7.9.2 The Long-Term Plan

7.9.2.1 Quantitative Risk Assessment for Resmethrin, Sumithrin, Permethrin, Piperonyl Butoxide, and Malathion

Much of the important background information for the risk assessment was presented in Section 4. In brief, application scenarios were developed in association with the County, and were intended to provide a range of reasonable applications that might be expected to occur in each of the four study areas. These application scenarios were based on past practices, and so are hoped to be an overestimate of potential impacts from the Long-Term Plan.

7.9.2.1.1 Introduction

The risk assessment discussion for larvicides (immediately above) provides information on the general methodology adopted for this work.

7.9.2.1.2 Compounds Evaluated

Adulticide applications were evaluated as a management tool to reduce public welfare impacts associated with adult mosquito populations (Vector Control applications), and to reduce public health risks when mosquito-borne diseases pose a public health emergency. Public health risk applications meet criteria published by CDC and NYSDOH.

The adulticides selected for consideration were:

- Pyrethroids. Pyrethroids are synthetic chemical insecticides that act by disrupting nerve cell activity in insects, which ultimately leads to insect paralysis. They are most commonly used in combination with the synergist PBO, which is added to increase potency, and thereby decrease the amount of pyrethroid used in the formulation. The pyrethroid products considered in this assessment are:
    - Resmethrin + PBO (Scourge)
    - Permethrin + PBO (Permanone, Aquareslin)
    - Sumithrin + PBO (Anvil).

- Organophosphates. Organophosphate (OP) pesticides consist of a broad class of chemicals that act through the inhibition of the neurotransmitter enzyme acetylcholinesterase (AChE). In insects, this inhibition interferes with the nerve-muscle communication, which ultimately causes paralysis. The pesticide malathion was the OP pesticide included in the quantitative risk assessment. Malathion is the active ingredient in the pesticide products Fyfanon, Atrappa, and Microflo. In addition, the evaluation also considered malaoxon and isomalathion, which can be present as impurities in the formulated product and/or formed via environmental degradation.

Chemical repellants also were evaluated as a management tool for broad-scale application. Chemical repellants are pesticides that are used to prevent or limit insect and other pest activity. Repellants are used in a variety of applications, including those associated with the protection of humans, pets, livestock and plants. Garlic oil was the repellant considered in the Evaluation Management Plan. Garlic oil products include Garlic Barrier AG+ and Mosquito Barrier.

Garlic oil was not selected for further evaluation because it is not likely to have any adverse effects on either human health or the environment (CA-IC 2004, CA-SCDHS 2005). Garlic is a naturally occurring substance that is widely distributed and commercially available for flavoring and seasoning. It is a "generally recognized as safe," or GRAS, substance under FDA regulations[1]. Further, garlic is not persistent in the environment and USEPA (1992) has determined that no significant adverse effects to humans or the environment are associated with its use as a pesticide. In fact, garlic is currently on the USEPA's exempted products list as stipulated under the USEPA Federal Insecticide, Fungicide and Rodenticide Act (FIFRA), Section Sec 25(b). This list contains "minimum risk" pesticides which may be used freely without regulation owing to their demonstrated safety. Based on this collective information, use of garlic oil as a barrier treatment is assumed to pose no unacceptable human health or environmental risks, and therefore, this product is not evaluated further in this risk assessment.

Potential exposures and risks also were not evaluated for isomalathion or malaoxon. Both of these chemicals can occur as impurities in technical formulations of malathion at low levels. ATSDR (2003a) reported that isomalathion was present in technical malathion formulations at a

[1] See 21 CFR 182.10, 182.20 and 184.1317

level of 0.2 percent and that malaoxon was present at a level of 0.1 percent. In addition, malaoxon is formed in vivo and in the environment via oxidative desulfuration of malathion (ATSDR, 2003; USEPA, 2000a). Activation may be achieved by photo-oxidation, chemical oxidation, or biological activation via enzymatic pathways. Malaoxon is the toxicologically active agent associated with malathion's AChE effects, and given this, is more potent than the parent malathion compound.

Limited data exists with respect to the environmental fate of malaoxon and isomalathion (ATSDR, 2003), and USEPA (2000a) has identified the environmental fate data gap for these chemicals, with special attention on the need for data development with respect to malaoxon. Relevant toxicological data are similarly sparse.

Overall, insufficient data are available to support quantitative exposure and risk assessment for either malaoxon or isomalathion. USEPA (2000a) discussed potential exposures to malaoxon as part of its evaluation of hypothetical human dietary exposures to malathion and concluded that:

> Although aware of the possible formation of malaoxon, there is insufficient information currently available to perform a quantitative exposure assessment without a large degree of uncertainty. Therefore, an assessment of the potential post application exposure to malaoxon has not been performed, and to do so would require the results from malathion/malaoxon residue dissipation studies for representative crops.

Given these collective considerations, evaluations of potential human and ecological impacts associated with malaoxon and isomalathion cannot be evaluated at this time with any acceptable degree of quantitative certainty. The limited data that are available suggest that in many terrestrial situations, exposures to malaoxon or isomalathion will be small relative to malathion due to degradation. For example, in its most recent risk assessment for malathion conducted as part of the RED review, USEPA (2000a) concluded that the primary route of dissipation of malathion in surface soils is microbially mediated soil metabolism (half-life less than one to 2.5 days) and hydrolysis (at pH 7, half-life was 6.12 days, and at pH 9, half-life was 12 hours). Studies of microbially mediated soil metabolism report that just 0.6 to 1.8 percent of malathion degrades to malaoxon when sufficient moisture, light, and microbial activity are present, and pH is not elevated. However, in situations where microbially mediated soil metabolism is less likely to occur (e.g., in aquatic environments, on anthropomorphic surfaces such as asphalt or

concrete), potential exposures can be higher. For example, under some dry and microbially inactive environmental conditions, malaoxon is formed from malathion at levels up to 10.7 percent of the total applied (USEPA, 2000a). Malaoxon also has been detected in runoff water and in leachate and soil extracts at greater than or equal to 12 percent of malathion application (USEPA, 2000a).

No further risk evaluations can occur, however, until additional data are published on the formation, persistence, and toxicity of these two malathion product compounds. Therefore, these compounds were not evaluated further in this risk assessment.

The application sites and scenarios that will be considered have been previously discussed in Section 4.

7.9.2.1.3 Conceptual Model

Most aspects of the conceptual model that apply to this evaluation of adulticides were presented just above, in the discussion of larvicides. Two aspects need to be redeveloped for the adult control assessment:

- Pesticide Characteristics – Release and Fate in the Environment
- Toxicological Effects of Target Pesticides

Two aspects do not need to be discussed again:

- Potentially Exposed Populations
- Endpoints

Please refer to the larvicides discussion above for pertinent information on these latter two parts of the conceptual model.

7.9.2.1.4 Pesticide Characteristics

Resmethrin

Resmethrin is a broad spectrum pyrethroid insecticide used for control of flying and crawling insects in homes, greenhouses, indoor landscapes, mushroom houses, and industrial sites; insects

that infest stored products; and mosquito control (Extoxnet, 1996c). Resmethrin commonly appears under the trade name Scourge, which is used to control adult mosquitoes.

Resmethrin is classified as relatively insoluble in water, but is highly soluble in organic solvents (Westchester, 2001). It binds tightly to soil and is not expected to be mobile in soil or to contaminate groundwater (Extoxnet, 1996c). In addition to binding to soil, resmethrin is likely to sorb to sediments and suspended particles, and possibly to plants. Biodegradation, hydrolysis, and photodegradation are the most rapidly acting degradation pathways for resmethrin, with environmental half-lives ranging from 15 minutes to just over a month, depending on the environmental setting (Extoxnet, 1996c; WHO-FAO, 1996). Resmethrin's photodegradation half-life on surfaces is approximately three hours, while half-lives in soil and sediment have been reported to be 30 and 36.5 days, respectively (Westchester, 2001). No degradation products have been identified as more toxic than parent resmethrin.

Resmethrin has a low to moderate potential to bioaccumulate in aquatic organisms.

Sumithrin[2]

Sumithrin is a broad spectrum pyrethroid insecticide registered for use against mosquitoes in swamps, marshes, and recreational areas. Sumithrin is the active ingredient in the product Anvil 10 + 10.

Sumithrin degrades readily, with a half-life of less than one day on plants and other surfaces (CA-IC, 2004). In soil, sumithrin has a half-life of one to two days under dry, sunny conditions. Under flooded conditions, the half-life increases to two to four weeks for the trans isomer and one to two months for the cis isomer. In general, the degradative processes that occur in the environment lead to less toxic products (WHO-FAO 1990a).

Sumithrin has a low to moderate potential to bioaccumulate in aquatic organisms.

Permethrin

Permethrin is a broad spectrum pyrethroid insecticide which is used against a variety of insect pests in addition to mosquitoes. Permethrin commonly appears under the trade names Permanone, Pounce, Nix, Torpedo, and Dragnet.

[2] Sumethrin is a synonym for sumithrin and is used throughout the appendices associated with the risk assessment.

Permethrin is readily degraded in most soils (except highly organic types). Field dissipation studies performed for permethrin indicate a half-life of 42 days in soil[3]. Degradation in soil is largely attributable to microbial biodegradation (Westchester, 2001; Extoxnet, 1996b). On surfaces, permethrin is degraded by sunlight, but may remain insecticidally active for up to 26 days (WHO-FAO, 1990b). The half-life in water has been reported to be less than 2.5 days (Extoxnet, 1996b). The half-life in sediment has been reported to be 2.5 days (HSDB, 2005).

Permethrin is tightly bound by soils, especially to organic matter. It is relatively immobile in a wide range of soil types and is nearly insoluble in water. Therefore, permethrin is not expected to leach or to contaminate groundwater (Extoxnet, 1996b). No degradation products have been identified as more toxic than parent permethrin.

Permethrin has been found to bioaccumulate to a low to moderate degree in aquatic organisms; most notably in aquatic insect larvae, aquatic plants (e.g., duckweed), and some fish (Westchester, 2001). However, its rapid degradation in aquatic systems likely limits the importance of this mechanism in ambient systems.

Malathion

Malathion is a nonsystemic broad-spectrum OP pesticide that is used in agriculture and horticulture applications (ATSDR, 2003; USEPA, 1999b; USEPA, 2000a). Malathion is the active ingredient in mosquito control products including Fyfanon and Atrapa. These products contain over 95 percent malathion and are often applied undiluted. However, in some cases they may be diluted with a petroleum solvent similar to kerosene before application. In such cases the petroleum solvent will make up most of the pesticide solution (ATSDR, 2003).

Malathion is degraded in the environment through three main pathways: activation, degradation, and isomerization. Activation of the compound involves oxidative desulfuration, yielding malaoxon. Degradation of malathion occurs through both chemical and biological means, with hydrolysis being the most important process for each (ATSDR 2003a, USEPA 2000a).

[3] Field dissipation studies determine how fast a pesticide disappears from the upper soil layers after the pesticide is applied to bare soil at a known application rate. Such studies simulate the disappearance of the pesticide under naturally-occurring environmental conditions. Disappearance can be due to any or all of the following: degradation due to water, sunlight and/or microbial activity, leaching from soil due to downward movement of rain water, and evaporation into the air.

Malathion can be broken down via microbial activity and photodegradation under various settings. Its half-lives can range from five hours to 25 days, depending on the medium (i.e. water, soil, air) (Westchester, 2001). Isomerization results in the formation of isomalathion (ATSDR, 2003). Malathion can persist for greater periods of time on surfaces if not subject to hydrolysis, such as might occur during a rain event (USEPA, 2000a).

The majority of the data available on the bioaccumulation of malathion suggest that, while malathion may be bioconcentrated, it is rapidly metabolized or depurated from the tissue of aquatic organisms and is, therefore, not likely to be biomagnified in an aquatic-based food chain (CA-IC, 2004).

Piperonyl Butoxide (PBO)

PBO is a derivative of piperic acid and is utilized as a chemical synergist in pyrethroid formulations. Pyrethroid products containing PBO are used to control mosquitoes in outdoor residential and recreational areas, as well as indoors to control insects such as fleas, ticks, and ants.

PBO prevents metabolic enzyme activity (specifically that of Cytochrome P450 enzymes), through microsomal enzyme inhibition in insects, and thereby allows the active ingredients in pyrethroids to remain available and cause enhanced toxic effects (HSDB, 2003; Klaasen et al., 1986; NPIC, 2000).

PBO is rapidly degraded in soil with a half-life of 14 days in aerobic soils. If released to soil, PBO is expected to have moderate to low mobility. If released into water, PBO is expected to adsorb moderately to suspended solids and sediment. Although stable to hydrolysis under sterile, dark conditions, PBO is degraded by sunlight in aqueous solutions (HSDB, 2003). No degradation products have been identified as more toxic than parent PBO.

PBO has a low to moderate potential to bioaccumulate in aquatic organisms (HSDB, 2003; NPIC, 2000).

Table 7-15. Physical and Chemical Properties of Adulticides and Synergist

	Parameter	Value	Units	Source	Notes
Permethrin					
	CAS No.	52645-53-1	--	CAS	--
	Molecular weight	391.29	--	ARS PPDB	--
	Molecular formula	C21H20CL2O3	--	ARS PPDB	--
	Melting Point	34	^oC	Extoxnet	--
	Solubility (in water)	0.0061	ppm	ARS PPDB	pH=7.3 @ 20^oC
	Henry's Law Constant (solubility)	1.89E-06	atm. m^3/mol	ARS PPDB	@ 20-25^oC
	Vapor Pressure (in mPA)	0.0029	mPA	ARS PPDB	@ 25^oC
	Vapor Pressure (mm Hg)	--	mm Hg	--	--
Resmethrin					
	CAS No.	10453-86-8	--	CAS	--
	Molecular weight	338.4	--	ARS PPDB	--
	Molecular formula	C22H26O3	--	ARS PPDB	--
	Melting Point	43	^oC	Extoxnet	--
	Solubility (in water)	0.038	ppm	ARS PPDB	@ 25^oC
	Henry's Law Constant (solubility)	1.3E-07	atm. m^3/mol	HSDB	--
	Vapor Pressure (in mPA)	--	mPA	--	--
	Vapor Pressure (mm Hg)	1.1E-08	mm Hg	HSDB	@ 30^oC
Sumithrin					
	CAS No.	26002-80-2	--	CAS	--
	Molecular weight	350.46	--	HSDB	--
	Molecular formula	C23H26O3	--	HSDB	--
	Melting Point	43	^oC	Extoxnet	value for resmethrin
	Solubility (in water)	<0.0097	ppm	HSDB	@ 25^oC
	Henry's Law Constant (solubility)	6.8E-06	atm. m^3/mol	HSDB	--
	Vapor Pressure (in mPA)	--	mPA	--	--
	Vapor Pressure (mm Hg)	1.43E-07	mm Hg	HSDB	@ 21^oC
Piperonyl butoxide (PBO)					
	CAS No.	51-03-6	--	CAS	--
	Molecular weight	338.43	--	HSDB	--
	Molecular formula	C19H30O5	--	HSDB	--
	Melting Point	25	^oC	USEPA 2005a	--
	Solubility (in water)	14.3	ppm	HSDB	@ 25^oC
	Henry's Law Constant (solubility)	8.9E-11	atm. m^3/mol	HSDB	--
	Vapor Pressure (in mPA)	--	mPA	--	--
	Vapor Pressure (mm Hg)	2.60E-07	mm Hg	HSDB	@ 25^oC
Malathion					
	CAS No.	121-75-5	--	CAS	--
	Molecular weight	330.4	--	ARS PPDB	--
	Molecular formula	C10H19O6PS2	--	ARS PPDB	--
	Melting Point	2.85	^oC	Extoxnet	--
	Solubility (in water)	145.0	ppm	ATSDR 2003a	@ 20^oC
	Henry's Law Constant (solubility)	4.9E-09	atm. m^3/mol	ATSDR 2003a	@ 25^oC
	Vapor Pressure (in mPA)	0.45	mPA	HSDB	@ 25^oC
	Vapor Pressure (mm Hg)	--	mm Hg	--	@ 25^oC

= Not available or not applicable
CAS = Chemical Abstract Service
ARS PPDB = USDA's Agricultural Research Service Pesticide and Properties Database (USDA 2005)
Extoxnet = Extension Toxicology Network (1996b,c,h)
HSDB = National Library of Medicine's Hazardous Substances Data Bank (HSDB, 2005)

All of the target pesticides are proposed for direct release into the environment and thereby have the potential to reach human or ecological receptors. The likelihood, magnitude, and duration of any potential exposure are dependent to a large degree on how the compound is released, where it is released and how it behaves once it is released.

Adulticides are applied in and near residential areas for public welfare or public health concerns. The target adulticides are released as liquid formulations to air using a suite of application methods (e.g., truck, helicopter, backpack sprayers).

Once released to air, the pesticides can remain airborne or can deposit on a variety of surfaces within the study area. This includes direct deposition onto people or wildlife, but also onto soils, indoor dust, vegetation, fruits and vegetables, and impermeable and non-reactive surfaces (e.g., parking lots, roads, outside lawn furniture). Adulticides present on soil or indoor dust also can be re-suspended into the air.

Adulticides deposited on surfaces can be transported through the environment via surface runoff following rain events, either in a dissolved form or sorbed to soil or other fine-grained particles, eventually reaching nearby water bodies where they can partition to sediments or remain dissolved. Degradation can occur throughout the entire transport process, and because none of the adulticides are likely to persist to any significant degree in surface environments, longer-term transport to surface waters is not likely to be a quantitatively important transport mechanism. Also, given low persistence in soil and the low quantities used, leaching to groundwater is not an important pathway for these compounds.

Adulticides also can indirectly deposit into surface waters that exist within the application area or its buffer.

The chemical concentrations that ultimately reach a given receptor are the result of a complex combination of fate, transport and degradation process. Repeated application of adulticides could result in a slight increase in target pesticide concentrations over time, if degradation is not complete prior to the repeat application.

As a chemical class, pyrethroids have very low volatility, are very poorly soluble in water, and have a tendency to bind very tightly to organic particles in soil (NYCDOH, 2001). Given these characteristics, pyrethroids are not expected to leach to groundwater. In aquatic settings,

pyrethroids strongly adsorb on sediments, and once adsorbed, are difficult to remove with water (WHO-FAO, 1990a). Pyrethroids that do persist in surface waters in bioavailable (non-sorbed) forms can bioaccumulate to a low to moderate degree in aquatic organisms. These compounds are non-volatile, so transport to air via this pathway will not be important.

Based on its physical-chemical properties, malathion is moderately to extremely mobile, shows little persistence in soil and water, and is unlikely to accumulate in aquatic food webs to any significant degree. If no water is present, malathion can persist for longer periods of time on surfaces (USEPA, 2000a). It is relatively non-volatile and is not expected to be released to air via this mechanism.

PBO is expected to have moderate to low mobility in soil and be rapidly degraded. In water, PBO is expected to adsorb to suspended solids and sediment, and can be degraded by sunlight. PBO that persists in a bioavailable form in water can accumulate in aquatic organisms. PBO is non-volatile.

7.9.2.1.5 Toxicological Effects of Target Pesticides

When present at sufficiently high concentrations, the target pesticides can potentially cause a variety of toxic effects in both humans and wildlife. A detailed review of the toxicology of these compounds is presented separately in CA-SCDHS (2005) and CA-IC (2004). A brief summary is presented below to support development of the conceptual model.

Pyrethroids – Ecotoxicity

Permethrin, resmethrin, and sumithrin induce toxicity by disrupting sodium transport at the nerve axon in both the peripheral and central nervous system (CA-IC, 2004). Initially, they cause nerve cells to discharge repetitively; and later, they cause paralysis. When applied alone, pyrethroids may be swiftly detoxified by enzymes in the insect, and for this reason pyrethroids are typically applied along with a synergist, such as PBO, that inhibits enzyme degradation and thus enhances efficacy (USEPA, 2002b).

Overall, pyrethroids are low in toxicity to mammals, and are practically nontoxic to birds. However, at sufficiently high concentrations, laboratory data indicate that pyrethroids are toxic to aquatic life and non-target insects, including honeybees (USEPA, 2002b).

Pyrethroids – Human Health

Pyrethroids interfere with nerve and brain function. Exposure to very high levels of these compounds for a short period in air, food, or water may cause dizziness, headache, and other neurological effects in people during the period of exposure and for short-time following exposure. There is no evidence that pyrethroids affect reproduction in humans, but some animal studies have shown reduced fertility in males and females.

WHO (2005) has concluded that there is "no clear indication of carcinogenicity relevant for human risk assessment" for pyrethroid pesticides. Permethrin also has been evaluated by IARC (1991), and classified in Group 3, indicating that it is not classifiable as a carcinogen in humans. USEPA has classified permethrin as a possible human carcinogen based upon limited data from animal studies.

Malathion – Ecotoxicity

Malathion's insecticidal activity is due to is inhibition of the neuroenzyme AChE. At sufficiently high concentrations, malathion can cause toxicity in aquatic and terrestrial wildlife. In general, aquatic life exhibits greater sensitivity to malathion than terrestrial wildlife. USEPA (2000a, 2002c) has reviewed extensive data and has classified malathion as very highly to moderately toxic for both fresh water and estuarine/marine fish species.

Malathion exhibits generally low to moderate toxicity to terrestrial wildlife (USEPA, 2000a; USEPA, 2002c). Malathion has been shown to result in slight toxicity to mammals (USEPA, 2000c). High acute doses in the range of 150 to 2,100 mg/kg bw-d may cause death. Malathion can affect the central nervous system, the immune system, adrenal glands, liver and blood following chronic exposure to lower dosages. Reproductive effects are not expected unless exposure to high dosages (500 to 1000 mg/kg) occurs for extended periods of time (USEPA, 2000a).

Malathion is considered to be highly toxic to bees on an acute contact basis either through exposure to direct spray or through foliar residue contact within eight hours after spray is applied. Field incidents of extensive honeybee mortality following malathion applications have also been documented (USEPA, 2000a).

Malathion – Human Health

Malathion is a cholinesterase inhibitor and, therefore, its primary toxic effect following human exposures to sufficiently high concentrations is on the nervous system (CA-SCDHS, 2005). Inhibition of cholinesterase can lead to various forms of toxicity affecting muscles, the central nervous system, and endocrine glands. Exposure to the skin or eyes may produce some irritant effects. Some animal studies have shown that under certain conditions, malathion may cause allergic reactions and affect the endocrine system. USEPA (2000a) considers malathion to have evidence suggestive of carcinogenicity, but not sufficient evidence to assess human carcinogenic potential. IARC and ATSDR consider the evidence to be insufficient to determine carcinogenic potential (CA-SCDHS, 2005).

PBO – Ecotoxicity

PBO's synergistic action is due to its ability to inhibit metabolic enzyme activity in insects, thereby allowing the active ingredients to remain available and cause enhanced toxic effects (CA-IC, 2004). Overall, PBO has limited toxicity to terrestrial wildlife. It is considered to be moderately to acutely toxic in fish, and highly toxic in aquatic invertebrates (CA-IC, 2004).

PBO – Human Health

As in insects, PBO also can inhibit metabolic enzyme in mammals, however higher doses are required relative to insects (CA-SCDHS, 2005).

Studies in animals indicate the liver to be the primary target organ for toxicity. Exposures through ingestion and inhalation have been shown to lead to enlarged livers, and at some doses, enlarged kidneys in laboratory animals. Developmental and reproductive effects, including behavior effects in offspring and fetotoxicity, have been noted in animal studies; however these effects have been observed at relatively high doses. No information on developmental or reproductive effects in humans was found. Some studies in animals indicate that PBO depletes immune system T-cells in the spleen and thymus. These immune system cells have been implicated in some autoimmune diseases, such as multiple sclerosis (CA-SCDHS, 2005).

IARC considers there to be insufficient evidence to classify PBO as to its carcinogenic potential. USEPA classifies PBO as a probable carcinogen (CA-SCDHS, 2005).

7.9.2.1.6 Human Health Risk Assessment

The human health risk assessment (HHRA) was conducted to evaluate the potential for adverse health effects in people from use of the target pesticides. The HHRA was conducted using methods, protocols, and data developed by national and international agencies and organizations. Overall, the evaluation adopted the general risk assessment paradigm developed by the NAS in 1981, and relied largely on protocols and procedures developed by USEPA to assess exposure, toxicity, and risk (e.g., USEPA, 1989; USEPA, 1991b; USEPA, 1997a; USEPA, 2002d; USEPA, 2003a; USEPA, 2003b; USEPA,2004a; USEPA, 2004b; USEPA, 2004c; USEPA, 2004d,; USEPA, 2004e). In addition, the risk assessment utilized toxicity data and evaluations conducted by expert non-regulatory organizations, including the Agency for Toxic Substances and Disease Registry (e.g., ATSDR 2005) under CDC, WHO (1984, 1999, 2005), and the International Agency for the Research of Cancer (IARC, 1991).

The conceptual model developed jointly for human and ecological receptors was used as the starting foundation of the human health risk assessment. That model showed that target pesticides could be released and move throughout the environment and potentially reach residents, workers, and recreational users in Suffolk County. From this broad conceptualization, additional analyses were conducted to quantify the potential exposures in these receptor groups, define toxic response as a function of exposure, and characterize risk as a function exposure and toxicity.

Methods

Human health risks were evaluated for short-term (acute) exposures immediately following an application event, and for longer-term exposures that hypothetically could occur for an extended period of time after such an event. Acute exposures were modeled to represent instantaneous maximum concentrations. Longer-term exposures were evaluated to represent concentrations that occur during the five month application season, taking into account degradation and other fate processes.

Both acute and chronic risks were assessed using USEPA methods. Under this approach, risk assessment consists of:

1) exposure assessment

2) dose-response assessment

3) risk characterization

An exposure assessment characterizes the likelihood and extent of potential human contact with target chemical in the environment.

USEPA (1989) uses the concept of exposure pathways to characterize the way in which a chemical can move through the environment and contact a receptor population. For an exposure pathway to be considered "complete" there must be a source, a release mechanism of the constituent from the source, a transport medium, a potentially exposed population (receptor), an exposure point, and an exposure route. An exposure pathway that does not contain each of these elements is incomplete, and is not evaluated in risk assessment.

The conceptual model detailed in the previous section identified the source of the target pesticides (i.e., vector control activities), and discussed transport in a variety of environmental media. The other elements of exposure pathways are discussed below.

The HHRA evaluated potential risks to a total of 13 receptor populations across four distinct age groups:

- birth to less than six years ("young child")
- six to less than 12 years ("older child")
- 12 to less than 18 years ("adolescent")
- greater than or equal to 18 years ("adult").

The three child groupings allow the evaluation of three critical developmental timeframes of equal length, and are appropriate and conservative based on USEPA guidance (2003a) and on the specific characteristics of this exposure assessment.

The receptor populations evaluated were:

Acute

- Young child and adult residents

Longer-term

- Young child and adult residents
- Young child, older child, adolescent and adult park visitors
- Adult community gardeners
- Older child and adolescent school attendants
- Adult school workers
- Adult homeless
- Adult commercial/industrial workers
- Adult public workers.

These receptor groups were assumed to occur in each of the study areas evaluated and are considered representative of receptor populations across the County as a whole. Other receptor populations potentially exist; however, these receptors are subsets of the receptors identified or would have lower exposures. For example, while boaters and swimmers could be viewed as additional receptor populations, these receptors are actually subsets of the park visitor group. Similarly, exposures to older child and adolescent residents were not specifically examined, as these receptors would have lower exposures than young child residents.

Potential exposure media were identified based on consideration of the release of the target pesticides into air followed by the likely deposition onto surfaces, and subsequent transport.

Acute exposures were evaluated for inhalation of aerosols during an application event and dermal contact with turf immediately following the application, for adults and children. Additional child exposures were evaluated for incidental ingestion exposures as a result of post-spray hand-to-mouth, object-to-mouth, and soil ingestion.

For longer-term exposures, the exposure media considered were soil, non-organic surfaces (e.g., lawn furniture, playground equipment), tap water, swimming/wading water, produce, fish/shellfish, and air. The exposure routes are ingestion, dermal contact, and inhalation.

For the purpose of this evaluation, the tap water assessment assumed that drinking water in the County is obtained from a surface water source. County drinking water is in fact obtained from groundwater, and given that none of the target adulticides is likely to be transported to groundwater, the tap water pathway is likely not complete. It was included here more as an upper-bound evaluation of potential changes in the quality of surface water to serve as a drinking water source, rather than as representative of potential human health risks. Tap water ingestion and dermal contact were evaluated. Inhalation exposures during in-home tap water use were not evaluated because none of the target pesticides is sufficiently volatile.

All exposure media and exposure routes were assumed to occur in each study area, although in reality, this might not be the case. For example, fishing/shellfishing, swimming, or home or community gardens might not exist in each study area. They were nonetheless evaluated here for each study area to provide theoretical upper bound estimates of risk and so that the study area-specific risks could be used as representative surrogates for risks in other parts of the County (where any and all exposure pathways could exist) that could be subject to target pesticide application in the future.

Table 7-16 presents the acute and chronic exposure pathways that were evaluated in this assessment.

Table 7-16. - Exposure Pathways Evaluated in HHRA

Exposure Medium & Route	Resident		Park Visitor				Com. Gardener	School Attendant / Worker			Homeless	Worker	
	Young child	Adult	Young child	Older child	Adolescent	Adult	Adult	Older Child	Adolescent	Adult	Adult	Adult comm./ ind.	Adult public
Incidental ingestion of surface soil	?	?	?	?	?	?	?	?	?	?	?	?	?
Dermal contact with surface soil	?	?	?	?	?	?	?	?	?	?	?	?	?
Ingestion of residues on hands via surfaces	?	?	?	?	?	?	?	?	?	?	?	?	?
Dermal contact with residues on surfaces	?	?	?	?	?	?	?	?	?	?	?	?	?
Ingestion of tap water	?	?	--	--	--	--	--	--	--	--	?	--	--
Dermal contact with tap water	?	?	--	--	--	--	--	--	--	--	--	--	--
Ingestion of swimming/wading water	?	?	?	?	?	?	--	--	--	--	--	--	--
Dermal contact with swimming/ wading water	?	?	?	?	?	?	--	--	--	--	?	--	?
Ingestion of produce	?	?	--	--	--	--	?	--	--	--	?	--	--
Ingestion of fish/ shellfish	?	?	--	--	--	--	--	--	--	--	--	--	--
Inhalation of residues on particulates in air	?	?	?	?	?	?	?	?	?	?	?	?	?
Inhalation of aerosols (Acute only)	?	?	--	--	--	--	--	--	--	--	--	--	--
Oral ingestion of spray on soils/objects (Acute – 3 separate pathways)	?	--	--	--	--	--	--	--	--	--	--	--	--
Dermal contact with spray (Acute)	?	?	--	--	--	--	--	--	--	--	--	--	--

Human intake resulting from exposures to the target pesticides were estimated using exposure algorithms and assumptions developed by the USEPA.

Acute dose calculations for inhalation exposures were assessed using the same equations and exposure algorithms discussed below for longer-term exposures. Acute exposures via other media were evaluated based on application rate using USEPA methods. HHRA Appendix C (in Cashin Associates, 2005c) presents acute intake equations and assumptions.

Dose associated with longer-term exposures were calculated as average daily intake, which represent the daily dose of a chemical taken into the body, averaged over the appropriate exposure duration. For exposures via ingestion or dermal absorption, doses are typically expressed in milligrams chemical (mg) per kilogram body weight (kg) per day (mg/kg-day).

The primary sources for the exposure algorithms selected for this HHRA are USEPA risk assessment guidance documents (USEPA, 1989; USEPA, 2004c; USEPA, 2004d). The generalized equation for calculating chemical intakes via ingestion or dermal absorption for the longer-term pathways in this risk assessment is as follows:

$$ADD = \frac{EPC * CR * EF * ED * FI}{BW * AT * CF}$$

whereby

ADD	=	average daily dose, the amount of chemical taken in by the receptor (mg chemical per kg body weight/day)
EPC	=	exposure point concentration, the chemical concentration contacted over the exposure period at the exposure point (e.g., mg chemical/kg soil)
CR	=	contact rate, the amount of affected medium contacted per unit time or event (e.g., soil ingestion rate, mg soil/day)
EF	=	exposure frequency, how often an exposure occurs (days/year)
ED	=	exposure duration, how long an exposure occurs (years)

FI = fractional intake, fraction of medium contacted that is assumed to be from the potentially contaminated source (unit less)

BW = body weight, the average body weight over the exposure period (kg)

AT = averaging time, period over which exposure is averaged (days)

CF = conversion factor (e.g., 10^{-3} kg soil/mg soil).

The parameters shown in the above equation are called exposure factors or exposure parameters. The values assumed for any given factor/parameter vary depending on the receptor population being evaluated. For some exposure pathways, the equation may also vary from the generalized format shown above to include parameters that describe chemical-specific features.

For example, for the inhalation pathway, exposure is not a simple function of the inhalation rate and body weight (USEPA, 2004c). The physicochemical characteristics of the inhaled agents are key determinants to their interaction with the respiratory tract and ultimate deposition. Therefore, current USEPA methodology uses the principals of inhalation dosimetry to determine the human equivalent concentration (HEC) for evaluating inhalation risk and calculating inhalation toxicological criteria. According to these procedures, it is unnecessary to calculate inhaled dose, but instead to evaluate health concerns, inhalation risk assessments require only an average air concentration adjusted to continuous exposure (USEPA, 2004c). Inhalation exposures were evaluated by adjusting the modeled concentration of the chemical in the inspired air to a continuous exposure, yielding an effective exposure concentration (EEC). The generalized equation used to calculate longer-term EECs is as follows:

$$EEC = \frac{EPC * EF * ED}{AT}$$

whereby

EEC = effective exposure concentration (mg/m^3)

EPC = exposure point concentration, the chemical concentration in particulates in air over the exposure period at the exposure point (mg/m^3)

EF = exposure frequency for air (days/year)

ED = exposure duration (years)

AT = averaging time (days).

Two exposure calculations were developed in this assessment to satisfy the County's request to assess exposures for a most exposed individual and a more typical individual. For evaluation purposes and to be consistent with USEPA terminology, the varying exposure assumptions associated with these two exposure scenarios are termed the reasonable maximum exposure (RME) case and the central tendency exposure (CTE) case. RME and CTE exposures are often dually evaluated in risk assessments to bound the uncertainty that is inherent in quantitative estimates of risk. For RME scenarios, the values used to calculate exposures include values that represent the high end of the range of all possible values. CTE exposure parameters are typically based on averages or means derived from a range of values.

Exposure pathway-specific intake equations and all parameter assumptions used to calculate longer-term exposures are detailed in HHRA Appendices D, E and G (all found in Cashin Associates, 2005c).

The longer-term exposure assessment employed some screening techniques to identify the receptors and pathways that could be associated with the greatest risk. This was done to focus the assessment on those elements most relevant to defining the magnitude of potential risks associated with vector control pesticide use and to facilitate a more concise presentation of the risk assessment results without eliminating any potentially important and risk-driving receptors or pathways. Two screening approaches were employed.

Potential risks in adult and child residents associated with longer-term risks were evaluated for all relevant exposure pathways identified in Table 7-15. These receptor groups and pathway combinations were fully evaluated since risks in these two groups are always some of the highest risks evaluated for any receptor group. For all other receptor groups and pathways identified in

the table, however, a screening technique was first employed to focus the longer-term risk calculations in this HHRA. Risks in these other groups were evaluated by comparing potential dose in these receptor groups compared to child residents (adult resident exposures are lower than child resident). A quantitative scaling factor was derived for each receptor group-pathway combination that represents the relative potential dose compared to a child resident. Any receptor-pathway combination for which the relative potential dose exceeded a factor of one was selected for additional quantitative evaluation along with child and adult residents. Any receptor-pathway combination, for which the relative potential dose was less than one, was not selected for further quantitative assessment. Instead, risks associated with these other receptor-pathway combinations can be calculated by applying the relative potential dose scaling factor to the risks calculated for a child resident for a comparable pathway. In this way, all 90 longer-term exposure pathways are evaluated, but without the need to present nearly 3,000 individual exposure and risk tables.[4]

Table 7-17 summarizes the results of the relative potential dose and identifies the receptor groups, in addition to child and adult residents that are were carried through subsequent quantitative risk calculations. Risks in other groups are evaluated by analogy to child resident risks.

[4] [90 pathways]*[4 study areas]*[4 adulticides]*[2 exposure cases] = 2,880 exposure and risk tables.

Table 7-17. Relative Potential Dose Summary: Ratios of Potential Doses for Non-residential Receptors in RME Scenarios as Compared to Young Child Resident RME Potential Doses.

	Young child park visitor	Older child park visitor	Adolescent park visitor	Adult park visitor	Adult community gardener	Older school child	Adolescent school child	Adult school worker	Adult homeless	Adult commercial/ industrial worker	Adult public worker
Incidental ingestion of surface soil	43%	19%	5%	2%	1%	23%	7%	3%	5%	4%	4%
Dermal contact with surface soil	43%	30%	39%	7%	5%	23%	47%	9%	17%	35%	35%
Incidental ingestion of residues on hands via surfaces	43%	31%	26%	22%	15%	39%	32%	27%	52%	37%	37%
Dermal contact with surfaces	43%	31%	26%	22%	15%	39%	32%	27%	52%	37%	37%
Ingestion of tap water	--	--	--	--	--	--	--	--	37%	--	--
Dermal contact with tap water	--	--	--	--	--	--	--	--	--	--	--
Ingestion of swimming water	151%	67%	38%	31%	--	--	--	--	--	--	--
Dermal contact with swimming/ wading water	151%	111%	89%	84%	--	--	--	--	43%	30%	--
Ingestion of produce	--	--	--	--	309%	--	--	--	35%	--	--
Ingestion of fish/shellfish	--	--	--	--	--	--	--	--	--	--	--
Inhalation of particulates	43%	43%	43%	43%	28%	53%	53%	53%	100%	71%	71%

Notes:
-- = Incomplete or negligible exposure pathway.
.

For all the selected pathways, a phased assessment approach was additionally used to assess longer-term human health risks associated with potential exposure to target pesticides. A first phase (Tier I) analysis was conducted as a screening-level evaluation to identify those receptors, exposures, and target pesticides that would not be associated with unacceptable chronic human health risks under any situation. The Tier I analysis employed bounding assumptions regarding chemical fate, transport, persistence and human contact patterns to estimate exposures. The second phase (Tier II) was conducted for any receptor, exposure, and pesticide that could not be eliminated through Tier I analysis. The Tier II analysis employed more realistic assumptions to calculate exposures.

A series of assumptions were made to ensure that Tier I risks calculations provided upper-bound estimates of longer-term risk. Importantly, this analysis used EPCs that represented the highest accumulated residue predicted to occur throughout the application season, and assumed that a person would be exposed to that maximum residue throughout the entire season,[5] and only at that single location from the study area (one out of more than 220 modeled locations) where the predicted concentration would be highest. All lower predicted concentrations from the remainder of the study area were ignored. Further, the Tier I analysis used modeled air deposition rates calculated for Davis Park, which were higher than those calculated for any other study area. Davis Park also had the highest number of pesticide applications and frequency – up to 14 events conducted every seven days, which ensured that the calculated EPCs would be higher than those similarly calculated for any other study area.

Given this approach, the calculated longer-term risks for all pathways, receptors, and chemicals considered in this assessment will not exceed the Tier I estimate. Therefore, if a chemical or pathway did not pose an unacceptable risk during the Tier I analysis, it was concluded that it does not represent a longer-term risk to human health under any condition or in any study area, and thus was not evaluated further.

Those pathways or chemicals that remained were selected for further (Tier II) analysis, employing more realistic and representative assumptions regarding longer-term exposure.

[5] Vector control activities are focused on a three to five month period because this is the period of greatest mosquito abundance and human health risk.

EPCs are the concentrations of the target pesticides to which people might be exposed. They are calculated to represent the target pesticide concentration at the exposure point throughout the duration of exposure.

The air dispersion and deposition modeling conducted by RTP was the starting basis for the acute inhalation EPC and all longer-term EPC calculations. Both hourly air concentration data and deposition data were provided for each of the target adulticides and the synergist PBO. Results were generated by RTP for each study area to include the study area and quarter mile buffer zones surrounding each of the study areas. In this way, both target application impacts as well as off-target drift impacts could be addressed in the risk assessment. Results were modeled for individual model receptor points located across the entire study area and buffer zone. More than 200 model receptor points were defined for each study area/buffer zone combination.

Raw data sets generated by RTP consisted of all model results for each modeled receptor location (e.g., 200) and each modeled hour (i.e., 1,553 total modeled hours), resulting in more than 300,000 individual model predictions. These data offer insight on impact conditions across the study areas and buffer areas, and provide the basis for the characterization of spatial and temporal averages.

Acute EPCs were calculated using one hour post-spray concentrations modeled by RTP for inhalation exposures and, consistent with USEPA pesticide assessment methods (e.g., USEPA, 2000a), the product application rates to assess dermal and ingestion pathways. This inhalation EPC was based on the maximum modeled air concentration across all modeled receptor locations and across all modeled times. Thus, this was the maximum value reported out of the more than 300,000 individual modeled results. These maximum one hour values represent the worst-case hourly meteorological condition. Inhalation EPCs and risks were evaluated for each of the individual study areas. Other acute exposure pathways were modeled based on application rate and therefore, are not study-area specific. HHRA Appendix C (Cashin Associates, 2005c) presents details all the methods and assumptions used to quantify acute exposures.

For the Tier I and Tier II analysis of chronic exposure, longer-term EPCs were calculated as the concentrations that could occur throughout the spray season, taking into account degradation, multiple spray events, fate and transport processes, and the specific spray period associated with

each study area. As mentioned above, for the Tier I analysis, the EPC represented the highest accumulated residue predicted to occur at any time during the spray season. For the Tier II analysis, the EPC was the calculated average concentration that could occur throughout the spray period.

The RTP model results were categorized in two ways to support calculation of longer-term EPCs.

- Maximum-average one hour concentration (Max-avg). This EPC was based on the temporal average of 1,553 model-predicted one hour air concentrations and deposition values for each modeled receptor location. The highest calculated temporal average was referred to as the maximum-average (max-avg) EPC.
- Study area-average. This EPC was based on the temporal and spatial average of all modeled values for a given study area across all locations and all times.

The maximum one hour average value noted above (Max-avg) was used as the starting point for calculation of RME exposures for the Tier I and Tier II analyses of longer-term risk. The study-area average was used as the starting point for calculation of CTE exposures for the Tier II analysis of longer-term risks.

HHRA Appendix A (Cashin Associates, 2005a) presents a detailed summary of the modeling approaches used to calculate EPCs for longer-term exposures.

Dose-response assessment in human health risk assessment most typically relies on the use of numerical toxicity criteria against which predicted exposures can be evaluated. Toxicity criteria can be developed for the evaluation of both carcinogenic and non-carcinogenic risks, and are typically established specific to the pathways by which a receptor population may be exposed (USEPA, 1989). This assessment focused on evaluation of non-cancer risks. As discussed earlier (and see CA-SCDHS [2005]), the collective data do not provide strong evidence that the target adulticides or synergist are human carcinogens, and a variety of organizations, including USEPA, WHO, IARC, and ATSDR have historically recommended that these compounds not be evaluated for carcinogenicity.

Recently (August 31, 2005), however, USEPA (2005b) released a preliminary draft of a RED and accompanying risk assessment for permethrin in which the pesticide was evaluated for carcinogenic effects. This recent USEPA risk assessment is provisional and has not been finalized or subject to public review or comment. For these reasons, this HHRA focuses on evaluation of permethrin for non-cancer effects. The potential for carcinogenic effects is evaluated and discussed in later sections of this HHRA, as part of the overall uncertainty analysis.

In the case of exposure via either ingestion or dermal absorption, non-carcinogenic risks are typically evaluated by comparing the estimated ingested or absorbed chemical dose to a reference dose (RfD), below which adverse health effects are not expected (USEPA, 1993a). Similarly, non-carcinogenic risks from exposure to inhaled chemicals are typically evaluated by comparing the concentration of the chemical in the air to a reference concentration (RfC), which represents an air concentration to which a population could be exposed on a daily basis without experiencing any adverse health effects.

RfDs and RfCs are most commonly developed from experimental animal studies where laboratory animals are exposed to a range of doses or concentrations of a specific chemical. The lowest dose or concentration resulting in an adverse health effect is termed the Lowest-Observable-Adverse-Effect-Level (LOAEL), and usually corresponds to the most sensitive toxic effect in the most sensitive species tested. One dosage or concentration level lower (i.e., the highest tested dose or concentration not resulting in any adverse health effects) is termed the No-Observable-Adverse-Effect-Level (NOAEL), and it is this NOAEL that typically is used as the basis for RfD/RfC selection. RfDs and RfCs are derived from the NOAEL by incorporating appropriate safety and uncertainty factors. Often, NOAELs are adjusted by two ten-fold factors, one that accounts for uncertainty in extrapolating from animals to humans (interspecies variability) and one that accounts for varying sensitivity within the human population (intraspecies variability). This typically results in an RfD/RfC that is two orders of magnitude smaller (more protective) than the NOAEL. Other uncertainty factors may also be applied as appropriate. For example, if adverse effects occur at the lowest tested dose or concentration, a ten-fold uncertainty factor may be employed to extrapolate from the LOAEL to the NOAEL. Additionally, if there is evidence to suggest that developing fetuses or newborns are particularly

sensitive to the effects of a chemical, an additional ten-fold safety factor may be applied to ensure protectiveness for this population.

For certain chemicals, the USEPA does not recommend using RfDs/RfCs to evaluate non-cancer risk, but instead utilizes a Margin of Exposure (MOE) approach. A MOE is the ratio of a NOAEL to an estimated dose/exposure level (USEPA, 1989). The calculated MOE is compared to a target MOE which reflects the assumed safe level of exposure. If the calculated MOE is equal to or greater than the target MOE, then the exposure is not expected to result in adverse health effects. USEPA uses an MOE approach for evaluation of PBO inhalation exposures and acute malathion exposures.

Non-carcinogenic toxicity criteria are developed specific to the amount of time that a receptor could ingest, absorb, or inhale a chemical. Three distinct timeframes are typically considered. For short exposures (less than one to two weeks), the dose or concentration of a chemical is compared to an acute RfD, or RfC (or NOAEL for MOE chemicals), a value usually developed based on an animal study with a similarly short duration (USEPA, 1989). For intermediate length exposures (those lasting more than a few weeks but less than seven years), subchronic RfDs and RfCs are employed (USEPA, 1989; USEPA, 2004c). These intermediate length criteria are typically developed from animal studies occurring for 10 percent of the animals' lifespan. Finally, longer exposures (those lasting between seven years and a lifetime), are compared to chronic toxicity criteria, which are developed from animal studies occurring for time periods of between three months up to the entire length of the animal's lifespan (USEPA, 1989).

In this assessment, acute RfDs and RfCs (and NOAELs for MOE chemicals) were used to assess risks following short-term exposures immediately following an application. Ideally, to assess these types of immediate post-application risks, the acute criteria would be derived from studies that examined exposures that occurred for a few hours. However, most of the acute toxicity criteria were based on studies of longer duration, and as such, probably overestimate potential acute toxicity for the lengths of acute exposures examined in this HHRA. Given the short atmospheric residence times of these chemicals, in many cases the test animals were exposed to the chemicals for much longer than could actually occur following an application. For example,

the resmethrin acute toxicity value was based on a 90 day study in laboratory animals. Higher dosages would likely have been required to result in the same health effects over the shorter, more realistic exposure durations.

Chronic toxicity criteria were used to assess longer-term exposures across all pathways, even in those situations (e.g., childhood exposures) where the predicted exposure duration is less than seven years. This is a conservative (protective) approach because subchronic RfDs are typically up to a factor of 10 higher than chronic RfDs.

Table 7-18 presents the toxicity criteria used in the HHRA. HHRA Appendix B (Cashin Associates, 2005c) provides a complete description of the basis of these values.

Table 7-18. Non-Cancer Toxicity Criteria

		Malathion	Permethrin	Resmethrin	Sumithrin	Piperonyl Butoxide
Ingestion	Acute	MOE = 100; NOAEL = 50 mg/kg-d	0.3 mg/kg-day	0.1 mg/kg-day	0.55 mg/kg-day	6.3 mg/kg-day
	Chronic	0.02 mg/kg-day	0.05 mg/kg-day	0.03 mg/kg-day	0.071 mg/kg-day	0.16 mg/kg-day
Dermal	Acute	MOE = 100; NOAEL = 50 mg/kg-d	0.3 mg/kg-day	10 mg/kg-day	0.55 mg/kg-day	--
	Chronic	0.02 mg/kg-day	0.05 mg/kg-day	0.03 mg/kg-day	0.071 mg/kg-day	--
Inhalation	Acute	MOE = 1000; LOAEL = 25.8 mg/kg-d	1.05 mg/m^3	0.1 mg/m^3	0.26 mg/m^3	MOE = 100; NOAEL = 630 mg/kg-day
	Chronic	0.02 mg/m^3	0.18 mg/m^3	0.01 mg/m^3	0.25 mg/m^3	MOE = 1000; NOAEL = 4 mg/kg-day

-- = USEPA (2005a) recommends that dermal PBO exposure not be evaluated due to a lack of systemic toxic effects following dermal exposures.

Risks for non-carcinogenic effects are calculated by comparing the estimated exposure to the toxicity criterion. The approach is called the Hazard Quotient (HQ) approach. The HQ is simply the ratio of the predicted exposure to the toxicity criterion. HQs less than one indicate that health effects are unlikely under the conditions evaluated. HQs greater than one indicate an increased potential for health effects.

For exposures via ingestion and dermal absorption, HQs were calculated as follows:

$$HQ = \frac{ADD}{RfD}$$

For exposures via inhalation, HQs were calculated as the quotient of the EEC and the corresponding chronic daily reference concentration (RfC):

$$HQ = \frac{EED}{RfC}$$

For those chemicals evaluated using an MOE, a threshold is set for safe exposure, termed the target MOE. The target MOE is then compared to the actual MOE, which is defined as the ratio of the appropriate toxicological endpoint (e.g., NOAEL, LOAEL) to the average daily dose or effective exposure concentration (USEPA, 1989). For example:

$$MOE = \frac{NOAEL}{EEC}$$

If the actual MOE is greater than the target MOE, human health risks are unlikely. In this way, the MOE is analogous to the HQ.

Consistent with USEPA (2000a, 2005a) recommendations, acute and chronic inhalation and acute ingestion exposures to PBO and all acute exposures to malathion were evaluated using an MOE approach To evaluate these exposures, the ratio of the estimated (EEC) was compared to appropriate toxicity criterion (see HHRA Appendix B, in Cashin Associates [2005c]). The resulting MOE was then compared to the target MOE.

HQs were calculated for each adulticide/synergist product:

- Sumithrin + PBO
- Resmethrin + PBO

- Permethrin + PBO
- Malathion.

Because a receptor population would be exposed to the parent pyrethroid and PBO simultaneously, HQs for each pyrethroid and PBO product were added for each pathway to yield a product-specific HQ representing the entire mixture, as opposed to the individual agent. The summation is complicated by the fact that pyrethroids health effects are evaluated using an RfD to calculate an HQ, while PBO was evaluated using the MOE approach with a NOAEL. As discussed earlier, the RfD and RfC values directly incorporate any assumptions regarding uncertainty or modifying factors in the toxicity study used to derive the value. The MOE analysis uses a NOAEL value and accounts for uncertainty in the toxicity study by setting a target MOE value. An effective HQ was calculated for PBO to provide a common risk metric to combine the hazard results. An effective HQ for PBO was calculated for each pathway as follows:

$$HQ_{effective} = \frac{MOE_{Target}}{MOE_{Predicted}} = \frac{MOE_{Target}}{\frac{NOAEL}{ADD}}$$

The pyrethroid HQ and the PBO $HQ_{effective}$ were then combined to derive the HQ for the pesticide formulation.

HQs for individual chemicals typically are not summed unless the individual compounds are known to operate via a similar mechanism of action (USEPA, 1989). Because PBO operates via a different mechanism than the parent pyrethroid, this assumption of additivity might overestimate potential risk. Theoretically, this approach also could underestimate risks, given that PBO enhances pyrethroid toxicity for neurological endpoints. Too few data are available to more fully evaluate these potential interactions in humans.

Risks were calculated assuming that each of these products was used exclusively throughout the spray season and not in combination with any other product. Therefore, HQs are not summed across adulticide products.

In the Tier II analysis, HQs were summed across all pathways, under the assumption that a receptor could be exposed via all exposure pathways.[6] This is a very conservative assumption, because no one receptor is likely to be exposed across all theoretically possible exposure pathways.

In this assessment, the sum of the HQs is termed a Hazard Index (HI) and is calculated as follows:

$$HI = HQ_1 + HQ_2 + HQ_3 + HQ_4 + ... \text{HQ}_n$$

whereby

HQ_1 = hazard quotient for pathway 1 (e.g., ingestion via soil) (unit less)

HQ_2 = hazard quotient for pathways 2 (e.g., dermal absorption via soil) (unit less)

HQ_3 = hazard quotient for pathway 3 (e.g., ingestion of residues on hands via surfaces) (unit less)

HQ_4 = hazard quotient for pathway 4 (e.g., dermal absorption of residues on hands via surfaces) (unit less)

HQ_n = hazard quotient for the n^{th} pathway (unit less).

HIs that are less than one indicate that human health risks are unlikely. HIs greater than one indicate that there may be concern for potential health effects under the conditions of exposure evaluated (USEPA, 1989). There are a number of limitations to this approach. Individual HQs are derived based on critical effects of varying toxicological significance (USEPA, 1989). However, summation of these values gives equal weight to each potential endpoint. In addition, exposures via differing routes (e.g., dermal absorption, inhalation) could result in effects occurring by different mechanisms. Summing together risks that occur via different mechanisms could lead to an overestimation of potential for effects. Therefore, the existence of an HI greater than one does not necessarily indicate a true increase risk of adverse health effects. In addition, simultaneous exposure via all pathways is considered highly unlikely for any receptor, and

[6] Pathway-specific HQs were not summed in the Tier I analysis, because the focus of that analysis was to identify the particular receptors and target pesticide combinations that were most likely to be associated with unacceptable risks and that should be further evaluated in Tier II. Furthermore, the bounding nature of the Tier I risk calculations makes calculation of an HI inappropriate.

therefore is considered a conservative assessment approach. Potential risks for any individual are likely well below those estimated here.

Quantitative Risk Results

Quantitative risk results were generated for acute risk exposure scenarios, as well as longer-term exposure scenarios under a screening-level assessment (Tier I) and a more refined assessment (Tier II).

Acute Risks

Acute risks were evaluated for inhalation exposures of aerosols during an application, dermal contact with turf following an application, and incidental ingestion exposures as a result of post-application hand-to-mouth, object-to-mouth, and soil ingestion behavior in children. The results of the acute risk assessment are provided in HHRA Appendix C (Cashin Associates, 2005c).

The acute risk assessment indicates that none of the target pesticides is likely to cause any health effect following short-term exposures during and immediately after an application.

Tier I Analysis

The Tier I analysis utilized EPCs derived for Davis Park, as these were the highest of the four study areas (see HHRA Appendix A [Cashin Associates, 2005c]). Risks for all other study areas will thus be lower than those predicted for Davis Park in the Tier I analysis. HHRA Appendix F (Cashin Associates, 2005c) presents a more complete description of the Tier I analysis and presents all inputs and results. Table 7-19 summarizes the HQ results.

Table 7-19. Summary of Pathway-Specific HQs > 1 from the Tier I Analysis

	Resident		Park Visitor		Community Gardener
	Young child	Adult	Young child	Older child	
Malathion	2 - ingestion-surface residues 2– ingestion of produce	--	--	--	7 – ingestion of produce
Permethrin + PBO (Permanone)	--	--	--	--	--
Resmethrin + PBO (Scourge)	--	--	--	--	--
Sumithrin + PBO (Anvil)	--	--	--	--	--

-- = no pathway HQ > 1

The Tier 1 analysis showed the following:

- None of the synthetic pyrethroid products in combination with PBO poses a chronic human health risk under the bounding conditions associated with the evaluation management plan
- Malathion Tier I HQs exceeded 1 for the young child resident and the community gardener.
    - Young child resident risks were due to potential produce ingestion and ingestion of residues on hands via surfaces
    - Community gardener risks were driven by produce ingestion.

Based on this Tier I analysis, potential malathion exposures in young child residents and community gardeners were selected for further evaluation in a Tier II analysis.

Synthetic pyrethroids were assessed not to pose a human health risk and were not evaluated further.

Tier II Analysis

Based on the results of that Tier I evaluation, the Tier II evaluation focused on malathion exposures to young child residents and community gardeners.

HHRA Appendix G (Cashin Associates, 2005c) presents a more complete description of the Tier II analysis and presents all inputs and results. Table 7-20 summarizes the HI results.

Table 7-20. Summary of Hazard Indices from the Tier II Analysis

Chemical	Young child resident		Community gardener	
Study Area	CTE	RME	CTE	RME
Malathion				
Dix Hills	0.01	0.03	0.01	0.06
Manorville	0.01	0.03	0.01	0.06
Davis Park	0.3	1	0.4	3
Mastic-Shirley	0.1	0.4	0.2	0.9

Overall, the Tier II analysis indicates that malathion does not pose a significant health threat to study area receptors. No unacceptable health risks were predicted for any receptor group in any study area under the CTE conditions. Under RME conditions, HIs were less than 1 for all study

areas receptors and pathways except for the community gardener at Davis Park, where an HI of 3 was predicted due to produce ingestion. This risk was predicted based on modeled air deposition of malathion from one out of more than 200 modeling receptor points, and is not truly representative of study area exposure conditions. More typical exposures throughout the remainder of the study area are not predicted to be associated with a human health risk. Even at this one location, the predicted risk could be easily mitigated by simply washing produce prior to consumption.

Uncertainty Evaluation

As in any risk assessment, there are uncertainties associated the predictions presented here regarding the probability or magnitude of adverse health effects. The predicted risks are based on many assumptions about ways in which pesticides can persist and move in the environment and how people might contact them. Many of these assumptions are based on general scientific studies, but some uncertainty still exists regarding how accurately the available data reflect the ways in which residents could actually be exposed to the target pesticides.

The accuracy of a risk assessment is described in an uncertainty evaluation. The uncertainty evaluation includes a list of critical assumptions and an evaluation of the possibility that the assumptions used in the risk assessment may over-predict risk or under-predict risk. The key sources of uncertainty in this risk assessment and their potential impact on the risk assessment are outlined below.

Exposure Pathways

This assessment examined a suite of possible exposure scenarios for a range of potentially exposed populations in each study area selected for evaluation. This was done universally, regardless of whether the exposure was deemed likely to occur in that particular study area. For example, fishing/shellfishing, swimming, or home or community gardens might not exist in each study area, but were nonetheless evaluated. This was done to provide theoretical upper bound risk estimates so that the study-area specific risks could be used as representative surrogates for risks in other parts of the County (where any and all exposure pathways could exist) that could be subject to target pesticide application in the future. Risks for any one study area would need to be defined based on a representative survey of the particular activity patterns within the study area rather than on general assumptions. Given that no one study area is likely to support all

assumed activities, the risks presented here likely overestimate risks for any given study area. The overall magnitude of overestimate could be high or low, depending on activity patterns in the study area.

In the Tier II assessment, the assumption was also made that any one receptor could be exposed via all possible pathways during each application season. This is a very conservative assumption, because no one receptor is likely to be exposed across all theoretically possible exposure pathways, and therefore, the Tier II risk results likely overestimate risk. This likely has an overall low impact on the results reported here, however, given that the risks were driven largely by only a few pathways (e.g., produce ingestion).

Exposure Intake Parameters

A number of assumptions were made in this risk assessment about how often and how much people could contact the target pesticides. For any parameter, there is a range of possible values. This is tied directly to the inherent variability of our world. In this assessment, variability in exposure intake parameters was addressed in the acute risk scenario and in the Tier I longer-term evaluation by selecting the upper end of the range of possible exposure parameter values. For the Tier II assessment, variability in exposure parameters was addressed by selecting parameter values that represent different points in the distribution of all possible value (CTE and RME). This approach is commonly used in exposure assessment to bound the range of possible exposures. It is probable the RME estimates overestimate exposure for the majority of the population, but are potentially representative of exposures in a small proportion of the population.

Air Modeling

Uncertainty is inherent in any modeling process, and therefore, identifying areas of uncertainty is critical in understanding the model results and limitations. In air dispersion modeling, the accuracy is limited by the input data and the ability of the model to characterize the transport, dispersion, and deposition of emissions to the atmosphere.

RTP conducted air dispersion and deposition modeling of target pesticide aerial application using a combination two models – AgDISP and ISCST3 models. A field study was conducted in August 2004 to verify model predicted results for a helicopter ULV spraying of Scourge.

Because analytical detection limits for the field collected samples were not sufficiently low, the model-predicted air concentrations could not be verified with the field collected data. For deposition, the model-predicted values at each deposition sampler location show inconsistent results. In most cases, the models over-estimated deposition by at least a factor of two, and up to a factor of 10 higher for any given model receptor location, with an average over-prediction of about three. As a result, subsequent exposures and risks based on deposition modeling for spray application of the vector control agents could be overestimated by a similar magnitude. There were instances, however, where the model under-predicted deposition for a given location. In the HHRA, the temporal maximum air concentration and deposition reported for any given receptor location was used as the basis of the RME risk calculations under the Tier II assessment. Based on RTPs verification data, it is possible that the results for any given location could be higher or lower than those predicted. In most cases, RTP's verification data suggest that this would be expected to lead to an over-estimate of risk, but it is possible that at any given location, the true concentration and deposition could be underestimated. This source of uncertainty cannot be further evaluated without additional field collected data.

The degree to which the model verification efforts hold true for other target pesticides also is not known. For example, the model-predicted deposition rates reported by RTP for malathion were up to nearly two orders of magnitude greater than that reported by Westchester (2001) as part of its modeling efforts. The reason for this magnitude of difference is not known.

There also is uncertainty surrounding variability in the model inputs, either due to stochastic factors or default assumptions for model inputs. For example, RTP did not account for any chemical or physical degradation of the pesticides during the modeling, and some physical properties such as impingement on trees or buildings were also discounted. This means that the simulations were conservative, and will tend to overestimate the actual concentrations that might affect receptors. There also is variability in the meteorological conditions associated with potential application events. This was accounted for by using average meteorological conditions in the model as based on readings at Islip Airport over a five-year period (1987 to 1991). This could result in under- or overestimates of risk, but neither is suspected to be sufficiently large to significantly affect risk results.

Fate and Transport Parameters

A number of assumptions were needed to predict the environmental persistence, partitioning, and transport of each to the target pesticides. The parameter values used in this risk assessment were derived largely based on data published in the peer-reviewed or other literature. Most often a range of values was available, and the mean or other mid-point estimate of the parameter value was used in the assessment in an attempt to best represent these fate and transport characteristics. However, the values assumed in this assessment could lead to under- or overestimates of exposure and risk, depending upon how well they truly represent actual characteristics. The certainty in the selected values is considered greater for parameters for which multiple data points are available. The magnitude of this uncertainty and its importance on risk is considered low to moderate, depending upon the chemical and the fate parameter of interest. Overall, there are important data gaps in understanding the likely fate and transport of all the targeted pesticides. For example, persistence of the pyrethroids on food crops or non-reactive surfaces has not been well studied, nor have the overall fate and transport of malathion degradates, isomalathion and malaoxon, been well-established. These uncertainties can be best addressed by consideration of new data as they become available.

Exposure Point Concentrations (EPCs)

The EPCs used in this assessment were rooted in the air dispersion and deposition modeling and so therefore have the associated uncertainties noted above for the air modeling results. In addition, there is uncertainty in the concentration to which an individual would be exposed within the range of modeled results. In this assessment, generally conservative assumptions were used to select the starting point for the EPC calculations. For example, for the acute risk assessment the overall maximum concentrations were used, which reflects the highest one hour concentration predicted at any time or place within a study area. In the Tier I and RME Tier II assessment, the maximum temporal average deposition that was predicted to occur at a single location was used as the starting point for the EPC calculations. This represents the concentrations for one out of more than 200 modeled locations, and just based on that, represents a low-probability exposure condition. The average concentrations that could occur throughout the study area over time are likely much more representative of the concentrations to which receptors could be exposed.

Toxicological Hazard

There is some uncertainty surrounding the types of toxicological effects that can be induced by the target pesticides. All target pesticides have been shown to cause non-cancer effects in either experimental animals or humans when administered at sufficiently high dosages. Data relating to carcinogenic potential is not definitive, however. For example, USEPA (2000a) considers malathion to have evidence suggestive of carcinogenicity, but not sufficient evidence to assess human carcinogenic potential. IARC and ATSDR have rendered similar opinions. For the pyrethroid pesticides, WHO (2005) has concluded that there is "no clear indication of carcinogenicity relevant for human risk assessment," and IARC has reached a similar conclusion for permethrin.

Recently, USEPA (2005b) released a preliminary draft of a RED and accompanying risk assessment for permethrin in which the pesticide was evaluated for carcinogenic effects. The cancer toxicity criterion utilized in the assessment was based on evidence of reproducible but benign tumor types (in lung and liver) in laboratory mice, equivocal evidence of carcinogenicity in rats, and supportive information based on structure activity relationships (USEPA, 2004j). This recent USEPA assessment is provisional and has not been finalized or subject to peer review or public comment. For these reasons, this HHRA focused on evaluation of permethrin for non-cancer effects.

To address the uncertainty surrounding the cancer classification for permethrin, the cancer risk evaluations presented by USEPA (2005e) were reviewed. USEPA evaluated potential cancer risks in residents potentially exposed to permethrin following ULV spray via truck foggers and via aerial application for vector control. USEPA also evaluated a number of exposure scenarios associated with permethrin use directly by residents in their home and in agricultural settings. While these latter scenarios are certainly not directly applicable to potential exposures following exposure to vector control ULV sprays, they can provide some perspective on the potential magnitude of risk.

USEPA estimated cancer risks for a single exposure event assuming exposure occurred on the application day. USEPA then calculated the number of application day exposures it would take to reach a 10^{-6} risk level, which equates to a chance of one in a million that an exposed person

could develop cancer as a result of the exposure. For risk management purposes, USEPA typically considers risks in the range of 10^{-6} to 10^{-4} (1 in 10,000) to be acceptable.

Table 7-21 summarizes these results for exposure scenarios of potential relevance to this HHRA of vector control activities in Suffolk County. This table also presents the application rates that were assumed in the USEPA assessment and compares them to the permethrin application rate of 0.007 pounds (lb) active ingredient (AI) per acre (A) potentially used by Suffolk County to support vector control activities.

Table 7-21. Summary of USEPA Cancer Risk Assessment Results for Residential Exposures to Permethrin under a Variety of Use Scenarios and Application Rates and Comparison to Suffolk County Application Rates

Exposure Scenario[a]	Exposure Route[a]	Appli-cation Rate (lb AI/ acre)[a]	Cancer Risk – Application Day*	Number of Application Day Exposures per Year to Reach 1E-06 Risk[a]	USEPA-assumed Application Rate Compared to Suffolk County[b]	Approximate Number of Exposure Days per Year to Reach 1E-06 Using Suffolk County Application Rate[b]
Residential turf (high contact activities)	Dermal	0.87	7.10E-08	14	124	1,751
Residential turf (mowing)	Dermal	0.87	2.40E-09	# 417	124	51,786
Home garden (fruit & nut harvesting)	Dermal	0.4	2.80E-08	37	57	2,114
Home garden (vegetable harvesting)	Dermal	0.23	6.70E-08	15	33	493
Mosquitoes (ULV truck fogger)	Inhalation	0.1	5.20E-08	20	14	286
Mosquito (ULV aerial)	Inhalation	0.1	8.50E-16	# 1.E+09	14	2.E+10
Agricultural use	dietary (food/ water)	2	1.80E-06	0.56	286	159

[a] = as reported in USEPA (2005e), except as noted.
[b] = calculated ratio of USEPA assumed application rate to Suffolk County's rate.
# = Integral-calculated values. USEPA did not report any calculated value that exceed 365 application days per year.

As can be seen, the application rates under the USEPA scenarios are significantly higher than the application rates potentially used by Suffolk County for vector control, being between 14 and

286 times larger. Even with that, the predicted cancer risks are well below the target risk level of 10^{-6} in virtually all cases.

Under the two estimated mosquito ULV application scenarios evaluated by USEPA, inhalation cancer risks are predicted to be in the range of 10^{-8} to 10^{-16}, or two to more than ten orders of magnitude below the target risk level of 10^{-6}. Under USEPA scenarios, it would take between 20 to a billion application day exposures in any given year to result in a 10^{-6} inhalation cancer risk following ULV applications for mosquito control. Assuming the Suffolk County application rate and accepting all other USEPA assumptions, 286 to more than 10 billion application day exposures would have to occur in any one year to result in a 10^{-6} cancer risk. Clearly, these are not plausible. Even recognizing that exposure assumptions used by USEPA could differ from those included in this risk assessment for Suffolk County, and that there are inherent uncertainties associated with any risk evaluation, this magnitude of difference clearly indicates that permethrin ULV application for mosquito control in Suffolk County would not associated with unacceptable inhalation cancer risks.

A similarly large number or application day exposures would be necessary to result in unacceptable cancer risks under the other residential use scenarios evaluated by USEPA when considering the Suffolk County application rates and using all other assumptions used by USEPA. Therefore, permethrin application for mosquito control in Suffolk County will not be associated with unacceptable cancer risks for these additional residential exposure scenarios.

USEPA predicted a risk in the range of 10^{-6} for dietary exposures to permethrin when used in agricultural applications. These risks were predicted using surveys of permethrin residues in foodstuffs as reported by USDA, and so were not calculated directly as a function of application rate. Permethrin application rates in agricultural settings range up to two lb AI per acre. This is 286 times higher than the application rate potentially used by Suffolk County for vector control activities. Based on this, permethrin application for mosquito control is not predicted to be associated with unacceptable cancer risks for dietary exposure scenarios.

Overall, collective consideration of the recent USEPA assessment indicates that vector control application of permethrin in Suffolk County will not be associated with an increased cancer risk. While the USEPA results are not directly transferable to Suffolk County, given differences in the exposure routes and scenarios evaluated, the magnitude by which Suffolk County application

rates fall below those assumed by USEPA is sufficiently large to conclude that permethrin risk for mosquito control in Suffolk County does not pose a cancer risk.

Dose-Response Criteria

There also is uncertainty associated with the toxicity criteria used to evaluate dose-response in this assessment.

Some of this uncertainty stems from the lack of toxicity data in humans. The toxicity criteria used in this assessment were all derived based on the results of toxicological studies in animal studies. The degree to which these data are representative of potential effects in humans is not truly known. Uncertainty factors are typically applied to NOAELs or LOAELs to account for a potentially increased sensitivity of human receptors. In this assessment, uncertainty factors between 100 and 10,000 were used. This could result in significant over-estimates of risk if humans are not substantially more sensitive than animals to the target pesticides.

Additional uncertainty stems from the lack of toxicity data from studies conducted over time periods relevant to the exposure periods evaluated in this assessment. This is particularly important for the acute toxicity criteria used in this assessment, which were extrapolated largely from longer term studies. For example, the acute resmethrin toxicity criterion was based on a 90-day study in laboratory animals. This probably significantly overestimates potential acute toxicity because the test animals were exposed to the chemicals for much longer than could actually occur following vector control applications. Higher dosages would likely have been required to result in the same health effects over the shorter, more realistic exposure durations.

Table 7-22 summarizes the findings of the HHRA for the adulticides.

Table 7-22. Summary of the Human Health Risk Analysis for Adulticides

Agents Considered	Most Critical Endpoint Considered	Pathway Considered Potential Risk	Locations with Potential Risk	Conclusion in Risk Assessment
Adulticides				
Resmethrin	incr. liver wgt, blood chemistry changes, behavioral effects	No pathways or populations presented acute or chronic risks of concern	No locations had risks of concern	The use of resmethrin products for vector control does not pose a health risk under study conditions
Sumithrin	increased liver wgt and adrenal cortex toxicity	No pathways or populations presented acute or chronic risks of concern	No locations had risks of concern	The use of sumithrin products for vector control does not pose a health risk under study conditions
Permethrin	neurological impairment	No pathways or populations presented acute or chronic risks of concern	No locations had risks of concern	The use of permethrin products for vector control does not pose a health risk under study conditions
Malathion	cholinesterase inhibition, maternal toxicity	no acute risks, some risks to RME child resident and adult community gardener	Davis Park only	Malathion does not pose a significant health threat to study area receptors
Degradates				
Malaoxon	NA	NA	NA	
Isomalathion	NA	NA	NA	
Synergist				
PBO	reproductive and developmental toxicity liver and body wgt dec., laryngeal hyperplasia	No pathways or populations presented acute or chronic risks of concern	No locations had risks of concern	The use of PBO-containing products for vector control does not pose a health risk under study conditions

7.9.2.1.7 Other Evaluations of Potential Human Health Risks

CDC Nine-State Survey

CDC surveyed acute insecticide-related illness associated with mosquito control efforts in nine states. The report covers the time period between April, 1999 and September, 2002. The researchers concluded that insecticide application "posed a low risk for acute, temporary health effects among persons in areas that were sprayed and among workers handling and applying insecticides" (CDC, 2003b). In 2000, the total population of the states included in the CDC report was 118 million, and the total number of acute insecticide-related illnesses was 133. Of these, just two (1.5 percent) were classified as definite. Twenty-five (18.8 percent) were defined as probable, meaning that the evidence of exposure either did not come from laboratory, clinical, or environmental results but from a written or verbal report, or that the post-exposure abnormal

symptoms were reported but not documented by a licensed health care professional (CDC, 2000). The majority of the cases (106 [79.7 percent]) were identified as possible, which means that the evidence of exposure did not come from a laboratory, clinical, or environmental source and that the post-exposure abnormal symptoms were reported by someone other than a licensed health care professional.

The CDC report also classified illness based on severity, and only one of the 133 cases was identified as highly severe (CDC, 2003). This category includes life threatening illnesses usually requiring hospitalization (CDC, 2001). The remaining cases were either categorized as moderate (33.8 percent) or low (65.4 percent). Cases of moderate severity generally involve systemic manifestations, but no residual impairment is present. Low severity health effects were by far the most common, and these effects most often manifest as skin, eye or upper respiratory irritation. Typically, a low severity illness or injury resolves itself without treatment. It should also be noted that a significant number of the events reported by CDC were isolated incidences of inappropriate applications, such as the accidental spraying of 29 spectators and players at a softball game by mosquito-control truck workers. Overall, the findings of the report support the conclusion that serious adverse outcomes potentially related to public health insecticide application were extremely unlikely (CDC, 2003).

New York City Study

A similar conclusion was drawn in a study examining the association between ground spraying of pyrethroids in residential neighborhoods in New York City and emergency department (ED) visits for asthma (Karpati et al., 2004). Researchers compared the dates and locations of pyrethroid spraying for vector control with the number of asthma ED visits to public hospitals between October 1999 and November of 2000. The number of asthma visits during the three-day periods before spray events was not significantly different than the number of visits in the three-day periods after spray events, when adjusted for season, day of week, and daily temperature, precipitation, particulate, and ozone levels. Multivariate analysis further showed that daily rates of asthma visits were not associated with pesticide spraying.

Cape Cod Studies

In addition to acute symptoms, some researchers have examined the potential for chronic health effects resulting from exposure to pesticides used in mosquito control programs. McKelvey et al. (2003) suggested that increased incidence of breast cancer among residents of Cape Cod, Massachusetts could be due to pesticides. However, this finding has not been borne out in more specific epidemiological research. A recent study employed GIS technology to conduct a large case-control study of women residing in Cape Cod who were diagnosed with breast cancer between 1988 and 1995. Exposures to pesticides applied for a variety of purposes were assessed, including wetland applications of temephos, dichlorodiphenyltrichloroethane (DDT), and methoxychlor for mosquito control. Overall, no pattern of association between pesticide use in general and breast cancer was found. Specifically, no consistent pattern between breast cancer and residential proximity to mosquito control locations was found (Brody et al., 2004).

Malathion ULV Studies

Field studies of downwind drift and deposition during ULV sprays have also been conducted using human volunteers. One such study evaluated ULV sprays of malathion (Fyfanon). The adulticide was sprayed using a truck-mounted ULV aerosol generator, and concentrations were measured at selected positions on stationary human subjects placed along a transect at right angles to the path of the truck (Moore et al., 1993a). It was concluded that more than 100 years of direct daily exposure to the maximum allowable rates of ULV aerosols would be required to accumulate a dose equivalent to the acute mammalian LD_{50} for malathion. Another study of health risks to applicators engaged in ULV aerial applications of malathion for mosquito control found no specific toxicity signs and symptoms with the exception of nausea and irritation of eyes in a few cases (Gupta et al., 1980).

Mississippi, North Carolina, and Virginia Studies

Several biomonitoring studies following vector control program spray events were recently conducted by CDC in Mississippi, North Carolina, and Virginia. The results of these studies support the findings described above.

In 2002, Mississippi experienced high incidences of WNV, and, in some areas, responded with greater than normal applications of the permethrin with PBO (Luber et al., 2003; CDC, 2005).

These agents were applied using ULV truck-mounted foggers. Based on concerns of the potential health effects of their vector control program, the Mississippi Department of Health asked CDC to assess exposure to mosquito control pesticides to area residents. In response, CDC conducted a study comparing pesticide metabolite levels in the urine of residents living in sprayed areas (n = 125) and that of residents in control areas (n = 67) (CDC, 2005).

Questionnaires and spot urine samples were collected from each study participant. The researchers found no difference in permethrin metabolite levels between mosquito control and non-mosquito control regions. There was also no association found between the number of days since exposure to mosquito control agents and permethrin metabolite levels. In addition, CDC attempted to identify and investigate any reports of pesticide poisonings during the spray period. However, the Mississippi Poison Control Center received no reports of adverse health effects during this time (CDC, 2005).

In September, 2003, hurricane-induced rain and flooding in Virginia and North Carolina were expected to create a five- to ten-fold increase in mosquito populations (Azziz-Baumgartner, 2004; CDC, 2005). Public health response in the two states included continued truck-mounted ULV spraying of permethrin in North Carolina and sumithrin in Virginia, and additional ULV aerial spraying with naled in both states. CDC was again invited to assess human pesticide exposure to residents during these planned spray events.

CDC compared urinary pesticide metabolite levels in residents both before and after spraying to determine if spraying caused an overall increase in exposure to these residents. Prior to the event, researchers recruited a random sample of adults and children living in areas to be sprayed (North Carolina: n = 75; Virginia: n = 83). Questionnaire data and urine samples were collected both before and after spray occurred, and levels of the urinary pesticide metabolites dimethylphosphate (DMP, a metabolite of naled) and 3-phenoxybenzoic acid (3-pba, a metabolite of permethrin and sumithrin) were compared pre- and post-spray. Generalized estimating equations (GEE) indicated no statistically significant difference in the urinary concentrations of DMP or 3-pba before and after spraying. It was concluded that surface spraying with sumithrin and permethrin and large-scale aerial spraying with naled for mosquito control did not significantly increase urinary pesticide metabolite levels in humans. CDC has concluded that the results of the Mississippi and North Carolina and Virginia studies combined

suggest that ULV application of naled, permethrin, and sumithrin is safe to humans as part of integrated vector control.

Generic Risk Assessment of Pesticides Used for West Nile Virus Management

Very recent work (Peterson et al., 2005) evaluated potential human health risks associated with permethrin, natural pyrethrum, resmethrin, sumithrin, malathion, and naled, together with PBO. The risk assessment, especially the air modeling, used many simplifications compared to the analysis completed in this DGEIS. It used the New York City DGEIS as a stepping stone to support the risk assessment. Assumptions used in this more generic review included:

- truck applications were the only mode tested (because the New York City DGEIS suggested ULV deposition rates were much less than truck deposition rates)
- each chemical was assumed to have a 24 hour half life (except naled, given a 48 hour half life)
- a Tier-1 air dispersion model was used (AERMOD v. 1.0), with concentrations computed for 25 ft and 300 ft above ground surface, and ISCST3 was used to compute deposition rates
- 1988 Albany, NY, at 9 pm, weather was assumed
- concentrations were computed for one and six hours post application
- receptors were set at 25 m intervals to 125 m from the application site for air concentrations
- receptors were set at 25 foot intervals to 300 feet from the source for deposition amounts; however, the analysis used mean values over the interval, and potential maximum rates at 300 feet
- an exponential decay program was used to determine persistence on surfaces, with repeated applications over a 90 day season (10 events, on days one, four, 14, 17, 27, 30, 40, 53, and 56), using an aerobic soil decay rate and a photolysis decay rate
- deposition on a garden was also computed

- routes of exposure were inhalation, dermal contact, hand-to-mouth ingestion by infants and toddlers, and ingestion of garden produce
- exposed populations were adult males, adult females, infants (0.5 to 1.5 years old), toddlers (two to three years old), and two other classes of children (five to six years old, and 10 to 12 years old), using body weights derived from USEPA mean values
- exposure was assumed to occur 25 ft from the application for six hours following the application
- acute effects were calculated for the day of the application
- subchronic effects were computed for each day over the 90 day season

The study used a 100-fold safety factor based on USEPA-published toxicity endpoints.

Integration of the risks for each mode of exposure for each exposed population found that none of the compounds had acute or subchronic risks that indicated potential impacts. Sumithrin and PBO risks to adults were the lowest acute risk quotients, sumithrin risks to adults had the lowest risk quotients under the subchronic evaluation, naled had the highest acute risk quotient, and malathion exposure for infants was the highest risk quotient under the subchronic evaluation.

7.9.2.1.8 Ecological Risk Assessment

The conceptual model developed jointly for human and ecological receptors was used as the starting foundation of the ecological risk assessment. That model showed that target pesticides could be released and move in the environment and potentially reach a variety of ecological receptors in terrestrial and aquatic habitats in Suffolk County. From this broad conceptualization, additional analyses were conducted to quantify the potential exposures in these receptor groups, define toxic response as a function of exposure, and characterize risk as a function of exposure and toxicity.

The particular methods used to evaluate ecological risk are dependent on the type of habitat and receptor of interest, but broadly followed a similar framework consisting of receptor identification, exposure assessment, dose-response assessment, and risk characterization (including uncertainty analysis). The general framework is outlined above in Section 7.8.

Terrestrial Wildlife Risk Evaluation

Potential ecological risks were evaluated for terrestrial wildlife, which includes avian, mammalian and reptilian wildlife species present within Suffolk County. ERA Appendix G (Cashin Associates, 2005c) provides a detailed overview of the theoretical and numerical approaches used to model terrestrial exposures and concomitant terrestrial wildlife risks associated with mosquito control agent use.

Terrestrial wildlife evaluated in this risk assessment included the following:

- insectivorous birds
- seed-eating birds
- fruit-eating birds
- terrestrial mammalian wildlife
    - field, forest and hedgerow grazers (e.g., deer)
    - opportunistic foraging mammals (e.g. raccoon)
    - seed-eating small mammals (e.g., field mice)
    - insect-eating mammals (e.g., bats).

Avian wildlife was used as a surrogate receptor group for reptiles given a relative lack of toxicological data for reptiles (Sparling et al., 2000) and because, of all taxa evaluated, reptiles are most closely linked to birds. Such taxonomic similarity is often a factor considered in selecting surrogate receptor species for ERA (e.g., Suter, 1993)

Terrestrial wildlife could be potentially exposed to the primary control agents following application. The assessment endpoint was identified as maintenance of abundance of terrestrial wildlife populations, including mammals, birds, and reptiles that utilize habitats potentially impacted by application of primary list control agents. Terrestrial wildlife exposures and risks were characterized for adulticides following a variety of application scenarios, including aerial (i.e., helicopter), truck ULV, and hand applications (inclusive of backpack sprayers).

For the purposes of this evaluation, dietary exposures were considered the principal route of exposure and were therefore the focus for the assessment of potential terrestrial wildlife risks.

Other potential routes of exposure include dermal exposure through direct interception of control agents or incidental contact with surfaces onto which the target pesticides had deposited, exposure via inhalation, and exposure through consumption of water (e.g., surface water, puddles, dew, or other water on the surfaces of treated vegetation). These routes of exposure were not more fully evaluated since they are unlikely to result in higher estimates of potential risks, and/or they can not be fully quantified based upon limitations and uncertainties in exposure and/or toxicity information.

For example, some recent USEPA research suggests that certain dermal exposures for select pesticides may result in potential avian risks. However, because birds and mammals are largely covered with feathers and fur, dermal exposures are considered to be negligible relative to dietary exposures. In addition, available measured data related to wildlife dermal contact are considered extremely limited (USEPA, 2004b) and toxicity data reflective of potential dermal exposures to wildlife in the field was not identified for the majority of the primary control agents. In the case of the inhalation route, available data suggest that exposures at the time of pesticide application are not likely to be an appreciable route of exposure for birds and mammals given the short atmospheric residence time predicted for the applied target pesticides. Further, virtually no inhalation toxicity data are available for the target pesticides in wildlife species, particularly in the case of birds. In the case of drinking water from puddles, the limited persistence and high degree of partitioning to sediments characteristic of each control agent suggest that this route of exposure would be negligible relative to dietary exposures.

Avian and mammalian terrestrial wildlife dietary exposures were estimated based upon predicted residues in prey and food items following control agent application. Direct application onto prey and food items was addressed. Factors potentially mitigating resultant post-application residues, namely forest canopy interception, were not factored into the assessment.

For acute dietary exposures, maximum average deposition rates (as provided by the RTP modeling as described in ERA Appendix C [Cashin Associates, 2005c]) were used to predict worst-case, instantaneous residues. For chronic exposures, average deposition rates were used to predict 90-day average residues. Deposition rates were adjusted to account for multiple applications based upon relative concentration scaling factors presented in ERA Appendix G (Cashin Associates, 2005c). ERA Appendix B (Cashin Associates, 2005c) provides a detailed

summary of the methods used to calculate relative concentrations to account for multiple applications and degradation under terrestrial settings.

Predicted residues in food and prey items were subsequently calculated based upon empirically derived nomogram values developed by Hoerger and Kenaga (i.e., Hoerger-Kenaga nomograms) and modified by Fletcher et al. (Hoerger and Kenaga, 1972; Fletcher et al., 1994). For assessing acute exposures, the maximum nomogram residues range from 15 to 240 mg/kg (based on a one pound active ingredient/acre application [lb AI/acre]). This range represents potential post-application residues in short grass, tall grass, broad-leaved/forage plants and small insects, and fruits, pods, seeds, and large insects categories (Fletcher et al., 1994). For the purposes of this evaluation, the highest residue concentration of 240 mg/kg for the short grass category was conservatively selected. For the assessment of chronic exposures, the mean nomogram value for short grass of 85 mg/kg was used. Maximum and average residues following control agent application were calculated by adjusting the nomogram values to study area-specific deposition rates (converted to lbs AI/acre).

Acute and chronic risks to terrestrial wildlife species were characterized by comparing estimates of dietary exposure concentrations to wildlife TRVs under the HQ approach.

For acute risks, the selected wildlife TRVs represent the lowest of available acute dietary LD_{50} or LC_{50} values for mammalian and avian species. Preference was given to toxicity tests based on the shortest duration under the assumption that concentrations in dietary items are predicted under this scenario at time zero. For the chronic risks, selected wildlife TRVs represent the lowest reported laboratory or derived dietary chronic NOEC or NOEL for mammalian or avian species. In instances where laboratories reported no effect and toxicity values were not available, NOECs were developed by applying toxicity uncertainty factors. Resultant acute and chronic risks were calculated under the hazard quotient method.

Supporting calculations and predicted HQs for mammalian and avian wildlife are presented in ERA Appendix G (Cashin Associates, 2005c). No dietary risks were predicted for any mammalian or avian wildlife species. Neither acute HQs nor chronic HQs for mammalian and avian wildlife are predicted to exceed a value of one for any control agent applied in the four study areas. Based on these results, it is concluded that the maintenance of abundance of

terrestrial wildlife populations will not be negatively impacted as a result of terrestrial applications of adulticides.

Terrestrial Non-target Insect Risk Evaluation

Terrestrial non-target insects could be potentially exposed to the primary control agents following application. The assessment endpoint was identified as maintenance of abundance of terrestrial non-target insects that utilize habitats potentially impacted by application of primary list control agents. Because toxicological information for other terrestrial insects is generally limited, honeybees were used as a surrogate for other non-target insects, such as butterflies, damselflies, and dragonflies. Terrestrial non-target insect exposures and risks were characterized for adulticides following a variety of application scenarios, including aerial (i.e., helicopter), truck ULV, and hand applications (inclusive of backpack sprayers).

Exposure to a honeybee was conservatively estimated by assuming a honeybee's integument is coated with an adulticide at the time of application (i.e., an instantaneous exposure event assuming zero degradation). Application events that occur when honeybees are inactive (i.e., at night) would preclude direct exposure and results in lower estimates of risk. Instantaneous maximum average deposition rates and instantaneous average deposition rates following application were evaluated. In both instances, deposition rates are adjusted by the maximum relative concentrations to account for multiple applications. Total dose received by a honeybee was calculated as a function of its area and the total control agent mass assumed to cover a bee based upon a given deposition rate. Risks were characterized by comparing the resultant dose to similarly expressed honeybee toxicity reference values. Resultant risks were calculated under the hazard quotient method.

For exposures, both the instantaneous maximum average at any one location and the spatial average across all locations were used to assess potential risks. The former concentration is representative of the concentrations to which a few individuals might be exposed, whereas the latter is more representative of the concentrations to which the population as a whole could be exposed.

Supporting calculations and predicted HQs for honeybees are presented in ERA Appendix G (Cashin Associates, 2005c). Under instantaneous, maximum average conditions, honeybee HQs were predicted to be above one for all adulticides in all study areas, ranging from four to 200,

with the highest HQ of 200 occurring for malathion applied in Davis Park using a golf cart sprayer.

Under instantaneous, average conditions, honeybee HQs range from one to 30, with the highest HQ of 30 predicted for malathion applied to Mastic-Shirley by helicopter. Under the instantaneous average condition, permethrin + PBO and malathion have predicted HQs above one for all study locations (permethrin + PBO HQ range is two to seven; malathion HQ range is eight to 30). Sumithrin + PBO has predicted HQs of greater than one for Davis Park, Dix Hills, and Mastic-Shirley under aerial application scenarios, with all HQs less than or equal to four. Resmethrin + PBO has predicted HQs above one for Davis Park and Mastic-Shirley aerial applications (HQs of three).

Under both maximum average and average conditions, potential risks could also exist for sensitive insect species, such as adult threatened dragonfly species and adult and caterpillar stages of endangered or threatened butterfly species.

Although HQs were generally predicted to be above one for honeybees and other flying terrestrial insects, a number of key factors may act to mitigate and in some cases entirely remove the potential for risks to honeybees and other non-target insects. For example:

- Exposures in this evaluation are predicted assuming that adulticides are applied when honeybees and other non-target insects are active. Honeybees and a number of other non-target insects are predominantly active during the daytime. Actual risks would be most likely to occur when insect activity coincides with the application timing, with risks being largely mitigated for daytime insects if spraying were to occur at night. This would necessitate a consideration for potential risks that could occur for insects active at night, such as moths.

- Exposures and risks are predicted based upon instantaneous conditions. However, adulticides are generally not persistent in terrestrial environments. The use of instantaneous exposure conditions precludes the incorporation of degradation of adulticides, which in turn would likely reduce the potential for risk.

- Laboratory experiments used to derive honeybee benchmarks are based upon the application of an adulticide using a micropatch affixed to a bee's integument. This

experimental regimen may be applicable for the evaluation of non-target insects receiving direct deposition (i.e., flying through or resting within a swath). However, under post application scenarios, where non-target insects may walk across a sprayed surface, the dose received is expected to be far less, and therefore, risks would be expected to be lower.

- Additional habitat preferences, activity patterns, and behavior could result in lower risks for certain non-target insects than those predicted in this evaluation. For example, many insects are active on the ground and may be below vegetation, which may intercept applied adulticides. Many insects, such as crickets, beetles, ants, and millipedes, spend a portion of their life cycle underground. If this period does not temporally coincide with the spray season, the potential for exposure could be significantly mitigated. Some flying insects, such as certain moths and dragonflies, rest at nighttime underneath plants or other structures, and therefore would be less likely to be exposed during nighttime applications. Certain insects may actively avoid sprayed areas. For example, Gerig (1985) reported that permethrin had a strong repellant effect on honeybees in the field.

No further risk evaluation can be conducted to refine these risk estimates, given a lack of additional toxicity data or more refined exposure methods. Overall, aerial application of the target adulticides could be associated with an increased risk of adverse effects in non-target terrestrial insects. These risks could be minimized or eliminated by not applying vector control pesticides in certain habitat areas, such as butterfly gardens and designated wildflower meadows, where butterflies and honeybees may congregate. Additionally, risk to day-active insects such as butterflies, honeybees, and many species of dragonfly could be eliminated or substantially reduced by limiting adulticide applications to evening.

Some considerations may bear on the finding of potential impacts to flying insects. Tables 7-23 and 7-24 list the risk quotients (predicted dose divided by effect level) for bee exposure to the tested pesticides.

Table 7-23. Bee Risk Quotients, Study Area Maximum Average Pesticide Concentrations

Pesticide	Davis Park	Dix Hills	Manorville	Mastic-Shirley (aerial)	Mastic-Shirley (truck)
Permethrin	200	8	9	20	90
Resmethrin	90	4	4	8	40
Sumithrin	100	5	6	10	60
Malathion	200	30	20	50	100

(PBO effects included)

Table 7-24. Bee Risk Quotients, Study Area Mean Pesticide Concentrations

Pesticide	Davis Park	Dix Hills	Manorville	Mastic-Shirley (aerial)	Mastic-Shirley (truck)
Permethrin	7	3	2	7	2
Resmethrin	3	1	1	3	1
Sumithrin	4	2	1	4	1
Malathion	20	20	9	30	8

(PBO effects included)

Verification of the air modeling data showed that under "normal" atmospheric conditions, there was typically a three to one difference between predicted PBO values and measured PBO values; with unusual atmospheric conditions, the agreement was less good (an average of 14:1). The model overpredicts the pesticide concentrations. Conservatively, it seems reasonable to assert a slight overprediction on the basis of the air modeling, which suggests that under most atmospheric conditions resmethrin has little potential for impact to bees, using the study area mean concentrations as a basis for understanding impacts. The same would follow for sumithrin, and a similar conclusion can be drawn for permethrin – although the permethrin risk quotients of seven in Table 7-24 may be a little high to assert the air modeling validation studies imply a lack of potential impact.

Because of the difficulty in measuring resmethrin concentrations in the field, it was conservatively assumed that the resmethrin to PBO ratio would remain constant. However, deposition samples collected on solid media and aqueous samples collected within 30 minutes of the pesticide applications all found that the resmethrin had significantly decreased in concentration relative to PBO. The combination of this degradation and the overprediction by the air modeling makes it possible to assert that there is at least an order of magnitude safety factor associated with resmethrin deposition concentrations. This suggests there is not likely to be a potential impact for resmethrin to flying insects under the more conservative assumptions in Table 7-23 for any of the aerial application scenarios. Because sumithrin has been found to behave similarly to resmethrin in laboratory experiments, it may be that it, too, degrades very

quickly relative to PBO. If that were the case, then aerial applications of sumithrin would likewise be of much less concern, even under the more conservative modeling scenario.

The quantitative risk assessment just above identified a potential for impacts to non-target insects from adulticide applications, based on HQs derived for exposure of bees to the adulticides. The exposure pathway causing the impacts was direct contact with the pesticide while it is suspended in the air. Therefore, given the application techniques proposed in the Long-Term Plan, it is clear that insects most at risk are flying insects, especially those that are airborne at night.

Nearly 95 percent of all described animal species are invertebrates (approximately five million species). These fall into some major groupings, of which the most well known are arthropods, mollusks, annelids, and cnidarians. Nearly four million species are arthropods (comprised primarily of insects, spiders, centipedes, millipedes, and crustaceans). Arthropods all have jointed legs, body segments, and a chitin exoskeleton (which is typically thick and rigid) (NYCDOH, 2001).

The Class Insecta has some 26 to 34 orders, depending on the source (as some merge similar orders while others prefer to distinguish them). Insects have six legs, and most have three body segments (head, thorax, abdomen) and two pairs of wings. They also have antennae and two eyes. Over one million different species of insects have been distinguished. Wingless insects are called Apterygotes, and winged insects are Pterygotes (Pterygotes are more numerous). Winged insects are further distinguished between those that cannot fold or flex their wings (Paleopters) and those that can (Neopters). All insects have larval stages, which can allow young to exploit different ecological niches than adults do (NYCDOH, 2001).

34 orders of insects are found in North America. These include 90,968 described species, and at least 70,000 undescribed species, as of 1990. Coleoptera (beetles) have the most described species (23,640), but many Diptera (flies, including mosquitoes) (19,652 species), Hymenoptera (ants, bees, wasps) (17,429 species) and Lepidoptera (butterflies and moths) (11,300) have been named. Undescribed species are thought to be mostly Diptera (41,000 species) and Hymenoptera (19,000 species). Only 8,668 immature stages of insects had been well described as of 1990, which, since adult insects are often short-lived, means little is known about most of the life span of most North American insects (Hodges, undated).

According to Hodges (undated), state faunal lists for insects exist only for New York (Leonard, 1926) and North Carolina (Brimley, 1938; Brimley, 1942; Wray, 1950; Wray, 1967). Apparently, Connecticut has also made a systematic effort to classify its insects (Britten and Walden, 1911; Britten, 1916; Britton, 1920; Britton, 1923; Garman, 1927; Britton and Kasten, 1938; Crampton, 1942; Mattheson, 1945; Fairchild and Brues, 1950; Johannsen and Townes, 1952; Hardy and Pritchard, 1958; Quate, 1960; Cook et al., 1963; Stone, 1964; Hitchcock, 1974). Such lists, while helpful in determining what is known about a region's insects, have not enabled complete identification of any place's insects. Hodges (undated) asserts that all surveys of particular settings have not been completed, and part of the difficulty in making such a complete assessment is that numerous taxa are not identified.

It thus has proved difficult to identify the insects that might be affected by evening or night applications of insecticides. One approach to this issue was begun in the New York City EIS. There, the insect orders were parsed into those generally larger, generally about the same size, and generally smaller than mosquitoes; in addition, certain broad generalizations were made regarding adult habits (such as night flying) (NYCDOH, 2001). CA therefore has adapted several tables from that document below.

Table 7-25. Flying Insects that are Generally Larger than Mosquitoes (0.15 inches)

Order	Notes	Examples
Coleoptera	The most numerous kind of insect, worldwide; generally nocturnal	Beetles, such as Colorado potato beetle, fireflies, featherwing beetles, Asian long-horned beetle, many, many others
Dermaptera	Generally nocturnal	Earwigs
Embioptera	Only adult males fly; attracted to lights	Web spinners
Hemiptera	Some fly at night; often quite large	True bugs, such as water skimmers
Hymenoptera	Generally fly during the day	Bees, wasps
Isoptera	If fliers, tend to be day active	Termites
Lepidoptera	Some are nocturnal	Butterflies are usually day fliers, moths are night fliers
Neuropter	Some are poor fliers; tend to be day active	Antlions, lacewings
Odonata	Tend to be day fliers	Dragonflies
Orthoptera	Tend to be day active; feed on plants	Grasshoppers, crickets
Plecoptera	Many nocturnal; tend to be poor fliers and short-lived	Stoneflies
Tricoptera	Many nocturnal, weak flying, short-lived	Caddisflies

Table 7-26. Flying Insects that are Generally Similar in Size to Mosquitoes or Smaller (0.15 inches)

Order	Notes	Examples
Diptera	NYC classified as larger than mosquitoes	True flies – black flies, midges, fruit flies, houseflies
Ephemeroptera	Often attracted to lights; shortlived; Paleoptera; classified by NYSDOH as larger than mosquitoes	Mayflies
Homoptera	Important herbivores	Aphids, scale insects, leaf hoppers, cicadas
Mecoptera	Seldom common; insect predators	Scorpionflies
Proscoptera	Many wingless; effective dispersers (often first colonizers of islands)	Bark lice
Strepsiptera	Only males fly; insect parasites	
Thysanoptera		Thrips
Zoraptera	Termite-like; rare; winged individuals may be dispersal form	

Indications are that larger insects will tend not to be impacted by mosquito control applications. Tests of bee exposure to mosquito control pesticides tend to find losses more than those experienced at unexposed hives, but within ranges of natural mortality (as when one study reported statistically elevated bees deaths for exposed hives, but mortalities that were less than the "100 bee per day" apiary mortality standard) (Zhong, 1999; Hester et al., 2001; Caron, 1979; Smith and Stratton, 1986). The maximum application of malathion for mosquito control is 0.23 lbs AI per acre (USEPA, 2005g), but for grasshopper control the dose is allowed to be from 0.58 to 0.87 lbs AI per acre (although population reductions of up to 75 percent were also achieved using 0.3 lbs AI per acre with an encapsulated version) (Reuter and Foster, 2000) and for treatments for crops such as brussel sprouts, cauliflower, ciollards, kale, kohlrabi, peppermint, and trefoil, it can be as high as 0.94 to 1.25 lbs AI per acre (Birchfield et al., undated). This is an indication that other insects require larger doses than mosquitoes do for control to ensue.

Nonetheless, it is clear that impacts to certain insects could occur from mosquito control applications. There appears to be only one study of the effect of a pyrethroid on non-target insects (Jensen et al., 1999). In that study, biomass trapped following a pyrethroid application was less than before the application. However, biomass trapped at a control site was also reduced on the evening following an application. UV light traps were used for this experiment, and so only species attracted to UV light were tested. Both the control and the test site recovered to pre-application biomasses within one week. Similar results occurred following application of dibrom in Cicero Swamp, New York (near Syracuse), down to the coincidental reduction of

biomass at the control site, and recovery to pre-application biomass within one week (O'Brien & Gere, 1995). Neither test suggests that mosquito control pesticides have large impacts on night-flying insects, but neither determined that they did not. Recruitment and dispersal by insects makes it difficult to tell the absolute effects of the application.

Impacts could be greater for repeated applications over short time spans, applications that are made over very large areas that would inhibit recruitment from outside of the application area, or incidences where short-lived, susceptible insects are treated as they emerge. Mitigation may be to avoid applications for vector control purposes at times or in areas where mayfly emergences are predictable, for instance.

New York State has identified insects of concern (NYSDEC, 2005). They are listed below, with their Long Island/Suffolk County status, as is known, included.

Table 7-27. NYSDEC-identified insect species of concern in the Long Island region

Species	Suffolk County Status
American burying beetle	Believed not present; may be on Gardiner's Island
Barrens buck moth	Pine barrens
Northeastern beach tiger beetle	Believed extirpated (may not have been native)
Karner blue butterfly	Historical; no longer present
Odonates of bogs/fens/ponds	yellow-sided skimmer; southern sprite
Odonates of brackish marshes/lakes/ponds	Rambur's forktail, Needham's skimmer
Odonates of coastal plain lakes/ponds	scarlet bluet, little bluet, pine barrens bluet
Odonates of lakes/ponds	spatterdock darner, comet darner, mantelled baskettail, New England bluet
Odonates of rivers/streams	Historical presence of brook sanketail, common sanddragon, Appalachian jewelwing, sparkling jewelwing, russet-tipped clubtailbut statuses are unknown
Odonates of seeps/rivulets	gray petaltail, arrowhead spiketail, and seepage dancer (historical and current staus unknown)
Odonates of small forest streams	mocha emerald
Butterflies (other)	mottled duskywing, persius duskywing, Southern grizzled skipper, Arogos skipper, Brazilian skipper, Hessel's hairstreak, frosted elfin, Henry's elfin, northern oak hairstreak, Northern metalmark, regal fritillary, checkered white,

Species	Suffolk County Status
Moths (other)	*Lepipolys perscripta*, gray woodgrain, *Eucoptocnemis fimbriaris, Euxoa plueritica, Richia acclivis, Abagrotis barnesi*, coastal heathland cutworm, *Schinia bifascia*, *Heterocampa varia*, Herodias underwing, Jair underwing (historically present, current status unknown), barrens dagger moth (historically present, current status unknown), *Amphipoea erepta ryensis*, toothed apharetra (historically present, current status unknown), bay underwing (historically present, current status unknown), the consort underwing (historically present, current status unknown), Jersey Jair underwing, *Catocala sp. 3*, broadlined catopyrrha, *Chaetaglaea cerata*, *Chytonix sensilis*, Melsheimer's sack bearer (historically present, current status unknown), regal moth (historically present, current status unknown), *Dantana ranaeceps*, brown-bordered geometer, Phyllira tiger moth, coastal barrens buckmoth, Buchholz's gray, barrens itame, pale green pinion moth (historically present, current status unknown), wooly gray, Doll's merolonche (historically present, current status unknown)barrens metarranthis moth, *Nemoria bifilata*, *Orgyia detrita* (historically present, current status unknown), yellow stoneroot borer (historically present, current status unknown), Culvers root borer (historically present, current status unknown)ostrich fern borer moth, chain fern borer moth, *Phoberia orthosioides*, pink sallow, *Semiothisa banksianae*, Gordian sphinx, black-bordered lemon moth, dimorphic gray, Acadian swordgrass moth
Pine barrens tiger beetles	Pine barrens: three species (*Cicendela patreula*, *C. unipunctata*, *C. abdominalis*) historically present, current status unknown
Stoneflies/mayflies of uncertain habitat	*Procloeon simile* (historically present, current status unknown)

Recommended actions for all present species include maintaining as high quality habitat as possible. For most species, the needed action is comprehensive surveys to determine its actual status. Specific actions to be avoided, if possible, included the use of pesticides for mosquito control (this was especially noted for the odonata) (NYSDEC, 2005). Mitigation is difficult to determine, absent any firm information on the distribution of most of these insects, and little to no good information available on specifics of their life histories.

Aquatic Life Risk Evaluation

Potential ecological risks were evaluated for aquatic life species present within fresh water and marine/estuarine surface waters of Suffolk County. Aquatic life could be potentially exposed to the primary control agents following application. The assessment endpoint was identified as maintenance of abundance of fish, invertebrate, and amphibian populations that utilize aquatic habitats potentially impacted by application of primary list control agents.

Three levels of analyses were conducted to evaluate potential risks to aquatic life. Multiple levels of analyses were conducted given the complexity of fate and transport modeling and risk estimation techniques required to provide perspective on the full continuum of potential aquatic risks:

- Level 1 – worst case aquatic life exposures and risk – acute exposures only
- Level 2 – refined evaluation of aquatic life exposures and risk – acute and chronic exposures
- Level 3 – evaluation of potential aquatic community level responses.

The methodologies used for Level 1 and Level 2 were described in Section 7.8. Level 3 is described below.

Level 1

Under the first level of assessment, simplistic and conservative modeling was used to provide upper-bound estimates of potential surface water concentrations and acute aquatic life risks associated with adulticides and PBO in each of the four study areas.

The supporting calculations and results of the Level 1 assessment for a generic open water body are presented in ERA Appendix E (Cashin Associates, 2005c) and are summarized as follows

- Based upon maximum average deposition rates, HQs greater than one were predicted for permethrin + PBO (HQ range is two to 20) under both fresh water and marine/estuarine settings within each of the four study areas. Highest risks were predicted for Davis Park. HQs for all other adulticides are below 1.
- Based upon average deposition rates, HQs greater than one were predicted for permethrin + PBO (HQ range is two to three) in Davis Park and Mastic-Shirley (aerial), and for malathion (HQ range is two to five) under both fresh water and marine/estuarine settings within each of the four study areas with the exception of potential impacts to marine/estuarine species following truck application in Mastic-Shirley (HQs less than one).
- HQs for all other adulticides were below one.

The supporting calculations and results of the Level 1 assessment for a shallow wetland are presented in ERA Appendix E (Cashin Associates, 2005c) and are summarized as follows:

- Based upon maximum instantaneous surface water concentrations, HQs greater than one were predicted for malathion (HQ range is 30 to 300) under both fresh water and

marine/estuarine settings within each of the four study areas. Highest risks are predicted for Davis Park. HQs for all other adulticides were below one.

- Based upon average instantaneous surface water concentrations, HQs greater than one were predicted for malathion (HQ range is seven to 70) under both fresh water and marine/estuarine settings within each of the four study areas. Highest risks were predicted for Davis Park and Dix Hills. HQs for all other adulticides were below one.

The supporting calculations and results of the Level 1 analysis to address runoff from impervious surfaces are presented in ERA Appendix E (Cashin Associates, 2005c) and are summarized as HQs greater than one were predicted for malathion (HQ range is two to seven) under both fresh water and marine/estuarine settings within each of the four study area. HQs greater than one were not predicted for the remaining adulticides.

An aquatic food chain evaluation was performed based upon upper-bound and conservative estimates of food chain exposure conditions. Food chain exposures were evaluated for three mid- to upper-trophic level consumers:

- raccoon
- sandpiper
- belted kingfisher.

The supporting calculations and results of the Level 1 assessment of an aquatic food chain are presented in ERA Appendix F (Cashin Associates, 2005c). Based upon this evaluation, predicted HQs for raccoon, sandpiper and belted kingfisher are below one for all adulticides in each of the four study areas.

Based on this worst-case and conservative evaluation, the following were determined:

- Potential acute risks were identified for malathion following application and resultant drift to open surface water bodies and shallow wetlands under fresh water and marine/estuarine settings
- Acute risk associated with runoff from impervious surfaces was also identified for malathion

- Some acute risks were additionally identified for permethrin + PBO following application and resultant drift to open surface water bodies under fresh water and marine/estuarine settings
- Potential acute risks could additionally exist for malathion and permethrin + PBO under the above scenarios for sensitive aquatic life, such as larval or nymph forms of threatened dragonfly species
- Inclusion of important factors such as degradation and chemical partitioning would likely result in lower estimates of risks
- No acute aquatic life risks were identified for:
    - Resmethrin + PBO
    - Sumithrin + PBO
- No risks associated with aquatic food chain exposures were identified.

Level 2

Under the second level of assessment, refined surface water modeling and aquatic life risk characterization were performed to more accurately characterize potential acute risks, as well as chronic aquatic life risks associated with adulticide and larvicide applications.

ERA Appendix E (Cashin Associates, 2005c) presents complete and detailed technical documentation on the theoretical and numerical approaches used to perform refined modeling of potential surface water concentrations and concomitant aquatic life risks water body types present in each of the study areas.

The supporting calculations and results of the Level 2 refined acute aquatic risks are presented in Appendix E (Cashin Associates, 2005c). Only malathion presents predicted acute HQs greater than one, for all receptor groups except aquatic plants, with highest risks predicted for crustaceans and aquatic insects/larvae. Ranges of acute HQs greater than one are predicted for each of the study areas as follows:

- Davis Park (golf cart sprayer application): eight to nine for fresh water species; two to 100 for marine/estuarine species

- Dix Hills (helicopter application): eight to 100 for fresh water species
- Manorville (helicopter application): three to 50 for fresh water species
- Mastic-Shirley (helicopter application): 10 to 100 for fresh water species; three to 100 for marine/estuarine species
- Mastic-Shirley (truck application): three to 50 for fresh water species; four to 40 for marine/estuarine species].

Overall, predicted acute risks from malathion are typically highest in shallow water bodies, such as inland and coastal wetlands/marshes and streams. Malathion risks are predicted to be generally highest in Mastic-Shirley following helicopter application. Risks are generally highest for crustaceans and aquatic insects/larvae. .

A summary of acute risks is presented in Table 7-28.

Table 7-28. Summary of Level 2 Refined Acute Aquatic Life risks (HQs > 1 denoted by shading)

Chemical	Freshwater Aquatic Life Receptors	Marine/Estuarine Aquatic Life Receptors	Davis Park - HQs_{acute}			Dix Hills - HQs_{acute}		Manorville - HQs_{acute}				Mastic Shirley$_{Aerial}$ - HQs_{acute}								Mastic Shirley$_{Truck}$ - HQs_{acute}							
			Aquatic Setting																									
			FW	M/ES		FW		FW				FW				M/ES				FW				M/ES			
			Freshwater Pond/Depression in Target + Runoff	Coastal Wetland/Marsh in Buffer - Drift + Runoff	Coastal Embayment in Buffer - Drift + Runoff	Freshwater Pond/Depression in Target + Runoff	Freshwater Wetland in Target + Runoff	Freshwater Pond/Depression in Target + Runoff	Freshwater Wetland in Target + Runoff	Freshwater Lake in Target + Runoff	Freshwater Stream in Target + Runoff	Freshwater Pond/Depression in Target + Runoff	Freshwater Wetland in Target + Runoff	Freshwater Stream in Buffer - Drift + Runoff	Freshwater Stream in Target + Runoff	Coastal Wetland/Marsh in Target + Runoff	Coastal Embayment in Buffer - Drift + Runoff	Tidal Stream in Buffer - Drift + Runoff	Tidal Stream in Target + Runoff	Freshwater Pond/Depression in Target + Runoff	Freshwater Wetland in Target + Runoff	Freshwater Stream in Buffer - Drift + Runoff	Freshwater Stream in Target + Runoff	Coastal Wetland/Marsh in Target + Runoff	Coastal Embayment in Buffer - Drift + Runoff	Tidal Stream in Buffer - Drift + Runoff	Tidal Stream in Target + Runoff
Permethrin + PBO (Permanone)																											
	Fish	Fish	3E-04	2E-03	1E-04	1E-04	7E-04	9E-05	6E-04	8E-05	1E-04	3E-04	2E-03	3E-04	4E-04	2E-03	2E-04	4E-04	5E-04	1E-04	6E-04	7E-05	1E-04	8E-04	5E-05	1E-04	2E-04
	Amphibians	Crustaceans	7E-04	7E-02	3E-03	3E-04	3E-03	3E-04	3E-03	2E-04	5E-04	8E-04	8E-03	1E-03	1E-03	8E-02	8E-03	1E-02	2E-02	2E-04	3E-03	3E-04	4E-04	3E-02	2E-03	3E-03	6E-03
	Crustaceans	Mollusks	2E-02	7E-04	3E-05	8E-03	4E-02	7E-03	4E-02	7E-03	9E-03	2E-02	1E-01	2E-02	3E-02	9E-04	4E-05	1E-04	2E-04	8E-03	4E-02	5E-03	9E-03	3E-04	1E-05	3E-05	5E-05
	Mollusks	Aquatic insects/larvae	9E-05	7E-03	4E-04	4E-05	4E-04	3E-05	3E-04	2E-05	5E-05	9E-05	1E-03	1E-04	2E-04	9E-03	9E-04	1E-03	2E-03	3E-05	3E-04	3E-05	5E-05	3E-03	2E-04	4E-04	7E-04
	Aquatic insects/larvae	Aquatic plants	9E-03	2E-06	1E-07	3E-03	2E-02	3E-03	1E-02	3E-03	3E-03	9E-03	5E-02	7E-03	1E-02	3E-06	3E-07	5E-07	7E-07	3E-03	2E-02	2E-03	4E-03	1E-06	7E-08	1E-07	2E-07
	Aquatic plants		5E-07			2E-07	1E-06	2E-07	9E-07	2E-07	2E-07	5E-07	3E-06	5E-07	7E-07					2E-07	1E-06	1E-07	2E-07				
Resmethrin + PBO (Scourge)																											
	Fish	Fish	8E-02	3E-02	1E-03	3E-02	3E-01	3E-02	2E-01	2E-02	4E-02	8E-02	7E-01	9E-02	1E-01	4E-02	2E-03	6E-03	8E-03	3E-02	2E-01	2E-02	4E-02	1E-02	6E-04	2E-03	2E-03
	Amphibians	Crustaceans	1E-03	3E-01	1E-02	5E-04	5E-03	4E-04	5E-03	3E-04	7E-04	1E-03	1E-02	2E-03	2E-03	4E-01	2E-02	5E-02	7E-02	4E-04	4E-05	5E-04	7E-04	1E-01	6E-03	1E-02	2E-02
	Crustaceans	Mollusks	1E-02	1E-03	5E-05	6E-03	5E-02	5E-03	4E-02	4E-03	8E-03	1E-02	1E-01	2E-02	2E-02	2E-03	8E-05	2E-04	3E-04	5E-03	4E-02	5E-03	8E-03	6E-04	2E-05	6E-05	9E-05
	Mollusks	Aquatic insects/larvae	2E-04	9E-04	4E-05	7E-05	7E-04	6E-05	6E-04	4E-05	1E-04	2E-04	2E-03	2E-04	3E-04	1E-03	6E-05	1E-04	2E-04	5E-05	8E-05	6E-05	9E-05	4E-04	2E-05	4E-05	6E-05
	Aquatic insects/larvae	Aquatic plants	1E-04	8E-06	3E-07	4E-05	5E-04	4E-05	4E-04	3E-05	7E-05	1E-04	1E-03	1E-04	2E-04	1E-05	6E-07	1E-06	2E-06	4E-05	2E-06	4E-05	6E-05	3E-06	1E-07	4E-07	6E-07
	Aquatic plants		1E-06			5E-07	4E-06	4E-07	3E-06	3E-07	6E-07	1E-06	1E-05	1E-06	2E-06					4E-07	3E-06	4E-07	6E-07				
Sumethrin + PBO (Anvil)																											
	Fish	Fish	1E-05	8E-05	3E-06	6E-06	6E-05	5E-06	5E-05	3E-06	9E-06	1E-05	2E-04	2E-05	3E-05	1E-04	4E-06	1E-05	2E-05	5E-06	5E-05	5E-06	8E-06	3E-05	1E-06	3E-06	5E-06
	Amphibians	Crustaceans	2E-04	9E-04	3E-05	9E-05	1E-03	8E-05	3E-06	5E-05	1E-04	2E-04	3E-03	3E-04	4E-04	1E-03	5E-05	1E-04	2E-04	7E-05	8E-04	9E-05	1E-04	4E-04	1E-05	4E-05	6E-05
	Crustaceans	Mollusks	1E-04	2E-04	8E-06	4E-05	5E-04	4E-05	2E-05	2E-05	6E-05	1E-04	1E-03	1E-04	2E-04	3E-04	1E-05	3E-05	4E-05	3E-05	4E-04	4E-05	6E-05	9E-05	3E-06	1E-05	1E-05
	Mollusks	Aquatic insects/larvae	2E-05	2E-04	7E-06	1E-05	1E-04	9E-06	5E-06	6E-06	2E-05	3E-05	3E-04	3E-05	4E-05	2E-04	1E-05	3E-05	4E-05	8E-06	9E-05	1E-05	1E-05	7E-05	3E-06	8E-06	1E-05
	Aquatic insects/larvae	Aquatic plants	2E-05	3E-11	9E-12	8E-06	9E-05	7E-06	4E-06	5E-06	1E-05	2E-05	2E-04	3E-05	4E-05	1E-10	4E-11	4E-11	8E-11	7E-06	7E-05	8E-06	1E-05	4E-11	8E-12	9E-12	3E-11
	Aquatic plants		8E-11			3E-11	5E-11	3E-11	4E-11	3E-11	3E-11	8E-11	1E-10	4E-11	8E-11					3E-11	4E-11	9E-12	3E-11				
Malathion (96.8%, Fyfanon)																											
	Fish	Fish	9E-02	***2E+00***	8E-02	9E-02	1E+00	3E-02	5E-01	2E-02	6E-02	9E-02	1E+00	1E-01	2E-01	***3E+00***	1E-01	3E-01	4E-01	3E-02	5E-01	4E-02	6E-02	1E+00	3E-02	9E-02	1E-01
	Amphibians	Crustaceans	3E-02	***3E+01***	1E+00	3E-02	5E-01	1E-02	2E-01	7E-03	2E-02	3E-02	5E-01	5E-02	6E-02	***4E+01***	1E+00	***4E+00***	***5E+00***	1E-02	2E-01	1E-02	2E-02	***1E+01***	4E-01	1E+00	***2E+00***
	Crustaceans	Mollusks	***8E+00***	2E-02	8E-04	***8E+00***	***1E+02***	***3E+00***	***4E+01***	***2E+00***	***6E+00***	***9E+00***	***1E+02***	***1E+01***	***2E+01***	3E-02	1E-03	3E-03	4E-03	***3E+00***	***4E+01***	***4E+00***	***5E+00***	1E-02	3E-04	1E-03	1E-03
	Mollusks	Aquatic insects/larvae	2E-03	***1E+02***	***3E+00***	2E-03	3E-02	7E-04	1E-02	5E-04	1E-03	2E-03	3E-02	3E-03	4E-03	***1E+02***	***5E+00***	***1E+01***	***2E+01***	7E-04	1E-02	1E-03	1E-03	***4E+01***	1E+00	***4E+00***	***5E+00***
	Aquatic insects/larvae	Aquatic plants	***9E+00***	8E-01	3E-02	***8E+00***	***1E+02***	***3E+00***	***5E+01***	***2E+00***	***6E+00***	***9E+00***	***1E+02***	***1E+01***	***2E+01***	1E+00	4E-02	1E-01	1E-01	***3E+00***	***5E+01***	***4E+00***	***6E+00***	3E-01	1E-02	3E-02	4E-02
	Aquatic plants		6E-02			6E-02	1E+00	2E-02	3E-01	1E-02	4E-02	7E-02	1E+00	1E-01	1E-01					2E-02	3E-01	3E-02	4E-02				

Acute hazard quotient of 1 is exceeded
Freshwater setting
Marine/estuarine setting
See Tables E-30 through E-36 for a complete presentation of surface water exposure concentrations, aquatic Life TRVs, and estimated acute aquatic life hazard quotients.

The supporting calculations and results of the Level 2 refined chronic aquatic risks are presented in ERA Appendix E (Cashin Associates, 2005c). Of the adulticides, malathion most frequently presents predicted chronic HQs greater than one. Risks greater than one are predicted for all receptors groups except fish and aquatic plants, with highest risks predicted for crustaceans and aquatic insects/larvae.

Some limited chronic risks are also predicted for permethrin for crustaceans and aquatic insects/larvae in shallow water bodies following helicopter application in Mastic-Shirley.

No HQs greater than one are calculated for resmethrin + PBO, or sumithrin + PBO.

Ranges of chronic HQs greater than one are predicted for each of the study areas as follows (HQs are for malathion unless as otherwise noted for permethrin + PBO):

- Davis Park (golf cart sprayer application): nine to 40 (fresh water species); two to 100 (marine/estuarine species)
- Dix Hills (helicopter application): eight to 400 (fresh water species)
- Manorville (helicopter application): two to 100 (fresh water species)
- Mastic-Shirley (helicopter application): eight to 400 (fresh water species), four to 300 (marine/estuarine species); permethrin + PBO, two to 20 (fresh water species), and 10 (marine/estuarine species)
- Mastic-Shirley (truck application): two to 100 (fresh water species); three to 100 (marine/estuarine species).

In summary, refined estimates of potential chronic risks to aquatic life were predicted predominantly for malathion, with some limited risks for permethrin + PBO in Mastic-Shirley following aerial application in fresh water and wetlands and streams (including off-target streams in the quarter mile buffer zone) and marine/estuarine wetlands. Overall, risks are typically highest in shallow water bodies, such as inland and coastal wetlands/marshes and streams. Risks are generally highest for crustaceans and aquatic insects/larvae. Chronic HQs greater than one are not predicted for the remaining adulticides.

A summary of chronic risks is presented in Table 7-29.

Table 7-29. Summary of Level 2 Refined Chronic Aquatic Life Risks (HQs > 1 denoted by shading)

Chemical	Freshwater Aquatic Life Receptors	Marine/Estuarine Aquatic Life Receptors	Davis Park - HQs_{acute}			Dix Hills - HQs_{acute}		Manorville - HQs_{acute}				Mastic Shirley$_{Aerial}$ - HQs_{acute}								Mastic Shirley$_{Truck}$ - HQs_{acute}							
			Aquatic Setting																								
			FW	M/ES		FW		FW				FW				M/ES				FW				M/ES			
			Freshwater Pond/Depression in Target + Runoff	Coastal Wetland/Marsh in Buffer - Drift + Runoff	Coastal Embayment in Buffer - Drift + Runoff	Freshwater Pond/Depression in Target + Runoff	Freshwater Wetland in Target + Runoff	Freshwater Pond/Depression in Target + Runoff	Freshwater Wetland in Target + Runoff	Freshwater Lake in Target + Runoff	Freshwater Stream in Target + Runoff	Freshwater Pond/Depression in Target + Runoff	Freshwater Wetland in Target + Runoff	Freshwater Stream in Buffer - Drift + Runoff	Freshwater Stream in Target + Runoff	Coastal Wetland/Marsh in Target + Runoff	Coastal Embayment in Buffer - Drift + Runoff	Tidal Stream in Buffer - Drift + Runoff	Tidal Stream in Target + Runoff	Freshwater Pond/Depression in Target + Runoff	Freshwater Wetland in Target + Runoff	Freshwater Stream in Buffer - Drift + Runoff	Freshwater Stream in Target + Runoff	Coastal Wetland/Marsh in Target + Runoff	Coastal Embayment in Buffer - Drift + Runoff	Tidal Stream in Buffer - Drift + Runoff	Tidal Stream in Target + Runoff
Permethrin + PBO (Permanone)																											
	Fish	Fish	3E-04	2E-03	1E-04	1E-04	7E-04	9E-05	6E-04	8E-05	1E-04	3E-04	2E-03	3E-04	4E-04	2E-03	2E-04	4E-04	5E-04	1E-04	6E-04	7E-05	1E-04	8E-04	5E-05	1E-04	2E-04
	Amphibians	Crustaceans	7E-04	7E-02	3E-03	3E-04	3E-03	3E-04	3E-03	2E-04	5E-04	8E-04	8E-03	1E-03	1E-03	8E-02	8E-03	1E-02	2E-02	2E-04	3E-03	3E-04	4E-04	3E-02	2E-03	3E-03	6E-03
	Crustaceans	Mollusks	2E-02	7E-04	3E-05	8E-03	4E-02	7E-03	4E-02	7E-03	9E-03	2E-02	1E-01	2E-02	3E-02	9E-04	4E-05	1E-04	2E-04	8E-03	4E-02	5E-03	9E-03	3E-04	1E-05	3E-05	5E-05
	Mollusks	Aquatic insects/larvae	9E-05	7E-03	4E-04	4E-05	4E-04	3E-05	3E-04	2E-05	5E-05	9E-05	1E-03	1E-04	2E-04	9E-03	9E-04	1E-03	2E-03	3E-05	3E-04	3E-05	5E-05	3E-03	2E-04	4E-04	7E-04
	Aquatic insects/larvae	Aquatic plants	9E-03	2E-06	1E-07	3E-03	2E-02	3E-03	1E-02	3E-03	3E-03	9E-03	5E-02	7E-03	1E-02	3E-06	3E-07	5E-07	7E-07	3E-03	2E-02	2E-03	4E-03	1E-06	7E-08	1E-07	2E-07
	Aquatic plants		5E-07			2E-07	1E-06	2E-07	9E-07	2E-07	2E-07	5E-07	3E-06	5E-07	7E-07					2E-07	1E-06	1E-07	2E-07				
Resmethrin + PBO (Scourge)																											
	Fish	Fish	8E-02	3E-02	1E-03	3E-02	3E-01	3E-02	2E-01	2E-02	4E-02	8E-02	7E-01	9E-02	1E-01	4E-02	2E-03	6E-03	8E-03	3E-02	2E-01	2E-02	4E-02	1E-02	6E-04	2E-03	2E-03
	Amphibians	Crustaceans	1E-03	3E-01	1E-02	5E-04	5E-03	4E-04	5E-03	3E-04	7E-04	1E-03	1E-02	2E-03	2E-03	4E-01	2E-02	5E-02	7E-02	4E-04	4E-05	5E-04	7E-04	1E-01	6E-03	1E-02	2E-02
	Crustaceans	Mollusks	1E-02	1E-03	5E-05	6E-03	5E-02	5E-03	4E-02	4E-03	8E-03	1E-02	1E-01	2E-02	2E-02	2E-03	8E-05	2E-04	3E-04	5E-03	4E-02	5E-03	8E-03	6E-04	2E-05	6E-05	9E-05
	Mollusks	Aquatic insects/larvae	2E-04	9E-04	4E-05	7E-05	7E-04	6E-05	6E-04	4E-05	1E-04	2E-04	2E-03	2E-04	3E-04	1E-03	6E-05	1E-04	2E-04	5E-05	8E-05	6E-05	9E-05	4E-04	2E-05	4E-05	6E-05
	Aquatic insects/larvae	Aquatic plants	1E-04	8E-06	3E-07	4E-05	5E-04	4E-05	4E-04	3E-05	7E-05	1E-04	1E-03	1E-04	2E-04	1E-05	6E-07	1E-06	2E-06	4E-05	2E-06	4E-05	6E-05	3E-06	1E-07	4E-07	6E-07
	Aquatic plants		1E-06			5E-07	4E-06	4E-07	3E-06	3E-07	6E-07	1E-06	1E-05	1E-06	2E-06					4E-07	3E-06	4E-07	6E-07				
Sumethrin + PBO (Anvil)																											
	Fish	Fish	1E-05	8E-05	3E-06	6E-06	6E-05	5E-06	5E-05	3E-06	9E-06	1E-05	2E-04	2E-05	3E-05	1E-04	4E-06	1E-05	2E-05	5E-06	5E-05	5E-06	8E-06	3E-05	1E-06	3E-06	5E-06
	Amphibians	Crustaceans	2E-04	9E-04	3E-05	9E-05	1E-03	8E-05	3E-06	5E-05	1E-04	2E-04	3E-03	3E-04	4E-04	1E-03	5E-05	1E-04	2E-04	7E-05	8E-04	9E-05	1E-04	4E-04	1E-05	4E-05	6E-05
	Crustaceans	Mollusks	1E-04	2E-04	8E-06	4E-05	5E-04	4E-05	2E-05	2E-05	6E-05	1E-04	1E-03	1E-04	2E-04	3E-04	1E-05	3E-05	4E-05	3E-05	4E-04	4E-05	6E-05	9E-05	3E-06	1E-05	1E-05
	Mollusks	Aquatic insects/larvae	2E-05	2E-04	7E-06	1E-05	1E-04	9E-06	5E-06	6E-06	2E-05	3E-05	3E-04	3E-05	4E-05	2E-04	1E-05	3E-05	4E-05	8E-06	9E-05	1E-05	1E-05	7E-05	3E-06	8E-06	1E-05
	Aquatic insects/larvae	Aquatic plants	2E-05	3E-11	9E-12	8E-06	9E-05	7E-06	4E-06	5E-06	1E-05	2E-05	2E-04	3E-05	4E-05	1E-10	4E-11	4E-11	8E-11	7E-06	7E-05	8E-06	1E-05	4E-11	8E-12	9E-12	3E-11
	Aquatic plants		8E-11			3E-11	5E-11	3E-11	4E-11	3E-11	3E-11	8E-11	1E-10	4E-11	8E-11					3E-11	4E-11	9E-12	3E-11				
Malathion (96.8%, Fyfanon)																											
	Fish	Fish	9E-02	***2E+00***	8E-02	9E-02	1E+00	3E-02	5E-01	2E-02	6E-02	9E-02	1E+00	1E-01	2E-01	***3E+00***	1E-01	3E-01	4E-01	3E-02	5E-01	4E-02	6E-02	1E+00	3E-02	9E-02	1E-01
	Amphibians	Crustaceans	3E-02	***3E+01***	1E+00	3E-02	5E-01	1E-02	2E-01	7E-03	2E-02	3E-02	5E-01	5E-02	6E-02	***4E+01***	1E+00	***4E+00***	***5E+00***	1E-02	2E-01	1E-02	2E-02	***1E+01***	4E-01	1E+00	***2E+00***
	Crustaceans	Mollusks	***8E+00***	2E-02	8E-04	***8E+00***	***1E+02***	***3E+00***	***4E+01***	***2E+00***	***6E+00***	***9E+00***	***1E+02***	***1E+01***	***2E+01***	3E-02	1E-03	3E-03	4E-03	***3E+00***	***4E+01***	***4E+00***	***5E+00***	1E-02	3E-04	1E-03	1E-03
	Mollusks	Aquatic insects/larvae	2E-03	***1E+02***	***3E+00***	2E-03	3E-02	7E-04	1E-02	5E-04	1E-03	2E-03	3E-02	3E-03	4E-03	***1E+02***	***5E+00***	***1E+01***	***2E+01***	7E-04	1E-02	1E-03	1E-03	***4E+01***	1E+00	***4E+00***	***5E+00***
	Aquatic insects/larvae	Aquatic plants	***9E+00***	8E-01	3E-02	***8E+00***	***1E+02***	***3E+00***	***5E+01***	***2E+00***	***6E+00***	***9E+00***	***1E+02***	***1E+01***	***2E+01***	1E+00	4E-02	1E-01	1E-01	***3E+00***	***5E+01***	***4E+00***	***6E+00***	3E-01	1E-02	3E-02	4E-02
	Aquatic plants		6E-02			6E-02	1E+00	2E-02	3E-01	1E-02	4E-02	7E-02	1E+00	1E-01	1E-01					2E-02	3E-01	3E-02	4E-02				

Acute hazard quotient of 1 is exceeded
Freshwater setting
Marine/estuarine setting
See Tables E-30 through E-36 for a complete presentation of surface water exposure concentrations, aquatic Life TRVs, and estimated acute aquatic life hazard quotients.

The refined aquatic life risk evaluation determined:

- Malathion potentially poses acute and chronic risks to aquatic life under the application scenarios evaluated. Predicted risks are typically highest in shallow water bodies, such as inland and coastal wetlands/marshes and streams following helicopter application. Risks are generally highest for crustaceans and aquatic insects/larvae.
- Permethrin + PBO potentially poses some chronic aquatic life risk following helicopter application, though predicted risks were lower in magnitude and prevalence than malathion risks.
- Potential acute and chronic risks could exist for permethrin + PBO under the above scenarios for sensitive aquatic life, such as larval or nymph forms of threatened dragonfly species.
- Resmethrin + PBO and sumithrin + PBO do not pose any unacceptable aquatic life risks under the application scenarios evaluated.

Level 3

Potential aquatic life risks were predicted for both malathion and permethrin under the more refined Level 2 analysis. The degree to which these predicted risks truly indicate a potential threat to the ecological community is not completely known. However, information from field studies examining community and population level ecological responses to vector control pesticide use can be used to provide perspective on the likelihood and/or potential ecological consequences of the predicted risks. Additionally, community level ecological modeling can provide additional insight.

The potential for long-term impacts to aquatic life populations and communities associated with pesticides used for insect control has been studied with increasing attention over the last 10 to 15 years. The theoretical ecological premise underlying this concern is that if repeated use of a pesticide results in the suppression of target insect populations, and if such insects represent an important food source or play other vital roles in aquatic food webs, then as a result other members of the food web relying on target insects could in turn be impacted. Ultimately, such impacts could lead to an altering of ecosystem structure and function. Explicit cause and effect relationships are difficult to establish in this respect, because in addition to chemical sensitivity,

a number of often poorly defined ecological traits of the exposed organisms are likely to play a role in determining population and subsequent community level impacts. Such traits include generation time, migration ability, and presence of aquatic stages during time of pesticide application (Liess and Von Der Ohe, 2005).

To date, the vast majority of experimental work on potential population and community level effects has focused on a narrow set of taxonomic groups (e.g., zooplankton communities). For example, Forbes and Cold (2005) observed that pulsed exposures of the pyrethroid esfenvalerate under field conditions could result in life cycle effects to chironomids, although long-term population level impacts appeared to be dependent upon other factors, such as initial population density and exposure conditions. Potential community-level impacts were not addressed. Relyea (2005) recently conducted mesocosm-based studies using various pesticides, including malathion, in a simulated aquatic ecosystem. Pesticides were added to mesocosms at the manufacturer's maximum application rates. The estimated resultant concentration of malathion (based upon addition to a 1,200 L polyethylene tank) of 320 µg/L was described as resulting in a 30 percent reduction in species richness and a reduction in zooplankton and predatory insect diversity. This concentration ranges from approximately 10 to 1,000 times higher than the instantaneous concentrations modeled in this ecological risk assessment conducted for Suffolk County.

A number of important studies have been conducted starting in the late 1980s and early 1990s on pesticides applied specifically for mosquito control. Charbonneau et al. (1994) conducted field and laboratory experiments to assess potential impacts to chironomids and other benthic invertebrates following application of Bti (as corn cob granules) at the Minnesota Valley NWR. Although laboratory toxicity tests resulted in observed toxicity for chironomids, under field conditions no reduction in major taxa was observed during the two years of study (1989, 1990). During 1991 to 1993, Hershey et al. (1998) conducted field experiments on the effects of Bti and methoprene on non-target macroinvertebrates in wetlands of Wright County, Minnesota. During the first year of study, minimal impacts were observed. In 1992 and 1993, Bti resulted in high reductions among dipteran species, including chrionomids, while minimal effects on non-insect macroinvertebrates were observed. Additionally, minimal effects were observed for methoprene.

As a series of follow-up studies to the Hershey et al. (1998) work, as well as the companion work reported by Neimi et al. (1999), the Metropolitan Mosquito Control District of St. Paul Minnesota conducted additional experiments in Wright County. This more detailed work showed no long-term impacts from Bti or methoprene on total macroinvertebrate density or biomass and no difference in overall chironomid numbers between treated and untreated areas. Additional analysis suggested that the earlier declines observed in 1992 and 1993 by Hershey et al. may have been attributable to higher than planned doses and to drought conditions which prevailed several years prior to the study (Balcer et.al., 1999; Read, 2001).

Jensen et al. (1999) evaluated potential impacts of permethrin and malathion ULV applications for mosquito control in seasonal wetlands in the Sutter and Colusa National Wildlife Refuges located in California. Abundance and biomass of aquatic macroinvertebrates, including chironomids, Odonata (dragonfly), mayfly, and Corixidae (water boatmen), were evaluated in treated and control wetlands. The results of this analysis showed no detectable reductions in abundance or biomass of aquatic macroinvertebrates in ULV-treated seasonal wetlands. The authors concluded that effective control of adult mosquitoes could be accomplished near wetlands using ULV applications of permethrin and malathion without substantially reducing the amount of aquatic invertebrates used by foraging wildlife. Of additional note is the fact that this study also looked for impacts to flying insects. Reductions in biomass were measured following one application, but a similar reduction was measured at the control site, and the difference was found to be not statistically significant. Recovery of flying insect biomass to pre-treatment date levels at both the treatment and control site occurred within several days.

Recent work has also been conducted to evaluate potential changes in benthic diversity in Suffolk County wetlands and marshes following the aerial application of resmethrin and methoprene, as part of the Long-Term Plan development (see Section 6, and Barnes, 2005). In addition to conducting cage experiments with grass shrimp, benthic community structure analysis was performed at the conclusion of the spray season. These analyses showed that no significant differences existed in benthic community structure or abundance between pesticide sprayed and non-sprayed sites.

The potential for aquatic life population and community level impacts was additionally assessed through the modeling of an adulticide application in Suffolk County. The focus of the

assessment was on potential long-term impacts on abundance of populations and potential resultant impacts on community structure and function following the aerial application of permethrin. Permethrin was selected for additional evaluation because some aquatic life risks were predicted in the Level 2 risk assessment and because synthetic pyrethroids have been relied upon by Suffolk County as part of past vector control activities. Although malathion aquatic life risks were also predicted, under SCVC practice, malathion is less likely to be relied on, and is not likely to be used at the same fequency or extent throughout the county as are synthetic pyrethroids.

The focus of this evaluation was on potential indirect deposition of permethrin into shallow water bodies, such as shallow wetlands, vernal pools, and shallow ponds, present in the Mastic-Shirley study area. This scenario was selected based upon the results of the Level 2 assessment, which demonstrated the highest potential for chronic risks to aquatic life from synthetic pyrethroid use exists under this scenario (i.e., HQs up to 20 for aquatic invertebrates).

For this modeling evaluation, potential long-term population and community-level impacts were evaluated using the USEPA AQUATOX model. AQUATOX is a process-based ecosystem model that predicts the fate of various pollutants, such as nutrients and organic toxicants, and their effects on aquatic populations and communities, including those for fish, invertebrates, and aquatic plants. Unlike most water quality models, AQUATOX treats aquatic organisms as integral to the chemical/physical system. Its potential applications include analyzing the cause and effect relationships between the chemical and physical environment and biological responses (USEPA, 2004g; USEPA, 2004h; Pastorok et al., 2003).

AQUATOX represents aquatic ecosystems in the schematic provided in Figure 7-7.

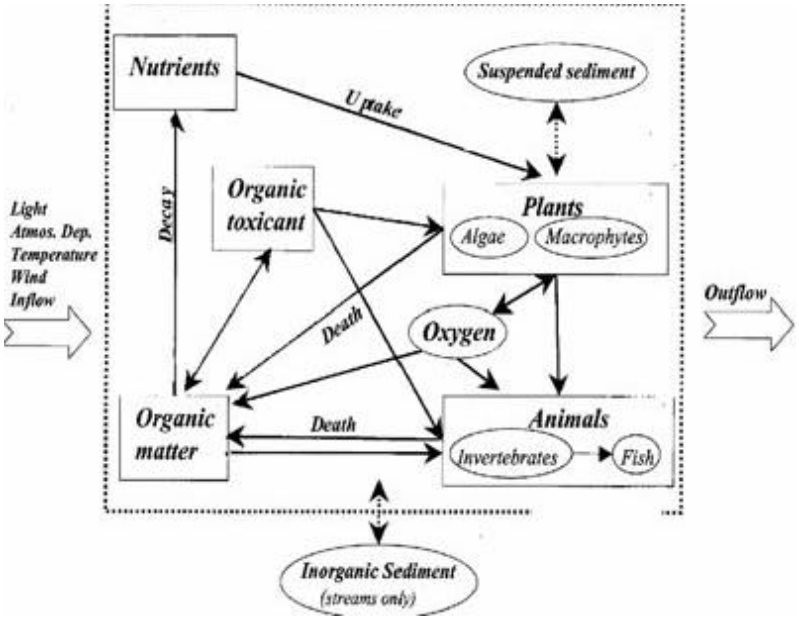

Figure 7-7. AQUATOX Aquatic Ecosystem Representation

The fate portion of the model, which is applicable especially to organic toxicants, includes:

- partitioning among organisms, suspended and deposited detritus, suspended and deposited inorganic sediments, and water;
- volatilization;
- hydrolysis;
- photolysis;
- ionization; and,
- microbial degradation.

The effects portion of the model includes toxicity to the various organisms modeled, and other indirect effects, which include:

- release of grazing and predation pressure
- increase in detritus and recycling of nutrients from killed organisms

- dissolved oxygen sag due to increased decomposition
- loss of food base for animals

(USEPA, 2004g; USEPA, 2004h)

Version 3.1.4, in a beta version (prior to formal release) was used for this simulation. This latest version offers a number of upgrades over previous versions, particularly with respect to the greater flexibility and dynamism added to the fate, transport and uptake components of the model and the inclusion of an estuarine modeling component.[7]

The focus of this modeling evaluation was on permethrin reaching a small water body, such as a shallow wetland, following aerial application in Mastic-Shirley. The results of the aquatic life risk assessment demonstrated that PBO, as formulated with permethrin in the product Permanone, does not contribute to observed risks. Therefore, permethrin alone was selected as the focus of this modeling.

The modeled shallow open surface water body was considered representative of fresh water mixed brackish environment. Aquatic species incorporated into the modeling included the following:

- benthic organisms (amphipods, chironomids)
- suspended feeders (daphnia, copepods)
- predatory invertebrate (Odonata)
- mollusks (mussel)
- gastropods (snail)
- small forage fish (silverside)
- large forage fish (perch)
- large bottom fish (catfish)

[7] The most current publicly available version of AQUATOX (Release 2) is available from USEPA, along with additional details and user's manuals, at http://www.epa.gov/waterscience/models/aquatox/about.html.

- small game fish (bass, young of year)
- large game fish (bass, adult).

Periphyhton and aquatic plants (e.g., diatoms, blue green algae) were also included as primary producers in the simulation. Permethrin, however, has very low toxicity to aquatic plants (e.g., blue-green algae 96 hour EC_{50} of 1,600 µg/L). Changes in abundances of aquatic plants attributable to permethrin were not anticipated and therefore aquatic plants were not included for detailed evaluation and interpretation.

An average deposition rate, as provided by RTP air modeling, of 2.3E-04 g/m^2 (see ERA Appendix C [Cashin Associates, 2005c]) was used to characterize input into the water body. Two applications in early June 2005, spaced seven days apart were used in the model simulation. For comparative purposes, a control simulation without treatment was also performed. The total modeling period for both treated and control simulations was May 2005 through April 2006.

ERA Appendix I (Cashin Associates, 2005c) provides a summary of the chemical and physical conditions used to characterize the shallow open surface water body used in the modeling, and complete details on the model inputs and model specifications used. Potential emigration and immigration among individual organisms were factored into the simulations. Complete details on the modeling approach, as well the technical and quantitative aspects of the model, are provided by USEPA (2004g, 2004h).[8] Post-processing of AQUATOX results were performed using Statistica v.7.1 statistical software. Observed differences between treated and control results were marked significant at $p \leq 0.05$.

The results of the fate and transport component of AQUATOX indicate good agreement between the model and Integral's refined surface water modeling described above. Based on the predicted aquatic persistence of permethrin, two peak concentrations were observed on the dates of applications followed by subsequent rapid drop-offs to nominal concentrations comparable to control conditions. No long-term persistence of permethrin was predicted. Good agreement was also reached between the model's predicted 14-day average concentration of 0.018 µg/L and the 14-day average concentration of 0.016 µg/L predicted under the refined surface water Level 2 risk assessment.

[8] Additional details on AQUATOX version 3.14 will be provided by USEPA upon completion of beta testing and review.

No long-term significant differences in abundance were observed among treated and control organisms. Summary descriptive statistics of annual abundance predictions for treated (organisms denoted with a "P") and control (organisms denoted with a "C") simulations are presented in Table 7-30. No long-term differences in the predicted annual abundances of Daphnia and copepods under treated and control simulations were observed. Some short-term reductions were predicted for Daphnia in the treated simulation, with recovery to pretreatment levels occurring within one to two months. No significant annual differences in abundances were observed for chironomids. Amphipods under treated conditions had a slightly lower average annual abundance than that predicted for the control (i.e., 1.0 versus 1.7 g/sq.m), but this difference was not statistically significant (see Figure 7-8). Some short-term reductions were predicted for both chironomids and amphipods in the treated simulation, with recovery to pretreatment levels occurring within two months to 10 weeks. No long-term significant differences in abundances were observed for mussels, gastropods, and dragonflies (Odonata) under treated and control simulations. Some short-term reductions were predicted for Odonata in the treated simulation, with recovery to pretreatment levels occurring within two to three months, possibly due to the inclusion of modeled immigration. No long-term significant differences in abundances of fish (i.e., silversides, white perch, catfish, largemouth bass) under treated and control simulations were observed.

Table 7-30. Summary Descriptive Statistics of AQUATOX Predicted Annual Abundances for Organisms Evaluated Under Treated and Control Simulations.

Organism‡	Mean	Min	Max	-95% CL	+95% CL	Variance	Std	SE
Peri, Green (g/sq.m) P	4.9453	0.2000	20.3088	4.4961	5.3945	19.0960	4.3699	0.2284
Peri, Green (g/sq.m) C	4.7152	0.2000	19.3508	4.2609	5.1694	19.5294	4.4192	0.2310
Phyt, Blue-Gre (mg/L) P	0.000005	0.000003	0.00002	0.000005	0.000006	0.0000001	0.000003	0.0000001
Phyt, Blue-Gre (mg/L) C	0.000005	0.000003	0.00002	0.000005	0.000006	0.0000001	0.000003	0.0000001
Myriophyllum (g/sq.m) P	40.2479	0.1000	60.4107	38.8211	41.6747	192.6800	13.8809	0.7256
Myriophyllum (g/sq.m) C	40.1604	0.1000	62.3070	38.7406	41.5802	190.7791	13.8123	0.7220
Chironomid (g/sq.m) P	0.3961	0.0000	6.5725	0.2763	0.5158	1.3569	1.1649	0.0609
Chironomid (g/sq.m) C	0.4310	0.0000	8.5166	0.2976	0.5644	1.6843	1.2978	0.0678
Amphipod (g/sq.m) P	1.0445	0.0434	5.8178	0.9312	1.1577	1.2135	1.1016	0.0576
Amphipod (g/sq.m) C	1.7498	0.0421	8.0540	1.5497	1.9500	3.7919	1.9473	0.1018
Daphnia (mg/L) P	0.0024	0.0000	0.0300	0.0018	0.0030	0.0000	0.0056	0.0003
Daphnia (mg/L) C	0.0024	0.0000	0.0300	0.0019	0.0029	0.0000	0.0047	0.0002
Copepod (mg/L) P	0.0015	0.0000	0.0359	0.0010	0.0021	0.0000	0.0051	0.0003
Copepod (mg/L) C	0.0013	0.0000	0.0359	0.0008	0.0018	0.0000	0.0049	0.0003
Mussel (g/sq.m) P	0.4686	0.0802	2.0000	0.4183	0.5188	0.2388	0.4887	0.0255
Mussel (g/sq.m) C	0.4592	0.0763	2.0000	0.4085	0.5099	0.2433	0.4932	0.0258
Gastropod (g/sq.m) P	3.5436	1.0000	6.1142	3.3901	3.6970	2.2280	1.4926	0.0780
Gastropod (g/sq.m) C	3.5318	1.0000	5.8914	3.3809	3.6827	2.1556	1.4682	0.0767
Odonata (g/sq.m) P	0.1062	0.0239	0.4479	0.0947	0.1176	0.0125	0.1118	0.0058
Odonata (g/sq.m) C	0.1015	0.0202	0.4479	0.0897	0.1132	0.0130	0.1142	0.0060
Silverside (g/sq.m) P	4.4197	0.8521	7.9172	4.1978	4.6417	4.6623	2.1592	0.1129
Silverside (g/sq.m) C	3.1173	1.2624	6.1384	2.9892	3.2453	1.5520	1.2458	0.0651
White Perch (g/sq.m) P	0.7124	0.2428	2.0193	0.6604	0.7643	0.2556	0.5055	0.0264
White Perch (g/sq.m) C	0.6826	0.2304	2.0193	0.6291	0.7361	0.2705	0.5201	0.0272
Catfish (g/sq.m) P	0.5916	0.5000	0.6293	0.5868	0.5963	0.0021	0.0463	0.0024
Catfish (g/sq.m) C	0.5910	0.5000	0.6311	0.5863	0.5958	0.0021	0.0461	0.0024
Largemouth Bass - YOY (g/sq.m) P	1.5971	0.9482	2.7340	1.5411	1.6531	0.2965	0.5445	0.0285
Largemouth Bass - YOY (g/sq.m) C	1.7957	1.0000	2.8195	1.7358	1.8557	0.3402	0.5832	0.0305
Largemouth Bass - A (g/sq.m) P	4.6181	0.5000	6.9188	4.3676	4.8687	5.9402	2.4372	0.1274
Largemouth Bass - A (g/sq.m) C	4.9589	0.5000	7.5089	4.6829	5.2349	7.2114	2.6854	0.1404

Notes:
‡ = Summary information presented for organisms present in surface water receiving permethrin (denoted as "P") and organisms present under control surface water (denoted "C").

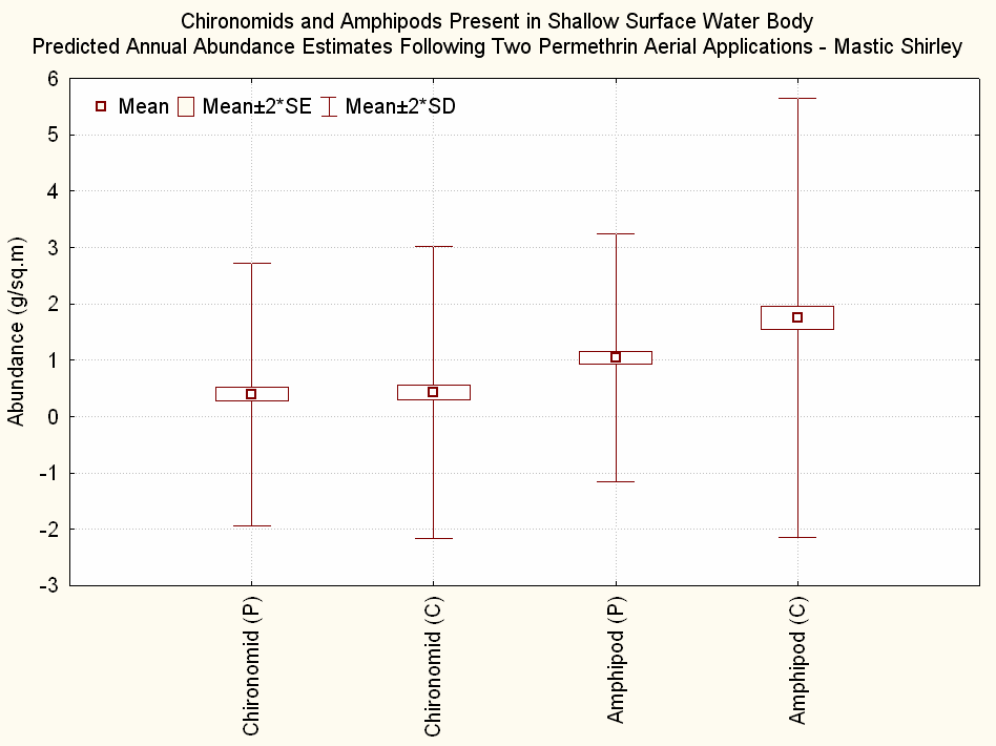

Figure 7-8. AQUATOX Predicted Annual Abundances for Chironomids and Amphipods in Treated and Control Simulations.

Based on the results of Integral's refined surface water modeling, some limited risks were predicted for aquatic invertebrates following aerial application to shallow surface waters in Mastic-Shirley. However, the results of the AQUATOX modeling indicate that although some short-term impacts to individual invertebrates may be possible, longer-term population-level impacts are not predicted. If population-level impacts are not predicted, then commensurate community-level impacts are, by proxy, also not predicted. Further, if no impacts are observed within one year, impacts attributable to application across years are unlikely. The absence of longer-term population and community level impacts for aquatic invertebrates predicted using AQUATOX is consistent with the findings previously summarized by Read (2001) for Minnesota wetlands, reported by Jensen et al. (1999) for seasonal wetlands in California, and reported by this project for wetlands and marshes in Suffolk County exposed to vector control pesticides.

Thus, ecological modeling suggests that pyrethroids will not have impacts on aquatic life beyond some potential short-term effects under limited conditions, for one compound only. The

potential impacts for malathion, although estimated to be possible under a broader set of conditions and potentially to a greater degree than the permethrin impacts, may similarly be limited.

Uncertainty Considerations

The results of the refined surface water modeling presented above and in ERA Appendix E (Cashin Associates, 2005c) indicate that potential aquatic risks may exist following the application of certain agents, particularly for aquatic invertebrates residing in shallow water settings (e.g., wetlands or small ponds). To gain further insight on the magnitude of these risks and their probability of occurrence, a Monte Carlo Analysis was conducted to evaluate variability and uncertainty in the models used to estimate surface water exposure concentrations.

Monte Carlo analysis is a method of estimating the probability of a model result given variability and/or uncertainty in the underlying inputs of the model. Monte Carlo analysis is frequently used in risk assessment to provide a more complete understanding of potential risk. Traditional risk assessments rely on single point estimates of model inputs and do not explicitly evaluate the intrinsic variability or uncertainty in those inputs. Such risk assessments, also referred to as deterministic risk assessments, provide a single point estimate of an exposure or risk result. Monte Carlo, or probabilistic risk assessments, explicitly account for intrinsic variability and uncertainty in model inputs and provide a distribution of possible exposure or risk estimates. Probabilistic risk assessments provide a more complete understanding of a risk estimate by offering insight on both the magnitude and probability of risk along the entire continuum of the risk estimate distribution.

Use of Monte Carlo and other varieties of uncertainty analysis procedures have seen increased use in risk assessment for both fate and transport exposure and toxicity-based evaluations. For example, Spurlock (2003) used Monte Carlo techniques to predict dissolved concentrations of pyrethroids in surface waters of California and demonstrated the importance of variability and measurement uncertainty in sediment partitioning. Other recent notable examples include the work of Citra (2004), Giddings et al. (2001), Maund et al. (2001), USEPA (2001c), Travis and Hendley (2001), Hall et al. (2000), ECOFRAM (1999), and Webster et al. (1998).

The focus of this Monte Carlo analysis was on the permethrin. Permethrin was selected for evaluation because some aquatic life risks were predicted in the Level 2 risk assessment and

because synthetic pyrethroids have been relied upon by Suffolk County as part of past vector control activities. Although malathion aquatic life risks were also predicted, under SCVC practice, it is typically not the primary choice for use, and it is anticipated that it will not be used at the same frequency or extent throughout the county as are synthetic pyrethroids.

The evaluation addresses variability and uncertainty in modeled 14 day average dissolved surface waters concentrations in shallow wetlands following aerial applications in the Mastic-Shirley study area. This scenario was selected based upon the results of the ecological risk assessment, which demonstrated the highest potential for chronic risks to aquatic life from synthetic pyrethroid use exists under this scenario.

The objective of the Monte Carlo analysis was to understand the degree to which variability in surface water model inputs influences the prediction of risk relative to the deterministic-based results. The Monte Carlo analysis was also performed to identify chief sources of uncertainty among surface water model inputs and characterize their associated influences upon the surface water model results.

The Monte Carlo analysis was conducted in accordance with established guidance available in the open literature, including that provided by Burmaster and Anderson (1994), Burmaster and Wilson (1996), Warren-Hicks and Moore (1998), Hoffman and Hammonds (1994), and Thompson et al. (1992), as well as guidance provided by USEPA (1997c, 2001c).

A 1-D Monte Carlo analysis was performed to incorporate and evaluate intrinsic variability in surface water model inputs.[9] A preliminary sensitivity analysis was performed to identify key inputs whose variability most greatly influences the surface water model predicted 14 day average dissolved concentration of permethrin. The key inputs were:

- chemical/physical properties which characterize the partitioning of permethrin between water, bed sediments and total suspended sediments in the water column
- RTP modeled deposition rates for permethrin.

With respect to chemical partitioning, the key inputs included the soil organic carbon-water partition coefficient (log K_{oc}) and the fraction of organic carbon (f_{oc}) present in benthic

[9] Analyses which address variability only are referred to as 1-D Monte Carlo analysis and those which address both variability and uncertainty are referred to as 2-D Monte Carlo analysis.

sediments, and total suspended solids (TSS) concentration in the water column. These inputs were used in the intermediary calculations of the bottom sediment-sediment pore water partition coefficient (K_{dbs}) and the suspended sediments-water partition coefficient (K_{dsw}) for permethrin. K_{dbs} and K_{dsw} in turn are used in the calculation of the fraction of permethrin present in the in the water column (f_{wc}) and in the final calculation of the dissolved phase concentration (C_{dw}) presented in Equations E-5 through E-8 of ERA Appendix E (Cashin Associates, 2005c). Variability among values used to represent log K_{oc} and f_{oc} for bed sediments and TSS was characterized using distributions (e.g., probability density functions [PDFs]). Information on and selected distributions used to represent each of the key inputs associated with permethrin partitioning are described in detail in ERA Appendix H (Cashin Associates, 2005c).

With respect to the RTP modeling, deposition rates were used as the starting point for characterizing the input of permethrin into a surface water body. Modeled surface water concentrations were therefore largely dependent on the RTP-modeled deposition rate. Variability among values used to represent modeled deposition rates was characterized using a distribution shape. Information on and the selected distribution used to represent the RTP modeled deposition rate for permethrin are described in detail in ERA Appendix H (Cashin Associates, 2005c).

Figure 7-9 presents the forecasted distribution for the 14 day average dissolved permethrin surface water concentration (in µg/l) for a wetland following aerial application in Mastic-Shirley.

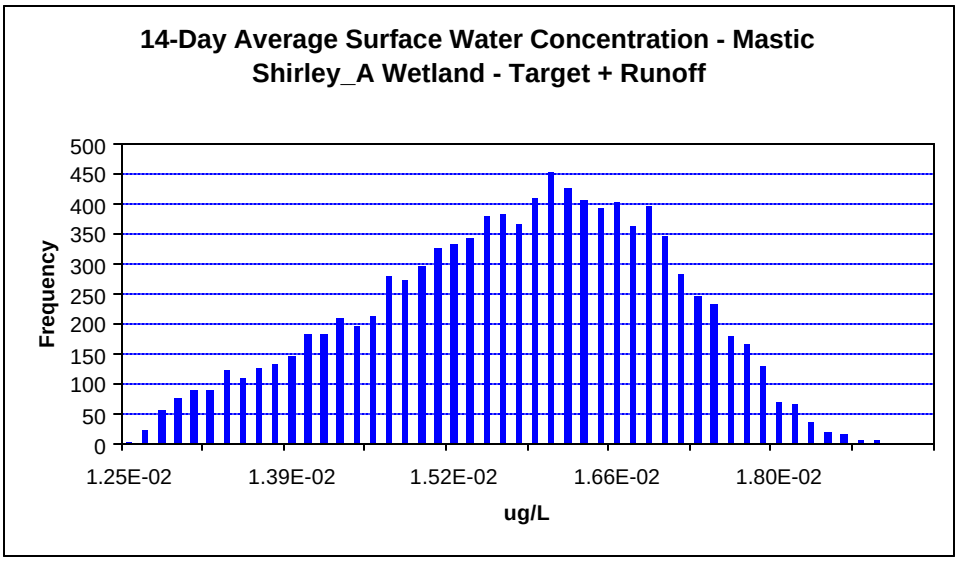

Figure 7-9. Monte Carlo-based Distribution Results for 14 Day Average Permethrin Surface Water Concentration

Thus, the results of the 1-D Monte Carlo analysis demonstrate that variability could exist in modeled estimations of the 14 day average dissolved permethrin surface water concentration.

Overall, agreement was reached between the deterministic-based concentration of 1.59E-02 µg/L used in the risk assessment and the 50th percent of the Monte Carlo-based results. This indicates that the deterministic based result suitably represents median or central tendency predicted concentrations of permethrin in surface water.

Based on the Monte Carlo analysis, concentrations could range by a factor of 1.3 lower (based on lower end values of 1.24E-02 µg/L) to a factor of 1.2 higher (based on the higher end values of 1.98E-02 µg/L) than the deterministic-based concentration used in the risk assessment. However, this range of possible exposure concentrations predicted under the Monte Carlo analysis is considered to be relatively narrow, and it is not anticipated to have any significant impact on the overall conclusions reached for permethrin in the ecological risk assessment.

A 2-D Monte Carlo analysis was performed to incorporate and evaluate intrinsic uncertainty in surface water model inputs. Based upon the preliminary sensitivity analysis, coupled with background information on the modeling approach used by RTP and the results of the 1-D

Monte Carlo analysis, potential uncertainty associated with the RTP modeled permethrin deposition rate also was evaluated.

Uncertainty results from lack of knowledge about a given input's true value. With respect to a modeled deposition rate, uncertainty exists with respect to how representative a single modeled rate may be for all atmospheric conditions and dependent deposition rates within a given study area.

For the purposes of conducting surface water modeling, variability in the RTP modeled deposition rate for permethrin was addressed in the 1-D Monte Carlo analysis. However, it is apparent that the deposition rate contains elements of both variability and uncertainty. Although a distribution was used to account for variability in the permethrin deposition rate, the parameters defining the distribution (e.g., mean, standard deviation) may be uncertain with respect to their representativeness. Model inputs which contain elements of both variability and uncertainty, such as the RTP modeled deposition rate for permethrin, are referred to as second order assumptions. In order to understand the effects of the potential uncertainty associated with the RTP modeled deposition rate for permethrin, a 2-D Monte Carlo analysis was performed.

In the second order assumption for the RTP modeled deposition rate, a series of second order distributions were used to characterize the parameters of the variability distribution used in the 1-D Monte Carlo analysis. Distributions for both the mean and standard deviations were developed based on the range of average values and standard deviations for all study areas modeled by RTP. Figure 7-10 presents the forecasted trend chart for the 14-day average dissolved permethrin surface water concentration for a wetland following aerial application in Mastic-Shirley.

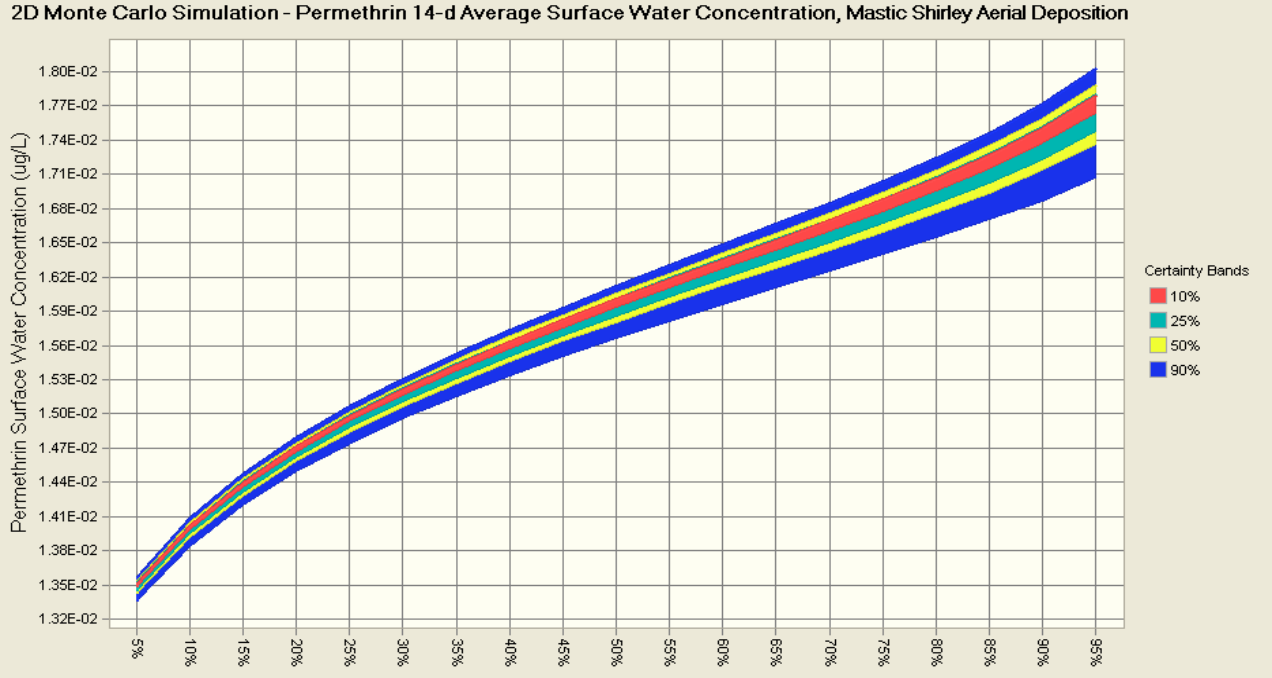

Figure 7-10. 2-D Monte Carlo Trend Chart with Accompanying Certainty Bands for 14 Day Permethrin Surface Water Concentration

Figure 7-10 depicts certainty bands associated with predicted percentiles of the modeled 14 day concentration. The lower percentile predictions (to the left) show a narrower range of uncertainty, while those for higher percentile predictions (to the right) show relatively larger bands of uncertainty. At the 5th percent, surface water concentrations could vary by a factor of 1.02. At the 95th percent, surface water concentrations could vary by a factor of 1.1. This indicates that upper-bound estimates of surface water concentrations when seen as a function of modeled deposition rates are prone to greater degrees of uncertainty than lower-bound estimates.

The second order distributions used to characterize uncertainty are based upon RTP modeled results for all study areas. To the extent that these results are representative of actual deposition rates occurring in Suffolk County, the above analysis indicates that in the case of upper-bound estimates, the surface water models are prone to relatively higher degrees of uncertainty compared to lower-bound estimates. While relatively higher uncertainties may exist in the upper-bound estimates, the corresponding estimated concentrations would only be a factor of 1.1 higher or lower, which is considered negligible. However, if the RTP modeled results cannot be considered representative of actual deposition rates, it is likely that much larger certainty bands would need to be prescribed to account for higher degrees of uncertainty in the surface water

modeling, as well as in the attendant predicted ecological risks. If not representative, uncertainty in deposition rates would likely result in upper-bound estimated surface water concentrations and attendant risks that could vary to a much higher, albeit unknown, extent.

Additional information on potential ecological impacts from pyrethroids became available following completion of the quantitative risk assessment (October, 2005). One was an article discussing the potential for toxic impacts to grass shrimp (*Palaemontes pugio*), which was one of the test organisms used in the Long-Term Plan Caged Fish experiment. It was found that grass shrimp were more sensitive to synergized resmethrin (as Scourge) than to unsynergized resmethrin, in that the LC_{50} for Scourge was nearly an order of magnitude lower than that of resmethrin, and that larvae were more sensitive than adult shrimp, so that the LC_{50} for Scourge for larvae was half that of adults. For adult shrimp, using 24 hour exposures resulted in much less mortality (the LC_{50} was five times higher than a 96 hour exposure), but larval shrimp were equally affected for shorter and longer durations. Adding sediment to the test made the resmethrin much less toxic, causing LC_{50}s to be 50 times higher (Key et al., 2005). The concentrations measured for even the lowest LC50 for Scourge were higher than those measured in all but some of the initial samples during the Caged Fish experiment. Key et al. pointed out that resmethrin concentrations should be expected to decrease rapidly in natural settings; their experiment emphasized the apparent importance of sediments and other organic material in the water column in making the resmethrin unavailable biologically, but that larval grass shrimp may be at risk for even relatively short exposures.

On February 4, 2006, Science News discussed the presentations made at the Society of Environmental Toxicology and Chemistry meeting in November, where one session focused on pyrethroids and their potential environmental impacts. A paper discussing the quantitative risk assessment from this project had been presented at that meeting (Preziosi et al., 2005). While that paper suggested that pyrethroid use in mosquito control presents little potential for environmental impact (as discussed above), other presentations disagreed with the finding that pyrethroids do not have environmental impacts (Raloff, 2006).

Most impacts associated with pyrethroid use are related to those pyrethroids used for at-home or agricultural pest control. With the exception of permethrin (which is probably the least toxic to aquatic organisms of all of the pesticides used for agricultural purposes), the pyrethroids that

were focused on for their potential to cause environmental impacts are not those proposed for use in the Long-Term Plan. The pyrethroids found to be of most concern were bifenthrin, cypermethrin, cyfluthrin, and esfenvalerate. Evidence was presented that showed why fish have difficulty detoxifying pyrethroids, and that other environmental stressors (such as cold weather for cold-blooded animals such as lizards, or viral infections in salmon) made the pesticides more toxic (Amweg et al., in press; Amweg et al., 2005; Clifford et al., 2005; Talent, 2005; Weston et al., 2005). USEPA announced that it would be completing a "comparative assessment" of pyrethroids in December, 2006 (Raloff, 2006).

A white paper published by the San Francisco Estuary Institute also found the potential for impact from pyrethroids. Permethrin was the only one of the Suffolk County pyrethroids discussed, as the focus was on agricultural pesticides used in the San Joaquin Valley and Sacramento Delta. It pointed out that other pyrethorids can be much more potent than permethrin, as cypermethrin is 20 times more toxic. The paper concluded that, for the purposes of determining impacts from agricultural uses to fish populations in San Francisco Bay, that not enough relevant toxicity information existed to support a quantitative risk assessment. The available information was suggestive that impacts could occur; however, neither exposure nor toxicity information was specific enough to determine if impacts were in fact occurring. In addition, important information regarding interactions between chemicals, or synergistic effects from environmental stressors, which have been suggested to be important, are also lacking (for this setting in particular) (Oros and Werner, 2005).

Table 7-31 summarizes the findings of the ecological risk assessment for the adulticide compounds.

Table 7-31. Summary of the Ecological Risk Assessment for Adulticides

Agents Considered	Terrestrial Birds, Mammals, Reptiles	Terrestrial Insects	Aquatic Life	Comments	Conclusion in Risk Assessment	Role in Management Plan
Adulticides						
Resmethrin	No risk*	Risks to non-target insects, such as butterflies, bees, dragonflies; all locations	No risk*	Terrestrial insect risks used honeybees as surrogate. Endpt was maintenance of abundance.	Terrestrial insect risks can be mitigated by timing applications approp.	Primary material for truck and aerial ULV, based on effectiveness and results of risk assessment.
Sumithrin	No risk*	Risks to non-target insects, such as butterflies, bees, dragonflies; all locations	No risk*	Terrestrial insect risks used honeybees as surrogate. Endpt was maintenance of abundance	Terrestrial insect risks can be mitigated by timing applications approp.	Primary material for hand held ULV. Would be first choice if resmethrin cannot be used.
Permethrin	No risk*	Risks to non-target insects, such as butterflies, bees, dragonflies; all locations	Only chronic risk to individual aquatic insects/larvae and crustaceans in shallow water (e.g., daphnid, opossum shrimp, mayfly)	Terrestrial insect risks used honeybees as surrogate. Endpt was maintenance of abundance	Terrestrial insect risks can be mitigated by timing applications approp. Aquatic risks will not result in community level impacts	Primarily will be used as an alternative for the other pyrethroids, due to setbacks and higher risks estimated in risk assessment.
Malathion	No risk*	Risks to non-target insects, such as butterflies, bees, dragonflies; all locations	Acute and chronic risk to individual aquatic insects and crustaceans in shallow water bodies (e.g., stonefly, amphipod, mysid shrimp)	Terrestrial insect risks used honeybees as surrogate. Endpt was maintenance of abundance	Terrestrial insect risks can be mitigated by timing applications approp. Aquatic, community level impacts not expected	Since a different class than the pyrethroids, could be used if pyrethroid resistance becomes an issue. Label restrictions make it less useful for ULV and risk assessment indicates higher risk.

Agents Considered	Terrestrial Birds, Mammals, Reptiles	Terrestrial Insects	Aquatic Life	Comments	Conclusion in Risk Assessment	Role in Management Plan
Degradates						
Malaoxon	NA	NA	NA	Not quantitatively evaluated due to lack of exposure, fate and toxicity data	NA	
Isomalathion	NA	NA	NA	Not quantitatively evaluated due to lack of exposure, fate and toxicity data	NA	
Synergist						
PBO	No risk*	Risks to non-target insects, such as butterflies, bees, dragonflies; all locations	No risk*	Based on evaluation of PBO containing products	Terrestrial insect risks can be mitigated by timing applications approp.	Combined with pyrethroids to maximize ULV effectiveness

* That is, predicted exposures were below levels of concern established by USEPA and/or others and so do not indicate that there is an increased risk of unacceptable ecological impacts from use of the pesticides under the conditions evaluated in this assessment

7.9.2.2 Qualitative Risk Assessment for Natural Pyrethrum

Pyrethrums are a mixture of pyrethrins, which are naturally occurring insecticides produced by certain species of the chrysanthemum plant. The flowers of the plant are harvested shortly after blooming. The flowers are either dried and powdered or the oils within the flowers extracted with solvents. The resulting pyrethrin containing dusts and extracts usually have an active ingredient content of about 30 percent (Extoxnet, 1996d). These active insecticidal components are collectively known as pyrethrins.

Pyrethrin is extremely toxic to aquatic life, such as bluegill, while it is slightly toxic to bird species, such as mallards. Toxicity increases with higher water temperatures and acidity. Additionally, these compounds are toxic to bees. Pyrethrum has a toxic potency similar to synthetic pyrethroids.

Exposure to pyrethrins can lead to coughing, wheezing and shortness of breath if inhaled. Symptoms noted in humans are more frequently related to allergic responses and irritation than neurotoxic effects (USEPA, 2005f). The pyrethrum extracts have been shown to cause allergic skin rashes and asthmatic responses. These allergic reactions may be in response to "impurities" present in the pyrethrum extract, since the more refined products available commercially today do not appear to have this property (National Pesticide Telecommunications Network, 1998).

Pyrethrum is inactivated and decomposed by exposure to light and air. It is also rapidly decomposed by mild acids and alkalis (Extoxnet, 1996d).

Overall, pyrethrum is likely to pose a lower risk than synthetic pyrethroids because it is more rapidly removed from the environment.

7.9.2.3. Special Considerations Regarding Human Breast Cancer

General Carcinogenicity Determinations

As discussed above, there is little evidence for carcinogenicity of the considered compounds. Some work has been done on pyrethrum and pyrethroids. In a two-year feeding study of rats, moderate to high doses of pyrethrum resulted in benign thyroid tumors in females, while high doses resulted in ovarian tumors and benign liver tumors (National Pesticide Telecommunications Network, 1998). USEPA has classified permethrin as a Group C Possible Human Carcinogen. The classification "Group C Possible Human Carcinogen" is assigned to a chemical when limited evidence of carcinogenicity in animals is available, but no human data is available. Pyrethrins have been classified by USEPA as likely to be a human carcinogen by the oral route (USEPA, 1999f). Deltamethrin and permethrin have both been determined to be "not classifiable" for carcinogenicity to humans by IARC (http://monographs.iarc.fr/monoeval/crthall.html).

Liver tumors developed in female mice fed high doses of piperonyl butoxide and in male mice fed middle and high doses (National Pesticide Telecommunications Network, 2000). However, in other feeding studies conducted in rats alone or in both rats and mice, investigators found no evidence of carcinogenicity for piperonyl butoxide (Butler et al., 1998; National Pesticide Telecommunication Network, 2000). USEPA has classified piperonyl butoxide as a Group C Possible Human Carcinogen (USEPA, 1999f).