adult life in the ocean and traveling into freshwater to spawn and complete their early life histories. Redfish Lake in Custer County, Idaho, supports the only remaining run of Snake River sockeye salmon.

Adult Snake River sockeye salmon usually enter Redfish Lake in August, and spawning occurs near shoreline shoals in October. Eggs hatch in the spring, and the juveniles remain in Redfish Lake for normally 2 years before migrating to the ocean. Migrants leave Redfish Lake from late

April through May. Smolts migrate almost 900 miles through the Salmon, Snake, and Columbia Rivers to the ocean where they usually spend 2 years. Adults begin their return migration to Redfish Lake in June and July in their 4^{th} or 5^{th} year of life.

Hydropower development, water withdrawal and diversions, water storage, harvest, predation, and inadequate regulatory mechanisms are factors contributing to the Snake River sockeye salmon's decline. These issues represent a continued threat to the Snake River sockeye salmon's existence.

2. Steelhead Life History and Description

Steelhead exhibit one of the most complex life histories of any salmonid species. Steelhead may exhibit anadromy or freshwater residency. Resident forms are usually referred to as rainbow trout, while anadromous life forms are termed steelhead. Like all trout, steelhead are positively separated from the various salmon species by having 8 to 12 rays in the anal fin. Steelhead are separated from brook trout, lake trout, and Dolly Varden by the complete absence of teeth at the base of the tongue. Coloration on the back is basically blue-green shading to olive, with black regularly spaced spots. The black spots also cover both lobes of the tail. Steelhead from the ocean are much more silver than resident rainbow trout. Spawning steelhead develop a distinct pink to red strip-like coloration that blends along the side, both above and below the lateral line. Juvenile steelhead trout are identical to rainbow trout until the period prior to their ocean migrations. Prior to migrating to the sea, juvenile steelhead become very silvery and resemble miniature adults. They are called smolt during this life phase. Steelhead typically migrate to marine waters after spending 2 years in freshwater. They then reside in marine waters for 2 to 3 years prior to returning to their natal stream to spawn. Within the range of west coast steelhead, spawning migrations occur throughout the year, with seasonal peaks of activity. Summer steelhead enter fresh water up to a year prior to spawning. Depending on water temperature, steelhead eggs may incubate in redds for 1 ½ to 4 months before hatching as alevins. The alevins remain within redds, living on the rich nutrients contained in the yolk sac. In 3 to 4 weeks, they emerge as fry and feed on small insects and drifting plankton. They then develop into parr (about 3 inches in length) feeding primarily on aquatic and flying insects, although small fish become an increasingly important part of their diet as they grow. Juveniles rear in freshwater from 1 to 4 years, then migrate to the ocean as smolts.

a. Upper Columbia River Steelhead, *Oncorhynchus mykiss*

Status: The Upper Columbia River steelhead was listed as an endangered species on August 18, 1997 (62 FR 43974). Critical habitat was designated on February 16, 2000 (65 FR 7764).

Pertinent Species Information: Critical habitat is designated to include all river reaches accessible to listed steelhead in Columbia River tributaries upstream of the Yakima River in Washington, and downstream of Chief Joseph Dam. Excluded are tribal lands and areas above specific dams or above long-standing, naturally impassable barriers (i.e., natural waterfalls in existence for at least several hundred years). The following counties lie partially or wholly within these basins (or contain migration habitat for the species): Oregon—Clatsop, Columbia, Gilliam, Hood River, Morrow, Multnomah, Sherman, Umatilla, and Wasco; Washington—

Benton, Chelan, Clark, Cowlitz, Douglas, Franklin, Grant, Kittitas, Klickitat, Okanogan, Pacific, Skamania, Wahkiakum, Walla Walla, and Yakima.

Habitat degradation, juvenile and adult mortality in the hydrosystem management structures, and unfavorable environmental conditions in both marine and freshwater habitats have contributed to the declines of this ESU, and represent risk factors for the future. Harvest in lower river fisheries and genetic homogenization from composite broodstock collection are other factors that may contribute significant risk to the Upper Columbia River steelhead

b. Snake River Basin Steelhead, *Oncorhynchus mykiss*

Status: The Snake River Basin steelhead was listed as a threatened species on August 18, 1997 (62 FR 43974). Critical habitat was designated on February 16, 2000 (65 FR 7764).

Pertinent Species Information: Critical habitat is designated to include all river reaches accessible to listed steelhead in the Snake River and its tributaries in Idaho, Oregon, and Washington. Excluded are tribal lands and areas above specific dams identified or above long-standing, naturally impassable barriers (i.e., Napias Creek Falls and other natural waterfalls in existence for at least several hundred years). The following counties lie partially or wholly within these basins (or contain migration habitat for the species): Idaho—Adams, Blaine, Boise, Clearwater, Custer, Idaho, Latah, Lemhi, Lewis, Nez Perce, and Valley; Oregon—Baker, Clatsop, Columbia, Hood River, Morrow, Multnomah, Sherman, Umatilla, Union, Wallowa, and Wasco; Washington—Asotin, Benton, Clark, Columbia, Cowlitz, Franklin, Garfield, Klickitat, Skamania, Wahkiakum, Walla Walla, and Whitman.

Widespread habitat blockage from hydrosystem management, potentially deleterious genetic effects from straying, and introgression from hatchery fish have contributed to the decline of Snake River Basin steelhead.

c. Middle Columbia River Steelhead, *Oncorhynchus mykiss*

Status: The Middle Columbia River steelhead was listed as a threatened species on March 25, 1999 (64 FR 14517). Critical habitat was designated on February 16, 2000 (65 FR 7764).

Pertinent Species Information: Critical habitat is designated to include all river reaches accessible to listed steelhead in Columbia River tributaries (except the Snake River) between Mosier Creek in Oregon and the Yakima River in Washington (inclusive). Excluded are tribal lands and areas above specific dams (Condit Dam and Pelton Dam), or above longstanding, naturally impassable barriers (i.e., natural waterfalls in existence for at least several hundred years). The following counties lie partially or wholly within these basins (or contain migration habitat for the species): Oregon—Clatsop, Columbia, Crook, Gilliam, Grant, Harney, Hood River, Jefferson, Morrow, Multnomah, Sherman, Umatilla, Union, Wallowa, Wasco, and Wheeler; Washington—Benton, Clark, Columbia, Cowlitz, Franklin, Kittitas, Klickitat, Pacific, Skamania, Wahkiakum, Walla Walla, and Yakima.

The recent and dramatic increase in the percentage of hatchery fish in natural escapement in the Deschutes River Basin is a significant risk to natural steelhead in this ESU.

B. Effects Determination for ESUs and Associated Critical Habitat

Based upon the lack of program activities within salmonid critical habitat, 20 of the 28 listed salmonids are assessed to be no effect determinations (table 5–1). If in the future the program expands to include areas where these ESUs are present, APHIS would re-initiate consultation with NMFS. APHIS would also re-initiate consultation in the event that new ESUs become listed or critical habitat is expanded into the action area for the program. APHIS will stay abreast of any new species or critical habitat designations and will contact NMFS if re-initiation of consultation if warranted.

Table 5-1. Effects Determination for Pacific Salmonid ESUs

Species	Location	Effects Determination
Sockeye Salmon (*Oncorrynchus nerka*)	Snake River	May affect- Not likely to adversely affect
	Ozette Lake	No effect
Chinook Salmon (*Oncorrynchus tshawytscha*)	Sacramento River Winter Run	May affect- Not likely to adversely affect
	Upper Columbia River Spring Run	May affect- Not likely to adversely affect
	Snake River Spring/Summer Run	May affect- Not likely to adversely affect
	Snake River Fall Run	May affect- Not likely to adversely affect
	Puget Sound	No effect
	Upper Willamette	No effect
	Lower Columbia River	No effect
	California Coastal	No effect
	Central Valley Spring Run	No effect
Coho Salmon (*Oncorrynchus kisutch*)	Central California Coastal	No effect
	South Oregon/N. California	No effect
	Lower Columbia River	No effect
	Oregon Coast	No effect
Chum Salmon (*Oncorrynchus keta*)	Hood Canal Summer Run	No effect
	Columbia River	No effect
Steelhead (*Oncorrynchus mykiss*)	Southern California	No effect
	Upper Columbia River	May affect- Not likely to adversely affect
	Central Coastal California	No effect
	South Central California Coast	No effect
	Snake River Basin	May affect- Not likely to adversely affect
	Lower Columbia River	No effect
	California Central Valley	No effect
	Upper Willamette River	No effect
	Middle Columbia River	May affect- Not likely to adversely affect
	Northern California	No effect
	Puget Sound	No effect

Effects determinations for those ESUs that occur within potential areas of a program insecticide treatment were listed as a may affect but not likely to adversely effect based on the risk characterization which integrates the response and exposure analysis, and implementation of program application restrictions which would result in potential effects that are considered discountable and insignificant. For the purpose of this BA, discountable was defined as those effects that are extremely unlikely to occur while insignificant effects relates to the size of the impact and should not reach the scale where take would be expected to occur. Discountable effects are determinations based on best judgment, where a person would not: (1) be able to meaningfully measure, detect, or evaluate insignificant effects; or (2) expect discountable effects to occur (FWS and NMFS, 1998).

Application buffers are one of the primary mitigation measures for reducing the potential for drift and runoff to occur in designated critical habitat for salmonids. The proposed buffers will be applied to all salmonid habitats, including any off-channel habitats when water is present.

Table 5–2. Proposed Application Buffers to Protect Listed Salmonids and Critical Habitat

Insecticide Treatment	Method of Application	Application Buffer (feet)
Carbaryl	Aerial Ultra Low Volume (ULV)	3500
	Aerial Bait	1000
	Ground ULV	350
	Ground Bait	200
Diflubenzuron	Aerial ULV	1500
	Ground ULV	150
Malathion	Aerial ULV	3500
	Ground ULV	500

In addition to the proposed application buffers, other mitigation measures are also required to further reduce the potential for exposure to listed salmonids and their designated critical habitat. These measures include:

- Avoid applications when winds speeds exceed 10 mph
- Avoid applications when wind direction is blowing towards salmonid critical habitat
- Reduced area agent treatments (RAATs) only are allowed adjacent to salmonid critical habitat
- Avoid applications under conditions where a temperature inversion is possible or when a storm event is imminent.

Due to the variability in wind speed and direction that can occur during an application the program will initiate treatments nearest the protected aquatic habitat to insure that wind mitigation measures are met for those application swaths that can contribute the greatest amount of drift to salmonid habitat. Comparison of the proposed mitigation measures in this program to the reasonable and prudent alternatives (RPAs) and reasonable and prudent measures (RPMs) in the biological opinions that contain malathion and carbaryl demonstrates expected differences in proposed mitigation measures to reduce exposure to listed salmonids (NMFS, 2008; NMFS, 2009). The intent of the biological opinion prepared by NMFS was in response to formal

consultation with the US EPA OPP regarding all registrations and uses of several insecticides including carbaryl and malathion. The protection measures proposed in this program for salmonids are based on program specific use patterns that were not considered in either biological opinion. Mitigation measures proposed in this biological assessment exceed mitigation measures proposed in the biological opinion for malathion and carbaryl. The intent of this biological assessment is to propose mitigation that would allow the program to effectively suppress grasshopper and Mormon cricket populations while not having an adverse impact on listed salmonids and their designated critical habitat. Measures proposed in the biological opinions for all malathion and carbaryl uses in addition to two other organophosphate and carbamate insecticides resulted in may effect determinations for most uses where ESUs and critical habitat were present.

Data regarding carbaryl and malathion effects attributed to direct risk to salmonids that were identified in the biological opinions were evaluated in this assessment. Where available, toxicity data regarding survival as well as sublethal responses such as olfactory and behavioral effects were included in the response analysis. Similar efforts were made to incorporate available acute sublethal and chronic fish response data for diflubenzuron. In relation to designated critical habitat, APHIS attempted to draft a risk characterization and propose mitigation measures that are protective of biological and physical features required for the conservation of salmonids that occur in potential areas where program applications could occur. These features identified in the carbamate biological opinion as primary constituent elements (PCEs) include (NMFS, 2009):

1. freshwater spawning sites with water quantity and quality conditions and substrate supporting spawning, incubation and larval development;

2. freshwater rearing sites with water quantity and floodplain connectivity to form and maintain physical habitat conditions and support juvenile growth and mobility; water quality and forage supporting juvenile development; and natural cover such as shade, submerged and overhanging large wood, log jams and beaver dams, aquatic vegetation, large rocks and boulders, side channels, and undercut banks;

3. freshwater migration corridors free of obstruction, along with water quantity and quality conditions and natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels, and undercut banks supporting juvenile and adult mobility and survival;

4. estuarine areas free of obstruction, along with water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh and saltwater; natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels; and juvenile and adult forage, including aquatic invertebrates and fishes, supporting growth and maturation;

5. nearshore marine areas free of obstruction with water quality and quantity conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation; and natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels; and

6. offshore marine areas with water quality conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation.

Emphasis was placed on PCEs listed under one through three based on the defined action area for the program. The current action area for the program is well inland and potential transport at the site of application of program insecticides to estuarine or marine areas would attenuate prior to reaching these types of habitats. Due to the complex life history of salmonids in general, they can occupy a variety of aquatic habitats throughout their range that support various life stages of development. One of the more sensitive habitat types that salmonids may occupy when relating potential chemical exposure, and was identified in both biological opinions, is shallow off-channel habitats. Both natural and constructed off-channel habitats have been shown to be important in salmonid development, in particular providing abundant invertebrate prey items to juveniles (Morley et al., 2005; Henning et al., 2006; NMFS, 2009). The water body used in the exposure analysis and risk characterization for this BA was considered representative of these types of habitats. Protection of isolated shallow waterbodies was assumed to also be protective of other aquatic habitats important for salmonids since these habitats would contain either a larger volume of water and/or flow that would result in reduced exposure. Results from the risk characterization for all three insecticides demonstrate that physical habitat as well as biological resources, such as aquatic invertebrate prey, would not be expected to be adversely impacted by program applications.

Another important food source for salmonids that should be considered in the evaluation of impacts to listed salmonids from program activites is effects to terrestrial invertebrates that serve as prey items and nitrogen inputs into salmonid habitat (Wipfli, 1997; Johnson and Ringler, 1980). Impacts to terrestrial invertebrates that occupy the riparian areas adjacent to salmonid habitat are not expected to be significantly impacted by program treatments due to the proposed mitigation measures that would be used in areas where salmonid-bearing waters occur and available published data regarding the effects of these chemicals to terrestrial invertebrate populations. Weiland et al. (2002) assessed the impacts of Sevin® XLR Plus applications at 750 g a.i./ha to several invertebrate groups within treatment blocks over a 21-day period. This rate equates to 0.67 lb ai/ac, which is above the maximum RAAT rate (0.375 lb ai/ac) proposed in the program. Results from the study demonstrated no negative effects on abundance in the following insect groups: Homoptera, Hymenoptera, Coleoptera, Hemiptera, Lepidpotera or Neuroptera. Smith et al. (2006) assessed changes in nontarget arthropod populations within treatment areas following applications of diflubenzuron, carbaryl, or malathion using RAAT treatments. In the 2-year study, post application surveys of the major insect fauna revealed that only ants were negatively affected by grasshopper applications within treatment areas. Previously conducted studies related to the grasshopper IPM project measured impacts to nontarget terrestrial invertebrates from diflubenzuron applications with minimal impact on ants, spiders, predatory beetles, and scavenger beetles reported. There was no significant reduction in populations of these species from 7 to 76 days after treatment. Although ant populations exhibited declines of up to 50%, these reductions were temporary, and population recovery was described as immediate (Catangui et al., 1996). Weiland et al. (2002) in Wyoming monitored the effects of Dimilin® 25W for 21 days post-application on terrestrial invertebrates after full treatment applications of 17.5 and 52.5 g a.i./ha. Based on high and low sweep net captures, there was no effect on invertebrates in the order Homoptera, Hymenoptera, Coleoptera, Hemiptera, Lepidoptera or Neuroptera. There was a statistically significant increase in Diptera

and a statistically significant decrease in Araneae (spiders); however, the authors question the spider analysis since untreated populations dropped dramatically during the study. Tingle (1996) assessed the impacts of diflubenzuron applications in two field trials occurring in two separate years with applications of 93 g a.i./ha (0.08 lb/ac). Based on an analysis of 28 taxomonic groupings, only two were affected and included nontarget grasshoppers and lepidopteran larvae. This effect only occurred in the sprayed areas but did not occur in the unsprayed buffer areas that were sampled.

C. Uncertainty

Several areas of uncertainty exist in characterizing the risk of program insecticides to listed salmonids and their critical habitat. Some of these uncertainties are specific for evaluating the impacts of chemicals, such as pesticides, to listed species, while other uncertainties are more general and apply to any potential action in salmonid-bearing watersheds.

The reliance on laboratory toxicity data in the effects analysis is an area of uncertainty in this assessment. Where data were available, APHIS used field collected information to validate methods and assumptions in the risk characterization to listed salmonids and their critical habitat; however, most of the response data from field studies for each insecticide had limited use since they dosed at levels and frequencies well above those proposed in this program. In using distributional toxicity data, the entire range of sensitivities was considered when evaluating the available toxicity data. For those species where multiple toxicity endpoints were available, the lowest value was selected compared to the more conventional estimate of a geometric mean.

Uncertainty in the evaluation of effects data for pesticides (e.g., toxicity of formulation, metabolites, and inert ingredients) was addressed to the extent possible based on available data. Based on the limited formulation toxicity data available for each insecticide, there was comparable toxicity to the technical active ingredient; however, a majority of those studies are based on acute exposures with lethality as the primary endpoint. Inerts are not typically listed on the label or material safety data sheet and are considered confidential business information; therefore, evaluation of the inerts individually is not possible. Of the three insecticides, uncertainty regarding formulation effects is the least for malathion since greater than 96% of the formulation is the technical active ingredient. Toxicity of metabolites is another area of uncertainty in the evaluation of pesticides. Similar to the formulation data, there is response data for the primary metabolites for each chemical as well as some secondary metabolite data which was considered in the effects analysis. As with the formulation response data, the available metabolite data primarily emphasizes acute exposure durations using lethality as an endpoint. Available metabolite response data were considered in this assessment as well as environmental fate for the primary metabolites to determine potential risk to listed salmonids.

The impacts of program activities, as they relate to other current and future activities in areas of listed salmonids are another area of uncertainty. Causes for salmonid decline, such as hydroelectric generation, acid mine drainage, climate change, habitat loss, harvesting, invasive species and other point and non-point pollution sources, are difficult to quantify in the context of the potential impacts to listed salmonids from program applications (NMFS, 2008; NMFS, 2009;

Sanderson et al., 2009). Temporal variability in the occurrence of multiple stressors, as well an understanding of their effects, is not well understood. As an example, available water quality monitoring data for areas where salmonid habitat occurs, as well as for the rest of the United States, indicate the presence of multiple natural and anthropogenic contaminants. Sources for these chemicals can occur from point and non-point sources, and the relative contribution from each is dependent on land use in a given watershed. Based on the most recent United States Geological Survey National Water Quality Assessment (USGS–NAWQA) data for pesticides, frequency of occurrence for two or more pesticides in surface water exceeds 80% nationally (USGS, 2006). When considering other organics and trace metals, the combination of mixtures that can occur can become extremely large, especially when spatial and temporal variability in mixtures that can occur in a given watershed are considered. The seasonal variability in mixtures has been well documented nationally, including areas that support listed salmonids (USGS, 1996; USGS, 2006). An analysis of all detections from agricultural streams indicated more than 6,000 unique mixtures of 5 pesticides (USGS, 2006). As would be expected, based on the large variability in mixtures, the response data for these types of exposure scenarios is very limited for all organic and inorganic chemicals including those proposed in the program. No aquatic toxicity data appears to be available regarding the interaction of diflubenzuron with other chemicals including pesticides. Some mammalian data has been generated that demonstrates additive toxicity in rats when diflubenzuron and carbon monoxide exposures occur simultaneously. There is considerably more pesticide mixture data when considering the more widely used organophosphate and carbamate insecticides, as well as other pesticides, which have shown a range of impacts to aquatic organisms including salmonids (Deneer, 2000; Belden, 2007; Tierney et al., 2008; NMFS, 2009).

Due to the uncertainties identified in this BA there was an attempt to account for some of this uncertainty and ensure protection of listed salmonids and their critical habitat by making conservative assumptions in the exposure analysis. The exposure analysis that was used in this BA had several assumptions that when integrated with the response data could be used to demonstrate that the risk to listed salmonids, and their designated critical habitat, would be discountable and insignificant. Input parameters for the drift modeling such as wind speed, release height, temperature, humidity and wind direction were simultaneously set at values to maximize the likelihood off-site deposition of program insecticides.

From an efficacy standpoint, an application with all of these parameters set to maximize drift would result in a low probability of grasshopper or Mormon cricket suppression given the use of reduced rates and/or reduced area applications compared to conventional rates and coverage. In addition to the input parameters, the model was set to assume no interception of drift from sloping terrain and to minimize drift interception from terrestrial and riparian vegetation. Vegetation between the spray block and the sensitive habitat as well as vegetation at the sensitive site can intercept drift and reduce exposure to aquatic habitats (Ucar and Hall, 2001; Dabrowski et al., 2005; Dabrowski et al., 2006). Sensitive sites such as off-channel habitats have been shown to have aquatic and riparian vegetation with canopy coverage ranging from 41 to 81 % which would also act to intercept drift (Morley et al, 2005; Beechie et al, 2005).

Other factors that demonstrate the residues estimated from this exercise are conservative are that the spray block is parallel to the aquatic habitat the entire length of the swath and that the estimated pesticide residues are instantaneous concentrations. The lack of consideration of

degradation and dissipation during transport as well as the environmental fate of each insecticide once it is deposited in water also result in estimates above those expected if these factors were considered. All three chemicals are not expected to persist in water based on their chemical properties and environmental fate, which are discussed in section two of this BA. A majority of the acute effects data was based on exposure durations of 96 hours while chronic effects data had much longer durations. The comparison of the instantaneous residues to longer exposure periods provides an additional conservative approach in the risk characterization for each insecticide.

In addition to the above assumptions, which apply to all proposed applications, the estimates from bait applications are very conservative. Limitations in model input for bait applications in the ground treatments resulted in having to rely on drift from a liquid application with a median volume diameter of 340.87 µm compared to the use of dry bait that ranges in size from 1500 to 3000 µm. AgDisp does allow for dry aerial bait applications; however, the bait size was chosen from a standard range with a median bait size diameter of 340.87 µm that is well below the two to three mm size range that would be used. The resulting drift values when factoring in all the above assumptions significantly overestimates off-site drift from bait applications.

The selection of the above inputs with the exception of those inputs that are limited by each drift model, and the proposed mitigation measures, were collectively designed to generate residues that could be used with reasonable certainty to demonstrate protection of listed salmonids and their designated critical habitat in the risk characterization from program insecticide applications.

D. Summary

USDA APHIS evaluated the available data regarding exposure and response of carbaryl, diflubenzuron, and malathion to listed salmonids and their essential breeding, feeding and sheltering habitats as it relates to use patterns defined for the Grasshopper and Mormon Cricket Suppression Program. Based on our integration and synthesis of the exposure and response analyses to characterize risk to listed salmonids and their habitat, APHIS established avoidance and minimization measures to ensure the use of Program insecticides will not adversely affect individual salmonids or the habitats upon which they depend. All Program activities within the range of listed salmonids and/or salmon supporting waters or critical habitat will incorporate all prescribed avoidance and minimization measures (RAATs, buffers, wind direction, etc.) to ensure the use of carbaryl, diflubenzuron and malathion will not result in any adverse affects to individual salmonids or the habitat upon which they depend. The analyses APHIS used to develop these measures considered all relevant exposure pathways to ensure the integrity of the biological, chemical and physical attributes of salmonid habitat would be protected. APHIS has determined that the use of the aforementioned measures will provide necessary and sufficient protection of salmonids and their habitats from any potential impacts associated with use and fate of Program pesticides.

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Appendix A-1. Carbaryl acute aquatic fish toxicity values

A. Test Organism	B. Endpoint/Length	C. Toxicity Value	D. Reference
Salmo salar	96-hour LC_{50}	250 µg/L	Mayer and Ellersiek, 1986
Perca flavescens	96-hour LC_{50}	350 µg/L	Mayer and Ellersiek, 1986
Salvelinus fontinalis	96-hour LC_{50}	680 µg/L	Mayer and Ellersiek ,1986
Salvelinus namaycush	96-hour LC_{50}	690 µg/L	Mayer and Ellersiek, 1986
Oncorynchus mykiss	96-hour LC_{50}	780 µg/L	Mayer and Ellersiek, 1986
Oncorynchus clarki	96-hour LC_{50}	970 µg/L	Mayer and Ellersiek, 1986
Oncorynchus kisutch	96-hour LC_{50}	1150 µg/L	Mayer and Ellersiek, 1986
Acipenser brevirostrum	96-hour LC_{50}	1810 µg/L	Dwyer et al., 2005
Ptychocheilus lucius	96-hour LC_{50}	1300 µg/L	Beyers et al., 1994
Oncorynchus apache	96-hour LC_{50}	1540 µg/L	Dwyer et al., 2005
Oncorynchus clarki stomias	96-hour LC_{50}	1550 µg/L	Dwyer et al., 2005
Fundulus similis	96-hour LC_{50}	1600 µg/L	Mayer 1987
Lepomis macrochirus	96-hour LC_{50}	1800 µg/L	Mayer and Ellersiek, 1986
Salmo trutta	96-hour LC_{50}	2000 µg/L	Mayer and Ellersiek, 1986
Etheostoma lepidum	96-hour LC_{50}	2014 µg/L	Dwyer et al., 2005
Etheostoma fonticola	96-hour LC_{50}	2020 µg/L	Dwyer et al., 2005
Gilea elegans	96-hour LC_{50}	2020 µg/L	Beyers et al., 1994
Oncorynchus clarki henshawi	96-hour LC_{50}	2250 µg/L	Dwyer et al., 2005
Oncorynchus tshawytscha	96-hour LC_{50}	2400 µg/L	Mayer and Ellersiek, 1986
Pomoxus nigromeculatus	96-hour LC_{50}	2600 µg/L	Mayer and Ellersiek, 1986
Cyprinodon variegatus	96-hour LC_{50}	2600 µg/L	EPA, 2003a
Hybopsis monacha	96-hour LC_{50}	3410 µg/L	Dwyer et al., 2005
Xyrauchen texanus	96-hour LC_{50}	4350 µg/L	Dwyer et al., 2005
Notropis mekistocholas	96-hour LC_{50}	4510 µg/L	Dwyer et al., 2005
Cyprinodon bovinus	96-hour LC_{50}	4540 µg/L	Dwyer et al., 2005
Cyprinus carpio	96-hour LC_{50}	5280 µg/L	Mayer and Ellersiek, 1986
Micropterus salmoides	96-hour LC_{50}	6400 µg/L	Mayer and Ellersiek ,1986
Cyprinodon macularius	96-hour LC_{50}	7710 µg/L	Dwyer et al., 2005
Pimepheles promelas	96-hour LC_{50}	7770 µg/L	Mayer and Ellersiek, 1986
Ictalurus puncatus	96-hour LC_{50}	7790 µg/L	Mayer and Ellersiek, 1986
Lepomis cyanellas	96-hour LC_{50}	9460 µg/L	Mayer and Ellersiek, 1986
Carassius auratus	96-hour LC_{50}	12,800 µg/L	Mayer and Ellersiek, 1986
Mystis vittatus	96-hour LC_{50}	17,500 µg/L	Arunachalam et al., 1980
Amelurus melas	96-hour LC_{50}	20,000 µg/L	Mayer and Ellersiek, 1986

Appendix A-2. Carbaryl acute aquatic invertebrate toxicity values

Test Organism	Endpoint/Length	Toxicity Value	Reference
Chironomus riparius	24-hour LC_{50}	1.2 µg/L	Karnak and Collins, 1974
Paneaus aztecus	48-hour LC_{50}	1.5 µg/L	Mayer, 1987
Pteronarcella badia	96-hour LC_{50}	1.7 µg/L	EPA, 2003a
Isogenus sp.	96-hour LC_{50}	3.6 µg/L	EPA, 2003a
Pteronarcys californica	96-hour LC_{50}	4.8 µg/L	Mayer and Ellersiek, 1986
Paleomenetes kadiankesis	96-hour LC_{50}	5.6 µg/L	Mayer and Ellersiek, 1986
Classenia sabulosa	96-hour LC_{50}	5.6 µg/L	EPA, 2003a
Daphnia magna	48-hour EC_{50}	5.6 µg/L	EPA, 2003a
Mysidopsis bahia	96-hour LC_{50}	5.7 µg/L	EPA, 2003a
Daphnia pulex	48-hour EC_{50}	6.4 µg/L	Mayer and Ellersiek, 1986
Chironomus tentans	24-hour LC_{50}	7.0 µg/L	Karnak and Collins, 1974
Simocephalus serrulatus	48-hour EC_{50}	7.6 µg/L	Mayer and Ellersiek, 1986
Chironomus plumosus	96-hour LC_{50}	10 µg/L	Sanders et al., 1983
Cynigma sp.	96-hour LC_{50}	11.1 µg/L	Peterson et al., 2001b
Calineura californica	96-hour LC_{50}	17.3 µg/L	Peterson et al., 2001b
Ameletus sp.	96-hour LC_{50}	24 µg/L	Peterson et al., 2001b
Gammarus lacustrus	96-hour LC_{50}	22 µg/L	Mayer and Ellersiek, 1986
Metapenaeus monoceros	96-hour LC_{50}	24.6 µg/L	Reddy and Rao, 1992
Gammarus fasciatus	96-hour LC_{50}	26 µg/L	Mayer and Ellersiek, 1986
Paleomenetes pugio	48-hour LC_{50}	28 µg/L	Mayer, 1987
Lepidistoma unicolor	96-hour LC_{50}	29.0 µg/L	Peterson et al., 2001b
Psyglypha sp.	96-hour LC_{50}	30.3 µg/L	Peterson et al., 2001b
Paneaus duorarum	48-hour EC_{50}	32 µg/L	Mayer, 1987
Brachycentrus americanus	96-hour LC_{50}	41.2 µg/L	Peterson et al., 2001b
Pseudochinus magellanicus	96-hour EC_{50}	92.5 µg/L	Hernandez et al., 1990
Cypridopsis vidua	48-hour EC_{50}	115 µg/L	Mayer and Ellersiek, 1986
Xanthocnemis zealandica	48-hour LC_{50}	156 ppb	Hardeson and Wratten, 2000
Aselius bravicaudus	96-hour LC_{50}	280 µg/L	Mayer and Ellersiek, 1986
Callinectes sapidus	48-hour LC_{50}	320 µg/L	Mayer, 1987
Procambarus sp.	96-hour LC_{50}	1900 µg/L	Mayer and Ellersiek, 1986
Crassostrea virginica	48-hour EC_{50}	2900 µg/L	EPA, 2003a
Corbicula striatella	96-hour LC_{50}	5100 µg/L	Jadhav et al., 1996
Mytilus edulis	96-hour LC_{50}	22,700 µg/L	CA DPR, 1998

Appendix A-3. Carbaryl acute sublethal and chronic aquatic toxicity values

Test Organism	Endpoint/Length	Toxicity Value	Reference
Oncorynchus clarki	6-hr NOEC (predator avoidance)	200 µg/L	Labenia et al., 2007
Oncorynchus clarki	6-hr NOEC (swimming performance)	500 µg/L	Labenia et al., 2007
Oncorynchus mykiss	96-hour NOEC (swimming capacity)	100 µg/L	Little et al., 1990
Oncorynchus mykiss	96-hour NOEC (swimming activity)	100 µg/L	Little et al., 1990
Oncorynchus mykiss	96-hour NOEC (Daphnia consumed)	100 µg/L	Little et al., 1990
Cyprinodon variegatus	96-hour NOEC	1100 µg/L	EPA, 2003a
Fathead minnow *Pimephales promelas*	7-day NOEC (Growth)	250 µg/L	Pickering et al., 1996
Mysidopsis bahia	96-hour NOEC	3.2 µg/L	EPA, 2003a
Pimepheles promelas	35-day NOEC (Reproduction)	210 µg/L	EPA, 2003a
Gilea elegans	32-day NOEC	650 µg/L	Beyers et al., 1994
Ptychocheilus lucius	32-day NOEC	445 µg/L	Beyers et al., 1994
Daphnia magna	21 day NOEC (Reproduction)	1.5 µg/L	EPA, 2003a
Chironomus riparius	28-day NOEC (Emergence/development)	500 µg/L	EPA, 2003a

Appendix A-4. Carbaryl toxicity values not used in the aquatic effects analysis

Test Organism	Endpoint	Justification	Reference
Uca minax	25-hour mortality, swimming behavior	No toxicity endpoint established	Capaldo, 1987
Oncorynchus mykiss	96-hour NOEC (% survival from predation)	No toxicity endpoint established	Little et al., 1990
Cyprinus carpio	ChE inhibition	No toxicity endpoint established	Gruber and Munn, 1998
Cirrhana mrigala	Reproduction, maturation time	No toxicity threshold established/concentrations not corrected for active ingredient	Kaur and Dhawan, 1996
Mystis vittatus	27-day exposure (feeding rates/growth)	No statistical analysis of sublethal data	Arunachalam et al., 1980
Puntius conchonius	Gill, liver, kidney pathology	No toxicity threshold established/no solvent control	Gill et al., 1988
Carassius auratus	Cellular pathology	No solvent control/No toxicity threshold established	Shea and Berry, 1983
Tilapia mossambica	Respiratory potential	No toxicity threshold established	Basha et al., 1984
Utterbackia imbecilis	Lethality and genotoxicity	No toxicity threshold established	Conners and Black, 2004

Appendix A-5. Diflubenzuron acute aquatic fish toxicity values

Test Organism	Endpoint/Length	Toxicity Value	Reference
Yellow Perch *Perca flavescens*	96-hour LC_{50}	25 mg/L	Johnson and Finley, 1980
Bluegill sunfish* *Lepomis macrochirus*	96-hour LC_{50}	135 mg/L	EPA, 1997
Rainbow trout *Onchorrynchus mykiss*	96-hour LC_{50}	140 mg/L	EPA, 1997
Cutthroat trout *Oncorynchus clarki*	96-hour LC_{50}	>60 mg/L	Mayer and Ellersiek, 1986
Atlantic salmon *Salmo salar*	96-hour LC_{50}	>50 mg/L	Mayer and Ellersiek, 1986
Brook Trout *Salvelinus fontinalis*	96-hour LC_{50}	>50 mg/L	Mayer and Ellersiek, 1986
Flathead catfish *Ictalurus punctatus*	96-hour LC_{50}	>100 mg/L	Johnson and Finley, 1980
Fathead minnow *Pimepheles promelas*	96-hour LC_{50}	>500 mg/L	US FS, 2004

* The lowest LC_{50} value for the bluegill sunfish is reported above. Values as high as 660 mg/L have been reported

Appendix A-6. Diflubenzuron acute aquatic invertebrate toxicity values

Test Organism	Endpoint/Length	Toxicity Value	Reference
Aedes nigromaculatum	48-hour EC_{50}	0.5 µg/L	Miura and Takahashi, 1974
Chironoumus plumosus	48-hour EC_{50}	0.56 µg/L	Julin and Sanders, 1978
Palaemontes pugio	96-hour LC_{50}	0.64 µg/L	EPA, 1997
Streptocephalus sudanicus	48-hour EC_{50}	0.74 µg/L	Lahr et al., 2001
Tanytarsus dissimilis	120-hour LC_{50}	1.02 µg/L	Hansen and Garton, 1982a
Ceriodaphnia dubia	48-hour EC_{50}	1.7 µg/L	US FS, 2004
Daphnia magna	48-hour EC_{50}	1.84 µg/L	Hansen and Garton, 1982a
Hyallela azteca	96-hour LC_{50}	1.84 µg/L	Hansen and Garton, 1982a
Mysidopsis bahia	96-hour LC_{50}	2.0 µg/L	EPA, 1997
*Eurytemora affinis**	48-hour LC_{50}	2.2 µg/L	Savitz et al., 1994
*Callinectes sapidus**	96-hour LC_{50}	18.5 µg/L	Rebach, 1996
Gammarus sp.	96-hour LC_{50}	30 µg/L	US FS, 2004
Gammarus pseudolimnaeus	96-hour LC_{50}	45 µg/L	EPA, 1997
Orthemis sp.	168-hour LC_{50}	50 µg/L	Miura and Takahashi, 1974
Hydrophilus triangularis	48-hour EC_{50}	100 µg/L	Miura and Takahashi, 1974
Anisops sardius	48-hour EC_{50}	1937 µg/L	Lahr et al., 2001
*Crassostrea virginica**	96-hour LC_{50}	130 mg/L	EPA, 1997

* Formulation studies

Appendix A-7. Diflubenzuron acute sublethal and chronic aquatic toxicity values

Test Organism	Endpoint/Length	Toxicity Value	Reference
Fundulus heteroclitus	96-hour NOEC	29.86 mg/L	Lee and Scott, 1989
Pimepheles promelas	35-day NOEC	0.10 mg/L	EPA, 1997
Onchorrynchus mykiss	30-day NOEC (Growth/Survival)	>45 µg/L	Hansen and Garton, 1982a
Daphnia magna	21- day NOEC (Reproduction)	0.04 µg/L	EPA, 1997
Mysidopsis bahia	28- day NOEC (Reproduction)	0.045 µg/L	EPA, 1997
Ceriodaphnia dubia	7-day NOEC (Reproduction)	0.25 µg/L	US FS, 2004

Appendix A-8. Malathion acute fish toxicity values

Test Organism	Endpoint/Length	Toxicity Value	Reference
Rainbow trout	96-hour LC_{50}	4.0 µg/L	EPA, 2006
Bluegill sunfish	96-hour LC_{50}	20.0 µg/L	EPA, 2006
Sheepshead minnow	96-hour LC_{50}	33.0 µg/L	EPA, 2006
Redear sunfish	96-hour LC_{50}	62.0 µg/L	EPA, 2006
Walleye	96-hour LC_{50}	64.0 µg/L	EPA, 2006
Striped bass	96-hour LC_{50}	60.0 µg/L	EPA, 2006
Lake trout	96-hour LC_{50}	76.0 µg/L	EPA, 2006
Brown trout	96-hour LC_{50}	101.0 µg/L	EPA, 2006
Coho Salmon	96-hour LC_{50}	170.0 µg/L	EPA, 2006
Cutthroat trout	96-hour LC_{50}	174.0 µg/L	EPA, 2006
Largemoth bass	96-hour LC_{50}	250.0 µg/L	EPA, 2006
Yellow Perch	96-hour LC_{50}	263.0 µg/L	EPA, 2006
Spot	96-hour LC_{50}	320.0 µg/L	EPA, 2006
Striped Mullet	96-hour LC_{50}	330.0 µg/L	EPA, 2006
Green sunfish	96-hour LC_{50}	1460.0 µg/L	EPA, 2006
Tilapia	96-hour LC_{50}	2000.0 µg/L	EPA, 2006
Carp	96-hour LC_{50}	6590.0 µg/L	EPA, 2006
Channel catfish	96-hour LC_{50}	7620.0 µg/L	EPA, 2006
Fathead minnow	96-hour LC_{50}	8650.0 µg/L	EPA, 2006
Goldfish	96-hour LC_{50}	10700.0 µg/L	EPA, 2006
Black bullhead catfish	96-hour LC_{50}	11700.0 µg/L	EPA, 2006
Colorado bonytal	96-hour LC_{50}	15300.0 µg/L	Beyers et al., 1994

Appendix A-9. Malathion acute aquatic invertebrate toxicity values

Test Organism	Endpoint/Length	Toxicity Value	Reference
Gammarus fasciatus	96-hour LC_{50}	0.5 µg/L	EPA, 2006
Simocephalus serrulatus	96-hour LC_{50}	0.69 µg/L	EPA, 2006
Isoperla sp.	96-hour LC_{50}	0.69 µg/L	EPA, 2006
Daphnia magna	96-hour LC_{50}	1.0 µg/L	EPA, 2006
Pteronarcella badia	96-hour LC_{50}	1.1 µg/L	EPA, 2006
Limnephalus sp.	96-hour LC_{50}	1.3 µg/L	EPA, 2006
Gammarus lacustris	48-hour EC_{50}	1.8 µg/L	EPA, 2006
Daphnia pulex	48-hour EC_{50}	1.8 µg/L	EPA, 2006
Neomysis mercedis	96-hour LC_{50}	2.2 µg/L	Brandt et al., 1993
Mysidopsis bahia	96-hour LC_{50}	2.2 µg/L	EPA, 2006
Claasenia sabulosa	96-hour LC_{50}	2.8 µg/L	EPA, 2006
Hydropsyche sp.	96-hour LC_{50}	5.0 µg/L	EPA, 2006
Lestes congener	96-hour LC_{50}	10.0 µg/L	EPA, 2006
Paleomenetes kadiankesis	96-hour LC_{50}	12.0 µg/L	EPA, 2006
Orconectes nais	96-hour LC_{50}	180.0 µg/L	EPA, 2006
Penaeus duorarum	48-hour LC_{50}	180.0 µg/L	EPA, 2006
Atherix variegata	96-hour LC_{50}	385 µg/L	EPA, 2006
Crassostrea virginica	96-hour LC_{50}	>1000 µg/L	EPA, 2006
Callinectes sapidus	48-hour LC_{50}	>1000 µg/L	EPA, 2006
Asellus brevicaudus	96-hour LC_{50}	3000 µg/L	EPA, 2006
Utterbackia imbecilis	96-hour LC_{50}	40 mg/L	Keller and Ruesler, 1997
Villosa lienosa	96-hour LC_{50}	74 mg/L	Keller and Ruesler, 1997
Villosa villosa	96-hour LC_{50}	180 mg/L	Keller and Ruesler, 1997

Appendix A-10. Malathion toxicity values not used in the aquatic effects analysis

Test Organism	Endpoint	Justification	Reference
Gambusia affinis	Gill histopathology	No NOEC value calculated	Cengiz and Unlu, 2003
Cyprinus carpio	Mortality/mixtures	No toxicity endpoint calculated/Unknown malation doses	Anwar et al., 2005
Seriola dumerilli	Brain cholinesterase activity	Fish were injected with malathion	Jebali et al., 2006
Lepomis macrochirus	Gill histopathology	No NOEC value calculated	Richmonds and Dutta, 1989
Cyprinus carpio	Oocyte maturation	In vitro exposure using one fish, no NOEC calculated, [ ] an order of magnitude above EEC's from Program applications	Haider and Inbaraj, 1988